National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

Compilation start date:	16 May 2013
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This compilation has been split into 22 volumes

Volume 1:	**sections 1–6, Schedules A and B**
Volume 2:	Schedule B1
Volume 3:	Schedule B2
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Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

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Introductory note

Section 14 of the National Environment Protection Council Act 1994 and the equivalent provision of the corresponding Act of each participating State and Territory provides for the making of Measures by the National Environment Protection Council and the matters to which they may relate. This Measure relates to the matters set out in paragraph 14(1)(d).

The Measure is to be implemented by the laws and other arrangements participating jurisdictions consider necessary: see Section 7 of the Commonwealth Act and the equivalent provision of the corresponding Act of each participating State and Territory.

Preliminary

1 Citation

This Measure may be cited as the National Environment Protection (Assessment of Site Contamination) Measure 1999.

2 Commencement

This Measure commences on the date of gazettal of this Measure.

3 Definitions

This clause defines particular words and expressions used in this Measure. Definitions of other terms that are used in particular guidelines in Schedule B are set out in the relevant guidelines.

In the context of this Measure the use of the word "should" does not imply obligation, but rather provides for general guidelines for the assessment of site contamination.

In this Measure, unless the contrary intention appears:

Agency means a body or bodies of a participating State or a participating Territory which that State or Territory has nominated for the purposes of this Measure.

Assessment of site contamination means a set of formal methods for determining the nature, extent and levels of existing contamination and the actual or potential risk to human health or the environment on or off-site resulting from that contamination.

Background concentrations means the naturally occurring, ambient concentrations of substances in the local area of a site.

Chemical substance means any organic or inorganic substance, whether liquid, solid or gaseous.

Commonwealth Act means the National Environment Protection Council Act 1994 of the Commonwealth.

Contamination means the condition of land or water where any chemical substance or waste has been added as a direct or indirect result of human activity at above background level and represents, or potentially represents, an adverse health or environmental impact.

Ecological Risk Assessment is a set of formal, scientific methods for defining and estimating the probabilities and magnitudes of adverse impacts on plants, animals and/or the ecology of a specified area posed by a particular stressor(s) and frequency of exposure to the stressor(s). (Stressors include release of chemicals, other human actions and natural catastrophes).

Epidemiology is the study of the distribution and determinants of disease in human populations.

Health Risk Assessment is the process of estimating the potential impact of a chemical, biological or physical agent on a specified human population system under a specific set of conditions.

Health Risk Management is the process of evaluating and implementing appropriate options to address risks identified from health risk assessments. The decision making will incorporate scientific, social, economic and political information. The process requires value judgements eg. on the tolerability and reasonableness of costs.

Investigation or Screening Level means the concentration of a contaminant above which further appropriate investigation and evaluation will be required.

Risk means the probability in a certain timeframe that an adverse outcome will occur in a person, a group of people, plants, animals and/or the ecology of a specified area that is exposed to a particular dose or concentration of a chemical substance, ie it depends on both the level of toxicity of the chemical substance and the level of exposure.

Site means the parcel of land being assessed for contamination.

Unless otherwise stated, a term used in this Measure and in the Commonwealth Act has the same meaning in this Measure as it has in the Commonwealth Act. The following terms are defined in subsection 6(1) of the Commonwealth Act:

Agreement means the Intergovernmental Agreement on the Environment made on 1 May 1992 between the Commonwealth, the States, the Australian Capital Territory, the Northern Territory and the Australian Local Government Association, a copy of which is set out in the Schedule to the Commonwealth Act.

Council means the National Environment Protection Council established by Section 8 of the Commonwealth Act and the equivalent provisions of the corresponding Acts of participating States and Territories.

National environment protection guideline means a guideline that gives guidance on possible means for achieving desired environmental outcomes.

National Environment Protection Measure (Measure) means a Measure made under section 14(1) of the Commonwealth Act and the equivalent provisions of the corresponding Acts of participating States and Territories.

Participating jurisdiction means the Commonwealth, a participating State or a participating Territory.

Participating State means a State:

(a) that is a party to the Agreement; and

(b) in which an Act that corresponds to the Commonwealth Act is in force in accordance with the Agreement.

Participating Territory means a Territory:

(a) that is a party to the Agreement; and

(b) in which an Act that corresponds to the Commonwealth Act is in force in accordance with the Agreement.

Head of power for making this Measure

4 Head of power

This Measure is made pursuant to section 14(1) of the Commonwealth National Environment Protection Council Act, and in particular, paragraph (d) of that section, and the equivalent provisions of corresponding Acts in participating States and Territories.

Purpose and desired environmental outcome of the Measure

5 Purpose and desired environmental outcome

(1) The purpose of the Measure is to establish a nationally consistent approach to the assessment of site contamination to ensure sound environmental management practices by the community which includes regulators, site assessors, environmental auditors, land owners, developers and industry.

(2) The desired environmental outcome for this Measure is to provide adequate protection of human health and the environment, where site contamination has occurred, through the development of an efficient and effective national approach to the assessment of site contamination.

Assessment of Site Contamination Policy Framework

6 Assessment of site contamination principles

The following principles should be observed in relation to the Assessment of Site Contamination:

(1) Individual responsibility

The primary responsibility for ensuring the assessment of site contamination rests with the States and Territories, excluding sites owned by the Commonwealth which are the responsibility of the Commonwealth.

(2) Implementation of jurisdictional responsibility

There should be a consistent approach to the assessment of site contamination across Australia but each participating jurisdiction may implement the necessary controls in its own manner.

(3) Prevention

Contamination, or further contamination, of a site should be prevented. Investigation or Screening Levels provided as part of this policy framework process should not be construed as desirable soil/water quality criteria or levels up to which contamination may be allowed to occur.

There should be no noticeable or measurable change in the characteristics of soil, or associated ground or surface waters. It is recognised that certain activities will lead to the addition of substances to the soil which raise the background levels of soils. These are valid and legitimate activities where they are undertaken in accordance with relevant laws and best practice guidelines.

(4) Regulatory control of site contamination

Contaminated soil and associated ground and surface waters should be categorised by the nature and concentration of contaminants and subject to appropriate controls over their use, storage, transport and ultimate disposal.

(5) Planning and development

Authorities of participating jurisdictions (at local and State government level) that consent to developments, or changes in land use, should ensure a site that is being considered for development or a change in land use, and that the authorities ought reasonably know if it has a history of use that is indicative of potential contamination, is suitable for its intended use.

(5A) Decommissioning of industrial activities

Industries, including mining and mineral processing industries, are responsible for ensuring that, when equipment on a site is dismantled or a site is otherwise decommissioned, appropriate measures are taken to leave the site in a safe and stable condition in order to prevent or, as far as practical, minimise adverse long-term environmental (physical, social and economic) impacts.

(6) Availability of site contamination information

Without detracting from any obligation of disclosure, which may exist at law, all relevant information on site contamination should be accessible to the community and particularly to those who need to make informed decisions, for example, potential land purchasers.

Without detracting from any obligation of disclosure, which may exist at law, the owner of a contaminated site should inform any person who proposes to purchase or lease the site, of information from the assessment of site contamination.

Prospective purchasers of land should also make appropriate enquiries to satisfy themselves regarding the condition of a site and any financial liabilities that may apply for the current use or the proposed future use of the land.

(7) Community engagement

If a community could reasonably have an interest in the potential site contamination, community engagement should start at an early stage of, and continue throughout, the process of assessment of site contamination.

(8) Cultural and spiritual significance

Due regard should be given to sites of cultural or spiritual significance, in particular, the significance that indigenous people attach to land.

(9) Education

Education programs should be implemented in the community, industry and all levels of government to raise awareness and understanding of site contamination issues, including the prevention of soil, air and water contamination.

(10) Site assessment process

The recommended general process for the assessment of site contamination is shown in Schedule A. The assessment should be conducted by professionals who have the relevant qualifications, competencies and experience.

(11) Human health

Human health should be a primary concern when assessing land use and exposure scenarios.

There should be appropriate occupational health and safety measures (including training) for personnel involved in assessment of site contamination.

Community health assessment and monitoring for specific health effects may be warranted where appraisal has indicated a significant risk of exposure to contamination.

(11A) Work health and safety

There should be appropriate work health and safety measures (including training) in place for any personnel involved in the assessment of site contamination, in accordance with the applicable work health and safety legislation.

(12) Environmental impact

The assessment of site contamination should include a consideration of risks to water resources and other ecological risks.

During the assessment, the on-site and off-site impacts of contaminants should be appropriately managed to prevent adverse impacts, particularly impacts relating to air emissions, surface water and groundwater.

(13) Data collection and chemical analyses

Site Assessors should develop data quality objectives and implement data quality assurance and quality control procedures that address sampling, contaminant identification and chemical analyses. These procedures should enable the evaluation of the precision and accuracy of results as part of the assessment of site risk. All other aspects of the risk assessment process should also be subject to quality assurance.

Chemical analyses should be performed using approved standard methods and should be performed by laboratories accredited for those analyses in the particular environmental medium. Field analytical methods should be performed by appropriately skilled personnel using approved standard methods.

Laboratories should be accredited for relevant analytical procedures by the National Association of Testing Authorities, Australia (NATA), or by an organisation recognised under NATA's Mutual Recognition Agreement (MRA) Network, or according to an appropriate standard dealing with laboratory quality assurance.

(14) Risk assessment

The initial assessment of human health risks and ecological risks may be undertaken by comparing levels of contaminants on the site with appropriate investigation or screening levels or, if necessary, by undertaking a site-specific risk assessment. The initial assessment may be followed by a more detailed assessment of human health risks and ecological risks.

An assessment of human health risks and ecological risks should, if practicable, take into account any additive, synergistic and antagonistic effects of mixing chemical substances.

(15) Objectives of assessment

The purpose of site assessment is to determine whether site contamination poses an actual or potential risk to human health and the environment, either on or off the site, of sufficient magnitude to warrant remediation appropriate to the current or proposed land use. In assessing that risk a balance is to be achieved between:

- optimising the current or intended use of the site; and
- adequately protecting human health and the environment.

The broader objective of assessment is to ensure:

- that the people of Australia enjoy the benefit of equivalent protection from air, water and soil pollution wherever they live;
- that the environmental values of water are maintained for future generations;
- that the capacity of the soil is maintained for future generations; and
- that there is consistency of approach between jurisdictions to aid government and business decision making.

(16) Attainment of environmental outcome

In general, to achieve the desired environmental outcome, the process of the assessment of site contamination should be placed within the context of the broader site assessment and management process. In particular, in assessing the contamination, the site assessor

and others should take into account the preferred hierarchy of options for site clean-up and/or management which is outlined as follows:

- on-site treatment of the contamination so that it is destroyed or the associated risk is reduced to an acceptable level; and
- off-site treatment of excavated soil, so that the contamination is destroyed or the associated risk is reduced to an acceptable level, after which soil is returned to the site; or,

if the above are not practicable,

- consolidation and isolation of the soil on site by containment with a properly designed barrier; and
- removal of contaminated material to an approved site or facility, followed, where necessary, by replacement with appropriate material;

or,

- where the assessment indicates remediation would have no net environmental benefit or would have a net adverse environmental effect, implementation of an appropriate management strategy.

When deciding which option to choose, the sustainability (environmental, economic and social) of each option should be considered, in terms of achieving an appropriate balance between the benefits and effects of undertaking the option.

In cases where no readily available or economically feasible method is available for remediation, it may be possible to adopt appropriate regulatory controls or develop other forms of remediation.

It should be emphasised that the appropriateness of any particular option will vary depending on a range of local factors. Acceptance of any specific option or mix of options in any particular set of circumstances is therefore a matter for the responsible participating jurisdiction.

(17) Specialist areas

In the assessment of site contamination the following sources are recognised as requiring specialised forms of assessment and initially, information should be sought from the relevant environmental protection agency for advice on assessing sites with:

(a) unexploded ordnance;
(b) radioactive substances;
(c) pathogenic materials and waste;
(d) contaminated sediments;
(e) explosive gas mixtures.

Consideration should be given to the physical, and/or chemical properties of the soil and associated ground and surface waters, including naturally elevated contaminant levels or acid sulfate characteristics, where they have the potential to adversely impact on the current or proposed land-use. In particular, the impact of such physical and/or chemical properties of the soil and associated ground and surface waters on the risk posed by such sites should include appropriate environmental impact assessment within relevant jurisdictional legislative requirements.

(18) Heritage sites

Heritage values should, wherever possible, be assessed prior to any physical assessment of contamination of a site. Where appropriate, advice should be sought from the local

representatives of the National Congress of Australia's First Peoples, the Australian Heritage Council, jurisdictional heritage bodies and local councils.

(19) Best practice

In observing the principles and guidelines in this Measure, each participating jurisdiction should give consideration to the most current advice and best practice.

Schedules to the Measure

7 Schedules

This Measure contains the following Schedules:

(1) Schedule A

Schedule A in this Measure identifies the general process for the Assessment of Site Contamination.

(2) Schedule B

Schedule B in this Measure identifies general guidelines for the Assessment of Site Contamination.

8 Stages of investigation

Schedule A shows the staged site assessment process indicating which general guidelines are applied to preliminary and detailed site investigations.

The preliminary investigation usually involves:

(a) establishing a site history to identify the characteristics of the site (such as the location and layout of the site, the building construction on the site, the geological setting, current and past activities at the site, current and past uses of the site, and heritage considerations); and

(b) inspecting the site; and

(c) interviewing representatives for the site.

Investigations are usually confined to areas where potentially contaminating activities have occurred and involve a site history-based sampling plan. The preliminary investigation and initial assessment of site contamination should consider the possibility of all forms of potential contamination based on past land use. The preliminary investigation should be sufficient to identify whether contamination exists on the site. Contamination may not be completely delineated at this stage.

A detailed investigation is required when the results of preliminary investigation are insufficient to enable site management strategies to be devised. Potential or actual contamination will need further evaluation. Potential contamination may have been indicated by the presence of unexpected underground structures (eg. underground fuel or chemical storage tanks) or by the presence of imported fill (eg. ash, odorous material or various types of refuse) or staining of soil. Actual contamination may have been detected in the form of contaminants which are not naturally occurring or as elements or compounds which are above background levels or exceed the applicable investigation or screening levels.

Depending on the proposed use and the results of initial site history investigations, the assessment of a site may involve both preliminary and detailed investigations.

Many site investigations proceed in multiple stages due to the complexity of the site and the discovery of unexpected contamination, or as investigation funds become available. Site investigators should obtain and consider all site information available to minimise the number of site visits and costs associated with the mobilisation of field investigation teams.

Reporting

9 Reporting requirements

(1) It is intended that each participating jurisdiction submit a report on the assessment of the implementation and effectiveness of the Measure, including compliance with the Measure, under Section 23 of the Commonwealth Act and similar provisions in the corresponding Acts of each participating State and Territory.

(2) It is intended that a report under subsection (1) be submitted to the Council by 30 September immediately after each reporting year.

(3) In this clause 'reporting year' means a year ending 30 June.

Review of the Measure

10 Review period

This Measure will be subject to a review every 10 years after the measure was last amended, or within any lesser period determined by the Council, which will consider:

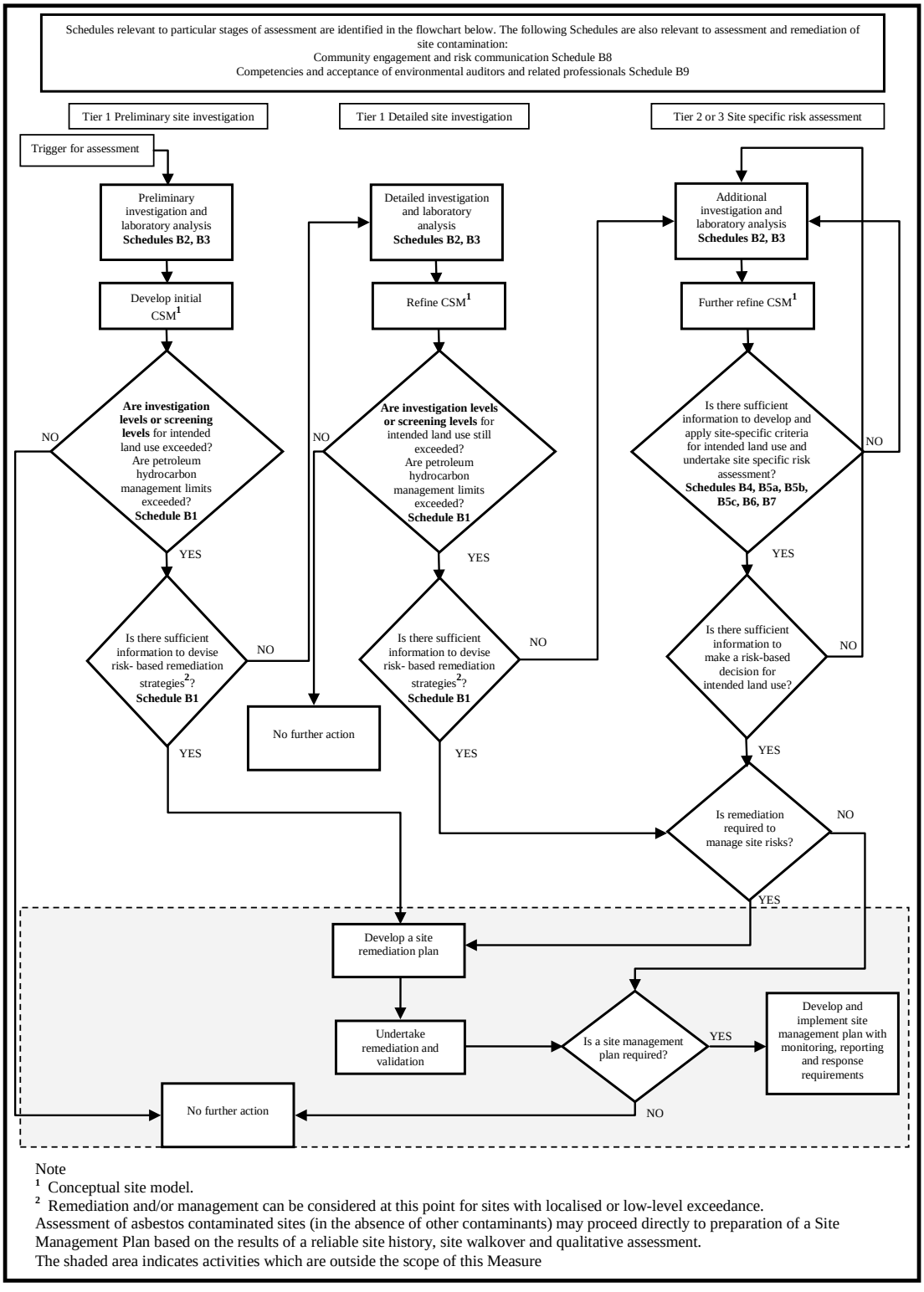
(1) the effectiveness of the Measure in achieving the desired environmental outcome set out within it;

(2) the resources available for implementing the Measure; and

(3) the need, if any, for amending the Measure (in accordance with the Act), including:

- whether any changes should be made to the Schedules; and
- whether any changes should be made to improve the effectiveness of the Measure in achieving the desired environmental outcome set within it.

Schedule A—Recommended general process for assessment of site contamination



Note

[1] Conceptual site model.

[2] Remediation and/or management can be considered at this point for sites with localised or low-level exceedance.

Assessment of asbestos contaminated sites (in the absence of other contaminants) may proceed directly to preparation of a Site Management Plan based on the results of a reliable site history, site walkover and qualitative assessment.

The shaded area indicates activities which are outside the scope of this Measure

Schedule B—General guidelines for the assessment of site contamination

The following general guidelines provide guidance on the possible ways of achieving the desired environmental outcome (PART 3 of the Measure) for the assessment of site contamination and should only be considered in relation to the assessment of site contamination.

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Schedule B1

GUIDELINE ON

Investigation Levels For Soil and Groundwater

Explanatory note

The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

The original Schedule B1 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document.

The National Environment Protection Council (NEPC) acknowledges the contribution of Queensland Department of Environment and Heritage Protection, Commonwealth Department of Health and Ageing, WA Department of Health, WA Department of Environment and Conservation, CRC Care and enHealth to the development of this Schedule.

Contents

Investigation levels for soil and groundwater

1 Introduction

1.1 Overview

The purpose of site assessment is to determine the human health and ecological risks associated with the presence of site contamination and to inform any remediation or management plan to make the site fit for the current or proposed land use. The appropriate use of investigation levels is an integral component of the assessment process.

This Schedule provides a framework for the use of investigation and screening levels. The framework is based on a matrix of human health and ecological soil and groundwater investigation and screening levels and guidance for specific contaminants. The derivation of health-based investigation levels is outlined in Schedule B7, and the risk assessment methodologies are detailed in Schedule B4. Schedule B5a outlines a risk-based framework for site-specific ecological risk assessment. The derivation of ecological investigation levels is outlined in Schedule B5c and the methodology is detailed in Schedule B5b. Reference is also made to the derivation and use of health and ecological screening levels in site assessment.

The selection of the most appropriate investigation levels for use in a range of environmental settings and land use scenarios should consider factors including the protection of human health, ecosystems, groundwater resources and aesthetics. The development of a conceptual site model is an essential element of site assessment and should inform the selection of appropriate investigation and screening criteria. A balance between the use of generic soil, soil vapour and groundwater criteria and site-specific considerations is essential practice in site assessment.

1.2 Prevention of site contamination

The National Environment Protection (Assessment of Site Contamination) Measure 1999 (NEPM) does not provide guidance on prevention of site contamination. Owners and occupiers of sites on which potentially contaminating activities are occurring are subject to the environmental protection legislation applying in each jurisdiction. Legislation provides for appropriate controls on potentially contaminating sources, including licensing of industrial activities, to minimise emissions and its application is the principal strategy for prevention of soil and groundwater contamination.

1.3 Specialised assessments

Specialised forms of assessment are required for sites affected by the following types of contaminants:

- radioactive substances
- unexploded ordnance
- pathogenic materials and waste
- explosive gas mixtures.

In situations where these materials occur on a site under assessment, guidance should be sought from the relevant jurisdictional environmental or health authority for assessment requirements. While the general principles of site assessment are applicable to these contamination types, compliance with specialised safety protocols and assessment guidance is essential to ensure protection of human health and the environment.

1.4 Acute hazards

Risk of explosion or other acute exposure hazards should be addressed immediately and are not within the scope of this guidance document.

Health effects can be broadly separated into acute and chronic effects. The distinction between acute and chronic exposure relates to the duration of exposure and the timing of onset of any health effects. Acute health effects occur within minutes, hours or days of a relatively short period of exposure, while

chronic health effects occur as a result of prolonged or repeated exposures over many days, months or years and symptoms may not be readily apparent.
Most contaminated land assessments will be focussed on chronic health effects; however, some sites may pose acute risks. Assessment of sites with petroleum hydrocarbon contamination will need to consider the potential for acute health risks and the risk of fire and explosion from the presence of light non aqueous phase liquids (LNAPLs).

Work health and safety issues should be considered for all sites and managed according to national and jurisdictional legislative requirements.

1.5 Mineralised areas

High levels of metals, metalloids and asbestos can be associated with ore bodies. Soils in mining areas may contain elevated levels of these materials due to natural mineralisation. Some urban areas may be affected by asbestos and various elements including lead, copper, zinc, cadmium and arsenic from the ore bodies, as well as activities associated with mining, smelting and metallurgical industries.

Due to the health concerns associated with asbestos, affected areas should be effectively managed in the short and long term. Naturally occuring asbestos is most likely encountered during exploration and mining operations. Management measures similar to those for free fibre usually apply.

These environments may require specific prevention measures and community awareness programs when human settlement has occurred, to enable appropriate precautions to be taken (for example, preventing the use of potentially contaminated soil or fill from a mining site for growing vegetables in the home garden, constructing driveways or filling private land and publicly accessible areas). Public information about preventing exposure to mineralised or contaminated soil is an essential component of public health programs to minimise community exposure to these contaminants.

Depending on the nature of the contaminants associated with the mining (or quarrying) activity, contaminated soil may be only one of a number of exposure pathways. Local health issues may be more effectively targeted by monitoring key community health parameters such as blood lead or by environmental monitoring of ambient air quality and dust.

2 Derivation of investigation and screening levels

2.1 Introduction

The purpose of this Schedule is to describe soil, soil vapour and groundwater criteria that can be used to evaluate potential risks to human health and ecosystems from site contamination. Investigation and screening levels are provided for commonly encountered contaminants which are applicable to generic land use scenarios and include consideration of, where possible, the soil type and the depth of contamination.

Investigation levels and screening levels are applicable to the first stage of site assessment. The selection and use of investigation and screening levels should be considered in the context of the iterative development of a conceptual site model (CSM) (refer Schedule B2 Section 4) to ensure appropriate evaluation of human health and ecosystem risks.

Site assessment should include consideration of all relevant human exposure pathways, ecological risks and risk to groundwater resources.

2.1.1 Definitions

Investigation levels and **screening levels** are the concentrations of a contaminant above which further appropriate investigation and evaluation will be required.

Investigation and screening levels provide the basis of Tier 1 risk assessment. A Tier 1 assessment is a risk-based analysis comparing site data with generic investigation and screening levels for various land uses to determine the need for further assessment or development of an appropriate management strategy. The application of investigation and screening levels is subject to a range of limitations.

Ecological investigation levels (EILs) have been developed for selected metals and organic substances and are applicable for assessing risk to terrestrial ecosystems. EILs depend on specific soil physicochemical properties and land use scenarios and generally apply to the top 2 m of soil. Further detail is provided in Section 2.5 and Schedule B5.

Ecological screening levels (ESLs) have been developed for selected petroleum hydrocarbon compounds and total petroleum hydrocarbon (TPH) fractions and are applicable for assessing risk to terrestrial ecosystems. ESLs broadly apply to coarse- and fine-grained soils and various land uses. They are generally applicable to the top 2 m of soil. Further detail on their use is provided in Section 2.6 and Warne (2010a, 2010b), available from the ASC NEPM Toolbox.

Groundwater investigation levels (GILs) are the concentrations of a contaminant in groundwater above which further investigation (point of extraction) or a response (point of use) is required. GILs are based on Australian water quality guidelines and drinking water guidelines and are applicable for assessing human health risk and ecological risk from direct contact (including consumption) with groundwater. Further information is provided in Section 2.8 and Schedule B6.

Health investigation levels (HILs) have been developed for a broad range of metals and organic substances. The HILs are applicable for assessing human health risk via all relevant pathways of exposure. The HILs are generic to all soil types and apply generally to a depth of 3 m below the surface for residential use. Site-specific conditions should determine the depth to which HILs apply for other land uses. Further detail is provided in Section 2.2 and Schedules B4 and B7.

Interim soil vapour health investigation levels (interim HILs) have been developed for selected volatile organic chlorinated compounds (VOCCs) and are applicable to assessing human health risk by the inhalational pathway. They have interim status pending further scientific work on volatile gas modelling from the sub-surface to building interiors for chlorinated compounds. Further detail on their use is provided in Section 2.3 and Schedule B4.

Health screening levels (HSLs) have been developed for selected petroleum compounds and fractions and are applicable to assessing human health risk via the inhalation and direct contact pathways. The HSLs depend on specific soil physicochemical properties, land use scenarios, and the characteristics of building structures. They apply to different soil types, and depths below surface to >4 m. Further detail on their use is provided in Section 2.4 and Friebel and Nadebaum (2011a, 2011b & 2011c).

'Petroleum hydrocarbon management limits' (**'management limits'**) are applicable to petroleum hydrocarbon compounds only. They are applicable as screening levels following evaluation of human health and ecological risks and risks to groundwater resources. They are relevant for operating sites where significant sub-surface leakage of petroleum compounds has occurred and when decommissioning industrial and commercial sites. Further detail on their use is provided in Section 2.9, including factors to be considered in determining the depth to which they apply.

2.1.2 Inappropriate use of investigation levels and screening levels

Investigation and screening levels are not clean-up or response levels nor are they desirable soil quality criteria. Investigation and screening levels are intended for assessing existing contamination and to trigger consideration of an appropriate site-specific risk-based approach or appropriate risk management options when they are exceeded. The use of these levels in regulating emissions and application of wastes to soil is inappropriate.

The use of investigation and screening levels as default remediation criteria may result in unnecessary remediation and increased development costs, unnecessary disturbance to the site and local environment, and potential waste of valuable landfill space. Similarly, the inclusion of an investigation and screening level in this guidance should not be interpreted as condoning discharges of waste up to these levels.

2.2 Health investigation levels

The health risk assessment methodology that forms the basis for calculation of HILs is provided in Schedule B4. The derivation of the HILs is presented in Schedule B7 (and appendices) and uses the *Australian exposure factor guidance* (enHealth 2012). The derivation of the HILs is illustrated by two worked examples for cadmium and benzo(a)pyrene (refer Schedule B7 Appendix B). The spreadsheet for calculating HILs is included in the ASC NEPM Toolbox (www.scew.gov.au/nepms/assessment-of-site-contamination.html).

The HILs are listed in Table 1A(1), found at the end of this Schedule.

HILs are scientifically based, generic assessment criteria designed to be used in the first stage (Tier 1 or 'screening') of an assessment of potential risks to human health from chronic exposure to contaminants. They are intentionally conservative and are based on a reasonable worst-case scenario for four generic land use settings:

- HIL A – residential with garden/accessible soil (home grown produce <10% fruit and vegetable intake, (no poultry), also includes children's day care centres, preschools and primary schools
- HIL B – residential with minimal opportunities for soil access includes dwellings with fully and permanently paved yard space such as high-rise buildings and flats
- HIL C – public open space such as parks, playgrounds, playing fields (e.g. ovals), secondary schools and footpaths. It does not include undeveloped public open space (such as urban bushland and reserves) which should be subject to a site-specific assessment where appropriate
- HIL D – commercial/industrial such as shops, offices, factories and industrial sites.

The land use scenarios are described in detail in Section 3 of Schedule B7. To make generic estimates of potential human exposure to soil contaminants, scientifically based assumptions are made about the environment, human behaviour, the physicochemical characteristics of contaminants, and the fate and transport of contaminants in soil within each of these land use categories. The HILs are derived by integrating these exposure estimates with toxicity reference values, that is, tolerable daily intakes (TDI), acceptable daily intakes (ADI), and reference doses (RfD), to estimate the soil concentration of a substance that will prevent exceedence of the toxicity reference value under the defined scenario. The toxicity reference values are generally based on the known most sensitive significant toxicological effect. Where toxicity reference values come from multiple sources, their underlying assumptions, defaults and science policy should be compatible and generally similar.

HILs establish the concentration of a contaminant above which further appropriate health investigation and evaluation will be required. Levels slightly in excess of the HILs do not imply unacceptability or that a significant health risk is likely to be present. Exceeding a HIL means further investigation is required and not 'risk is present, clean-up required'.

The HILs are referred to by regulators, auditors and consultants in the process of assessing soil contamination. HILs apply generally to the top 3 m of soil for residential use. Site-specific conditions should determine the depth to which HILs apply for other land uses.

HILs are not intended to be clean-up levels. The decision on whether clean-up is required, and to what extent, should be based on site-specific assessment triggered by an exceedence of the HIL. Health risk assessment is the primary driver for making site decisions. Other considerations such as practicality, timescale, effectiveness, cost, sustainability and associated ecological risk assessment are also relevant.

2.3 Interim HILs for volatile organic chlorinated compounds

Interim HIL soil vapour levels for specific volatile organic chlorinated compounds (VOCCs) have been developed (see Table 1A(2) at the end of this Schedule) to assess the vapour inhalation pathway (also known as the 'vapour intrusion' pathway when referring to indoor exposure). The derivation of the interim HILs is presented in Schedule B7 and Appendix A6. The methodology employs a simple though conservative approach using an attenuation factor that relates the concentration of a volatile contaminant in indoor air to the concentration in soil gas immediately below a building foundation slab.

The interim HIL values derived for volatile compounds are driven by the vapour intrusion pathway (that contributes >99% of the total risk when all pathways are considered). However, it is noted that there are limitations and uncertainties associated with the assessment of volatile contaminants on the basis of soil concentrations. As these limitations are significant for volatile organic chlorinated compounds, interim HILs for soil have not been derived. Rather it is recognised that where indoor/ambient air data cannot be collected (or the data is adversely affected by background sources), the most relevant approach to the assessment of this pathway is through the collection of soil vapour data. On this basis, interim HILs have been developed for soil vapour.

The interim HILs provide Tier 1 guidance for health risks from soil contamination sources and groundwater plumes associated with this group of compounds. The values may be applied for general site assessment and sub-slab environments for evaluation of potential health risks for the 0–1 m sub-slab profile. The interim HILs broadly apply to the same generic land use categories as do the HILs, though the values for residential A and B are combined as they are based on the same exposure conditions (i.e. the same amount of time spent indoors) for the vapour inhalation pathway. In addition, secondary school buildings should be treated as residential for the purposes of evaluating risks from vapour intrusion.

Biodegradation of VOCCs has not been included in the development of the interim HILs. The biodegradation approach developed for petroleum hydrocarbons (refer Section 2.4.10) is not applicable to the degradation of VOCCs as the mechanism by which degradation occurs is different for most chlorinated hydrocarbons compared with petroleum hydrocarbons.

2.4 Health screening levels for petroleum hydrocarbon compounds

2.4.1 Introduction

Site contamination by petroleum hydrocarbon compounds is frequently encountered. The complex mixtures of aliphatic and aromatic compounds that comprise petroleum hydrocarbon products present human health concerns predominantly through inhalation of vapours from contaminant sources and by direct contact with affected soils and groundwater. Assessment of petroleum impacts should include evaluation of risks via the groundwater pathway (e.g. consumption of contaminated groundwater that is not considered in the HSLs), the risk to groundwater resources and appropriate consideration of aesthetics. The application of relevant ecological and 'management' criteria for petroleum compounds is discussed in Sections 2.6 and 2.9.

Health Screening Levels (HSLs) for various petroleum hydrocarbon compounds were developed by the Cooperative Research Centre for Contamination Assessment and Remediation of the Environment (CRC CARE). The principal reference for the HSL methodology is Friebel and Nadebaum (2011a). In addition to the documentation of the methodology, a detailed application report (Friebel & Nadebaum 2011b) and a sensitivity analysis of the main parameter inputs ((Friebel & Nadebaum 2011c) are available.

Predictive modelling of sub-surface vapour movement in soil and penetration of building structures is a field of intensive data collection and research. The most recent research and derivation approaches adopted in developed international jurisdictions have been considered and adapted, as far as is practicable, for Australian conditions, to derive Tier 1 screening criteria for evaluating human health risk from petroleum hydrocarbons.

The HSLs' development was guided by a project advisory group with health, environmental, assessment and remediation, petroleum industry and regulatory expertise. A specialised technical working group provided technical support and review throughout the development process. The HSL methodology was subject to international peer review during its development.

Copies of the technical reports can be found in the ASC NEPM Toolbox. Additional information on the development phases of the project, including responses to peer review comments, can be found on the CRC CARE website:
http://www.crccare.com/publications/technical_reports/hsl_tech_report.html

Assessment of vapour risks is a specialist area. It is the responsibility of contaminated land professionals to become familiar with the limitations of the HSLs and their correct application in site assessment (Friebel & Nadebaum 2011a, 2011b, 2011c).

2.4.2 HSL methodology

The HSLs were developed to be protective of human health by determining the reasonable maximum exposure from site sources for a range of situations commonly encountered on contaminated sites. As there are many parameter inputs to the methodology, very conservative assumptions have not been made for every parameter as this would result in an unrealistic result arising from the compounding of conservatism. Typically the parameter values selected correspond to the mean or median of the available information, with some parameters corresponding to the 95th percentile. For further information on the rationale for each parameter selected, refer to Friebel and Nadebaum (2011a).

The HSLs apply to the same land use settings as for the interim HILs for VOCCs and include additional consideration of soil texture and depth to source to determine the appropriate soil, groundwater and soil vapour criteria for the exposure scenario. As with all modelling approaches, the assumptions made regarding the exposure scenario limit the extent of their reasonable application. The main limitations for the HSLs are summarised in Section 2.4.13.

HSLs for soil (Table 1A(3)), groundwater (Table 1A(4)) and soil vapour (Table 1A(5)) apply to exposure to petroleum hydrocarbons through the dominant vapour inhalation exposure pathway only. Direct contact HSLs have been developed for the incidental soil ingestion, dermal and inhalation exposure pathways. The direct contact HSLs are generally not the risk drivers for further site assessment for the same contamination source as the HSLs for vapour intrusion. Direct contact exposure should be considered where relevant to the site-specific scenario e.g. an external source in near-surface soils in a residential or recreational setting. Further details can be found in Friebel and Nadebaum (2011a, 2011b, and 2011c).

There are many site-specific, soil-specific and building-specific variables that affect the level of the HSLs and these factors should be considered in the site assessment. Detailed information on the model inputs and assumptions (for example, soil properties, sub-slab attenuation factor, organic carbon content, chemical properties, building parameters) and overall limitations are provided in Friebel and Nadebaum (2011a). A sensitivity analysis was used to evaluate the effect that these parameters have on the derived HSLs (Friebel & Nadebaum 2011c).

A review of vapour models was undertaken by CSIRO as a precursor project to the development of the HSLs (Davis et al. 2009c). As a result of this review, a modified Johnson and Ettinger vapour exposure model (US EPA 2004) was selected to derive HSLs for the vapour inhalation pathway. The model has been used assuming a finite source for soils equivalent to a source thickness of 2 m which avoids the extreme conservatism associated with assuming an infinite source and reflects empirical field observations. For groundwater and soil vapour, an infinite source (i.e. steady state model) has been assumed as replenishment of vapours may occur by contaminated groundwater flowing beneath the site.

It is noted that the Johnson and Ettinger model and other similar vapour intrusion models do not adequately address vapour risk issues where there are preferential vapour migration pathways, where the building structure extends into a saturated contaminated zone (i.e. into the groundwater table) or where biodegradation is of significance (see section 2.4.10 for further information).

The soil and groundwater HSLs are based on three-phase equilibrium theory and soil vapour is limited by the maximum solubility limit of the chemical in the soil pore water phase or the groundwater. The soil saturation concentration of a particular contaminant is the condition where pore water is at its solubility limit and soil vapour is at the maximum vapour concentration. When a calculated HSL in soil or groundwater exceeds this limit, the vapour in the soil or above groundwater cannot result in an unacceptable vapour risk and is denoted as NL (not limiting) in the HSL tables (Tables 1 A(3) – 1A(5)). Soil vapour HSLs are based on the vapour pressures of individual chemicals. Calculated soil vapour HSLs that exceed the possible maximums are similarly denoted as NL.

The HSLs have been derived using accepted approaches to assessment for non threshold (cancer) risk and threshold (non-cancer) risk. Exposure factors for the individual carcinogenic and non-carcinogenic compounds of concern were derived from a near-final draft of enHealth (2012).

2.4.3 Sub-slab to indoor air attenuation factor

Unlike the derivation of the soil vapour interim HILs, the attenuation factor adopted for petroleum hydrocarbon compounds is not used directly to calculate indoor air concentrations from soil gas concentrations (or vice versa); rather it is used to calculate one of the many input parameters

(advective air flow) in the Johnson & Ettinger model. For further information refer to section 7.3.2 of Friebel and Nadebaum (2011a).

As for other input parameters, the selected value for the attenuation factor is based on a reasonable assumption rather than the maximum possible exposure and is equivalent to the median of the US EPA 2008 attenuation factor database (US EPA 2008) and lies within the 75th to 95th percentiles of the updated database published in 2012 (US EPA 2012). The selected value of 0.005 was considered to represent the upper value not affected by indoor air sources, background air or other confounding factors.

2.4.4 Petroleum fuel composition

The soil saturation and water solubility limits used in the derivation of the HSLs assume a fixed fuel composition based on fresh petrol and diesel fuels typical of those available in Australia. The HSLs may be applied to other fuel types (e.g. kerosene, aviation fuel and fuel oil) providing that the aliphatic/aromatic speciation is similar to that assumed in the derivation of the HSLs (80:20). Further information on these fuel types can be found in TPHCWG (1998). There are a number of fuel additives, such as MTBE and ethanol, for which HSLs have not been derived. Where these are identified as potential contaminants of concern, then a site-specific risk assessment for these chemicals should be considered.

The HSLs apply to petroleum contamination sources and are not applicable to pure compound solvents, as solubility limits incorporated into the HSLs were derived based on typical petrol and diesel fuel mixtures. Equivalent values to the HSLs applicable to pure compounds (rather than fuel mixtures) are available in Friebel and Nadebaum (2011a Appendix C).

2.4.5 The Total Recoverable Hydrocarbons analytical method

The Total Recoverable Hydrocarbons (TRH) method is recommended for the analysis of petroleum hydrocarbon compounds in soil. Detailed information is provided in Schedule B3.

The term TRH is equivalent to the previously used total petroleum hydrocarbons (TPH) and represents extracted biogenic (biological) and petrogenic (petroleum) hydrocarbons by selected solvents. The TRH analysis is non-specific and will extract organic compounds such as ethanol, biodiesel compounds (esterised long chain fatty acids), organic acids, sterols and n-alkanes from plant waxes, as well as petroleum hydrocarbons. The sample extraction process may also extract other industrial organic chemicals. When used in the context of a screening assessment for petroleum hydrocarbon contamination, TRH analyses are likely to be conservative when non-petroleum compounds are present.

The potential for inclusion of non-petroleum compounds in the results may be relevant for site-specific assessment of petroleum hydrocarbon contamination. For example, the TRH analytical results may be overly conservative if soil organic matter is unusually high, for example from heavy applications of mulch, manure, compost or other natural organic material, or the presence of other synthetic organic compounds which are extractable in the analytical process. To assess potential false positive results, it is recommended that equivalent soil from the site, unaffected by petroleum hydrocarbon contamination, is analysed for comparison.

Where there is reasonable doubt as to the nature of the contamination, the sample may be subjected to a silica gel clean-up and analysed by gas chromatography mass spectrometry (GC-MS) (or other appropriate analytical method) to assist with the identification of contamination of petroleum origin. In these cases, an analyst report should be obtained with an interpretation of the chromatogram and the nature and extent of contamination present in the sample.

2.4.6 Petroleum hydrocarbon compounds and fractions

HSLs have been developed for BTEX and naphthalene plus four carbon chain fractions based on the fractions adopted in the *Canada-wide standard for petroleum hydrocarbons (PHC) in soil* (CCME 2008). The fractions are listed in Table 1 below:

Table 1. HSL fractions and corresponding equivalent carbon range

Fraction number	Equivalent carbon number range
F1	C_6 – C_{10}
F2	>C_{10} – C_{16}
F3	>C_{16} – C_{34}
F4	>C_{34} - C_{40}

The HSLs are provided in Tables 1A(3) – 1A(5)).

BTEX results should be subtracted from the TRH C_6 – C_{10} analytical results for comparison with the HSL for F1. Likewise, naphthalene should be subtracted from >C_{10} – C_{16} for comparison with the HSL for F2.

Chemicals in the >C_{16}-C_{34} and >C_{34}-C_{40} fractions are non-volatile and therefore not of concern for vapour intrusion, however, exposure can be via direct contact pathways (dermal contact and incidental ingestion and inhalation of soil particles). Direct contact HSLs for these fractions can be found in Friebel and Nadebaum (2011a).

2.4.7 Soil texture

HSLs for soil, groundwater and soil vapour have been developed for sand, silt and clay soils based on the US soil texture classification system (Friebel & Nadebaum 2011a). The HSLs assume a uniform soil profile and the soil texture making up the greatest proportion of the soil profile should be used in selecting the appropriate HSLs (Friebel & Nadebaum 2011a and 2011b).

For Tier 1 soil assessment, the HSL classifications of sand, silt and clay may be broadly applied to the soil texture classification in Table A1 of Standard AS 1726.

Table 2. HSL soil classification and equivalent soil classification in AS 1726

HSL soil classification	AS 1726 Equivalent
Sand	Coarse-grained soil
Silt	Fine-grained soil - silts and clays (liquid limit <50%)
Clay	Fine-grained soil - silts and clays (liquid limit >50%)

Where there is reasonable doubt as to the appropriate soil texture to select, either a conservative selection should be made (i.e. select coarsest applicable grain size such as sand) or laboratory analysis carried out to determine particle size and hence soil texture sub-class (refer Section 7.3.1 in Friebel and Nadebaum 2011b). If particle size analysis is undertaken then laboratory measurement of

additional parameters used in site-specific risk assessment (such as soil moisture content, organic carbon content and saturation porosity - refer Friebel & Nadebaum 2011b for further information) could also be considered if further assessment is possible. If laboratory measurement is undertaken, sufficient samples should be obtained and analysed to determine a representative value for each soil unit of interest for the assessment.

2.4.8 Land use

The HSLs are derived for various depths to source and for the same generic land uses as for the HILs (described in detail in Schedule B7). The values for residential A and B are combined in the HSL tables as they are based on the same exposure conditions for the vapour inhalation pathway (i.e. the same amount of time spent indoors).

The HSLs are applicable to ground floor land use. If the vapour exposure is acceptable at ground level, it can be assumed that it is also acceptable for floors above ground level. For multistorey buildings where non-residential uses (e.g. car parking or commercial use) exist in a basement or at ground level, then land use category D (commercial/industrial) should be applied.

Any sensitive land uses e.g. childcare or day care centre will require application of HSL A irrespective of their planning zoning. Secondary school buildings (as opposed to secondary school grounds) should also be assessed using HSL A.

2.4.9 Adjusting HSLs to site-specific circumstances

The HSL methodology enables parameter inputs to be changed to more accurately reflect local soil, site or building conditions. Input parameters should be selected to be representative of long-term stable conditions and appropriate to the soil unit/aquifer of concern e.g. moisture content may vary seasonally and may also be different beneath buildings. Where insufficient data is available to establish a representative value, a conservative approach should be taken, for example, by assuming dry soil moisture conditions in sand. The HSL application and sensitivity documents (Friebel & Nadebaum 2011b, 2011c) provide further details. Jurisdictions may also adopt policies to vary the HSLs to account for local conditions.

For example, air exchange rates have been set at 0.6 building volumes/hr which may not be appropriate for buildings designed for tropical and cold climates. Similarly, soil moisture has a significant effect on penetration of volatiles into buildings.

The HSL derivation has assumed a slab-on-ground construction. Elevated buildings on concrete supports or timber poles with no direct floor contact with the soil and clear underfloor ventilation are at lower risk of penetration of volatiles and the risk decreases with the elevation of the floor above ground. The state of the slab will require consideration if it has deteriorated, as cracks can act as preferential pathways.

2.4.10 Biodegradation

Recent research on underslab biodegradation of petroleum hydrocarbon contamination is reported in Davis et al. (2009a and 2009b). This research identified that the following site conditions are conducive to biodegradation of petroleum hydrocarbon compounds in the sub-surface:

- the presence of oxygen at concentrations greater than 5% in soil vapour at a depth 1 m below the surface immediately adjacent to the concrete slab

and

- a maximum slab width of less than 15 m, with oxygen access on both sides of the slab for Tier 1 screening purposes. A distance of 7–8 m from the exposed soil at the slab boundary is considered the maximum lateral underslab penetration of oxygen.

It is noted that the measurement of oxygen in the soil profile can be difficult and care should be taken when using this data to support biodegradation.

If these conditions are fulfilled, biodegradation factors can be applied to the vapour intrusion HSLs as follows:

- factor of x10 for depths to source of 2 to <4 m and
- factor of x100 for depths to source of 4 m and greater where the vapour source strength is 100 mg/L (100,000 mg/m^3) or less.

The biodegradation factors above are not applicable for depths of less than 2 m. For the purpose of this NEPM, assessment including biodegradation of petroleum hydrocarbons is considered a Tier 1 activity.

Application of the biodegradation factors described above may result in levels of TPH, BTEX and naphthalene that are acceptable for human health risk from the vapour exposure pathway for the specific land use but which may not be acceptable for protection of the environment or water resources or from an aesthetics perspective. Site results should be considered with reference to relevant ecological and 'management levels'(refer Sections 2.5 and 2.9) which may become the predominant risk driver. Management levels should be applied after human health, ecological risks and risks to groundwater resources have been assessed.

2.4.11 Direct contact HSLs

Direct contact HSLs have been developed for exposure through dermal contact, incidental oral ingestion and dust inhalation and then combined as a single HSL for direct contact with soil (Friebel & Nadebaum, 2011a). For most site assessments, the direct contact HSLs are unlikely to become drivers for further investigation or site management as the values are significantly higher than most other soil screening levels and consequently have not been included here. There are situations where the combined vapour and direct contact pathways can make a difference to the outcome of the assessment. For further information on considering combined vapour and direct contact exposure, refer to Section 3.3 of Friebel and Nadebaum (2011b). The combined HSLs for direct contact can be found in Appendix A of Friebel & Nadebaum (2011a).

Contamination at the levels of the direct contact HSLs are likely to present unacceptable aesthetic considerations which should be addressed in accordance with the discussion in Section 3.6. Exposure to a contaminated surface (other than of short and temporary duration) at the levels of the direct contact HSLs may also cause an unacceptable short-term vapour exposure risk.

2.4.12 HSLs and multiple-lines-of-evidence approach

For an assessor to conclude that the vapour intrusion/emission pathways are unlikely to be active or to present a significant risk, the assessor should undertake a multiple-lines-of-evidence approach. This requires the assessor to present several reasoned lines of evidence as to why the pathway is considered inactive or is unlikely to present a significant risk.

The soil and groundwater HSLs provide the principal assessment criteria for open excavations (such as tank removal operations) while greater emphasis is placed on soil vapour HSLs in assessing potential vapour intrusion risks from hydrocarbon sources and groundwater plumes adjacent to or under buildings. In general, evaluating all contaminant phases will provide greater confidence in the outcomes of the site assessment.

Soil vapour measurements can provide a more accurate representation of vapour risks (compared with the soil and groundwater HSLs), depending on site-specific conditions e.g. where soil vapour can be measured directly under conditions that are relevant to the future or continuing use of the site. In high moisture conditions, however, such as occur within the capillary fringe or as a result of seasonal watertable fluctuations, it is not possible to obtain reliable soil vapour readings. In these conditions, consideration may be given to obtaining vapour headspace readings from appropriately constructed

groundwater monitoring wells fitted with a soil vapour monitoring cap that seals the groundwater well from the atmosphere.

Soil vapour measurements are also preferred where contaminated groundwater is present at less than 2 m below the ground or basement foundation, though in fine-grained soils the ability to obtain soil vapour measurements may be constrained by moisture conditions, as the thickness of the capillary fringe increases as the soil texture decreases.

Where the watertable rises seasonally to intersect basements or building foundations, indoor air measurements will be required to assess vapour risk. The assessment approach may also include soil vapour measurements taken in the dry season as part of a multiple-lines-of-evidence approach.

Additional information on vapour assessment and the multiple-lines-of-evidence approach is provided in Section 9.2 of Schedule B2 and Friebel and Nadebaum (2011a, 2011b).

2.4.13 Limitations of the HSLs

As with all generic screening levels, actual site-specific conditions may mean that the assumptions underpinning the derivation of the screening levels are not valid for the site and consequently a site-specific assessment will be required. The principal limitations applicable to the HSLs are listed in Table 3 below, together with suggested alternative assessment approaches.

Immediate action should be taken where potentially explosive or acutely toxic gas concentrations are present in buildings or in-ground services (e.g. utility trenches, sumps or drains) connecting a vapour source to a building. Emergency management actions, such as relocation of building occupants, should be implemented as necessary.

Table 3. Site scenarios where the application of the HSLs is limited and possible alternative assessment approaches

Site scenario	Alternative assessment approach
The identified contamination has an atypical petroleum composition	Site-specific risk assessment including assessment of cumulative effects of chemical constituents
Contaminated groundwater or LNAPL is entering or is in contact with a basement or building foundation	Consider indoor air sampling
Depth to groundwater impact is less than 2 m	Consider soil vapour measurements for vapour intrusion
The impacted soil source thickness is significantly greater than 2 m	HSLs may be conservative for thinner soil sources. For thicker soil sources, refer to Section 2.4.7 of the HSLs application document (Friebel & Nadebaum 2011b)
A preferential migration pathway is present that could connect a vapour source to a building interior	Site-specific assessment
Hydrocarbon odour present in buildings or in-ground services (not attributable to an indoor or ambient source) which indicates an active preferential migration pathway and potentially an immediate human health risk	Consider indoor air sampling or immediate action in the case of strong hydrocarbon odours

2.5 Ecological investigation levels

2.5.1 Introduction

Ecological investigation levels (EILs) for the protection of terrestrial ecosystems have been derived for common contaminants in soil based on a species sensitivity distribution (SSD) model developed for Australian conditions. EILs have been derived for As, Cu, CrIII, DDT, naphthalene, Ni, Pb and Zn.

Schedule B5a provides detailed guidance on the framework for ecological risk assessment. The methodology for deriving EILs is described in Schedule B5b and the detailed derivations of EILs for As, Cu, CrIII, DDT, naphthalene, Ni, Pb and Zn are presented in Schedule B5c. A spreadsheet, which may be used for calculating site-specific EILs is included in the ASC NEPM Toolbox.

2.5.2 EIL methodology

The detailed methodology, incorporated in Schedule B5b, was developed by CSIRO using data from various Australasian databases, the Australian National Biosolids Research Program and supplemented by data from the US EPA ecotoxicology database where necessary. The methodology is based on an SSD approach, which considers the physicochemical properties of soil and contaminants and the capacity of the local ecosystem to accommodate increases in contaminant levels (referred to as the 'added contaminant limit' or ACL) above ambient background. Where insufficient data is available for the SSD method to be used, a more conservative method using an assessment factor approach may be adopted.

The EILs are derived for specified levels of percentage species protection depending on land use. The approach is analogous to the methodology used for derivation of the Australian water quality guidelines (ANZECC & ARMCANZ 2000).

2.5.3 Land use

EILs have been developed for three generic land use settings:

- areas of ecological significance
- urban residential areas and public open space
- commercial and industrial land uses.

An area of ecological significance is one where the planning provisions or land use designation is for the primary intention of conserving and protecting the natural environment. This would include national parks, state parks, wilderness areas and designated conservation areas.

Urban residential/public open space is broadly equivalent to the HIL A, HIL B and HIL C land use scenarios (see Section 2.2 and Schedule B7).

EILs are not applicable to agricultural soils, which need evaluation in relation to crop toxicity, plant contaminant uptake and detailed consideration of soil type.

2.5.4 Levels of protection

The protection levels for the generic land use settings are:

- 99% for areas of ecological significance
- 80% for urban residential areas and public open space
- 60% for commercial and industrial land uses.

These protection levels are increased by 5% when biomagnification may occur (refer Schedule B5b).

2.5.5 Ecotoxicity data

The NEPM has adopted lowest observed effect concentration (LOEC) or effective concentration 30% (EC_{30}) data to derive EILs for the land use scenarios.

The LOEC is the lowest concentration used in a toxicity test that causes a toxic effect that is significantly different from the control. EC_{30} data is the concentrations of contaminants that cause an effect on 30% of the test group of an organism after a specified exposure time. The data is drawn from a range of species to derive individual EILs.

For further information see Schedule B5b.

2.5.6 Depth of application

EILs apply principally to contaminants in the top 2 m of soil at the finished surface/ground level which corresponds to the root zone and habitation zone of many species. In arid regions, where the predominant species may have greater root penetration, specific considerations may result in their application to 3 m depth.

2.5.7 Ambient background concentration

The methodology assumes that the ecosystem is adapted to the ambient background concentration (ABC) for the locality and that it is only adding contaminants over and above this background concentration which has an adverse effect on the environment.

The ABC of a contaminant is the soil concentration in a specified locality that is the sum of the naturally occurring background level and the contaminant levels that have been introduced from diffuse or non-point sources by general anthropogenic activity not attributed to industrial, commercial, or agricultural activities, for example, motor vehicle emissions. Methods to estimate background levels are provided in Schedule B5b.

Three methods for determining the ABC are presented in Schedule B5b. The preferred method is to measure the ABC at an appropriate reference site. This approach is essential in areas where there is a high naturally occurring background level such as will occur in mineralised areas.

In other situations where an appropriate reference site cannot be determined, the method based on urban metal levels in Olszowy et al. (1995) or the method from Hamon et al. (2004) may be used.

In the method of Hamon et al. (2004), the ABC varies (depending on the element) with the soil iron and/or manganese concentration; for example, the ABC for zinc varies from 3 to 62mg/kg in soils with soil iron concentrations between 0.1% and 20%. Alternatively, ABCs for old and new suburbs and high and low traffic areas for New South Wales, Queensland, South Australia and Victoria for Zn, Cu, Ni, Pb, and CrIII are included in Schedule B5b and are derived from Olszowy et al. (1995). Values for new suburbs would be appropriate to use for new suburbs or in areas with no known history of contamination for that metal. In old-established urban areas (i.e. suburbs more than 20 years old), it would be appropriate to use the 25^{th} percentile of the ABC values from Olszowy et al. (1995).

In some situations the ABC may be comparatively low and have a minor effect on the magnitude of the site EIL.

2.5.8 Added contaminant limits

An added contaminant limit (ACL) is the added concentration (above the ABC) of a contaminant above which further appropriate investigation and evaluation of the impact on ecological values is required. The EIL is derived by summing the ACL and the ABC.

ACLs are based on the soil characteristics of pH, CEC and clay content. Empirical relationships that can model the effect of these soil properties on toxicity are used to develop soil-specific values. These soil-specific values take into account the biological availability of the element in various soils. In this approach different soils will have different contaminant EILs rather than a single generic EIL for each contaminant.

ACLs apply to chromium III (CrIII), copper (Cu), nickel (Ni) and zinc (Zn) for site-specific EIL determination. The soil properties to be determined for each relevant soil type at the site, are shown in Table 4 below.

Table 4: Soil properties to be measured for site-specific derivation of ACLs for CrIII, Cu, Ni and Zn

Soil physicochemical property	CrIII	Cu	Ni	Zn
pH		✓		
CEC		✓	✓	✓
% clay	✓			

Insufficient data was available to derive ACLs for arsenic (As), DDT, lead (Pb) and naphthalene. As a result, the derived EILs are generic to all soils and are presented as total soil contaminant concentrations in Tables 1B(4) and 1B(5).

2.5.9 Ageing of contamination and soil properties

In general the toxicity of soil contaminants (both organic and inorganic) will reduce or age over time to a lower and more stable level by binding to various soil components and decreasing their biological availability. Hence, toxicity can be affected by the physicochemical or chemical properties of the soil including clay content, cation exchange capacity (CEC) measured in centi-mole charge/kg ($cmol_c/kg$), pH, iron and organic carbon content.

For the purposes of EIL derivation, a contaminant incorporated in soil for at least two years is considered to be aged for the purpose of EIL derivation. The majority of contaminated sites are likely to be affected by aged contamination. Fresh contamination is usually associated with current industrial activity and chemical spills.

In some cases insufficient data on aged contamination was available to apply the EIL methodology, and where possible, ageing factors based on relevant studies have been applied to determine a soil value for aged contamination.

EIL determination for fresh contamination (that is, present for less than two years) for the relevant contaminants should be site-specifically determined by reference to the relevant tables in Schedule B5c.

2.5.10 Determining site-specific EILs

Detailed information on the derivation of the EILs is provided in Schedule B5c. The following section describes the steps that are taken to derive site-specific EILs. A spreadsheet is included in the ASC NEPM Toolbox which can also be used for calculating site-specific EILs.

A. EILs for Ni, Cr III, Cu, Zn and Pb aged contamination (>2 years)

Steps 1–4 below describe the process for deriving site-specific EILs for the above elements using Tables 1B(1) – 1B(4), which can be found at the end of this Schedule.

1. Measure or analyse the soil properties relevant to the potential contaminant of concern (see Table 4). Sufficient samples need to be taken for these determinations to obtain representative values for each soil type in which the contaminant occurs.

2. Establish the sample ACL for the appropriate land use and with consideration of the soil-specific pH, clay content or CEC. The ACL for Cu may be determined by pH or CEC and the lower of the determined values should be selected for EIL calculation. Note that the ACL for Pb is taken directly from Table 1(B)4.
3. Calculate the contaminant ABC in soil for the particular contaminant and location from a suitable reference site measurement or other appropriate method.
4. Calculate the EIL by summing the ACL and ABC:

   EIL = ABC + ACL

B. EILs for As, DDT and naphthalene

EILs for aged contamination for DDT and naphthalene are not available and the adopted EIL is based on fresh contamination taken directly from Table 1B(5). The EILs for As, DDT and naphthalene are generic i.e. they are not dependent on soil type and are taken directly from Table 1B(5). Only EILs for fresh contamination are available for As, DDT and naphthalene due to the absence of suitable data for aged contaminants.

2.6 Ecological screening levels for petroleum hydrocarbon compounds

2.6.1 Introduction

Ecological screening levels (ESLs) are presented based on a review of Canadian guidance for petroleum hydrocarbons in soil and application of the Australian methodology (Schedule B5b) to derive Tier 1 ESLs for BTEX, benzo(a)pyrene and F1 and F2 (Warne 2010a, 2010b).

The Canadian Council of the Ministers of the Environment (CCME) has adopted risk-based TPH standards for human health and ecological aspects for various land uses in the *Canada-wide standard for petroleum hydrocarbons (PHC) in soil* (CCME 2008) (CWS PHC). The standards established soil values including ecologically based criteria for sites affected by TPH contamination for coarse- and fine-grained soil types.

The standard applies to the same four fractions (F1–F4) adopted for the HSLs (refer Section 2.4.5 of this Schedule).

2.6.2 ESL Methodology

The CWS PHC approach uses an SSD method and, when there is insufficient data for the SSD method, applies a weight-of-evidence approach to derive ecologically based 'Tier 1 eco soil contact' values for TPH fractions and specific compounds. The overall approach has similarities to the Australian EIL methodology by developing protective criteria based on EC_{25} toxicity for residential land use and EC_{50} for commercial/industrial land (cf. Australia EC_{30} and LOEC data).

The Australian EIL methodology was applied to the ecotoxicity data used to derive the Canadian F1 and F2 (eco soil contact) values (Warne 2010a) to produce comparable Tier 1 values for these fractions. Based on the data quality and applicability to the Australian environment, the derived values for F1 and F2 are adopted as moderate reliability ESLs (see Table 1B(6) at the end of this Schedule) and apply generically to fine- and coarse-grained soils.

Due to the limited ecotoxicity data for F3 and F4, the Australian methodology was not able to be applied. The data limitations were recognised in the Canadian guidance and an alternative weight-of-evidence approach was used to develop values for these fractions. Consequently, the adopted values for F3 and F4 (see Table 1B(6)) are considered low reliability ESLs for fine- and coarse-grained soils (Warne 2010a, 2010b).

A further review of Canadian soil quality guidelines was undertaken for BTEX and benzo(a)pyrene (Warne 2010b) and the Australian methodology applied to the ecotoxicological data as far as possible to derive equivalent ESLs. However, data limitations did not allow the full use of the EIL derivation methodology and the resulting values are adopted as low reliability ESLs in Table 1B(6). Values were derived using the Canadian data reduction methods, the Australian SSD method and employing the Australian levels of protection for various land uses.

ESLs for the adopted carbon fraction ranges are based on TRH analysis with F1 being obtained after subtraction of BTEX.

2.6.3 Depth of application

ESLs apply from the surface to 2 m depth below finished surface/ground level, which corresponds to the root zone and habitation zone of many species. In arid regions, where the predominant species may have greater root penetration, specific considerations may result in their application to 3 m depth.

2.6.4 Soil texture

The ESLs are applicable to coarse and fine textured soils equivalent to coarse-grained soils and fine-grained soils in Table A1 of Standard AS 1726:1993. Conservative Tier 1 values (i.e. values for coarse soils) should be applied where site-specific textural information is not available.

2.6.5 Fresh and aged contamination

ESLs were derived on the basis of fresh contamination. GC-MS analysis and examination of the gas chromatogram output can assist in differentiating between fresh and aged TPH contamination.

While aged contamination is generally of less human health and environmental concern, sub-surface conditions can preserve some petroleum hydrocarbons for extended periods of time. Consideration should be given to the realistic risk of material being excavated and causing an exposure risk.

2.7 Sediment quality guidelines

Investigation and screening levels developed for soils should not be applied directly to the assessment of sediments.

Interim sediment quality guidelines (ISQG) are available in the *Australian and New Zealand guidelines for Fresh and Marine Water Quality* (ANZECC & ARMCANZ 2000) for a number of common metal, metalloid and organometallic contaminants and organics, principally PAHs and organochlorine pesticides (OCPs). The ISQG have limitations relating to the availability of appropriate ecotoxicology data and the small number of species on which they are based.

Reference to these guidelines, balanced by consideration of their limitations, may have application in the site-specific assessment of sites where contamination may impact aquatic receptors. Guidance on the sampling of sediments can be found in AS/NZS 5667.12:1999 *Guidance on sampling of bottom sediments* and Simpson et al. (2005).

2.8 Groundwater investigation levels

Site assessment should consider the risks from contaminated groundwater to all potential receptors on and off the site of origin and potential effects on groundwater resources.

The Groundwater investigation levels (GILs) are based on the *Australian Water Quality Guidelines 2000* (AWQG), *Australian Drinking Water Guidelines 2011* (ADWG) and *Guidelines for Managing Risk in Recreational Waters 2008* (GMRRW). The GILs are adopted in the NEPM as investigation levels in the context of the framework for risk-based assessment of groundwater contamination (refer Schedule B6) i.e. levels above which further assessment is required.

The AWQG provide tabulated values based on percentage species protection for various aquatic environments and water uses. The appropriate settings for current and potential uses of groundwater need to be identified for the aquifer undergoing assessment. The guideline documents should be consulted for appropriate interpretation of guideline values, in consultation with relevant regulatory authorities if necessary.

Table 5. Groundwater environmental values and guidelines for their protection

Environmental value to be protected	Guidelines to apply
Raw drinking water source	ADWG
Agricultural use – stock watering	AWQG
Agricultural use – irrigation	AWQG
Fresh water aquatic ecosystem	AWQG
Marine water aquatic ecosystem	AWQG
Recreational use	GMRRW

The GILs provided in Table 1C at the end of this Schedule, define acceptable water quality for various contaminants at the point of use. Table 1C provides frequently used values for drinking water and protection of fresh and marine ecosystems. Additional GILs applicable to industrial use (aquaculture), agricultural use (stock watering and irrigation) and recreational waters are provided in the referenced documents.

The GMRRW recommend applying a multiplication factor of 10 to 20 to the ADWG for assessment of the acceptability of recreational water quality. GILs for other receptors should be obtained directly from the 'primary industries' section of the AWQG where relevant. Note that the recreational and aesthetics sections of the AWQG have been superseded by the GMRRW.

2.9 'Management limits' for petroleum hydrocarbon compounds

In addition to appropriate consideration and application of the HSLs and ESLs, there are a number of policy considerations which reflect the nature and properties of petroleum hydrocarbons:

- formation of observable light non-aqueous phase liquids (LNAPL),
- fire and explosive hazards and
- effects on buried infrastructure e.g. penetration of, or damage to, in-ground services by hydrocarbons.

The CWS PHC includes 'management limits' to avoid or minimise these potential effects and these values have been adopted as interim Tier 1 guidance. The values are included in Table 1B(7) at the end of this Schedule. A site-specific assessment (Tier 2 or 3) may be preferred where relevant site-specific information is available.

Application of the management limits will require consideration of site-specific factors such as the depth of building basements and services and depth to groundwater, to determine the maximum depth to which the limits should apply. The management limits may have less relevance at operating industrial sites (including mine sites) which have no or limited sensitive receptors in the area of potential impact. When the management limits are exceeded, further site-specific assessment and management may enable any identified risk to be addressed.

The presence of site TPH contamination at the levels of the management limits does not imply that there is no need for administrative notification or controls in accordance with jurisdiction requirements.

Further information on the consideration of aesthetics with respect to petroleum hydrocarbons is included in Section 3.6.

3 Application of investigation and screening levels

3.1 Recommended process for assessment of site contamination

The recommended site assessment process is shown in Schedule A of the NEPM. Refer to Schedule B2 for guidance on site characterisation.

Before comparing site data with investigation and screening levels, it is important that sufficient and appropriate characterisation of the site is carried out to ensure that the comparison is both meaningful and relevant for assessing potential risks to human health and the environment.

A number of cases studies which illustrate the application of the investigation and screening levels in site assessment are included in Section 5 of this Schedule.

3.2 Tier 1 assessment

A Tier 1 (or screening level) assessment comprises a comparison of representative site data with generic investigation levels and/or screening levels for protection of human health and the environment, together with an assessment of any limitations on their use in relation to site-specific conditions. A Tier 1 assessment provides an initial screening of the data to determine whether further assessment is required.

Contaminated sites may contain multiple contaminants in soil and groundwater and the risk posed is affected by site characteristics such as soil properties and the depth to the contamination. The selection of the appropriate investigation and screening levels to apply at a particular site should be determined using professional judgement and with reference to the CSM.

3.2.1 Comparison with investigation and screening levels

No single summary statistic will fully characterise a site and appropriate consideration of relevant statistical measurements should be used in the data evaluation process and iterative development of the CSM (refer to Schedule B2, Section 4).

The preferred approach is to examine a range of summary statistics including the contaminant range, median, arithmetic/geometric mean, standard deviation and 95% upper confidence limit (UCL). Further information is provided in Section 11 of Schedule B2.

At the very least, the maximum and the 95% UCL of the arithmetic mean contaminant concentration should be compared to the relevant Tier 1 screening criteria. However, where there is sufficient data available, and it is appropriate for the exposure being evaluated, the arithmetic mean (or geometric mean in cases where the data is log normally distributed) should also be compared to the relevant Tier 1 investigation or screening level. The implications of localised elevated values (hotspots) should also be considered. The results should also meet the following criteria:

- the standard deviation of the results should be less than 50% of the relevant investigation or screening level, and
- no single value should exceed 250% of the relevant investigation or screening level.

The maximum observed contaminant concentration generally provides a conservative assessment of exposure because if estimated risks from the maximum concentrations are not of concern, then the site should be suitable for use under the CSM considered. However, a maximum concentration may not be representative of the source as a whole and may result in an overestimation or underestimation of risk if the data is extremely limited.

The mean contaminant concentration can be a suitable metric provided that it can be shown that it adequately represents the source being considered. It is important that small areas of high concentrations or hot-spots are not ignored by averaging with lower values from other parts of the site.

The mean value may be more representative of the source as a whole than the maximum, and may provide a better estimation of the actual concentration that a population would be exposed to over a period of time.

The 95% UCL of the arithmetic mean provides a 95% confidence level that the true population mean will be less than, or equal to, this value. The 95% UCL is a useful mechanism to account for uncertainty in whether the data set is large enough for the mean to provide a reliable measure of central tendency. Note that small data sets result in higher values for the 95% UCL. Further guidance on the use of 95% UCLs can be found in NSW DECC (2006), US EPA (2006b) and US EPA (2007a).

Groundwater data being used to assess exposure should consider a relevant average at the site or off-site (as appropriate based on the CSM) together with a reasonable maximum based on understanding of seasonal and other trends in groundwater quality. Where trends are poorly defined in the early stages of an investigation, greater weight should be placed on the maximum concentration.

If air data or soil vapour data is available for the site, then the use of that data needs to be considered within the context of the CSM and the activities at the site or adjacent to the site that may affect the presence of substances in the air, including confounding substances. Consideration of both a reasonable maximum and a relevant average case should be considered where possible.

The effects of applying a multiplication factor to account for biodegradation to soil, soil vapour and groundwater HSLs where relevant should be considered in the data analysis. The data should be evaluated for trends and the presence of hot spots prior to the application of any biodegradation factors.

3.2.2 Exceedence of Tier 1 investigation and screening levels

The magnitude of the exceedence should be considered in the context of the CSM (that is, whether the exposure pathways are plausible and whether exposure will result in harm). In cases of minor exceedence of investigation or screening levels, a qualitative risk assessment may be sufficient to evaluate the potential impact.

Where exceedence of Tier 1 investigation and screening levels indicates that there is a likelihood of an adverse impact on human health or ecological values for that site, site-specific health and/or ecological risk assessment (Tier 2 or 3) should be carried out as appropriate. This will usually require the collection of additional site data.

Alternatively, appropriate management options may be considered such as engaging with landowners and occupants/site users regarding the nature of the contamination and implementing appropriate site management plans. Guidance on community engagement and risk communication is provided in Schedule B8.

The nature of the response should be determined on a site-specific basis and be proportional to the potential risk posed to human health and/or the environment.

3.2.3 Procedure if no generic investigation or screening levels are available

Site-specific investigation levels will need to be developed when:

- investigation or screening values are not available for the contaminants of concern and/or insufficient data is available for the derivation of generic guideline values
- site conditions, receptors and/or exposure pathways differ significantly from those assumed in the derivation of the generic investigation or screening levels.

Consult Schedules B4 and B7 for guidance on deriving site-specific HILs and on applying the HIL methodology to derive HILs for additional substances.

Consult Schedule B5b for guidance on applying the EIL methodology to derive EILs for additional substances. Schedule B5b Appendix B provides guidance using a method of soil–water partitioning coefficients for deriving EILs that are protective of aquatic ecosystems.

3.3 Specific considerations for petroleum hydrocarbons

The flowchart in Figure 1 (below) provides a general overview of the application of the HSLs and ESLs for petroleum hydrocarbons including linkage to the 'management limits' for TPH contamination. Information on these screening levels can be found in:

Human health concerns

- HSLs check list – ASC NEPM Toolbox
- Vapour inhalation pathway – HSLs – Section 2.4
- Direct contact pathways – HSLs – Section 2.4
- Consumption of groundwater – GILs – Section 2.8 and Schedule B6
- HILs – Benzo(a)pyrene, total PAH and lead – Section 2.2 and Schedule B7
- Aesthetics – Section 3.6
- 'Management limits' – Section 2.9.

Ecological concerns

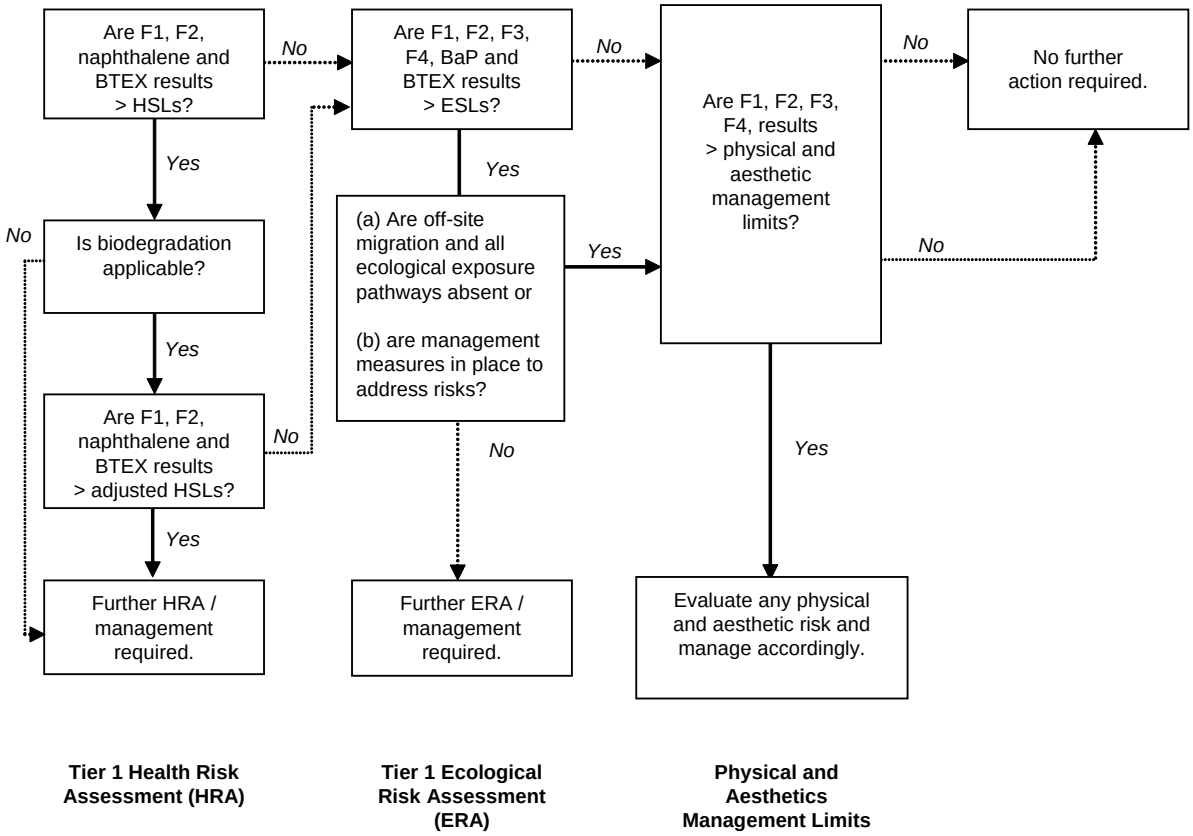
- ESLs – terrestrial ecosystems – Section 2.6
- AQWG – aquatic ecosystems - Section 2.8 and Schedule B6
- EILs – terrestrial ecosystems - lead – Section 2.5.

The application of these screening levels is illustrated by the case studies included in Section 5.

In many cases, sites assessed for petroleum hydrocarbon contamination are driven initially by human health concerns regarding volatile components (F1 and F2). In circumstances where the HSLs are modified by biodegradation factors or where the more volatile fractions are absent, then ecological considerations may become the predominant concern, particularly for the longer chain fractions (F3 and F4).

There are many HSLs that are denoted as non limiting or NL (refer Section 2.4.2, footnotes to HSL Tables and Friebel & Nadebaum (2011a)) and high levels of petroleum hydrocarbons, including observable LNAPL, may be present at the site without presenting a risk via the vapour inhalation pathway. The presence of observable and mobile LNAPL in test pits and bores will require careful consideration of health, environmental, fire and explosive risks and aesthetic concerns. This presentation of LNAPL may lead to active management depending on the current or proposed site use and the extent of the LNAPL. An immediate response may be required where there is penetration of in-ground services or detectable odours in building interiors. Dispersed droplets of LNAPL that are relatively immobile (e.g. in a clay-rich soil) that are assessed as low risk may not require active management.

Figure 1: Flowchart for Tier 1 human and ecological risk assessment of petroleum hydrocarbon contamination – Application of HSLs and ESLs and consideration of management limits



Notes

1. The CSM should inform the selection and application of human health and ecological screening levels and management limits. Relevant HSLs, GILs, HILs and EILs (e.g. PAHs and lead) should be considered for sites affected by petroleum hydrocarbons.
2. The limitations of the screening levels and investigation levels should be considered on a site-specific basis.
3. Petroleum hydrocarbon 'management limits' are used to consider the potential effects of LNAPL-related hazards. Refer to Section 2.9 for more information on depth of application. Jurisdictions may have policies applicable to the presence of LNAPL.
4. The potential for groundwater contamination and impacts on receptors including groundwater resources should be considered and assessed as appropriate in accordance with Schedule B6 and jurisdictional policies for the protection of groundwater resources.

3.4 Considerations for ecological assessment

3.4.1 General

Schedule A provides an overview of the site assessment process and the application of investigation and screening levels for human health and ecological risk assessment. While protection of human health often drives the first stages of assessment, protection of the environment (terrestrial and aquatic) should be a consideration for all site assessments.

In assessing the overall risk to the environment from soil contamination the following site-specific aspects should be considered:

- the location of the contamination in relation to any on-site and off-site sensitive receptors, e.g. watercourses, estuaries, groundwater resources, sensitive ecological areas
- the existing or proposed land use(s)
- the presentation of contaminants including areal extent, depth below finished ground level, the presence of barriers or containment that prevents or minimises the migration of contamination or exposure pathways
- the in situ leaching characteristics of contaminants of concern and the potential for leachate to adversely affect any accessible sensitive on-site and off-site receptors
- the potential for contaminants to be transported from the site at levels of concern by erosive forces.

3.4.2 Scope of ecological assessment

The relevance and scope of ecological assessment should be considered early in the development of the conceptual site model and data quality objectives. A pragmatic risk-based approach should be taken in applying EILs and ESLs in residential and commercial/industrial land use settings.

Site soils may have poor structure and drainage, low organic content, minimal topsoil depth and a limited ability to support plant growth and soil micro-organisms. In existing residential and urban development sites there are often practical considerations that enable soil properties to be improved by addition of ameliorants with a persistent modifying effect or by the common practice of backfilling or top dressing with clean soil. In other cases, all of the site soils will be removed during site development works or relocated for the formation of new land forms. Sites may also be backfilled with clean soil/fill and the fate of any excavated contaminated soil should be considered in the process.

Commercial and industrial sites may have large building structures and extensive areas covered with concrete, other pavement or hardstand materials and may have limited environmental values requiring consideration while in operational use.

3.4.3 Mobility of contaminants

When contamination is in a highly leachable form or is incorporated in exposed readily erodible soil, potentially adverse ecological effects may occur some distance from the contaminant source area. The potential for off-site environmental impacts should be considered in the development of the conceptual site model. Methods for determining leachability are discussed in Schedule B3.

It is common for established industrial areas to contain higher levels of soil contamination (such as metals) than surrounding areas. Receptors and soils immediately adjoining older industrial zones may be affected by the accumulation of soil contaminants caused by migration through subsurface contaminant movement and erosion of contaminated soils.

For example, a site with lead (Pb), zinc (Zn) and petroleum hydrocarbon concentrations in soil below EILs and ESLs for commercial/industrial land use (where a 60% or 65% species protection level would apply) would be acceptable for the site use. However, if the site adjoined an area of ecological significance, such as a protected wetland, the site assessment should also consider the possibility that contamination may migrate off-site and impact the wetland where 99% species protection limits would apply.

In other cases sites may have aged metals and metalloid contaminants with stable, cohesive soils and low in situ leachability and pose a low risk to the ecosystem.

3.5 Considerations for groundwater assessment

When groundwater from a monitoring well contains levels of contaminants above the appropriate investigation levels (Tier 1 assessment), then further investigation (Tier 2 assessment) is required. This may take the form of consideration of site-specific conditions and circumstances which may result in modification of the generic Tier 1 criteria. If no modification of the Tier 1 criteria is applicable, the assessment proceeds directly to Tier 3 where groundwater concentrations at the point of exposure (point of use) are compared with the generic GILs or site-specific response levels. If this indicates that the investigation levels are exceeded at the point of use, or in the discharge environment of the groundwater, then an appropriate response is required. The relevant guideline documents should be consulted for informed interpretation and application of GILs and modified GILs.

Groundwater protection may be a particular concern where contamination occurs in sandy soils containing naturally low levels of organic matter, clay and trace elements. In most situations, soil contaminants at levels below appropriate EILs or HILs do not pose a threat to local groundwater sources. However, possible impacts on groundwater should always be considered particularly for sites impacted by petroleum hydrocarbons and halogenated solvents. In some cases the soil may not reveal contaminants of concern while groundwater is affected.

It should be noted that some jurisdictions may have groundwater protection policies that require action even where levels do not exceed the AWQG values at the point of use.

3.6 Aesthetic considerations

3.6.1 Introduction

Aesthetic issues generally relate to the presence of low-concern or non-hazardous inert foreign material (refuse) in soil or fill resulting from human activity. Sites that have been assessed as being acceptable from a human health and environmental perspective may still contain such foreign material. Geotechnical issues related to the presence of fill should be treated separately to assessment of site contamination.

Various forms of refuse may be identified in bore or test pit logs, for example fragments of concrete, metal, bricks, pottery, glass, trivial amounts of bonded asbestos-containing-materials, bitumen, ash, green waste, rubber, plastics and a wide variety of other waste materials. These materials commonly occur in former industrial and filled sites. Similarly, construction and demolition waste materials, some of which are inert and non-hazardous, are widely distributed in urban areas.

Other sites may have some soil discolouration from relatively inert chemical waste (for example, ferric metals) or residual odour (for example, natural sulphur odour).

Care should be taken to ensure adequate site characterisation, particularly when there is a diverse range of foreign material and associated fill and an appreciable risk inferred from site history (or lack thereof) for the presence of hazardous contaminants. For example, some ash fill may contain PAHs and metals, while other ash deposits may contain no contaminants of concern.

3.6.2 Circumstances which would trigger an assessment of aesthetics

The following characteristics or presentations are examples of where site assessment may not have detected contamination above investigation or screening levels but where further assessment would be required:

- highly malodorous soils or extracted groundwater (e.g. strong residual petroleum hydrocarbon odours, hydrogen sulphide in soil or extracted groundwater, organosulfur compounds)
- hydrocarbon sheen on surface water

- discoloured chemical deposits or soil staining with chemical waste other than of a very minor nature
- large monolithic deposits of otherwise low-risk material, e.g. gypsum as powder or plasterboard, cement kiln dust
- presence of putrescible refuse including material that may generate hazardous levels of methane such as a deep-fill profile of green waste or large quantities of timber waste
- soils containing residue from animal burial (e.g. former abattoir sites).

3.6.3 Assessment process for aesthetic issues

There are no specific numeric aesthetic guidelines, however site assessment requires balanced consideration of the quantity, type and distribution of foreign material or odours in relation to the specific land use and its sensitivity. For example, higher expectations for soil quality would apply to residential properties with gardens compared with industrial settings.

General assessment considerations include:

- that chemically discoloured soils or large quantities of various types of inert refuse, particularly if unsightly, may cause ongoing concern to site users
- the depth of the materials, including chemical residues, in relation to the final surface of the site
- the need for, and practicality of, any long-term management of foreign material.

In some cases, documentation of the nature and distribution of the foreign material may be sufficient to address concerns relating to potential land use restrictions.

In arriving at a balanced assessment, the presence of small quantities of non-hazardous inert material and low odour residue (for example, weak petroleum hydrocarbon odours) that will decrease over time should not be a cause of concern or limit the use of a site in most circumstances. Similarly, sites with large quantities of well-covered known inert materials that present no health hazard such as brick fragments and cement wastes (for example, broken cement blocks) are usually of low concern for both non-sensitive and sensitive land uses.

Caution should be used for assessing sensitive land uses, such as residential, when large quantities of various fill types and demolition rubble are present.

4 Asbestos materials in soil

4.1 Scope of the guidance

This guidance applies to the assessment of known and suspected asbestos contamination in soil and addresses both friable and non-friable forms of asbestos. Most assessments will involve non-friable bonded forms of asbestos-containing-material (bonded ACM) as this is the most common type of asbestos soil contamination in Australia.

This guidance is not applicable to asbestos materials which are:

- wastes such as demolition materials present on the surface of the land or
- asbestos materials in buildings or structures including operational pipelines.

Transport and disposal of asbestos-contaminated soil should be carried out in accordance with state and territory legislation and guidelines. Soils that are known or suspected to be contaminated with asbestos should not be reused or recycled at other sites.

This guidance deals with assessment but is closely linked to remediation, management and protection of human health.

An overview of the assessment of asbestos contamination is presented here. More detailed information on site characterisation can be found in Schedule B2 Section 11 and WA DoH (2009, 2012).

Case studies illustrating the recommended approach for site assessment are included in Section 5.

4.2 Historical use of asbestos in Australia

Bonded asbestos products were first manufactured in Australia in the 1920s and were a common component of residential and commercial building materials from the mid-1940s until the late 1980s. Up to 90% of the asbestos mined or imported into Australia was used for the manufacture of these building products. Australia banned the use and import of building asbestos products in the mid-1980s and, in December 2003, banned import, manufacture and use of all asbestos products (e.g. automobile products).

Asbestos has been used in Australia as a reinforcing agent in cement sheeting for walls and roofs and in cement building products, such as pipes, gutters and flooring. Asbestos was also used in combination with other bonding compounds such as vinyl (e.g. for vinyl floor tiles and sheeting) and resin. Friable (non-bonded) asbestos products include low-density asbestos fibre board, insulating products such as lagging, sprays and asbestos rope gaskets.

Many older homes in all Australian communities still contain asbestos cement products, commonly in eaves or cladding of internal and external walls and roofs. When in good condition, bonded asbestos products do not release asbestos fibres into the air and are considered safe for people who are in contact with them, including when carrying and handling these materials (enHealth 2012). If asbestos materials can be maintained in good condition, enHealth (2005, 2012) recommends that these materials are best left alone and periodically checked to monitor their condition.

4.3 Work Health and Safety

Site assessors should be aware of (and where relevant comply with) the requirements of both national and jurisdictional work health and safety legislation and guidance relating to asbestos and its removal, such as:

- the national model Work Health and Safety Regulations and related jurisdictional legislation and guidelines
- *How to manage and control asbestos in the workplace Code of Practice* (Safe Work Australia 2011a)

- *How to safely remove asbestos Code of Practice* (Safe Work Australia 2011b)
- *Code of Practice for the Management and Control of Asbestos in Workplaces* (NOHSC: 2018 (2005))
- *Code of Practice for the Safe Removal of Asbestos 2nd edn* (NOHSC: 2002 (2005)).

State/territory agencies with responsibility for work health and safety should be consulted for specific guidance on what is required in that state or territory.

The final prohibition of asbestos in the workplace came into effect on 31 December 2003 but there are a number of exceptions including:

- genuine research and analysis
- sampling and identification in accordance with WHS Regulations
- where the regulator approves the method adopted for managing risk associated with asbestos.

Safe Work Australia (2011a) provides practical advice on how to manage risks associated with asbestos and asbestos-containing-material (ACM) in the workplace. It provides information on how to identify the presence of asbestos at the workplace and how to implement measures to eliminate or minimise the risk of exposure to airborne asbestos fibres.

Work involving asbestos-contaminated soil is permitted providing that a competent person has determined that the soil does not contain any visible ACM or friable asbestos; or if friable asbestos is visible, it does not contain more than trace levels of asbestos determined in accordance with AS4964:2004 *Method for the qualitative identification of asbestos in bulk samples.*

A competent person is defined in Safe Work Australia (2011a) as a person who has acquired through training, qualification or experience, the knowledge and skills to carry out the task.

A competent person in the context of asbestos and the NEPM is a person who has acquired through training, qualification or experience, the knowledge and skills to identify, investigate and assess asbestos in the context of an environmental site assessment. This includes identifying the potential for asbestos contamination from site history information.

If visible asbestos is present and it may be disturbed during work activities, it must be removed. This includes removing visible fragments of bonded ACM from exposed trench faces and those areas of the site where intrusive works may be carried out (e.g. to install utilities). The removal of visible asbestos should be appropriately managed and full details recorded (this information is required for assessing asbestos concentration in soil – refer Section 4.10). Visible asbestos should be removed prior to excavation/construction works commencing. Consult the relevant Code of Practice for more detailed information.

4.4 Terminology for asbestos contamination in soil

For the purpose of assessing the significance of asbestos in soil contamination, three terms are used in this Schedule which are based on guidance developed by the Western Australian Department of Health (WA DoH, 2009). The equivalent terms used in work health and safety legislation are listed in Table 6:

Table 6 Equivalency of terms used in the NEPM, WA DoH (2009) and Work Health and Safety legislation and guidelines

NEPM terminology (based on WA DoH 2009)	Work Health and Safety terminology
Bonded asbestos-containing-material or 'bonded ACM' *(referred to as ACM in WA DoH 2009)*	Bonded asbestos/non-friable asbestos
Fibrous asbestos, FA	Non-bonded/friable asbestos
Asbestos fines, AF	

Bonded asbestos containing material (bonded ACM)

Bonded ACM comprises asbestos-containing-material which is in sound condition, although possibly broken or fragmented, and where the asbestos is bound in a matrix such as cement or resin (e.g. asbestos fencing and vinyl tiles). This term is restricted to material that cannot pass a 7 mm x 7 mm sieve. This sieve size is selected because it approximates the thickness of common asbestos cement sheeting and for fragments to be smaller than this would imply a high degree of damage and hence potential for fibre release.

Bonded ACM is equivalent to 'non-friable' asbestos in Safe Work Australia (2011), which is defined therein as 'material containing asbestos that is not friable asbestos, including material containing asbestos fibres reinforced with a bonding compound'.

Fibrous asbestos (FA)

FA comprises friable asbestos material and includes severely weathered cement sheet, insulation products and woven asbestos material. This type of friable asbestos is defined here as asbestos material that is in a degraded condition such that it can be broken or crumbled by hand pressure. This material is typically unbonded or was previously bonded and is now significantly degraded (crumbling).

Asbestos fines (AF)

AF includes free fibres, small fibre bundles and also small fragments of bonded ACM that pass through a 7 mm x 7 mm sieve. (Note that for bonded ACM fragments to pass through a 7 mm x 7 mm sieve implies a substantial degree of damage which increases the potential for fibre release.)

From a risk to human health perspective, FA and AF are considered to be *equivalent to 'friable' asbestos in Safe Work Australia (2011), which is defined therein as 'material that is in a powder form or that can be crumbled, pulverised or reduced to a powder by hand pressure when dry, and contains asbestos'.*

4.5 Occurrence of asbestos contamination in soil

Bonded ACM is the most common form of asbestos site contamination across Australia, arising from:

- inadequate removal and disposal practices during demolition of buildings containing asbestos products
- widespread dumping of asbestos products and asbestos-containing fill on vacant land and development sites
- commonly occurring in historical fill containing unsorted demolition materials.

If identified early, i.e. prior to significant soil disturbance or earth movements, dumping and inadequate demolition practices usually only results in surface (or near surface) distribution of bonded ACM fragments.

Mining, manufacture or distribution of asbestos products may result in sites being contaminated by friable asbestos including free fibres. Severe weathering or damage (including by vehicle movements)

to bonded ACM may also result in the formation of friable asbestos (comprising fibrous asbestos (FA) and asbestos fines (AF)).

4.6 Asbestos soil contamination and health risk

Asbestos only poses a risk to human health when asbestos fibres are made airborne and inhaled. If asbestos is bound in a matrix such as cement or resin, it is not readily made airborne except through substantial physical damage.

This guidance emphasises that the assessment and management of asbestos contamination should take into account the condition of the asbestos materials and the potential for damage and resulting release of asbestos fibres.

Bonded ACM in sound condition represents a low human health risk. However, both FA and AF materials have the potential to generate, or be associated with, free asbestos fibres. As a result, FA and AF must be carefully managed to prevent the release of asbestos fibres into the air.

It is an inappropriate response to declare a site a human health risk on the basis of the presence of bonded ACM alone. However, if the bonded material is damaged or crumbling (that is, it has become friable), it may represent a significant human health risk if disturbed and fibres are made airborne.

The site-specific assessment of sites contaminated with asbestos in soil should be aimed at describing the nature and quantity of asbestos present in sufficient detail to enable a risk management plan to be developed for the current or proposed land use. The management plan should address potential scenarios for the relevant land use(s) whereby asbestos fibres may become airborne and pose a human health risk.

4.7 Basis for health screening levels for asbestos in soil

In 2009, the Western Australian Department of Health (WA DoH) released *Guidelines for the Assessment, Remediation and Management of Asbestos-Contaminated Sites in Western Australia* (WA DoH 2009). The WA DoH guidelines are based on research published by Swartjes & Tromp (2008), which is based on an extensive database of field and simulation trials using both bound and friable asbestos. The trial results indicated that a soil level of 0.01% for friable asbestos should keep asbestos fibre levels in air below 0.001 fibres per millilitre (f/ml) and probably to around 0.0001 f/ml. This corresponds to a lifetime risk of 10^{-6} to 10^{-5} in the exposed population from airborne asbestos fibres using WHO (2005) risk figures for mesothelioma (WA DoH 2009). The Netherlands (Swartjes & Tromp 2008) apply an investigation level of 0.01% weight for weight (w/w) for fibrous asbestos and 0.1% w/w asbestos for non-friable asbestos (i.e. bound asbestos in sound condition) in soil.

WA DoH has taken a more conservative approach (by a factor of 10) than the Netherlands to take account of the greater dryness and dust-generating potential of many local soils and the practice of treating all forms of asbestos (e.g. crocidolite, amosite, chrysotile and actinolite) as equivalent in terms of human health risk. The WA guidelines apply screening levels of:

- 0.01% w/w asbestos in soil for ACM (being asbestos in bonded ACM) to residential sites equivalent to land use setting HIL A. Additional criteria are provided for other land uses based on the default exposure ratios of the NEPM (1999)
- 0.001% w/w asbestos in soil for FA and AF for all site uses.

4.8 Health screening levels for asbestos in soil

Health screening levels for asbestos in soil, which are based on scenario-specific likely exposure levels, are adopted from the WA DoH guidelines and are listed in Table 7.

There are various acceptable means to provide confidence that the soil surface is free of visible asbestos including, but not limited to, multi-directional raking of soil to about 10 cm depth and hand-picking of asbestos fragments or covering with a durable hard cover. The requirement for the soil surface to be free of visible asbestos applies to both assessment and remediation phases.

Refer to sections 4.10 and 4.11 for guidance on determining asbestos concentration in soil and comparison with these screening levels.

Table 7. Health screening levels for asbestos contamination in soil

	Health Screening Level (w/w)			
Form of asbestos	**Residential A[1]**	**Residential B[2]**	**Recreational C[3]**	**Commercial/ Industrial D[4]**
Bonded ACM	0.01%	0.04%	0.02%	0.05%
FA and AF[5] (friable asbestos)	0.001%			
All forms of asbestos	No visible asbestos for surface soil			

1. Residential A with garden/accessible soil also includes children's day care centres, preschools and primary schools.
2. Residential B with minimal opportunities for soil access; includes dwellings with fully and permanently paved yard space such as high-rise buildings and apartments.
3. Recreational C includes public open space such as parks, playgrounds, playing fields (e.g. ovals), secondary schools and unpaved footpaths.
4. Commercial/industrial D includes premises such as shops, offices, factories and industrial sites.
5. The screening level of 0.001% w/w asbestos in soil for FA and AF (i.e. non-bonded/friable asbestos) only applies where the FA and AF are able to be quantified by gravimetric procedures (refer Section 4.10). This screening level is not applicable to free fibres.

4.9 Process for assessment of asbestos contamination

The recommended general process for assessment of site contamination, including for assessment of asbestos, is shown in Schedule A to this NEPM. The process starts with a Preliminary Site Investigation (PSI), which may lead to a Detailed Site Investigation (DSI). Depending on the site-specific circumstances and the proposed remediation approach, conservative management of presumed asbestos contamination may avoid the need for a DSI. Where remediation is required, appropriate validation sampling should be carried out to verify the effectiveness of the measures undertaken.

It is important to note that inadequate sampling strategies and/or inadequate documentation, rather than lack of accuracy in the adopted analytical methods, characteristically limit the effective evaluation of sites contaminated with asbestos.

Further information on the recommended assessment process is provided in Schedule B2.

A DSI is not necessary where there is a high degree of confidence that the asbestos contamination is confined to bonded ACM in superficial soil, i.e. the site history can be established with confidence and this clearly indicates that there is no reason to suspect buried asbestos materials and the site inspection confirms that any bonded ACM is in sound condition and only present on the surface/near surface of the site. In these circumstances the assessment can proceed directly to remediation (removal of bonded ACM fragments and ensuring that the soil surface is free of visible asbestos) and validation.

4.10 Determining asbestos in soil concentrations

Bonded ACM is the most common and the most readily quantifiable form of asbestos soil contamination due to its ease of visual detection. Where site circumstances are favourable, bonded ACM in sound condition can be used as the primary means of estimating contamination by subjecting soil samples to on-site sieving and gravimetric procedures as described below.

Assessment of bonded ACM is the recommended measure for total asbestos contamination where FA and AF (derived from bonded ACM only) are not likely to be significant as established by the PSI including the site inspection (as a guide, this may be taken to be where FA and AF are likely to make up less than 10% of the total amount of asbestos present).

Important considerations in determining asbestos concentrations in soil include:

- observations and calculations of surface asbestos occurrence/distribution should be recorded on a grid system (a grid of up to 10 m x 10 m is generally reasonable when large surface areas are impacted, however, non-impacted soils should be excluded from calculations to avoid dilution effects)
- where more than one distinct fill unit or soil stratum/unit is impacted by asbestos materials, separate asbestos determinations should be made for each stratum/unit
- averaging asbestos concentrations across all soils at a site is not appropriate
- for sub-surface samples, (e.g. boreholes and trenches) the calculation should be carried out per sample (i.e. not averaged over a grid square)
- the statistical procedures outlined in Section 3.2 (such as comparing mean concentrations with the screening level and no individual sample concentration exceeding 250% of the screening level) are not appropriate for asbestos
- a weight-of-evidence approach (refer 4.11), which takes into account field observations and methodology and relevant site history findings (e.g. location and nature of fill and demolished buildings etc.)' is recommended for determining whether individual or adjacent samples exceeding the relevant screening levels are of concern.

Asbestos in soil concentration by gravimetric approach

Guidance on recommended sampling methods is given in Schedule B2 and is based on the WA DoH guidelines (2009).

The asbestos concentration calculations are based on the amount of asbestos equivalent (i.e. asbestos in asbestos-containing-materials) in a measured/estimated amount of soil, expressed as a % weight for weight. The soil volume may be one or more individual 10 L samples from specific soil units or the area of a grid square multiplied by the investigation depth for raking and tilling methods (refer Schedule B2).

As outlined in enHealth *(*2005), the quantity of asbestos in soil may be estimated as follows:

$$\%\text{w/w asbestos in soil} = \%\text{ asbestos content} \times \text{bonded ACM (kg)} / \text{soil volume (L)} \times \text{soil density (kg/L)}$$

In the example included in enHealth (2005) it was assumed that:

> % asbestos content (within bonded ACM) = 15% and soil density (for sandy soils) = 1.65 kg/L

More representative results for asbestos concentration in soil can be calculated if the parameter values are analysed rather than assumed.

The assumption of 15% asbestos by weight in bonded ACM for sites contaminated with cement bonded ACM only is acceptable because typical compositions for bonded ACM products used in Australia are 10–15% asbestos by weight. However, other bonded products may contain much larger proportions of asbestos, e.g. asbestos vinyl floor tiles may contain 8–30% asbestos (Workplace Health

and Safety Queensland, 2011). The likely presence of bonded materials other than cement products should be addressed in the PSI and site inspection. If found during sampling, the calculation will need to be adjusted either by making a conservative assumption or based on laboratory analysis of representative material from the site.

Soil densities are typically greater than about 1.5 kg/L (1500 kg/m^3). The need to sample and analyse representative soil samples for soil density should be considered in the SAQP and will be required for dense and/or compacted soils.

The rationale for the calculation carried out, including the basis for all assumptions, should be documented in the site assessment report.

Depending on what is known of the site history and also the nature of the investigative methods used, the confidence in the calculation results will vary. In particular, hand-picking (using multi-directional raking and hand removal of fragments) and tilling surveys (mechanical turning over of surface soils to assist identification and collection of fragments) may provide less confidence compared with large volume mechanical screening (separation of fragments by automated sieving). Likewise, if the bonded ACM weight is estimated rather than measured, such as by estimating bonded ACM sheet area, then confidence in the results will be reduced. (Note that when considering which technique(s) to use that the increased confidence in results from mechanical methods should be considered in the context of the possible increased risk of releasing fibres associated with bulk screening.)

A comparable gravimetric assessment approach may be applied to FA when large discrete pieces (e.g. asbestos gaskets and pieces of asbestos 'rope') are present in soil, however care should be taken during their removal to minimise potential fibre release.

If bonded ACM is in poor condition or site conditions are likely to result in degradation (e.g. due to acidic soil conditions) then the bonded ACM should be assumed to be FA for the purposes of comparing with the relevant screening level.

Schedule B2 and WA DoH 2009 (Section 4) provide more detailed guidance for sampling soil and determination of the %w/w asbestos in soil by gravimetric procedure.

Laboratory analysis

As yet there is no validated method, readily available in Australia, of reliably estimating the concentration of free asbestos fibres in soil. Soil contamination by free asbestos fibres should therefore be simply determined according to the presence or absence of fibres, in accordance with *AS4964 – 2004: Method for the Qualitative identification of asbestos in bulk samples* (Standards Australia 2004) by a laboratory accredited by NATA (or its mutual recognition agreement partners) for this method.

AS4964-2004 sets out a tiered approach to detecting the presence of asbestos (amosite, crocidolite and chrysotile forms) in soil samples using polarised light microscopy and dispersion staining techniques. If evidence of asbestos fibres is not found in the greater than 2 mm sieved fraction, a trace analysis is required of the residue (sub-2 mm fraction). Depending on the nature and size of the soil sample, the sub-2 mm residue material may need to be sub-sampled for trace analysis.

The nominal detection limit of the AS4964 method is around 0.01%. The examination of large sample sizes (at least 500 ml is recommended) may improve the likelihood of identifying asbestos material in the greater than 2 mm fraction.

Care should be taken in selecting samples for laboratory analysis to ensure that they comprise representative samples, as far as practicable, of the soil units to be tested or material from suspect areas. This may be difficult to achieve because of the complexity of the soil unit or large size of soil particles.

In the case of co-located bonded ACM, FA and AF, where significant asbestos may be present as fibrous asbestos or asbestos fines (greater than 10% (in total for FA and AF) of that present in the bonded ACM alone), then laboratory analysis may be necessary to assist with impact delineation. It may be possible in the initial AS4964 procedure to obtain an estimate of the weight of asbestos (such as small ACM fragments and fibre bundles) which does not pass through the 2 mm sieve. Depending on site circumstances, this information may be useful as part of a weight-of-evidence approach to assessment of asbestos soil concentrations relative to the appropriate screening levels.

As a general guide, where sites are contaminated with bonded ACM only (i.e. no insulation materials or other non-bonded asbestos products) assessment for the presence/absence of free fibres by laboratory analysis is only warranted where greater than 10% of the total bonded ACM is significantly damaged i.e. present as small pieces less than 7 mm x 7 mm or can be crushed/crumbled with hand pressure (significant FA and/or AF is present).

4.11 Assessment against asbestos screening levels and procedure for exceedences

A tiered approach to risk assessment of asbestos contamination is recommended, including development of an appropriate CSM (refer Section 2.4 in Schedule B4).
A weight-of-evidence approach is recommended with consideration given to factors such as the distribution of different fill types, the heterogeneity of the contamination and the uncertainty associated with the sampling methodology. The evaluation and discussion of results should consider any trends across the investigated area including variability and change in asbestos type and condition. For buried asbestos contamination, the impacted units should be identified and discussed separately.

For Tier 1 analysis, the contamination concentrations are compared with the screening levels presented in Section 4.8. If the Tier 1 screening levels are not exceeded, and an appropriate level of investigation has been carried out, then no contamination management actions are required except for ensuring the surface soil is free of visual asbestos. This may be achieved by multidirectional raking or tilling and hand-picking of exposed fragments of bonded ACM. Final visual inspection of the assessment and remediated areas should not detect any visible asbestos.

When cohesive soils (such as firm clay) or a large surface area is involved it may be more practical to skim the top 5–10 cm of soil for disposal in accordance with jurisdictional requirements. The exposed surface of the site can then be further visually assessed by an appropriately qualified and experienced professional/competent person on a systematic basis. If bonded ACM fragments are found to be present after skimming, some localised hand-picking or additional earthworks may be required until no visible bonded ACM is present.

If exceedences of the Tier 1 screening levels are present, either a Tier 2 analysis should be carried out or a conservative management response implemented. The Tier 2 assessment will comprise a qualitative assessment of risk in many cases and should take into account the nature and extent of contamination; the site-specific exposure scenario(s) including the intensity of relevant site activities; the impact of any mitigating factors such as soil type and soil moisture conditions (and likely variation); the proposed remediation and management measures; and the final use of the site.

Remediation options which minimise soil disturbance and therefore public risk are preferred. Management of asbestos in situ is encouraged, which may include covering the contamination with uncontaminated fill or other protective or warning layers. It should be noted that the common alternative of complete removal of asbestos from a site often involves extensive and costly investigative and validation sampling and may not be effective or necessary for the protection of human health.
Regulatory authorities may consider statutory management controls to land with substantial asbestos contamination to ensure that appropriate management conditions, including land use limitations, apply to the site. These controls may include notation on title, approved management and listing on public

site contamination registers or ongoing controls under audit statements and planning controls, as relevant for the jurisdiction.

Additional information on the assessment approach is provided in Schedule B2 and WA DoH (2009 and 2011).

Further information on risk assessment, remediation and management procedures can be found in Section 5 of the WA DoH Guidelines (2009).

The recommended approach for circumstances involving bonded ACM (the commonest form of asbestos contamination) is illustrated by the included case studies.

5 Case Studies

Case study 1	**Assessment of asbestos contamination in soil –** poor demolition practice at a residential site
Case study 2	**Assessment of asbestos contamination in soil –** redevelopment of an industrial site for residential use
Case study 3	**Application of petroleum hydrocarbon screening levels –** redevelopment of an industrial site for residential use
Case study 4	**Application of soil vapour interim VOCC HILs and HSLs –** vapour intrusion assessment for a commercial building adjacent to industrial premises
Case study 5	**Application of HILs and EILs –** redevelopment of an industrial site for residential use

For the purposes of illustration, selected summary data only is presented and it can be assumed that the raw data has been evaluated in accordance with the guidance in Section 11 Schedule B2 and that the data has been assessed as being accurate and representative of the site.

Case study 1 - Assessment of asbestos contamination in soil –
poor demolition practices at a residential site

Site scenario

Typical low density residential site (individual house site) where poor demolition practices have resulted in fragments of bonded ACM being scattered over discrete area(s) of the site surface. In this scenario, there are no substantial fill materials or other sources of potential contamination present at the site. The demolition has occurred in the recent past and no further soil disturbing activities, including removal of sub-surface utilities, have taken place since the buildings were demolished.

Response

Conduct a PSI and a grid-based site inspection survey (walkover) including detailed notes of bonded ACM distribution and condition and nature of surface soils.

A DSI is not necessary provided that the contamination is only at surface/near surface and the bonded ACM is in good condition (non-friable).

The extent of the affected area(s) should be carefully documented and all visible asbestos removed. As the site walkover confirmed that the surface soils were sandy, fragments of bonded ACM can be removed effectively by raking and hand-picking (refer WA DoH (2009) for details of recommended methodology). Sufficient raking passes should be conducted to ensure that the raked depth (approximately 10 cm) is free of visible asbestos.

In this scenario, it is not necessary to sample and analyse surface soils to confirm that no asbestos fibres are present given that the only type of asbestos present is bonded ACM and that it is not severely weathered.

Case study 2 - Assessment of asbestos contamination in soil – redevelopment of an industrial site for residential use

Site Scenario

A former industrial site is proposed for redevelopment for high density residential land use. The site was historically filled in some areas with material containing bonded ACM to approximately 3 m depth (possibly as a result of poor demolition practices). More recently, the contaminated fill was covered by approximately 0.5 m of clean soil as an interim management measure. The proposed development will require that there is major site excavation to >3 m as well as alteration of the land form. Although broken, the bonded ACM fragments appear in reasonable condition and are not easily crumbled i.e. not fibrous asbestos. There is no evidence from the site history or direct observation during the initial site walkover that other fibrous asbestos materials (such as insulation or woven materials) are present on the site. Other non-asbestos soil contaminants may be present.

Response

In this redevelopment scenario, there are two potential options:

- Option A - excavate all the affected fill (and validate the work undertaken including that no visible asbestos is present on the site surface) and either manage by containment on-site or off-site disposal at an appropriate waste facility
- Option B - carry out a DSI to delineate the volume of contaminated soil requiring on-site containment or off-site disposal

The size of the site, the potential volume of affected fill and the practicality (including regulatory requirements) of containing asbestos-contaminated soil on-site, are likely to influence the decision taken.

The following steps outline Option B - the DSI approach:

1 ***Preliminary site investigation*** – desktop study and detailed site inspection
   - collect information on the location, condition and amount of bonded ACM present on the site surface to inform the SAQP for the DSI.

2 ***Preparation of the Sampling and Analysis Quality Plan (SAQP)*** - A conceptual site model (CSM) and data quality objectives (DQOs) should be developed which identify all the site-specific contaminants of concern including relevant forms of asbestos and the potential human health risks (refer Schedule B2).
   - the sampling program should account for the potentially non-homogenous distribution and condition of bonded ACM in soil, for example using judgemental sampling involving a detailed test pit and trenching program to identify the lateral and vertical distribution
   - photographic logging of test pits and trenches will assist documentation for site assessment
   - qualitative laboratory analysis may be required to confirm that representative pieces of suspect bonded ACM and other suspect material (if found during the site walkover or during test pitting and trenching) contain asbestos.
   - soil sampling for the detection of asbestos fibres released from fragments of bonded ACM is not required where the bonded ACM is in good/reasonable condition.
   - if fibrous asbestos (such as severely weathered bonded ACM or insulation materials) is not observed during the field sampling program or indicated by the laboratory

analysis of the selected suspect materials, no further consideration or action for this form of asbestos is required

- if asbestos fibres are detected by qualitative laboratory analysis, appropriate remediation and management action will be required (a conservative management approach which does not rely on extensive soil sampling for the presence/absence of asbestos fibres is recommended).

3 ***Intrusive investigation to delineate impacted area***

- gravimetric analysis of each fill area will be required and the bonded ACM results for each area compared with the relevant screening level
- a weight-of-evidence approach should be adopted for the assessment with consideration given to the distribution of different fill types, the heterogeneity of the contamination (including condition of bonded ACM) and the uncertainty associated with the sampling methodology
- if there is uncertainty that the screening level is exceeded, additional systematic sampling and gravimetric determination could be undertaken or a conservative approach to management adopted
- areas where the screening levels are not exceeded require no further action or assessment in relation to asbestos other than ensuring that no visible asbestos is present at surface.

4 ***Management/remediation*** of areas of elevated levels of bonded ACM and/or fibrous asbestos by

- bulk screening of impacted site soils to remove bonded ACM (only feasible for sandy soils)
- on-site containment in accordance with jurisdictional requirements[1] or
- disposal to an appropriate waste facility
- no visible asbestos should be present at the completion of remediation works.

1 These requirements will consider human health risks arising from current and potential future land uses. They may include mandating of barrier layers, containment cells, depth of burial, ongoing monitoring and other statutory conditions of site use (e.g. as listed in a site management plan).

Case study 3 - Application of petroleum hydrocarbon screening levels – redevelopment of an industrial site for residential use

Site Scenario - Former small-scale regional fuel depot proposed for low-density residential use

After the site ceased to operate as a fuel depot in the mid-1980s, all tanks were removed and the site is understood to have been filled with clean silt to 2 m depth shortly after the depot was decommissioned. For the past 15 years the site has been used for storage of motor vehicles and agricultural equipment.

Response

A PSI was carried out and an initial CSM developed. The site has been investigated (including the 'clean' fill) according to an appropriate SAQP informed by the CSM. A source of contamination has been identified in the unsaturated zone which has an associated contaminated groundwater plume. Depth to groundwater is approximately 6 m. Soil and groundwater samples have been analysed for TRH fractions, BTEX, PAHs and lead.

It can be assumed for the purpose of this case study that the maximum slab width for the proposed residential dwellings is less than 15 m.

Summary of site contamination

- A preliminary screening step (refer Section 9.2.1 of Schedule B2) has determined that an assessment of potential vapour intrusion risks is necessary as receptors (residents of houses) are to be located within 30 m of an identified volatile source.
- The HSL assumptions and limitations were checked with the aid of the HSL checklist (ASC NEPM Toolbox) and the HSLs confirmed to be applicable for the site-specific conditions.
- BTEX and naphthalene were subtracted from TRH fractions C_6 – C_{10} and >C_{10} – C_{16} to obtain F1 and F2 respectively.
- Fill layer 0–2 m below ground level – 95% UCL for all the identified contaminants of concern was less than the appropriate investigation and screening levels.
- Soil Type: The borelogs indicate silt and silty clay, predominant soil type determined to be silt (the HSLs for silt are more conservative than those for clay).
- The geometric mean (GM) for TRH and BTEX in soil for 2–4 m is tabulated below.
- All individual soil results are less than 2.5x the relevant investigation and screening levels; hotspots, if present, would need to be considered separately.
- No contamination of concern was found below 4 m in soil.
- Poor quality groundwater was found at 6 m in three wells MW1, MW2, MW3 (saline, TDS >5000 mg/L, low yield <2 L/sec)
- The maximum concentrations (based on quarterly monitoring results carried out over one year) for TRH and BTEX in groundwater are listed below.
- Soil vapour oxygen measurements of 9–10% were measured at 1 m depth at five locations above the soil source.

Step 1: Document results and select relevant soil and groundwater HSLs

Soil GM values mg/kg, refer Table 1A(3) for soil HSLs (silt, 2–<4 m)

	F1	F2	F3	F4	B	T	E	X
Soil GM	**130**	160	1100	260	**1.5**	80	70	60
HSL A	***100***	***NL***	***N/A***	***N/A***	***1.0***	***NL***	***NL***	***NL***

Notes

NL indicates the HSL is not limiting (see Footnote 5, Table 1A(3)).

N/A not applicable as these fractions are not volatile and hence are not of concern for vapour intrusion

Shaded and bold font for sample value indicates relevant HSL exceeded

Groundwater (site maximum concentration) values mg/L, refer Table 1A(4) for groundwater HSLs (silt, 4 m to <8 m)

	F1	F2	F3	F4	B	T	E	X
MW1	1.3	0.9	<LOR	<LOR	**7**	16	12	35
MW2	0.5	<LOR	<LOR	<LOR	<LOR	<LOR	<LOR	4
MW3	2.7	1.1	<LOR	<LOR	**8**	17	23	42
HSL A	***6***	***NL***	***N/A***	***N/A***	***5***	***NL***	***NL***	***NL***

Notes

LOR is the limit of reporting.

NL indicates the HSL is not limiting (see Footnote 4, Table 1A(4)).

N/A not applicable as these fractions are non-volatile and hence are not of concern for vapour intrusion

Shaded and bold font for sample value indicates relevant HSL exceeded

Step 2: Tier 1 risk assessment

1. **Are site values greater than soil and groundwater HSLs for assessing vapour intrusion risks**? *YES, elevated F1 and benzene in soil and elevated benzene in groundwater.*
2. **Is biodegradation applicable?** *YES (from consideration of likely slab size for a typical residential house and oxygen content of soil vapour at 1m) Adjust soil HSLs x10 (soil depth 2 – 4 m) and groundwater HSLs x100 (depth to groundwater 6 m) (see Notes Table 1A(3) and Table 1A(4)and Friebel & Nadebaum 2011a).*

Adjusted HSL values for soil and groundwater –biodegradation factors applied

HSL	F1	F2	F3	F4	B	T	E	X
Soil 2-4 m	*NL*	*NL*	*N/A*	*N/A*	*10*	*NL*	*NL*	*NL*
Groundwater 4 m to <8 m	***NL***	***NL***	***N/A***	***N/A***	***50***	***NL***	***NL***	***NL***

Notes

Confirmation of soil oxygen > 5% at 1 m depth allows a biodegradation factor of x10 for vapour sources from 2 m–<4 m. Similarly, a biodegradation factor of x100 applies to groundwater vapour sources >4 m which takes adjusted HSLs to above the non limiting threshold value except for benzene in the example above.

NL indicates the HSL is not limiting (see Footnote (5) Table 1A(3) and Footnote (4) Table 1A(4) .

N/A not applicable as these fractions are non-volatile and hence not of concern for vapour inhalation

3. **Are site values greater than adjusted HSLs for vapour intrusion**? *NO*
4. **Are direct contact HSLs relevant?** *YES* ***(proposed low density residential land use)*** however there is no soil contamination at surface (95% UCL for all the identified contaminants of concern was less than the appropriate investigation and screening levels).

5. **Are ecological considerations relevant?** ***YES*** *(proposed low density residential development with exposed areas of soil).* Site summary information indicates that the soil GM for 0–2 m was less than the applicable ESLs. A comparison with soil data for deeper horizons is not relevant as the ESLs are applicable to the top 2 m of soil.

6. **Are management limits relevant?** ***YES (decommissioned industrial site proposed for sensitive land use).*** Compare soil results with the relevant management limits for residential use. ***NO exceedences indicated.***

'Clean fill' and soil values mg/kg, refer Table 1B(7) for management limits (fine soil)

	F1	F2	F3	F4
Fill GM 0 – 2 m	<10	< 10	<50	<100
Soil GM 2 – 4 m	130	160	1,100	260
Management limit	***800***	***1,000***	***3,500***	***10,000***

7. **Are aesthetics relevant?** ***YES (sensitive land use proposed)*** As 2 m of clean fill is present across the site issues of soil staining or odours are unlikely. The assessor will also need to consider the likelihood of uncontrolled excavations exposing contaminated material at depth.

Outcome	• No exceedences are indicated from the comparisons with the relevant HSLs, ESLs and management limits for the proposed residential land use. • Evaluation of the data for naphthalene, BaP, total PAHs and lead would also be required.

Evaluation and conclusion

A multiple-lines-of-evidence approach is recommended for the evaluation of vapour intrusion risks. Although no unacceptable vapour intrusion risks were identified in the assessment above, the assessor would need to take into account the level of uncertainty associated with the data and whether a sufficient margin of safety was present, particularly in relation to the adjusted groundwater HSLs.

Further consideration should be given to the confidence in the site CSM particularly with regards to seasonal trends in groundwater quality and possible variation in depth to the water table. If the watertable is likely to rise by more than 2 m, then the maximum concentrations of benzene recorded in MW 1 (7 mg/L) and MW3 (8 mg/L) would be close to/at the level of the adjusted HSL of 8 mg/L for 2 m – <4 m depth (Table 1A(4) with x10 adjustment).

Given the sensitivity of the proposed land use (low density residential), consideration should be given to collecting further data such as conducting a soil vapour survey of the source area.

The level of groundwater contamination present is of concern. The groundwater quality is unacceptable for human consumption and should be restricted for use by site occupants. Potable use is unrealistic given the poor groundwater quality and yield, however, it could cause adverse effects on potential ecosystem receptors. Further consideration should be given to groundwater contamination regarding any potential receptors off-site and any realistic future use potential. State and local groundwater protection policies would take effect in applying controls over the presence, extraction and use of impacted groundwater.

Note, it would not be an appropriate approach to install a thickness of fill to cover hydrocarbon contamination to enable the use of less stringent HSLs or to enable the application of a 'x10' or 'x100' biodegradation factor.

Case study 4: Application of soil vapour interim VOCC HILs and HSLs – vapour intrusion assessment for a commercial building adjacent to industrial premises

Site Scenario

A drum reconditioning works is located beside a four-storey office building in an industrial estate. A drum pre-clean area attached to the works has leaked chemical wastes to the subsurface. Limited soil and groundwater sampling, constrained by existing infrastructure and land uses, have detected TCE, PCE and derivatives, BTEX and TRH fractions in soil and groundwater bores. Initial results suggested a potential human health risk to ground floor occupants of the office block. The surface of the site comprises sealed hardstand.

Response

The soil and groundwater sampling has been followed up with soil vapour samples at 0–1 m depth located in the bituminised area immediately adjacent to the office block at four locations to further assess the human health risk.

- A preliminary screening step (refer Section 9.2.1 of Schedule B2) has determined that an assessment of potential vapour intrusion risks is necessary as receptors (occupants of office block) are located within 30 m of an identified volatile source.
- The HSL assumptions and limitations were checked with the aid of the HSL checklist – (ASC NEPM Toolbox) and the HSLs confirmed to be applicable for the site-specific conditions.
- BTEX and naphthalene were subtracted from TRH fractions C_6 – C_{10} and >C_{10} – C_{16} to obtain F1 and F2 respectively (note F2 data not presented here).
- Soil Type - Predominant soil type determined to be sand.
- Biodegradation is not a consideration as the office block concrete slab and contiguous bituminised area is >15 m wide.
-

Step 1: Document results and select interim VOCC HILs and soil vapour HSLs

Soil vapour values mg/m³, refer Table 1A(2) for interim VOCC HILs and Table 1A(5) for soil vapour HSLs for 0-1 m (sand)

Sample	TCE	PCE	Vinyl chloride	B	T	E	X	F1
SG1	**22**	**110**	**6**	**7**	25	44	60	120
SG2	**30**	**130**	**17**	**9**	60	52	40	**200**
SG3	**7**	**75**	**1.5**	**5**	8	18	20	80
SG4	**4**	**30**	**1.3**	**3**	10	21	25	70
Interim HIL or HSL	***0.02***	***2***	***0.03***	***1***	***1,300***	***330***	***220***	***180***

Note: Shaded and bold font for sample value indicates relevant interim VOCC HIL or HSL is exceeded.

Step 2: Tier 1 risk assessment

1. **Are results greater than the soil vapour HSLs?** ***YES****, benzene exceeds the HSL in all locations and F1 in one location.*
2. **Are results greater than the soil vapour interim VOCC HILs?** ***YES****, TCE, PCE and vinyl chloride exceed the interim HILs for VOCCs in all sampling locations.*
3. **Are ecological considerations relevant?** ***NO for on-site only*** *(commercial industrial development with no exposed areas of soil and continuing industrial use).*
4. **Are petroleum management limits exceeded?** ***No*** **(***Compare results to management limits in Table 1B(7). F1 results do not exceed the management limits.)*

Outcome	• Results of Tier 1 assessment show exceedences of the HSL for benzene and interim VOCC HILs for TCE, PCE and vinyl chloride. Further assessment (Tier 2) or management action is required for these contaminants.

Evaluation and conclusion

The results indicate a potentially serious human health risk via the vapour inhalation pathway from benzene, TCE, PCE and VOCC derivatives to ground floor occupants of the office building. As the exceedences are 2–4 orders of magnitude above the interim soil vapour HILs, an immediate response is required to protect human health such as indoor air sampling to determine actual exposure and/or implementing mitigation measures.

This example is limited to consideration of health risks from selected petroleum hydrocarbons and VOCC inhalation exposure. Additional assessment would be required for other petroleum hydrocarbons and also to evaluate any off-site ecological risks for example via infiltration of contaminated groundwater into sewer or stormwater drainage systems and/or discharge into a sensitive receptor.

Case study 5 - Use of HILs and EILs – redevelopment of an industrial site for residential use

Site Scenario

The site is a former electroplating works and is proposed for residential townhouse development with individual gardens. Prior to industrial use, the whole site was filled from 0 to 1 m with imported clay/soil fill of uniform characteristics.

Response

The site was assessed by a detailed sampling program based on a well-documented site history and no contamination of concern was found below 2 m. Based on site history, CrVI was included in the sampling and analysis but not detected. Representative samples of site soils were analysed for cation exchange capacity (CEC), clay content and pH to assist with ecological assessment.

For the purposes of illustration, the generalised geometric mean (GM) data shown below is assumed to be sufficient from a statistical basis to describe and evaluate the condition of the site. All relevant contaminants of concern were identified and the original surface stratum has uniform characteristics across the site. Hot spots, if present, would need to be considered separately.

Step 1: Document soil results and select HILs

Depth (m)	Cu mg/kg	Zn mg/kg	Ni mg/kg	CrIII mg/kg	CEC (cmol/kg)	pH pH units	% Clay %
0–1	540	890	**660**	1100	9	6.0	10
1–2	170	470	380	400	17	6.5	12
HIL A	***6,000***	***7,400***	***400***	***N/A***	-	-	-

Note: N/A= not applicable due to the low human toxicity of CrIII.

Shaded and bold font for sample value indicates HIL exceeded.

Step 2: Tier 1 health risk assessment

1. **Are site values greater than HILs?** ***YES****, elevated Ni level requires further health risk assessment.*

Step 3: Tier 1 ecological risk assessment

2. **Determine site EILs (EIL = ABC + ACL) or use the EILs spreadsheet in the ASC NEPM Toolbox.**

3. **Determine the added contaminant limits (ACLs)**

   Examination of the site history indicates that the contamination has been present for over 2 years and therefore ACLs for aged contamination are appropriate. To determine site ACLs, refer Table 1B(1) for Zn, Table 1B(2) for Cu and Table 1B(3) for CrIII and Ni. Establish the site ACL for the appropriate land use and with consideration of the soil-specific pH, clay content or CEC as required. Select the nearest ACL value in the CEC table. The ACL for Cu may be determined by pH or CEC and the lower of the determined values should be selected for EIL calculation.

	Depth (m)	Cu	Zn	Ni	CrIII
Site ACLs (mg/kg)	**0–1**	190	400	170	400
	1–2	210	590	270	400

4. **Measure the ambient background (ABC) at an appropriate reference location.**

- 0–1 m clay/soil fill, sampled from filled area at rear of property known to be unaffected by subsequent industrial activity.
- 1–2 m, sample of uncontaminated strata from adjacent site.

	Depth (m)	Cu	Zn	Ni	CrIII
ABC (mg/kg)	**0–1**	4	65	2	7
	1–2	1.5	8	0.5	10

5. Calculate the site EILs (ABC + ACL)

	Depth (m)	Cu	Zn	Ni	CrIII
Site EILs (mg/kg)	**0–1**	194	465	172	407
	1–2	211.5	598	270.5	410

- Round results for reasons of consistency and avoidance of false accuracy[1]

	Depth (m)	Cu	Zn	Ni	CrIII
Site EILs (mg/kg)	**0–1**	190	465	170	410
	1–2	210	600	270	410

6. **Compare site data with EILs**

	Depth (m)	Cu	Zn	Ni	CrIII
Site data (mg/kg)	**0–1**	**540**	**890**	**660**	**1100**
	1–2	170	470	**380**	400

Note: Shaded and bold font for sample values indicates EIL exceeded

Are results greater than EILs? ***YES****, Cu, Zn, Ni and CrIII exceed EILs – further investigation required.*

Tier 1 outcome	Exceedences of the HIL for Ni and EILs for Cu, Zn, Ni and Cr III.

Evaluation and conclusion

[1] The following rounding rules are applicable to the EILs

Nos < 1	to nearest 0.1
1 - <10	to nearest integer
10 - < 100	to nearest 5
100 - <1000	to nearest 10
≥1000	to nearest 100

The geometric mean exceedences are 2–3 times the relevant screening level hence further investigation or management is required.

This example is limited to consideration of risks from exposure to metals in soil. Additional assessment would be required to evaluate groundwater issues at the site.

6 Tabulated investigation and screening levels

ROUNDING APPLIED TO INVESTIGATION AND SCREENING LEVELS

Tables 1A (HILs and interim HILs)

Rounded to 1 or 2 significant figures (see Schedule B7 Appendix C for details)

Tables 1A (HSLs) and 1B (EILs and ESLs) rounding rules

< 1	**to nearest 0.1**
1–<10	**to nearest whole number**
1–< 100	**to nearest 5**
100–<1,000	**to nearest 10**
1,000–<10,000	**to nearest 100**
≥10,000	**to nearest 1,000**

Numbers ending in '5' are rounded up, for example:

0.05 rounded to 0.1

1.5 rounded to 2

115 rounded to 120

Table 1A(1) Health investigation levels for soil contaminants

Chemical	Health-based investigation levels (mg/kg)			
	Residential[1] A	Residential[1] B	Recreational[1] C	Commercial/ industrial[1] D
Metals and Inorganics				
Arsenic[2]	100	500	300	3 000
Beryllium	60	90	90	500
Boron	4500	40 000	20 000	300 000
Cadmium	20	150	90	900
Chromium (VI)	100	500	300	3600
Cobalt	100	600	300	4000
Copper	6000	30 000	17 000	240 000
Lead[3]	300	1200	600	1 500
Manganese	3800	14 000	19 000	60 000
Mercury (inorganic)[5]	40	120	80	730
Methyl mercury[4]	10	30	13	180
Nickel	400	1200	1200	6 000
Selenium	200	1400	700	10 000
Zinc	7400	60 000	30 000	400 000
Cyanide (free)	250	300	240	1 500
Polycyclic Aromatic Hydrocarbons (PAHs)				
Carcinogenic PAHs (as BaP TEQ)[6]	3	4	3	40
Total PAHs[7]	300	400	300	4000
Phenols				
Phenol	3000	45 000	40 000	240 000
Pentachlorophenol	100	130	120	660
Cresols	400	4 700	4 000	25 000
Organochlorine Pesticides				
DDT+DDE+DDD	240	600	400	3600
Aldrin and dieldrin	6	10	10	45
Chlordane	50	90	70	530
Endosulfan	270	400	340	2000
Endrin	10	20	20	100
Heptachlor	6	10	10	50
HCB	10	15	10	80
Methoxychlor	300	500	400	2500
Mirex	10	20	20	100
Toxaphene	20	30	30	160
Herbicides				
2,4,5-T	600	900	800	5000
2,4-D	900	1600	1300	9000
MCPA	600	900	800	5000

Chemical	Health-based investigation levels (mg/kg)			
	Residential[1] A	Residential[1] B	Recreational[1] C	Commercial/ industrial[1] D
MCPB	600	900	800	5000
Mecoprop	600	900	800	5000
Picloram	4500	6600	5700	35000
Other Pesticides				
Atrazine	320	470	400	2500
Chlorpyrifos	160	340	250	2000
Bifenthrin	600	840	730	4500
Other Organics				
PCBs[8]	1	1	1	7
PBDE Flame Retardants (Br1–Br9)	1	2	2	10

Notes:

(1) Generic land uses are described in detail in Schedule B7 Section 3

HIL A – Residential with garden/accessible soil (home grown produce <10% fruit and vegetable intake (no poultry), also includes childcare centres, preschools and primary schools.

HIL B – Residential with minimal opportunities for soil access; includes dwellings with fully and permanently paved yard space such as high-rise buildings and apartments.

HIL C – Public open space such as parks, playgrounds, playing fields (e.g. ovals), secondary schools and footpaths. This does not include undeveloped public open space where the potential for exposure is lower and where a site-specific assessment may be more appropriate.

HIL D – Commercial/industrial, includes premises such as shops, offices, factories and industrial sites.

(2) Arsenic: HIL assumes 70% oral bioavailability. Site-specific bioavailability may be important and should be considered where appropriate (refer Schedule B7).

(3) Lead: HIL is based on blood lead models (IEUBK for HILs A, B and C and adult lead model for HIL D where 50% oral bioavailability has been considered. Site-specific bioavailability may be important and should be considered where appropriate.

(4) Methyl mercury: assessment of methyl mercury should only occur where there is evidence of its potential source. It may be associated with inorganic mercury and anaerobic microorganism activity in aquatic environments. In addition the reliability and quality of sampling/analysis should be considered.

(5) Elemental mercury: HIL does not address elemental mercury. A site-specific assessment should be considered if elemental mercury is present, or suspected to be present,

(6) Carcinogenic PAHs: HIL is based on the 8 carcinogenic PAHs and their TEFs (potency relative to B(a)P) adopted by CCME 2008 (refer Schedule B7). The B(a)P TEQ is calculated by multiplying the concentration of each carcinogenic PAH in the sample by its B(a)P TEF, given below, and summing these products.

PAH species	TEF	PAH species	TEF
Benzo(a)anthracene	0.1	Benzo(g,h,i)perylene	0.01
Benzo(a)pyrene	1	Chrysene	0.01
Benzo(b+j)fluoranthene	0.1	Dibenz(a,h)anthracene	1
Benzo(k)fluoranthene	0.1	Indeno(1,2,3-c,d)pyrene	0.1

Where the B(a)P occurs in bitumen fragments it is relatively immobile and does not represent a significant health risk.

(7) Total PAHs: HIL is based on the sum of the 16 PAHs most commonly reported for contaminated sites (WHO 1998). The application of the total PAH HIL should consider the presence of carcinogenic PAHs and naphthalene (the most volatile PAH). Carcinogenic PAHs reported in the total PAHs should meet the B(a)P TEQ HIL. Naphthalene reported in the total PAHs should meet the relevant HSL.

(8) PCBs: HIL relates to non-dioxin-like PCBs only. Where a PCB source is known, or suspected, to be present at a site, a site-specific assessment of exposure to all PCBs (including dioxin-like PCBs) should be undertaken.

Table 1A(2) Interim soil vapour health investigation levels for volatile organic chlorinated compounds

Chemical	Interim soil vapour HIL (mg/m^3)			
	Residential[1] A	Residential[1] B	Recreational[1] C	Commercial / Industrial[1] D
TCE	0.02	0.02	0.4	0.08
1,1,1-TCA	60	60	1200	230
PCE	2	2	40	8
cis-1,2-dichloroethene	0.08	0.08	2	0.3
Vinyl chloride	0.03	0.03	0.5	0.1

Notes:

1. Land use settings are equivalent to those described in Table 1A(1) Footnote 1 and Schedule B7, though secondary school buildings should be assessed using residential 'A/B' for vapour intrusion purposes.
2. Interim HILs for VOCCs are conservative soil vapour concentrations that can be adopted for the purpose of screening sites where further investigation is required on a site-specific basis. They are based on the potential for vapour intrusion using an indoor air-to-soil vapour attenuation factor of 0.1 and an outdoor air-to-soil vapour attenuation factor of 0.05.
3. Application of the interim HILs is based on a measurement of shallow (to 1 m depth) soil vapour (or deeper where the values are to be applied to a future building with a basement) or sub-slab soil vapour.
4. The applicability of the interim HILs needs to be further considered when used for other building types such as homes with a crawl-space and no slab, which may require site-specific assessment.
5. Use of the interim HILs requires comparison with data that has been collected using appropriate methods and meets appropriate data quality requirements.
6. Oral and dermal exposure should be considered on a site-specific basis where direct contact exposure is likely to occur.

Table 1A(3) Soil HSLs for vapour intrusion (mg/kg)

	HSL A & HSL B Low – high density residential				HSL C recreational / open space				HSL D Commercial / Industrial				
CHEMICAL	**0 m to <1 m**	**1 m to <2 m**	**2 m to <4m**	**4 m+**	**0 m to <1 m**	**1 m to <2 m**	**2 m to <4 m**	**4 m+**	**0 m to <1 m**	**1 m to <2 m**	**2 m to <4 m**	**4 m+**	**Soil saturation concentration (Csat)**
SAND													
Toluene	160	220	310	540	NL	NL	NL	NL	NL	NL	NL	NL	560
Ethylbenzene	55	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	64
Xylenes	40	60	95	170	NL	NL	NL	NL	230	NL	NL	NL	300
Naphthalene	3	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	9
Benzene	0.5	0.5	0.5	0.5	NL	NL	NL	NL	3	3	3	3	360
F1[9]	45	70	110	200	NL	NL	NL	NL	260	370	630	NL	950
F2[10]	110	240	440	NL	NL	NL	NL	NL	NL	NL	NL	NL	560
SILT													
Toluene	390	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	640
Ethylbenzene	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	69
Xylenes	95	210	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	330

	HSL A & HSL B Low – high density residential				HSL C recreational / open space				HSL D Commercial / Industrial				
Naphthalene	4	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	10
Benzene	0.6	0.7	1	2	NL	NL	NL	NL	4	4	6	10	440
F1[9]	40	65	100	190	NL	NL	NL	NL	250	360	590	NL	910
F2[10]	230	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	570
CLAY													
Toluene	480	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	630
Ethylbenzene	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	68
Xylenes	110	310	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	330
Naphthalene	5	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	10
Benzene	0.7	1	2	3	NL	NL	NL	NL	4	6	9	20	430
F1[9]	50	90	150	290	NL	NL	NL	NL	310	480	NL	NL	850
F2[10]	280	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	NL	560

Notes:

(1) Land use settings are equivalent to those described in Table 1A(1) Footnote 1 and Schedule B7. HSLs for vapour intrusion for high density residential assume residential occupation of the ground floor. If communal car parks or commercial properties occupy the ground floor, HSL D should be used,

(2) The key limitations of the HSLs should be referred to prior to application and are presented in Friebel and Nadebaum (2011b and 2011d).

(3) Detailed assumptions in the derivation of the HSLs and information on how to apply the HSLs are presented in Friebel and Nadebaum (2011a and 2011b).

(4) Soil HSLs for vapour inhalation incorporate an adjustment factor of 10 applied to the vapour phase partitioning to reflect the differences observed between theoretical estimates of soil vapour partitioning and field measurements. Refer Friebel & Nadebaum (2011a) for further information.

(5) The soil saturation concentration (Csat) is defined as the soil concentration at which the porewater phase cannot dissolve any more of an individual chemical. The soil vapour that is in equilibrium with the porewater will be at its maximum. If the derived soil HSL exceeds Csat, a soil vapour source concentration for a petroleum mixture could not exceed a level that would result in the maximum allowable vapour risk for the given scenario. For these scenarios, no HSL is presented for these chemicals and the HSL is shown as ‘not limiting’ or ‘NL’.

(6) The HSLs for TPH C_6-C_{10} in sandy soil are based on a finite source that depletes in less than seven years, and therefore consideration has been given to use of sub-chronic toxicity values. The >C_8-C_{10} aliphatic toxicity has been adjusted to represent sub-chronic exposure, resulting in higher HSLs than if based on chronic toxicity. For further information refer to Section 8.2 and Appendix J in Friebel and Nadebaum (2011a).

(7) The figures in the above table may be multiplied by a factor to account for biodegradation of vapour. A factor of 10 may apply for source depths from 2 m to <4 m or a factor of 100 for source depths of 4 m and deeper. To apply the attenuation factor for vapour degradation, a number of conditions must be satisfied. Firstly the maximum length of the shorter side of the concrete slab and surrounding pavement cannot exceed 15 m, as this would prevent oxygen penetrating to the centre of the slab. Secondly, measurement of oxygen in the subsurface is required to determine the potential for biodegradation. Oxygen must be confirmed to be present at >5% to use these factors.

(8) For soil texture classification undertaken in accord with AS 1726, the classifications of sand, silt and clay may be applied as coarse, fine with liquid limit <50% and fine with liquid limit>50% respectively, as the underlying properties to develop the HSLs may reasonably be selected to be similar. Where there is uncertainty, either a conservative approach may be adopted or laboratory analysis should be carried out.

(9) To obtain F1 subtract the sum of BTEX concentrations from the C_6-C_{10} fraction.

(10) To obtain F2 subtract naphthalene from the >C_{10}-C_{16} fraction.

Table 1A(4) Groundwater HSLs for vapour intrusion (mg/L)

	HSL A & HSL B Low – high density residential			HSL C recreational / open space			HSL D Commercial / industrial			
CHEMICAL	**2 m to <4 m**	**4 m to <8 m**	**8 m+**	**2 m to <4 m**	**4 m to <8 m**	**8 m+**	**2 m to <4 m**	**4 m to <8 m**	**8 m+**	**Solubility limit**
SAND										
Toluene	NL	NL	NL	NL	NL	NL	NL	NL	NL	61
Ethylbenzene	NL	NL	NL	NL	NL	NL	NL	NL	NL	3.9
Xylenes	NL	NL	NL	NL	NL	NL	NL	NL	NL	21
Naphthalene	NL	NL	NL	NL	NL	NL	NL	NL	NL	0.17
Benzene	0.8	0.8	0.9	NL	NL	NL	5	5	5	59
F1[7]	1	1	1	NL	NL	NL	6	6	7	9.0
F2[8]	1	1	1	NL	NL	NL	NL	NL	NL	3.0
SILT										
Toluene	NL	NL	NL	NL	NL	NL	NL	NL	NL	61
Ethylbenzene	NL	NL	NL	NL	NL	NL	NL	NL	NL	3.9
Xylenes	NL	NL	NL	NL	NL	NL	NL	NL	NL	21
Naphthalene	NL	NL	NL	NL	NL	NL	NL	NL	NL	0.17
Benzene	4	5	5	NL	NL	NL	30	30	30	59

	HSL A & HSL B Low – high density residential			HSL C recreational / open space			HSL D Commercial / industrial			
F1[7]	6	6	6	NL	NL	NL	NL	NL	NL	9.0
F2[8]	NL	NL	NL	NL	NL	NL	NL	NL	NL	3.0
CLAY										
Toluene	NL	NL	NL	NL	NL	NL	NL	NL	NL	61
Ethylbenzene	NL	NL	NL	NL	NL	NL	NL	NL	NL	3.9
Xylenes	NL	NL	NL	NL	NL	NL	NL	NL	NL	21
Naphthalene	NL	NL	NL	NL	NL	NL	NL	NL	NL	0.17
Benzene	5	5	5	NL	NL	NL	30	30	35	59
F1[7]	NL	NL	NL	NL	NL	NL	NL	NL	NL	9.0
F2[8]	NL	NL	NL	NL	NL	NL	NL	NL	NL	3.0

Notes:

(1) Land use settings are equivalent to those described in Table 1A(1) Footnote 1 and Schedule B7. HSLs for vapour intrusion for high density residential assume residential occupation of the ground floor. If communal car parks or commercial properties occupy the ground floor, HSL D should be used,

(2) The key limitations of the HSLs are presented in Friebel and Nadebaum (2011d) and should be referred to prior to application.

(3) Detailed assumptions in the derivation of the HSLs and information on the application of the HSLs are presented in Friebel and Nadebaum (2011a and 2011b).

(4) The solubility limit is defined as the groundwater concentration at which the water cannot dissolve any more of an individual chemical based on a petroleum mixture. The soil vapour that is in equilibrium with the groundwater will be at its maximum. If the derived groundwater HSL exceeds the water solubility limit, a soil vapour source concentration for a petroleum mixture could not exceed a level that would result in the maximum allowable vapour risk for the given scenario. For these scenarios, no HSL is presented for these chemicals and the HSL is shown as 'not limiting' or 'NL'.

(5) The figures in the above table may be multiplied by a factor to account for biodegradation of vapour. A factor of 10 may apply for source depths from 2 m to <4 m or a factor of 100 for source depths of 4 m and deeper. To apply the attenuation factor for vapour degradation, a number of conditions must be satisfied. Firstly, the maximum length of the shorter side of the concrete slab and surrounding pavement cannot exceed 15 m, as this would prevent oxygen penetrating to the centre of the slab. Secondly, measurement of oxygen in the subsurface is required to determine the potential for biodegradation. Oxygen must be confirmed to be present at >5% to use these factors.

(6) For soil texture classification undertaken in accord with AS 1726, the classifications of sand, silt and clay may be applied as coarse, fine with liquid limit <50% and fine with liquid limit >50% respectively, as the underlying properties to develop the HSLs may reasonably be selected to be similar. Where there is uncertainty, either a conservative approach may be adopted or laboratory analysis should be carried out.

(7) To obtain F1 subtract the sum of BTEX concentrations from the C_6-C_{10} fraction.

(8) To obtain F2 subtract naphthalene from the >C_{10}-C_{16} fraction.

Table 1A(5) Soil vapour HSLs for vapour intrusion (mg/m³)

	HSL A & HSL B Low – high density residential					HSL C recreational / open space					HSL D Commercial / Industrial				
CHEMICAL	**0 m to <1 m**	**1 m to <2 m**	**2 m to <4 m**	**4 m to <8 m**	**8 m+**	**0 m to <1 m**	**1 m to <2 m**	**2 m to <4 m**	**4 m to <8 m**	**8 m+**	**0 m to <1 m**	**1 m to <2 m**	**2 m to <4 m**	**4 m to <8 m**	**8 m+**
SAND															
Toluene	1300	3800	7300	15 000	29 000	NL	NL	NL	NL	NL	4800	16 000	39 000	84 000	NL
Ethylbenzene	330	1100	2200	4300	8700	NL	NL	NL	NL	NL	1300	4600	11 000	25 000	53 000
Xylenes	220	750	1500	3000	6100	NL	NL	NL	NL	NL	840	3,200	8000	18 000	37 000
Naphthalene	0.8	3	6	10	25	410	NL	NL	NL	NL	3	15	35	75	150
Benzene	1	3	6	10	20	360	2400	4700	9500	19 000	4	10	30	65	130
F1[8]	180	640	1,300	2600	5300	86 000	NL	NL	NL	NL	680	2800	7000	15 000	32 000
F2[9]	130	560	1200	2400	4800	NL	NL	NL	NL	NL	500	2400	NL	NL	NL
SILT															
Toluene	1400	14 000	32 000	69 000	140 000	NL	NL	NL	NL	NL	5700	63 000	NL	NL	NL
Ethylbenzene	380	4200	9700	21 000	43 000	NL	NL	NL	NL	NL	1500	19 000	54 000	NL	NL
Xylenes	260	2900	6800	15 000	30 000	NL	NL	NL	NL	NL	1000	13 000	38 000	NL	NL
Naphthalene	0.9	10	25	60	120	NL	NL	NL	NL	NL	4	50	150	350	750
Benzene	1	10	25	55	110	1800	12 000	24 000	48 000	97 000	4	50	140	320	670
F1[8]	210	2600	6000	13 000	26 000	NL	NL	NL	NL	NL	850	11 000	33 000	77 000	160 000
F2[9]	160	2300	5400	NL	NL	NL	NL	NL	NL	NL	670	NL	NL	NL	NL

	HSL A & HSL B Low – high density residential					HSL C recreational / open space					HSL D Commercial / Industrial				
CLAY															
Toluene	1600	23 000	53 000	110 000	NL	NL	NL	NL	NL	NL	6500	100 000	NL	NL	NL
Ethylbenzene	420	6800	16 000	35 000	NL	NL	NL	NL	NL	NL	1800	31 000	NL	NL	NL
Xylenes	280	4800	11 000	24 000	50 000	NL	NL	NL	NL	NL	1200	21 000	NL	NL	NL
Naphthalene	1	20	45	95	200	NL	NL	NL	NL	NL	4	85	240	560	1200
Benzene	1	15	40	90	180	3000	20 000	40 000	81 000	160 000	5	80	230	530	1100
F1[8]	230	4200	9900	21 000	44 000	NL	NL	NL	NL	NL	1000	19 000	55 000	130 000	270 000
F2[9]	180	3,800	NL	NL	NL	NL	NL	NL	NL	NL	800	NL	NL	NL	NL

1. Land use settings are equivalent to those described in Table 1A(1) Footnote 1 and Schedule B7. HSLs for vapour intrusion for high density residential assume residential occupation of the ground floor. If communal car parks or commercial properties occupy the ground floor, HSL D should be used,
2. The key limitations of the HSLs should be referred to prior to application and are presented in Friebel and Nadebaum (2011b and 2011d).
3. Detailed assumptions in the derivation of the HSLs and information on how to apply the HSLs are presented in Friebel and Nadebaum (2011a and 2011b).
4. The maximum possible soil vapour concentrations have been calculated based on vapour pressures of the pure chemicals. Where soil vapour HSLs exceed these values a soil-specific source concentration for a petroleum mixture could not exceed a level that would result in the maximum allowable vapour risk for the given scenario. For these scenarios, no HSL is presented for these chemicals and the HSL is shown as 'not limiting' or 'NL'.
5. Soil vapour HSLs should be compared with measurements taken as laterally close as possible to the soil or groundwater sources of vapour (i.e. within or above vapour sources). Consideration is required of where the sample is taken, the current condition of the site and the likely future condition of the site. Shallow gas measurements in open space (less than 1 m below ground surface) may be subject to influences of weather conditions and moisture.
6. The figures in the above table may be multiplied by a factor to account for biodegradation of vapour. A factor of 10 may apply for source depths from 2 m to <4 m or a factor of 100 for source depths of 4 m and deeper. To apply the attenuation factor for vapour degradation, a number of conditions must be satisfied. Firstly, the maximum length of the shorter side of the concrete slab and surrounding pavement cannot exceed 15 m, as this would prevent oxygen penetrating to the centre of the slab. Secondly, measurement of oxygen in the subsurface is required to determine the potential for biodegradation. Oxygen must be confirmed to be present at >5% to use these factors.
7. For soil texture classification undertaken in accord with AS 1726, the classifications of sand, silt and clay may be applied as coarse, fine with liquid limit <50% and fine with liquid limit >50% respectively as the underlying properties to develop the HSLs may reasonably be selected to be similar. Where there is uncertainty, either a conservative approach may be adopted or laboratory analysis should be carried out.
8. To obtain F1 subtract the sum of BTEX concentrations from the C_6-C_{10} fraction.
9. To obtain F2 subtract naphthalene from the >C_{10}-C_{16} fraction.

Table 1B(1) Soil-specific added contaminant limits for aged zinc in soil

Zn added contaminant limits (ACL, mg added contaminant/kg)						
Areas of ecological significance						
pH[a]	***CEC[b] ($cmol_c$/kg)***					
	5	***10***	***20***	***30***	***40***	***60***
4.0	15	20	20	20	20	20
4.5	20	25	25	25	25	25
5.0	30	40	40	40	40	40
5.5	40	60	60	60	60	60
6.0	50	90	90	90	90	90
6.5	50	90	130	130	130	130
7.0	50	90	150	190	190	190
7.5	50	90	150	210	260	280
Urban residential/public open space[1]						
pH[a]	***CEC[b] ($cmol_c$/kg)***					
	5	***10***	***20***	***30***	***40***	***60***
4.0	70	85	85	85	85	85
4.5	100	120	120	120	120	120
5.0	130	180	180	180	180	180
5.5	180	270	270	270	270	270
6.0	230	400	400	400	400	400
6.5	230	400	590	590	590	590
7.0	230	400	700	880	880	880
7.5	230	400	700	960	1200	1300
Commercial/industrial						
pH[a]	***CEC[b] ($cmol_c$/kg)***					
	5	***10***	***20***	***30***	***40***	***60***
4.0	110	130	130	130	130	130
4.5	150	190	190	190	190	190
5.0	210	290	290	290	290	290
5.5	280	420	420	420	420	420
6.0	360	620	620	620	620	620
6.5	360	620	920	920	920	920
7.0	360	620	1100	1400	1400	1400
7.5	360	620	1100	1500	1900	2000

1. Urban residential/public open space is broadly equivalent to the HIL A, HIL B and HIL C land use scenarios in Table 1A(1) Footnote 1 and as described in Schedule B7.
2. Aged values apply to contamination present in soil for at least two years. For fresh contamination refer to Schedule B5c.
3. The EIL is calculated from summing the ACL and the ABC.

a = pH measured using the $CaCl_2$ method (Rayment & Higginson 1992).

b = CEC measured using the silver thiourea method (Chabra et al. 1972).

Table 1B(2) Soil-specific added contaminant limits for aged copper in soils

Cu added contaminant limits (ACL, mg added contaminant/kg)					
Areas of ecological significance					
CEC ($cmol_c/kg$)[a] based					
5	***10***	***20***	***30***	***40***	***60***
30	65	70	70	75	80
pH[b] based					
4.5	***5.5***	***6***	***6.5***	***7.5***	***8.0***
20	45	65	90	190	270
Urban residential/public open space[1]					
CEC ($cmol_c/kg$)[a] based					
5	***10***	***20***	***30***	***40***	***60***
95	190	210	220	220	230
pH[b] based					
4.5	***5.5***	***6***	***6.5***	***7.5***	***8.0***
60	130	190	280	560	800
Commercial/industrial					
CEC ($cmol_c/kg$)[a] based					
5	***10***	***20***	***30***	***40***	***60***
140	280	300	320	330	340
pH[b] based					
4.5	***5.5***	***6***	***6.5***	***7.5***	***8.0***
85	190	280	400	830	1200

Notes:

1. Urban residential/public open space is broadly equivalent to the HIL A, HIL B and HIL C land use scenarios in Table 1A(1) Footnote 1 and as described in Schedule B7.
2. The lower of the CEC or the pH-based ACLs for the land use and soil conditions is the ACL to be used.
3. Aged values apply to contamination present in soil for at least two years. For fresh contamination refer to Schedule B5c.
4. The EIL is calculated from summing the ACL and the ABC.

a = CEC measured using the silver thiourea method (Chabra et al. 1972).

b = pH measured using the $CaCl_2$ method (Rayment & Higginson 1992).

Table 1B(3) Soil-specific added contaminant limits for aged chromium III and nickel in soil

CHEMICAL	Clay content (% clay)	Added contaminant limits (mg added contaminant/kg) for various land uses		
		Areas of ecological significance	Urban residential and public open space	Commercial and industrial
Chromium III	*1*	60	190	310
	2.5	80	250	420
	5	100	320	530
	≥10	130	400	660
Nickel	CEC[a] ($cmol_c/kg$)	Areas of ecological significance	Urban residential and public open space[1]	Commercial and industrial
	5	5	30	55
	10	30	170	290
	20	45	270	460
	30	60	350	600
	40	70	420	730
	60	95	560	960

Notes:

1. Urban residential/public open space is broadly equivalent to the HIL A, HIL B and HIL C land use scenarios in Table 1A(1) Footnote 1 and as described in Schedule B7.
2. Aged values apply to contamination present in soil for at least two years. For fresh contamination refer to Schedule B5c.
3. The EIL is calculated from summing the ACL and the ABC.

a = CEC measured using the silver thiourea method (Chabra et al. 1972).

Table 1B(4) Generic added contaminant limits for lead in soils irrespective of their physicochemical properties

	Pb added contaminant limit (ACL, mg added contaminant/kg) for various land uses		
CHEMICAL	Areas of ecological significance	Urban residential and public open space[1]	Commercial and industrial
Lead	470	1100	1800

Notes:

1. Urban residential/public open space is broadly equivalent to the HIL A, HIL B and HIL C land use scenarios in Table 1A(1) Footnote 1 and as described in Schedule B7.
2. Aged values are applicable to lead contamination present in soil for at least two years. For fresh contamination refer to Schedule B5c.
3. The EIL is calculated from summing the ACL and the ABC.

Table 1B(5) Generic EILs for aged As, fresh DDT and fresh naphthalene in soils irrespective of their physicochemical properties

	Ecological Investigation Levels (mg total contaminant/kg)		
CHEMICAL	**Areas of ecological significance**	**Urban residential and public open space[1]**	**Commercial and industrial**
Arsenic[2]	40	100	160
DDT[3]	3	180	640
Naphthalene[3]	10	170	370

Notes:

1. Urban residential/public open space is broadly equivalent to the HIL-A, HIL-B and HIL-C land use scenarios in Table 1A(1) Footnote 1 and as described in Schedule B7.
2. Aged values are applicable to arsenic contamination present in soil for at least two years. For fresh contamination refer to Schedule B5c.
3. Insufficient data was available to calculate aged values for DDT and naphthalene, consequently the values for fresh contamination should be used.
4. Insufficient data was available to calculate ACLs for As, DDT and naphthalene. The EIL should be taken directly from Table 1B(5).

Table 1B(6) ESLs for TPH fractions F1 – F4, BTEX and benzo(a)pyrene in soil

CHEMICAL	Soil texture	ESLs (mg/kg dry soil)		
		Areas of ecological significance	Urban residential and public open space	Commercial and industrial
F1 C_6-C_{10}	*Coarse/ Fine*	125*	180*	215*
F2 >C_{10}-C_{16}		25*	120*	170*
F3 >C_{16}-C_{34}	*Coarse*	-	300	1700
	Fine	-	1300	2500
F4 >C_{34}-C_{40}	*Coarse*	-	2800	3300
	Fine	-	5600	6600
Benzene	*Coarse*	10	50	75
	Fine	10	65	95
Toluene	*Coarse*	10	85	135
	Fine	65	105	135
Ethylbenzene	*Coarse*	1.5	70	165
	Fine	40	125	185
Xylenes	*Coarse*	10	105	180
	Fine	1.6	45	95
Benzo(a)pyrene	*Coarse*	0.7	0.7	0.7
	Fine	0.7	0.7	0.7

Notes:

(1) ESLs are of low reliability except where indicated by * which indicates that the ESL is of moderate reliability.

(2) '-' indicates that insufficient data was available to derive a value.

(3) To obtain F1, subtract the sum of BTEX concentrations from C_6-C_{10} fraction and subtract naphthalene from >C_{10}-C_{16} to obtain F2.

Table 1 B(7) Management Limits for TPH fractions F1–F4 in soil

TPH fraction	Soil texture	Management Limits[1] (mg/kg dry soil)	
		Residential, parkland and public open space	Commercial and industrial
F1[2] C_6- C_{10}	*Coarse*	700	700
	Fine	800	800
F2[2] >C_{10}-C_{16}	*Coarse*	1000	1000
	Fine	1000	1000
F3 >C_{16}-C_{34}	*Coarse*	2500	3500
	Fine	3500	5000
F4 >C_{34}-C_{40}	*Coarse*	10 000	10 000
	Fine	10 000	10 000

[1] Management limits are applied after consideration of relevant ESLs and HSLs

[2] Separate management limits for BTEX and naphthalene are not available hence these should not be subtracted from the relevant fractions to obtain F1 and F2.

Table 1C Groundwater Investigation Levels (GILs)

Substance		Groundwater Investigation Levels		
		Fresh Waters[A]	Marine Waters[A]	Drinking Water[B]
		(µg/L)	(µg/L)	(mg/L)
Metals and Metalloids				
Aluminium, Al pH>6.5		55	-	-
Antimony		-	-	0.003
Arsenic		24 as As(III) 13 as As(V)	-	0.01
Barium		-	-	2
Beryllium		-	-	0.06
Boron		370[C]	-	4
Cadmium	H	0.2	0.7[D]	0.002
Chromium, Cr (III)	H	-	27	-
Chromium, Cr (VI)		1[C]	4.4	0.05
Cobalt		-	1	-
Copper	H	1.4	1.3	2
Iron, (Total)		-	-	-
Lead	H	3.4	4.4	0.01
Manganese		1900[C]	-	0.5
Mercury (Total)		0.06[D]	0.1[D]	0.001
Molybdenum		-	-	0.05
Nickel	H	11	7	0.02
Selenium (Total)		5[D]	-	0.01
Silver		0.05	1.4	0.1
Tributyl tin (as Sn)		-	0.006[C]	-
Tributyl tin oxide		-	-	0.001
Uranium		-	-	0.017
Vanadium		-	100	-
Zinc	H	8[C]	15[C]	-
Non-metallic Inorganics				
Ammonia[E] (as NH_3-N at pH 8)		900[C]	910	-
Bromate		-	-	0.02
Chloride		-	-	-
Cyanide (as un-ionised Cn)		7	4	0.08
Fluoride		-	-	1.5
Hydrogen sulphide (un-ionised H_2S measured as S)		1	-	-
Iodide		-	-	0.5
Nitrate (as NO_3)		refer to	refer to	50

Substance	Groundwater Investigation Levels		
	Fresh Waters[A]	Marine Waters[A]	Drinking Water[B]
	(µg/L)	(µg/L)	(mg/L)
	guideline	guideline	
Nitrite (as NO_2)	refer to guideline	refer to guideline	3
Nitrogen	refer to guideline	refer to guideline	-
Phosphorus	refer to guideline	refer to guideline	-
Sulphate (as SO_4)	-	-	500
Organic alchohols/other organics			
Ethanol	1400	-	-
Ethylenediamine tetra-acetic acid (EDTA)	-	-	0.25
Formaldehyde	-	-	0.5
Nitrilotriacetic acid	-	-	0.2
Anilines			
Aniline	8	-	-
2,4-Dichloroaniline	7	-	-
3,4-Dichloroaniline	3	150	-
Chlorinated Alkanes			
Dichloromethane	-	-	0.004
Trichloromethane (chloroform)	-	-	0.003
Trihalomethanes (total)	-	-	0.25
Tetrachloromethane (carbon tetrachloride)	-	-	0.003
1,2-Dichloroethane	-	-	0.003
1,1,2-Trichloroethane	6500	1900	-
Hexachloroethane	290[D]	-	-
Chlorinated Alkenes			
Chloroethene (vinyl chloride)	-	-	0.0003
1,1-Dichloroethene	-	-	0.03
1,2-Dichoroethene	-	-	0.06
Tetrachloroethene (PCE) (Perchloroethene)	-	-	0.05
Chlorinated Benzenes			
Chlorobenzene	-	-	0.3
1,2- Dichlorobenzene	160	-	1.5
1,3- Dichlorobenzene	260	-	-
1,4- Dichlorobenzene	60	-	0.04

Substance	Groundwater Investigation Levels		
	Fresh Waters[A]	Marine Waters[A]	Drinking Water[B]
	(µg/L)	(µg/L)	(mg/L)
1,2,3- Trichlorobenzene	3[D]	-	0.03 for individual or total trichlorobenzenes
1,2,4- Trichlorobenzene	85[D]	20[D]	
1,3,5-Trichlorobenzene	-	-	
Polychlorinated Biphenyls (PCBs)			
Aroclor 1242	0.3[D]	-	-
Aroclor 1254	0.01[D]	-	-
Other Chlorinated Compounds			
Epichlorohydrin	-	-	0.1
Hexachlorobutadiene	-	-	0.0007
Monochloramine	-	-	3
Monocyclic Aromatic Hydrocarbons			
Benzene	950	500[C]	0.001
Toluene	-	-	0.8
Ethylbenzene	-	-	0.3
Xylenes	350 (as o-xylene) 200 (as p-xylene)	-	0.6
Styrene (Vinyl benzene)	-	-	0.03
Polycyclic Aromatic Hydrocarbons (PAHs)			
Naphthalene	16	50[C]	-
Benzo[a]pyrene	-	-	0.00001
Phenols			
Phenol	320	400	-
2-Chlorophenol	340[C]	-	0.3
4-Chlorophenol	220	-	-
2,4-Dichlorophenol	120	-	0.2
2,4,6-Trichlorophenol	3[D]	-	0.02
2,3,4,6-Tetrachlorophenol	10[D]	-	-
Pentachlorophenol	3.6[D]	11[D]	0.01
2,4-Dinitrophenol	45	-	-
Phthalates			
Dimethylphthalate	3700	-	-
Diethylphthalate	1000	-	-
Dibutylphthalate	10[D]	-	-
Di(2-ethylhexyl) phthalate	-	-	0.01

Substance	Groundwater Investigation Levels		
	Fresh Waters[A]	Marine Waters[A]	Drinking Water[B]
	(µg/L)	(µg/L)	(mg/L)
Pesticides			
Acephate	-	-	0.008
Aldicarb	-	-	0.004
Aldrin plus Dieldrin	-	-	0.0003
Ametryn	-	-	0.07
Amitraz	-	-	0.009
Amitrole	-	-	0.0009
Asulam	-	-	0.07
Atrazine	13	-	0.02
Azinphos-methyl	-	-	0.03
Benomyl	-	-	0.09
Bentazone	-	-	0.4
Bioresmethrin	-	-	0.1
Bromacil	-	-	0.4
Bromoxynil	-	-	0.01
Captan	-	-	0.4
Carbaryl	-	-	0.03
Carbendazim (Thiophanate-methyl)	-	-	0.09
Carbofuran	0.06	-	0.01
Carboxin	-	-	0.3
Carfentrazone-ethyl	-	-	0.1
Chlorantraniliprole	-	-	6
Chlordane	0.03[D]	-	0.002
Chlorfenvinphos	-	-	0.002
Chlorothalonil	-	-	0.05
Chlorpyrifos	0.01[D]	0.009[D]	0.01
Chlorsulfuron	-	-	0.2
Clopyralid	-	-	2
Cyfluthrin, Beta-cyfluthrin	-	-	0.05
Cypermethrin isomers	-	-	0.2
Cyprodinil	-	-	0.09
1,3-Dichloropropene	-	-	0.1
2,2-DPA	-	-	0.5
2,4-D [2,4-dichlorophenoxy acetic acid]	280	-	0.03
DDT	0.006[D]	-	0.009
Deltramethrin	-	-	0.04

Substance	Groundwater Investigation Levels		
	Fresh Waters[A]	Marine Waters[A]	Drinking Water[B]
	(µg/L)	(µg/L)	(mg/L)
Diazinon	0.01	-	0.004
Dicamba	-	-	0.1
Dichloroprop	-	-	0.1
Dichlorvos	-	-	0.005
Dicofol	-	-	0.004
Diclofop-methyl	-	-	0.005
Dieldrin plus Aldrin	-	-	0.0003
Diflubenzuron	-	-	0.07
Dimethoate	0.15	-	0.007
Diquat	1.4	-	0.007
Disulfoton	-	-	0.004
Diuron	-	-	0.02
Endosulfan	0.03[D]	0.005[D]	0.02
Endothal	-	-	0.1
Endrin	0.01[D]	0.004[D]	-
EPTC	-	-	0.3
Esfenvalerate	-	-	0.03
Ethion	-	-	0.004
Ethoprophos	-	-	0.001
Etridiazole	-	-	0.1
Fenamiphos	-	-	0.0005
Fenarimol	-	-	0.04
Fenitrothion	0.2	-	0.007
Fenthion	-	-	0.007
Fenvalerate	-	-	0.06
Fipronil	-	-	0.0007
Flamprop-methyl	-	-	0.004
Fluometuron	-	-	0.07
Fluproponate	-	-	0.009
Glyphosate	370	-	1
Haloxyfop	-	-	0.001
Heptachlor	0.01[D]	-	-
Heptachlor epoxide	-	-	0.0003
Hexazinone	-	-	0.4
Imazapyr	-	-	9
Iprodione	-	-	0.1
Lindane (γ-HCH)	0.2	-	0.01

Substance	Groundwater Investigation Levels		
	Fresh Waters[A]	Marine Waters[A]	Drinking Water[B]
	(µg/L)	(µg/L)	(mg/L)
Malathion	0.05	-	0.07
Mancozeb (as ETU, ethylene thiourea)	-	-	0.009
MCPA	-	-	0.04
Metaldehyde	-	-	0.02
Metham (as methylisothiocyanate, MITC)	-	-	0.001
Methidathion	-	-	0.006
Methiocarb	-	-	0.007
Methomyl	3.5		0.02
Methyl bromide	-	-	0.001
Metiram (as ETU, ethylene thiourea)	-	-	0.009
Metolachlor/s–Metolachlor	-	-	0.30
Metribuzin	-	-	0.07
Metsulfuron-methyl	-	-	0.04
Mevinphos	-	-	0.006
Molinate	3.4	-	0.004
Napropamide	-	-	0.4
Nicarbazin	-	-	1
Norflurazon	-	-	0.05
Omethoate	-	-	0.001
Oryzalin	-	-	0.4
Oxamyl	-	-	0.007
Paraquat	-	-	0.02
Parathion	0.004[C]	-	0.02
Parathion methyl	-	-	0.0007
Pebulate	-	-	0.03
Pendimethalin	-	-	0.4
Pentachlorophenol	-	-	0.01
Permethrin	-	-	0.2
Picloram	-	-	0.30
Piperonyl butoxide	-	-	0.6
Pirimicarb	-	-	0.007
Pirimiphos methyl	-	-	0.09
Polihexanide	-	-	0.7
Profenofos	-	-	0.0003
Propachlor	-	-	0.07

Substance	Groundwater Investigation Levels		
	Fresh Waters[A]	Marine Waters[A]	Drinking Water[B]
	(µg/L)	(µg/L)	(mg/L)
Propanil	-	-	0.7
Propargite	-	-	0.007
Proparzine	-	-	0.05
Propiconazole	-	-	0.1
Propyzamide	-	-	0.07
Pyrasulfatole	-	-	0.04
Pyrazophos	-	-	0.02
Pyroxsulam	-	-	4
Quintozene	-	-	0.03
Simazine	3.2	-	0.02
Spirotetramat	-	-	0.2
Sulprofos	-	-	0.01
2,4,5-T	36	-	0.1
Tebuthiuron	2.2	-	-
Temephos	-	0.05[D]	0.4
Terbacil	-	-	0.2
Terbufos	-	-	0.0009
Terbuthylazine	-	-	0.01
Terbutryn	-	-	0.4
Thiobencarb	2.8	-	0.04
Thiometon	-	-	0.004
Thiram	0.01	-	0.007
Toltrazuril	-	-	0.004
Toxafene	0.1 [D]	-	-
Triadimefon	-	-	0.09
Trichlorfon	-	-	0.007
Triclopyr	-	-	0.02
Trifluralin	2.6[D]	-	0.09
Vernolate	-	-	0.04
Surfactants			
Linear alkylbenzene sulfonates (LAS)	280	-	-
Alcohol ethoxylated sulfate (AES)	650	-	-
Alcohol ethoxylated surfactants (AE)	140	-	-

A Investigation levels apply to typical slightly-moderately disturbed systems. See ANZECC &

Substance	Groundwater Investigation Levels		
	Fresh Waters[A]	Marine Waters[A]	Drinking Water[B]
	(µg/L)	(µg/L)	(mg/L)

ARMCANZ (2000) for guidance on applying these levels to different ecosystem conditions.

B Investigation levels are taken from the health values of the Australian Drinking Water Guidelines (NHMRC 2011).

C Figure may not protect key species from chronic toxicity, refer to ANZECC & ARMCANZ (2000) for further guidance.

D Chemical for which possible bioaccumulation and secondary poisoning effects should be considered, refer to ANZECC & ARMCANZ (2000) for further guidance.

E For changes in GIL with pH refer to ANZECC & ARMCANZ (2000) for further guidance.

H Values have been calculated using a hardness of 30 mg/L $CaCO_3$ refer to ANZECC & ARMCANZ (2000) for further guidance on recalculating for site-specific hardness.

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8 Glossary

Added contaminant limit (ACL) is the added concentration of a contaminant above which further appropriate investigation and evaluation of the impact on ecological values will be required. ACL values are generated in the process of deriving ecological investigation levels (EILs).

Ambient background concentration (ABC) of a contaminant is the soil concentration in a specified locality that is the sum of the naturally occurring background and the contaminant levels that have been introduced from diffuse or non-point sources by general anthropogenic activity not attributable to industrial, commercial or agricultural activities.

An **area of ecological significance** is one where the planning provisions or land use designation is for the primary intention of conserving and protecting the natural environment. This would include national parks, state parks, and wilderness areas and designated conservation areas.

Asbestos fines (AF) includes free fibres of asbestos, small fibre bundles and fragments of bonded ACM that pass a 7 mm x 7 mm sieve.

Bioavailability is a generic term defined as the fraction of a contaminant that is absorbed into the body following dermal contact, ingestion or inhalation.

Bonded asbestos-cement-material (bonded ACM) comprises bonded asbestos containing material which is in sound condition (although possibly broken or fragmented), and is restricted to material that cannot pass a 7 mm x 7 mm sieve. This sieve size is selected as it approximates the thickness of common asbestos cement sheeting and for fragments to be smaller than this would imply a high degree of damage and potential for fibre release.

Conceptual site model (CSM) is a description of a site including the environmental setting, geological, hydrogeological and soil characteristics together with the nature and distribution of contaminants. Potentially exposed populations and exposure pathways are identified. Presentation is usually graphical or tabular with accompanying explanatory text.

Contamination means the condition of land or water where any chemical substance or waste has been added as a direct or indirect result of human activity at above background level and represents, or potentially represents, an adverse health or environmental impact.

Ecological investigation levels (EILs) are the concentrations of contaminants above which further appropriate investigation and evaluation will be required. EILs depend on specific soil physicochemical properties and land use scenarios and generally apply to the top 2 m of soil. EILs may also be referred to as soil quality guidelines in Schedules B5b and B5c.

Ecological screening levels (ESLs) for petroleum hydrocarbons are the concentrations above which further appropriate investigation and evaluation will be required. ESLs broadly apply to coarse- and fine-grained soils and various land uses. They are generally applicable to the top 2 m of soil.

Environmental value is a value or use of the environment which is conducive to public benefit, welfare, safety or health and which requires protection from the effects of pollution, waste discharge and deposits.

Exposure scenario is a set of conditions or assumptions about sources, exposure pathways, concentration of contaminants involved and an exposed population (that is, numbers, characteristics, habits) used in the evaluation and quantification of exposure(s) in a given situation.

Fibrous asbestos (FA) includes loose fibrous material such as insulation products, severely weathered cement-bonded asbestos sheeting and damaged low density board (up to 70% asbestos in calcium silicate). For the purposes of site assessment, FA includes any asbestos–containing-material (ACM) that is easily powdered or made pasty with clear separation of asbestos fibres by moderate hand pressure.

Groundwater investigation level (GIL) is the concentration of a groundwater parameter at which further investigation (point of extraction) or a response (point of use) is required. Includes Australian water quality guidelines, drinking water guidelines, guidelines for managing risk in recreational water criteria and site-specific derived criteria.

Health investigation levels (HILs) are the concentrations of a contaminant above which further appropriate investigation and evaluation will be required. HILs are generic to all soil types and generally apply to the top 3 m of soil.

Health risk assessment (HRA) is the process of estimating the potential impact of a chemical, biological or physical agent on a specified human population system under a specific set of conditions.

Health screening levels (HSLs) for petroleum hydrocarbons are the concentrations above which further appropriate investigation and evaluation will be required. HSLs depend on physicochemical properties of soil, as these affect hydrocarbon vapour movement in soil, and the characteristics of building structures. HSLs apply to different soil types, land uses and depths below surface to >4 m and have a range of limitations.

Investigation levels and **screening levels** are the concentrations of a contaminant above which further appropriate investigation and evaluation will be required. Investigation and screening levels provide the basis of Tier 1 risk assessment.

Petroleum hydrocarbon '**management limits'** are limited to petroleum hydrocarbon compounds. They are maximum values that should remain in a site following evaluation of human health and ecological risks and risks to groundwater resources and apply to all soil depths based on site-specific considerations. These limits are to consider the formation of light non aqueous phase liquids, fire and explosion risks and damage to buried infrastructure.

Multiple-lines-of-evidence approach is the process for evaluating and integrating information from different sources of data and uses best professional judgement to assess the consistency and plausibility of the conclusions which can be drawn.

Risk means the probability in a certain timeframe that an adverse outcome will occur in a person, a group of people, plants, animals and/or the ecology of a specified area that is exposed to a particular dose or concentration of a chemical substance, that is, it depends on both the level of toxicity of the chemical substance and the level of exposure to it.

Risk assessment is the process of estimating the potential impact of a chemical, physical, microbiological or psychosocial hazard on a specified human population or ecological system under a specific set of conditions and for a certain timeframe.

Risk management is a decision-making process involving consideration of political, social, economic and technical factors with relevant risk assessment information relating to a hazard to determine an appropriate course of action.

Screening is the process of comparison of site data to screening criteria to obtain a rapid assessment of contaminants of potential concern.

A **Tier 1 assessment** is a risk-based analysis comparing site data with investigation and screening levels for various land uses to determine the need for further assessment or development of an appropriate management strategy.

9 Shortened forms

ABC	ambient background concentration
ACL	added contaminant limit
ACM	asbestos-containing-material
ADI	acceptable daily intake
ADWG	*Australian Drinking Water Guidelines*
AF	asbestos fines
AM	arithmetic mean
AS	Australian Standard
As	Arsenic
AWQG	*Australian Water Quality Guidelines 2000*
B(a)P	benzo(a)pyrene
Bonded ACM	bonded asbestos-containing-material
BTEX	benzene, toluene, ethylbenzene and xylenes
CCME	Canadian Council of the Ministers of the Environment
CEC	cation exchange capacity
CRC CARE	Cooperative Research Centre for Contamination Assessment and Remediation of the Environment
CSIRO	Commonwealth Scientific and Industrial Research Organisation
Cr III	Chromium
CSM	Conceptual Site Model
Cu	Copper
CWS PHC	*Canada Wide Standard for Petroleum Hydrocarbons (PHCs) in Soil*
DDT	dichlorodiphenyltrichloroethane
DQO	data quality objective
DSI	detailed site investigation
EC_{30}	effective concentration 30%
EIL	ecological investigation level
ESL	ecological screening level
FA	fibrous asbestos
GC-MS	gas chromatography-mass spectrometry
GIL	groundwater investigation level
GM	geometric mean
GMRRW	*Guidelines for Managing Risk in Recreational Water*
HIL	health investigation level
HSL	health screening level

IEUBK	integrated exposure uptake biokinetic model (for lead)
ISQG	*Interim Sediment Quality Guideline*
LNAPL	light non-aqueous phase liquid
LOEC	lowest observed effect concentration
MTBE	Methyl tert-butyl ether
N/A	not applicable
NATA	National Association of Testing Authorities
Ni	Nickel
NL	not limiting
OCP	organochlorine pesticide
PAH	polycyclic aromatic hydrocarbon
Pb	Lead
PCB	polychlorinated biphenyl
PCE	perchloroethene
PSI	preliminary site investigation
RfD	reference dose
SAQP	Sampling and Analysis Quality Plan
SD	standard deviation
SSD	species sensitivity distribution
TCE	tetrachlorethene
TDI	tolerable daily intake
TDS	total dissolved solids
TEF	toxicity equivalence factor
TEQ	toxicity equivalent quotient
TPH	total petroleum hydrocarbons
TRH	total recoverable hydrocarbons
UCL	upper confidence limit
US EPA	United States Environmental Protection Agency
VOCC	volatile organic chlorinated compound
WA DoH	Western Australian Department of Health
WHO	World Health Organization
WHS	work health and safety
Zn	Zinc

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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This compilation has been split into 22 volumes

Volume 1: sections 1-6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of

Schedule B2

GUIDELINE ON

Site Characterisation

Explanatory Note

The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

It aims to ensure consistency in characterisation of potentially contaminated soils, groundwater, vapour and soil gases in order to inform appropriate human health and ecological risk assessment. It should be read in conjunction with other Schedules to the Measure.

The original Schedule B2 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document.

The National Environment Protection Council (NEPC) acknowledges the contribution of a number of individuals and organisations towards the development of these guidelines; in particular, the WA Department of Environment and Conservation, CRC CARE, CSIRO Land and Water, WA Department of Health, and individual officers of the NSW Office of the Environment and Heritage, the QLD Department of Environment and Heritage Protection, EPA Victoria, and the Commonwealth Department of Health and Ageing.

Contents

Site characterisation

1 Introduction

Adequate site characterisation is the foundation for appropriate assessment of health and environmental risks associated with site contamination. This guideline provides information on the design and implementation of soil, groundwater and vapour sampling programs and the presentation of site assessment reports. Guidance is also provided on the minimum measures that should be adopted to ensure protection of the environment during site assessment. Site-specific management measures must ensure compliance with environmental management and protection legislation applying in each jurisdiction.

Risk of explosion or other acute exposure hazards should be addressed immediately and is not within the scope of this guidance document.

The investigation components of an assessment of site contamination are:

- establishing the objectives of the site assessment
- desktop study and detailed site inspection
- compiling a site history from relevant site-related information
- development of a conceptual site model (CSM)
- identification of data gaps
- development of data quality objectives (DQOs)
- design of a sampling strategy and optimisation of a sampling and analysis quality plan (SAQP)
- data collection (delineation of potential and known contamination)
- data validation, analysis and interpretation (including risk assessment and iterative development of the CSM)
- coherent presentation and reporting.

The characterisation of site contamination should only be conducted by professional environmental practitioners who are suitably qualified and experienced in the assessment of contaminated sites. For further information on suitable qualifications and experience, refer to Schedule B9.

2 Stages of investigation

Source: Davis et al. (2006) and Clements et al. (2009)

Schedule A of the National Environment Protection (Assessment of Site Contamination) Measure 1999 (NEPM) shows the staged site assessment process and indicates that this guideline applies to both preliminary and detailed site investigations.

Many site investigations proceed in multiple stages due to the complexity of site conditions and of contaminant properties and/or the discovery of unexpected contamination. Poorly planned and executed site investigations are likely to result in time delays and additional costs (both during the investigation and any subsequent remediation) and inadequate or misleading data which may result in risks to human health and/or the environment not being addressed.

Site investigation efforts should be purpose driven, adequate in scope and of sufficient quality to meet the purpose of the assessment. They should provide representative site data. In order to achieve these objectives, the recommended procedures are to clarify the purpose of the investigation, develop a CSM, develop DQOs and identify significant data gaps. An SAQP can then be designed and implemented to achieve the desired objective(s).

Depending on the proposed land use and the results of initial site history investigations, the preliminary and detailed investigations may be incorporated into a single phase of investigation. Proponents and site assessors may also wish to adopt an accelerated site characterisation approach whereby rapid and 'real-time' sampling and field analytical methods, and on-site interpretation and iteration of field data, are undertaken in order to expedite the characterisation process. Further information on accelerated site characterisation methods can be found in Clements et al. (2009), and at www.triadcentral.org/tech, as well as on the Environment Canada website at www.on.ec.gc.ca/pollution/ecnpd/contaminassist_e.html .

The CLU-IN website at www.clu-in.org/characterization, produced by the Technology Innovation and Field Services Division of the US EPA, contains a wide range of current information on site characterisation and monitoring techniques for gas/air, soil, sediment and water. The information includes performance specifications, advantages and limitations and indicative costs.
Regardless of the approach taken, the site investigation must cover all the components identified in Section 1, which enable an appropriate level of risk assessment for human health and the environment to be undertaken.

2.1 Preliminary site investigation

Preliminary site investigations (PSIs) usually include a desktop study to collect basic site information and identify the site characteristics (site location, land use, site layout, building construction, geological and hydrogeological setting, historical land uses and activities at the site), a site inspection and interviews with current and past owners, operators and occupiers of the site and preparation of a report.

The preliminary investigation should be sufficient to:

- identify potential sources of contamination and determine potential contaminants of concern
- identify areas of potential contamination
- identify potential human and ecological receptors
- identify potentially affected media (soil, sediment, groundwater, surface water, indoor and ambient air).

The findings of the PSI are used to develop an initial CSM (refer Section 4). The PSI report should clearly identify any significant data gaps and include an assessment of the accuracy of the information collected.

It is not necessary to delineate any contamination at the PSI stage. Limited sampling may be included in a PSI, providing sufficient information is available to compile an appropriate site health and safety plan. Any investigations undertaken, however, are usually confined to areas where potentially contaminating activities have occurred and involve a site history-based sampling plan.

This Schedule provides guidance on the scope of preliminary investigations. Reference may also be made to AS 4482 and more generally to ASTM E1527–05 for information on the various elements which may be included in a preliminary site investigation.

If thorough preliminary investigation shows a history of non-contaminating activities and there is no other evidence or suspicion of contamination, further investigation is not required.

2.2 Detailed site investigation

A detailed site investigation (DSI) is required when the results of the preliminary investigation indicate that contamination is present or is likely to be present and the information available is insufficient to enable site management strategies to be devised. Potential or actual contamination will usually require further delineation. Potential contamination may have been indicated by the presence of underground structures (for example, underground fuel or chemical storage tanks), the presence of fill (for example, ash, odorous material or various types of waste) or staining of soil. Actual contamination may have been detected in the form of contaminants that are not naturally occurring or as elements or compounds that are above background levels or exceed the investigation or screening levels (see Schedule B1 for more information).

The detailed investigation stage should identify the nature of the contamination and delineate its lateral and vertical extent to a sufficient degree that an appropriate level of risk assessment may be undertaken and, if necessary, provide the basis for the development of an appropriate remediation or management strategy.

3 Preliminary investigations

The purpose of collecting basic site information is to identify potential contaminants, potentially affected media and potential areas of contamination by reviewing the site history, physical setting including local geology and hydrogeology, and site conditions. The information collected is used to develop an initial CSM (refer Section 4) of the site.

A site inspection should be undertaken to complement the findings of the desktop study and site history and to identify any additional relevant site information. It is recommended practice to conduct interviews with current site owners and occupiers and, where practicable, previous site owners and occupiers.

It is essential that the location of the site and the significant features involved in its history be accurately and clearly identified. The PSI report should clearly identify any significant data gaps and include an assessment of the accuracy of the information collected.

3.1 Site identification

The current legal description (real property description, for example, lot number X on plan XX) of all affected parcels and the street number and name and suburb should be obtained, together with a copy of the current certificate of title. It is also useful to list any common name or description by which the site is or has been known.

Where multiple lots are involved, plans that show lot boundaries in relation to significant features should be obtained. Maps (including street maps), plans or diagrams should be used to clearly identify the location of all affected land parcels in relation to their surrounds, for example, street access, neighbouring property boundaries, parks, local watercourses and any areas of environmental significance.

3.2 Current and proposed use

The following details should be obtained:

- current uses of the site
- map and narrative description of proposed use(s) for the site
- current land zoning of the site, for example, industrial, mixed commercial, residential, educational
- type of proposed use—in the context of the categories detailed in Schedule B1
- density of residential use (if proposed)
- type of users, e.g. residents (adults and children), workers, ecological
- local government approval(s) for proposed use (and date).

3.3 Site history

Source: Edwards et al. (1994) & NSW EPA (2011)

A site history should contain, as far as practicable, all available information that assists in identifying the potential nature and extent of site contamination. It may also be useful for identifying features (for example, current and disused utilities) that may act as potential preferential contaminant migration pathways. It may include the use of video or photographic logs to assist with site documentation.

Sources of information for compiling a site history include but are not limited to:

- past and current owners and occupiers, operators or workers at the site and adjacent properties

- local knowledge of residents
- current and historical aerial and ground photographs
- past involvement with government authorities or consultants (environmental audits, notices etc.)
- trade and street directories
- historical societies and local, state or territory government libraries
- historical titles back to original deeds
- local literature, including newspapers
- technical literature, including plumbing and building permits/plans, flammable and combustible liquid storage and handling licences
- complaint history and information from environmental licences and trade waste permits held by local government or state government departments
- geological survey maps and reports
- groundwater/drinking water protection zones
- groundwater abstraction licences
- local government development approval records, sewer and underground service plans
- site layout plans.

To compile a site history, the assessor should consider the issues described below in Sections 3.3.1 to 3.3.19.

3.3.1 Site plan and historical maps and aerial photographs

It is essential to have a locality map and a current plan of the site, with scale bar, indicating the site orientation (including north) and general topography of the property, local water drainage and other environmentally significant features. A review of the site history with dates as deduced from current and historic aerial photographs and other historical information should be included (where available). In addition to historical aerial photographs, other historical maps and plans are at times available and can be of great value (for example, government department maps and plans, local council records, street directories, topographic maps, geological maps, mining plans, and records of the mining department (where appropriate) etc).

3.3.2 Land Use Zoning

Necessary records include previous, present and proposed zoning, and relevant development and building approval records.

3.3.3 Present owners, occupiers and current users of the site

If these are not the parties responsible for the assessment and management of the site then those who are (or are thought to be) responsible should also be identified if possible.

3.3.4 Previous owners and occupiers of the site

These should be listed chronologically, noting any periods during which ownership or tenancy is unknown or uncertain.

3.3.5 Previous activities/uses

A chronological list of land uses should be compiled, focusing on industrial uses or other potentially contaminating activities, and including any periods during which the land use is unknown or uncertain. While ‘small tannery’ may be seen as an imprecise description, it nonetheless provides some

information about the nature, severity and distribution of any potential contamination. Precise industrial capacities of properties should be cited if available. The chronology should include dates when areas of the site were sealed, for example, by concrete slabs, in relation to the occurrence of potentially contaminating activities to prevent unnecessary under-slab sampling, although the potential for the migration of contamination underneath hardstands from adjacent sources will need to be taken into account. Consideration should also be given to uses on adjacent sites that could be a source or receptor of contamination.

3.3.6 Services to the property (including sewer and underground services)

Site plans showing the location, elevation and size of sewers, stormwater drains and underground utilities (such as communications infrastructure) should be included, as these may assist in identification of preferential contamination migration pathways.

3.3.7 Previous and present building and structures

These are generally best illustrated by a series of annotated site maps showing the locations of permanent and semi-permanent structures, offices, sheds, reaction vessels, storage tanks, etc. These should be presented in chronological order to show how the site developed. Key building design features such as the nature of foundations, presence or absence of crawl spaces or basements should also be included. The age and nature of buildings and infrastructure should be considered in relation to potential occurrence and distribution of asbestos-containing-materials. Where infrastructure has been decommissioned, the site history should note whether any potentially contaminating contents are known to have been removed (for example, whether tanks and pipelines were drained or simply blocked off).

3.3.8 Industrial processes carried out on site and the products manufactured

A list should detail the products from the industries and activities identified as being relevant to the site.

3.3.9 Chemical storage and transfer areas

Locations should be indicated on the scaled site plan and chemicals stored and transferred at each area identified.

3.3.10 Raw materials used

A list of raw materials stored or used at the site should be compiled. Chemicals should be identified by systematic names as well as common or trade names.

3.3.11 Intermediate products

These are important in both batch and continuous production processes. Residual reaction components and intermediate products may have been discharged from reaction vessels prior to production runs. Quality assurance procedures may also have included sampling points from intermediate stages in the manufacturing process which may have been allowed to drain away or be otherwise discarded on site.

3.3.12 Product spills, losses, incidents and accidents (including fire)

These should be listed chronologically, together with an indication of the material spilled, estimates of quantity, extent of fire damage and structures affected.

3.3.13 Discharges to land and water

The types of waste currently and historically discharged should be identified. Where practicable, the quantities should also be established.

3.3.14 Wastes produced

This requires an understanding of the processes being performed in the industries and activities identified above. Wastes may be identified specifically (for example, waste degreasing solvents including carbon tetrachloride) or more generally (for example, acid slurry).

3.3.15 Power generation

Many historical activities required steam as part of the process or for power generation. Before the advent of electric power, generation of steam could have progressed from solid to liquid fuels requiring fuel storage and disposal of ash. This may have resulted in contamination by fuel and combustion products, for example, polycyclic aromatic hydrocarbons (PAHs). If the power requirement was large, a sub-station with a transformer(s) may have been on site with the attendant risk of polychlorinated biphenyls (PCBs) spills. In addition, fibrous asbestos may have been used for insulation purposes.

3.3.16 Waste disposal locations and imported fill

Locations of solid waste and liquid waste disposal areas and liquid waste lagoons, settling tanks, sumps and soak wells should be identified in the maps and figures described above. The location of any wells on site should be indicated as these may have been used historically for liquid waste disposal.

Historically, many industrial wastes and diverse contaminated fill were considered a low-cost source of material to level or elevate sites. Wastes may have originated from on-site industrial activities or have been introduced from unknown off-site sources. Residential and industrial/commercial areas around major industries (for example, coal gas works, power stations, and mineral processing plants) may have been filled with ash, coke, hydrocarbon impacted fill, metal waste and various wastes originating from the industrial activity.

Sites should be assessed for areas of fill, particularly if there are reasonable grounds to suspect the original land form has been altered such as by filling gullies and watercourses.

3.3.17 Earthmoving activities carried out on the site

This information will assist in determining the source and location of any imported fill. Consideration should also be given to the possibility that earthmoving activity may have resulted in redistribution and burial of contamination.

3.3.18 Interview information

Interviews with past property or business owners and occupiers and employees should be conducted where practicable. The objective of interviews is to confirm information collected in the desktop study and to gain additional relevant site information (for example, source of drinking water, presence of wells on-site, date of connection to sewer, history of spills and leaks, arrangements for liquid and solid waste disposal etc.). Owners and occupants of neighbouring properties may also be able to provide useful information.

3.3.19 Sources of information

A log of all sources consulted for site history information should be kept so that the completeness and reliability of the information collected, and hence confidence in the desktop study results, may be assured. Personal recollections and anecdotal records should be cross-checked where possible and any limitations of the information noted. This information should be clearly documented in the PSI report.

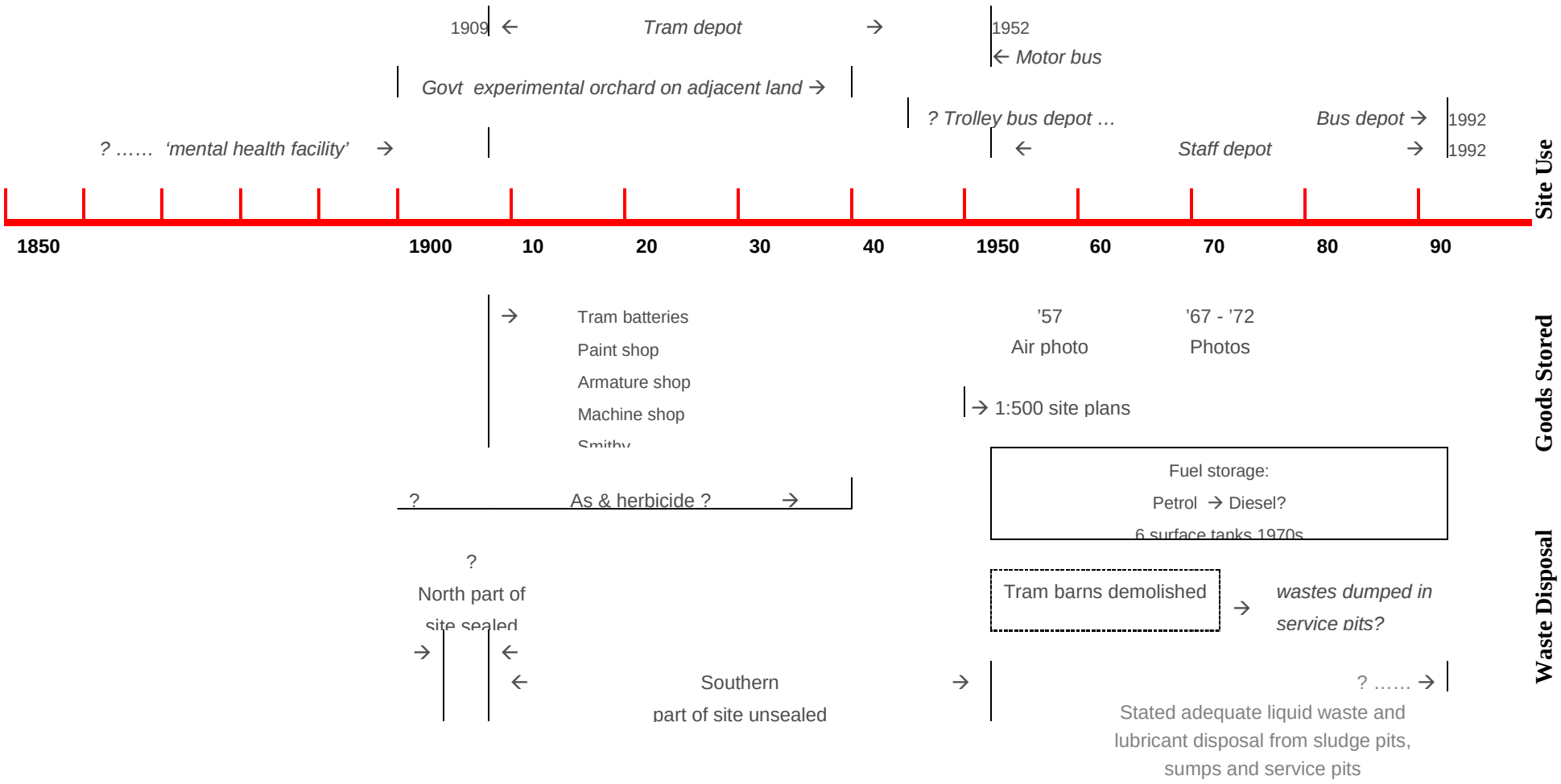
Table 1. An example of a site chronology table where the gaps in the data and inadequacies of information are readily identified

Date	Owner	Occupant	Industry or land use	Process equipment plant	Chemicals inputs by-products waste	Buildings, structures and services	Soil cover vegetation paved areas	Fill and excavation	Comments
1993 (to Mar)	PD Nominees	PD Nominees	Springwater bottling	Confidential					
1986 (from Sept)	PD Nominees	PD Nominees	Vinegar bottling		Acetic Acid	20 x 30 m Warehouse built Nov 1986	Site completely covered by a concrete slab		Soil logs available from the warehouse construction
1979 (11 Jun)	PD Nominees	R McLaren	Motor vehicle repair and car park		Oils solvents lubricants	No buildings on site unfenced	Half of site covered by 150 mm of coarse gravel	Coarse gravel …	Surface oil waste contamination
1979 (10 Jun) 1978 (5 Nov)	F Bath	F Bath	Electrical workshop		Solders, capacitors, mercury switches	Workshop destroyed in fire			Burning building associated with colourful flames
1979 (5 Nov) 1972	R Bath								
1972 1965	R Bath	R Bath and Sons	Process control and electrical motor maintenance	Burnt coatings off copper wire for scrap copper sales					(some complaints under the Clean Air Act)
1965 1958 (Sept)	R Bath and D Fergusson	R Bath and D Fergusson	Electrical motor rewinders			Tannery building converted to workshop		Tannery pits filled	

Date	Owner	Occupant	Industry or land use	Process equipment plant	Chemicals inputs by-products waste	Buildings, structures and services	Soil cover vegetation paved areas	Fill and excavation	Comments
1958 (Sept) 1958 (Feb)	D Muldoon Land being subdivided	Unoccupied due to closure of tannery				(property still fenced), drying shed removed			Cadastral survey records show ground level at 0.35 metres lower than in the 1979 survey

Source: van Alphen (1993)

Figure 1. An example of the representation of site history information on a time line, to enable a check of the completeness of available information. *This graphic illustrates 5 pages of site history text.*



Adapted from van Alphen (1993)

3.4 Environmental setting

An understanding of the environmental setting of the site is necessary for developing the CSM (refer section 4).

In general, the search radius should take into account the distance that contaminants could migrate to or from the site. A search radius of 500 m from the boundary of the site is suggested as a general guide for identification of potential ecological receptors such as surface water bodies, wetlands and areas of ecological significance.

If the site is located in low-lying land, consideration should also be given to whether the site is likely to be located in/affected by acid sulfate soils. Where there is the potential for acid sulfate soils to be present, this should be taken into account when preparing the sampling and analysis quality plan as appropriate procedures are required. WA DEC (2009) provides detailed information on the identification and sampling of acid sulfate soils.

3.5 Local geology and hydrogeology

The local and site-specific geological and hydrogeological settings influence the fate and transport of potential contaminants in the vicinity of and at the subject site.

The distribution of contaminants across a site is influenced by the local geology and natural or man-made/altered drainage features in the area or at the site. Their distribution within the sub-surface is influenced by geological structures, variations in the permeability of soil and rock (which may result in perched water tables), geochemical, biological and mineralogical variations and the presence of preferential pathways such as loose fill around services.

Certain sites may be located in areas that are naturally enriched with mineral resources and can appear to contain elevated levels of metals and metalloids in soil, surface water or groundwater. Consequently, it is essential to have an understanding of the background quality of these media and to evaluate potential contamination of this type of site in terms of the beneficial uses of the site and its water resources.

The geological/hydrogeological component of the desktop investigation may include review of the following types of published data:

- surface elevation
- regional and site-specific soil and geological records
- geophysical data
- drilling logs which clearly identify imported and locally derived fill (including refuse) and natural strata
- well logs including strata, casing or construction details, and water level, quality and pump/discharge rate information
- aquifer types (unconfined, semi-confined, confined) and aquitards/aquicludes present
- direction and rate of groundwater flow
- regional and site-specific hydrogeological information, including groundwater quality
- current usage/resource potential
- existing monitoring wells and records of registered production wells or survey of surrounding landholders to determine the existence of wells where the resource may potentially be used in the vicinity of the site.

For more comprehensive assessments, for example where groundwater fate and transport modelling is to be undertaken, desktop studies may also consider:

- values for soil bulk density and porosity
- aquifer storativity or storage
- soil organic matter content
- cation exchange capacity (CEC)
- soil pH and redox (Eh) potential measured in situ
- hydraulic and piezometric heads and hydraulic gradients
- hydraulic conductivity
- transmissivity
- other parameters as appropriate.

Appendix III of the *Guidelines for groundwater protection in Australia* (ARMCANZ & ANZECC 1995) gives helpful advice on hydrogeological desktop studies.

These data form the basis of an initial appraisal of the potential risk to a receptor. When the likelihood of an unacceptable groundwater impact is identified, Schedule B6 should be consulted.

3.6 Site inspection

A site inspection should be conducted by a professional who is suitably qualified and experienced in the assessment of contaminated sites. For further information on suitable qualifications and experience, refer to Schedule B9.

A comprehensive site inspection is a critical stage of the site assessment process. It validates anecdotal and historical information and can identify additional evidence of potential contamination.

The complexity and detail reported in a site inspection may vary depending on the level of historical information and anecdotal information relevant to the site and the complexity and detail of the site itself. The following features, among others, should be noted:

- current uses of the site and surrounding land
- disturbed, coloured or stained soil
- bare soil patches
- disturbed or distressed vegetation
- unusual odour
- quality of surface water
- sheens on water surfaces
- site topography and surface water drainage
- presence and type of groundwater bores on the site and adjacent landholdings
- condition of groundwater bore headworks
- measurement of groundwater (water table and/or piezometric) levels
- condition of buildings, concrete and bitumen floors and roads, etc.
- building construction (slab-on-ground or other, presence or absence of crawl spaces and basements)

- the means of heating (fuel type) and cooling buildings on the site
- presence or absence of bonded asbestos-containing materials (bonded ACM) on the ground surface
- presence of stockpiles, fill, containment areas, sumps, drains and waste disposal areas – operational and closed
- evidence of cut and fill activities
- presence of pits, ponds and lagoons
- presence and condition of chemical containers, holding tanks, bunds, etc.
- presence and condition of any underground storage tanks (USTs) and associated infrastructure
- underground structures that may be associated with sub-surface contamination
- condition of materials storage and handling facilities and any solid or liquid waste disposal areas
- any evidence of on-site spillage of dangerous goods and/or off-site migration.

For operating sites, an inventory of chemicals stored or used at the site and copies of Material Safety Data Sheets (MSDSs), dangerous good licences, operating licences, works approvals and notices, and results of environmental audits (e.g. audits conducted under ISO 14000) should be obtained where practicable.

4 Conceptual site models

4.1 Overview

A conceptual site model (CSM) is a representation of site-related information regarding contamination sources, receptors and exposure pathways between those sources and receptors. The development of a CSM is an essential part of all site assessments and provides the framework for identifying how the site became contaminated and how potential receptors may be exposed to contamination either in the present or the future.

Typically, the CSM should be presented in written format and illustrated with suitable graphics and flow diagrams. Example graphics can be found in Clements et al. (2009) and Davis et al. (2009a). An example CSM in the form of a flow diagram can be found in Schedule B4 (Figure 2).

The CSM can be a useful tool for informing discussions with stakeholders regarding the investigation and management of potential and known contamination impacts.

The complexity of the CSM should correspond to the scale and complexity of the known or potential contamination impacts.

4.2 Iterative development of conceptual site models

Source: Clements et al. (2009); SA EPA (2009) and Davis et al. (2009a)

The development of a CSM is a dynamic process and it is important that all the information and data from each stage of an assessment are reviewed in an integrated manner (using a multiple-lines-of-evidence approach where appropriate) to refine the CSM and used to inform subsequent decisions on whether further investigation or management is necessary. Note changes to the CSM may also involve revision of the data quality objectives (DQOs)—see Section 5.

The initial CSM is constructed from the results of the PSI and is used to identify data gaps and inform a decision on whether detailed investigation is required. The CSM should be continually challenged and updated throughout the assessment process.

The sub-optimal performance of many remediation systems can be traced back to the failure to undertake adequate site characterisation and to fully integrate the information gained into the CSM. For large and complex sites, 3-D imaging (visualisation) software may be useful for displaying and interpreting the results of the investigations and to refine the CSM.

4.3 Essential elements of conceptual site models

The CSM should identify complete and potential pathways between the known or potential source(s) and the receptor(s). Where the pathway between a source and a receptor is incomplete, the exposure to chemical substances via that pathway cannot occur but the potential for that pathway to be completed (for example, by abstraction of groundwater or a change in land use) should be considered in the assessment.

The essential elements of an initial CSM are:

- known and potential sources of contamination and contaminants of concern including the mechanism(s) of contamination (e.g. 'top down' spill or sub-surface release from corroded tank or pipe)
- potentially affected media (soil, sediment, groundwater, surface water, indoor and ambient air)
- human and ecological receptors
- potential and complete exposure pathways.

For the assessment of vapours, (refer Section 9.2.3) additional detail will be needed about preferential pathways for vapour migration and the design of buildings or planned buildings at the site — including the location of sub-surface utilities, foundation construction and condition, and ventilation and heating (Davis et al. 2009a).

4.4 Assessing data gaps and uncertainties in conceptual site models

Data gap identification and uncertainty assessment are key activities in developing and refining a CSM during site assessment. It is, therefore, important that the CSM addresses:

- how representative the available data is likely to be
- what the potential sources of variability and uncertainty are
- how important the identified gaps are to the objectives and reliability of the site assessment.

In developing the CSM, the assessor needs to distinguish between variability and uncertainty. Variability arises from true heterogeneity in the environment such as lateral variations in soil properties or lithology or changes in contaminant levels over time and space. Uncertainty represents lack of knowledge about factors, such as contaminant levels (which may be reduced with additional investigation).

The identification of data gaps should be carried out in a logical, structured manner, to facilitate the assessment of uncertainty and the significance of those data gaps to the assessment objectives. Subsequent investigative efforts should be focussed on addressing the critical data gaps in a manner that is proportional to the uncertainties identified and results in data which is representative of the assessment area.

A tool for assessing gaps and uncertainties in CSMs and assessing their level of significance can be found in Clements et al. (2009).

Further information about developing CSMs can be found in:

- API 2005, *Collecting and interpreting soil gas samples from the vadose zone*, API Publication no. 4741, American Petroleum Institute.
- ASTM E1689–95 (2008), *Standard guide for developing conceptual site models for contaminated sites*, ASTM International.
- ASTM E2531–06 (2006), *Standard guide for development of conceptual site models and remediation strategies for light non-aqueous-phase liquids released to the subsurface*, ASTM International.
- Clements et al. 2009, *Characterisation of sites impacted by petroleum hydrocarbons: guideline document*, CRC CARE Technical Report no. 11, CRC for Contamination Assessment and Remediation of the Environment, Adelaide, South Australia.
- Davis et al. 2009a, *Field assessment of vapours*, CRC CARE Technical Report no.13, CRC for Contamination Assessment and Remediation of the Environment, Adelaide, South Australia.
- EA 2000a, *Guide to good practice for the development of conceptual models and the selection and application of mathematical models of contaminant transport processes in the subsurface,* NC/99/38/3, Environment Agency, England and Wales.
- ITRC 2009, *Evaluating natural source zone depletion at sites with LNAPL*, LNAPL-1, LNAPL Team, Interstate Technology & Regulatory Council, Washington, DC.
- ITRC 2007a, *Vapor intrusion pathway: a practical guideline,* VI-1, ITRC Vapor Intrusion Team, Interstate Technology & Regulatory Council, Washington, DC.

- ITRC 2007b, '*Vapor intrusion pathway: investigative approaches for typical scenarios*', *a supplement to Vapor intrusion pathway: a practical guideline,* Technical and regulatory guidance supplement prepared by the ITRC Vapor Intrusion Team, Interstate Technology & Regulatory Council, Washington, DC.
- NJDEP 2005b, *Vapor intrusion guidance*, New Jersey Department of Environmental Protection, (Available online at www.nj.gov/dep/srp/guidance/vaporintrusion/vig.htm).
- ODEQ 2010, *Guidance for assessing and remediating vapor intrusion in buildings*, Report no. 10-LQ-007, Oregon Department of Environmental Quality, Portland, USA.
- SA EPA 2009, *Site contamination: guidelines for the assessment and remediation of groundwater contamination*, Environment Protection Authority, Adelaide, South Australia.

5 Systematic planning for collection of environmental data

5.1 Introduction

It is recommended that a systematic planning process is used for defining the objectives of a site assessment and to develop a sampling plan for the collection and evaluation of representative data to achieve those objectives. Without systematic planning, the site assessment may be ambiguous or inconclusive, which may lead to additional sampling requirements, resulting in increased costs and project delays.

In its simplest form, the planning process should consider:

- the overall objective of the site assessment
- the decision(s) to be made on the basis of the site assessment findings
- the constraints on the assessment (financial, time and logistical) and
- the degree of flexibility to conduct follow-up investigations.

This project level information can then be used to identify the specific site information needed to address the assessment objectives. The next step is to develop a sampling and analysis quality plan (SAQP) to obtain the necessary representative data for the study area.

5.2 Data quality objective process

The US EPA seven-step Data Quality Objective (DQO) process is one example of a suitable systematic planning approach (US EPA 2000a, 2000b and 2006a). The DQO process is recommended when site contamination data is being relied on to make a risk-based decision as part of a detailed site investigation, though a simplified planning process may be appropriate for straightforward screening assessments.

The DQO process is applicable at both the project level (for example, is the site suitable for development?) and at the investigation level. Further information can be found in US EPA (2006a).

At the investigation level, DQOs are qualitative and quantitative statements, developed in the first six steps of the DQO process that define the purpose of the site assessment to be undertaken and the type, quantity and quality of data needed to inform decisions relating to the assessment of site contamination. In the seventh step of the DQO process, the SAQP is developed to generate data to meet the DQOs. The SAQP should document the criteria that a sample design should satisfy, including when, where and how to collect samples or measurements, acceptance (performance) criteria and the samples or measurements that should be collected.

The process includes development of the following:

- a statement of the DQOs
- the SAQP to achieve the DQOs
- procedures to follow if the data does not meet the specified DQOs.

The development of the DQOs should be guided by identification of critical data gaps in the CSM. The objectives for sampling may include:

- determining the nature and extent of contamination
- delineating the lateral and vertical extent of contamination
- developing an understanding of the geology and hydrogeology
- the identification of potential and actual contaminant migration routes
- determining whether relevant investigation and/or screening levels are exceeded

- determining whether further investigation or management is required.

Subsequent objectives may be to determine whether relevant investigation levels are exceeded and whether further action is required (additional investigation or management of some form). As understanding of the site will evolve over time, the iterative development of the CSM may also have implications for the DQOs and the SAQP. Data quality assessment (refer Sections 5.6 and 13.1) is also an important part of this iterative process.

A summary of the DQO process is included in Appendix B. More detailed information can be found in US EPA (2000a, 2000b and 2006a). ODEQ (2010) provides a detailed case study of the DQO process applied to a benzene and TCE spill.

5.3 Sampling and analysis quality plans

A well-developed sampling and analysis quality plan (SAQP) has a critical role in ensuring that the data collected is representative and provides a robust basis for site assessment decisions. In order to meet this objective, an SAQP will generally include the following:

- site investigation objectives and a brief background including appropriate plans and diagrams
- a summary of the CSM
- a review of existing information indicating reliability and usability of any existing data (data gap analysis)
- DQOs including a quality assurance (QA) plan and details of quality control (QC) samples to be collected
- pre-mobilisation tasks (e.g. preparation of a site-specific health and safety plan)
- media to be sampled (soil, sediment, groundwater, vapour, NAPL (non-aqueous-phase liquids), biota, surface water, deposited dusts, indoor air, outdoor air)
- details of analytes and parameters to be monitored
- number, location (coordinates) and depth (elevation AHD) of sampling points
- frequency and pattern of sampling
- sampling methods and procedures
- field screening methods
- analysis methods
- the methods for analysing and interpreting field data obtained (for any dynamic or reactive sampling).

The scope and level of detail contained in the SAQP will vary according to the site-specific circumstances and the stage of the investigation.

Flexibility in the SAQP is advisable so that changes may be made during the course of the investigation in response to identified data gaps such as the specific location of sub-surface utilities (which can act as preferential pathways for volatile organic compounds and other gases or a physical hazard) or evidence of more widespread contamination than expected (for example, widespread distribution of contaminated fill).

Professional experience and judgement will be required to ensure that the SAQP contains adequate coverage (spatial and temporal) of all the relevant media to obtain representative samples capable of satisfying the DQOs. If the sampling pattern and density are adequate, a further increase in the density or frequency of sampling is unlikely to change the site-assessment outcomes.

Approaches and methods for assessing soils, groundwater, and vapours and gases are discussed throughout this Schedule.

5.4 Quality assurance and quality control

5.4.1 Overview

Quality assurance (QA) and quality control (QC) are essential elements of the systematic planning process and should be documented in the SAQP. Field QA and QC procedures are discussed in this section. Laboratory QA and QC procedures are discussed in Schedule B3.

Further information is presented in Appendix C, including a QA and QC checklist.

5.4.2 Field quality assurance procedures

Quality assurance involves all of the planned and systematic actions, procedures, checks and decisions undertaken to ensure the representativeness and integrity of samples collected for analysis, and the accuracy and reliability of the analytical results. In the field QA measures include:

- selection of appropriate sampling and preservation methods, sample containers and sample storage
- decontamination procedures such as cleaning of tools before sampling and between samples
- maintenance of the sample environment to minimise sample contamination and analyte losses
- delivery to the laboratory in good condition and within the timeframes required for the particular analytes.

Section 8 of the *Standard guide to the investigation and sampling of sites with potentially contaminated soil* (AS 4482.1-2005) provides a basis for developing a program of quality assurance. As many sites are small with limited sampling, the rate of blind replicates and split samples should be adjusted to an appropriate level to ensure sufficient quality assurance.

5.4.3 Field quality control procedures

Quality control involves those parts of an investigation which serve to monitor and measure the effectiveness of the QA procedures by comparison with the relevant DQOs. In the field, this may include checking of sampling equipment cleanliness by keeping rinses for analysis, duplicate sampling and inclusion of ‘field blanks’ and ‘field spikes’.

Adequate QA is achieved when QC results demonstrate that agreed objectives such as freedom from contamination, method accuracy and precision can be reliably achieved. Selecting an appropriate level of QC is imperative to ensure that DQOs are met.

Standard AS 4482.1-2005 recommends the use of a variety of QC samples including blind replicate samples and rinsate blanks collected in the field which are sent to the primary laboratory to determine the precision of the field sampling and laboratory analytical program, and split samples (collected in the field) which should be submitted to the laboratory as two individual samples without any indication to the laboratory of their common source.

As a general rule, the level of QC required is that which adequately measures the effects of all possible influences upon sample integrity, accuracy and precision, and which is capable of predicting their variation with a high degree of confidence.

5.4.4 Sample handling, storage and transport

The integrity of all samples must be considered, particularly when dealing with VOCs and SVOCs. Reference should be made to Standards AS 4482.1-2005, and AS 4482.2-1999.

Weathering and biodegradation by soil microorganisms will result in a loss of volatile hydrocarbon components from the surface and near-surface of affected sites. An example situation would be an underground fuel storage site where the tanks have been removed and the excavation has been left exposed for several months. In these circumstances, collecting samples from sub-surface layers (at least 500 mm below the surface of the excavation) may provide a more accurate representation of contamination.

Samples should be placed in appropriate sample containers, preferably prepared by a laboratory, with gas-tight, non-absorptive seals, allowing no headspace, and kept cool, preferably with ice bricks or a refrigerated cooler, until arrival at the laboratory. Arrangements should be made to ensure delivery of chilled samples to the laboratory within the holding time of the specified analysis. Samples must remain preserved and be analysed within the time limitations that apply for the analyte and laboratory method. Additional information on sample integrity and appropriate procedures is available in Standard AS 4482.1-2005.

5.4.5 Chain of custody

Site investigators must complete chain-of-custody documentation which details the following information:

- site identification
- the sampler
- nature of the sample
- collection time and date
- analyses to be performed
- sample preservation method
- departure time from site
- dispatch courier(s).

All parties in the chain (sampler, dispatcher, courier and laboratory) should complete the chain-of-custody documentation so that it gains the status of a valid record of sample transfer to the laboratory. An example of a chain-of-custody form can be found in NSW EPA (1994).

The assessment report should include a copy of the receiving laboratory's advice with respect to:

- the condition in which the samples and chain-of-custody documentation were received and the container type
- cross-checking information on sample identification numbers and paperwork received
- confirmation of preservation method.

5.5 Choice of analytes

Analyte choice should be informed by the site history findings and data gaps identified in the development of the CSM and the DQO process. Depending on the available history, potentially contaminated fill may require a more extensive suite of analytes. The appearance and odour of soil and groundwater samples may influence the selection of analytes.

Appendix A provides a list of possible analytes by contaminant grouping. Specific information on the assessment of asbestos and dioxins can be found later in this Schedule.

Additional information on the selection of possible analytes is available in the *Standard guide to the investigation and sampling of sites with potentially contaminated soil* (AS 4482.1-2005), WA DoE (2004), and Turczynowicz (1991).

5.6 Data quality assessment

Checking the validity and usability of the data collected assists with ensuring that only representative and reliable data meeting the specified requirements is considered in the assessment. Activities include verification of sampling procedures, data verification and validation and determination of data usability. The principal assessment measures (also known as Data Quality Indicators or DQIs) are precision, accuracy or bias, representativeness, completeness and comparability.

Further information is provided in Section 13.1, Appendix C and US EPA (2006a).

6 Sampling Design

6.1 Introduction

The site assessor should exercise professional judgement to select and develop an appropriate sampling design, based on accurate and reliable site-specific information (as integrated in the CSM) as far as practicable to obtain sufficient representative data to address the DQOs. For example, if the objective is to establish whether a site is contaminated, a limited number of samples located in those areas most likely to be contaminated may be sufficient, however, a greater number of samples and effort would be required to delineate known contamination. An explanation of, and justification for, the sampling design selected should be provided in the assessment report.

6.2 Categories of sampling designs

Source: US EPA (2002)

There are two main categories of sampling design: judgemental and probability-based sampling programs. The advantages and disadvantages of judgemental and probability-based sampling are listed in Table 2.

Table 2. Advantages and disadvantages of probability-based and judgemental sampling

	Probability-based	Judgemental
Advantages	• Designs are unbiased • Provides ability to calculate uncertainty associated with estimates • Provides reproducible results within uncertainty limits • Provides ability to make statistical inferences • Can handle decision error criteria	• Can be less expensive than probabilistic designs • Can be very efficient with a reliable and full site history • Easy to implement
Disadvantages	• Random locations may be difficult to locate and implement on the ground • An optimal design depends on an accurate CSM	• Depends on expert knowledge • Cannot reliably evaluate precision of estimates • Depends on subjective judgement to interpret data relative to study objectives • Designs are biased

Judgemental sampling designs involve selection of sampling locations based on expert knowledge or professional judgement. The value of judgemental sampling depends on the DQOs, the study size and scope, and the degree of professional judgement available to locate and interpret the data. When judgemental sampling is used in isolation, quantitative statements about the level of confidence in the results cannot be made.

Probability-based designs (such as random, systematic, grid, stratified, transect and composite sampling) apply statistical sampling theory and may involve random selection of sampling locations. An essential feature of this type of sampling is that each member of the population from which the sample is selected has a known probability of selection. When a probability-based design is used, quantitative conclusions (or statistical inferences) may be made about the sampled population from the analytical results. For example, the assessor may calculate a 95% upper confidence level (UCL) of the

arithmetic mean for the parameter of interest, say, lead concentrations in soil. If comparing this with the relevant investigation levels, the assessor can state whether the data indicates that the concentration exceeds or is below the investigation levels with a certain level of confidence (in this example 95%). Expert judgement is then used to draw conclusions about the study area based on the results of the sample data. Data analysis is discussed further in Section 13.

6.2.1 Judgemental sampling

In judgemental sampling, the selection of samples (number, location, timing, etc.) is based on knowledge of the site and professional judgement. Sampling is localised to known or potentially contaminated areas identified from knowledge of the site either from the site history or an earlier phase of site investigation. Judgemental sampling is commonly used to investigate sub-surface contamination issues in site assessment.

Although judgemental sampling can invalidate some statistical methods, particularly where the sampling size is small, alternative methods using non-parametric approaches can be used. Further information can be found in Gilbert (1987) and US EPA (2006b, 2007a). Judgemental sampling may be used in combination with other sampling designs to produce effective sampling for defensible decision-making.

6.2.2 Simple random sampling

In simple random sampling, the selection of samples (number, location, timing, etc.) is based on using random numbers, and all possible selections are equally likely. For example, a simple random sample of a set of drums containing soil for disposal can be taken by numbering all the drums and randomly selecting numbers from that list. Simple random sampling protects against bias (which may occur if sampling locations are subjective) providing that the sample size is not small (more than approximately 20 samples). Many commonly used statistical analysis methods assume that the data was obtained by using a simple random sampling design.

The method is most useful when the area of interest is relatively homogeneous and no major patterns or hotspots are expected. The main advantages of this design are:

- it provides statistically unbiased estimates of the mean and variability
- it is easy to understand and implement
- sample size calculations and data analysis are straightforward.

Information on implementing a simple random sampling approach may be found in US EPA (2006b). As most site investigations deal with non-uniform distributions of contamination, simple random sampling is usually combined with a stratified approach.

6.2.3 Systematic and grid sampling

In systematic and grid sampling, samples are taken at regularly spaced intervals over space or time. An initial location or time is chosen at random or based on a convenient site feature, and then the remaining sampling locations are defined so that all locations are at regular intervals over an area (e.g. grid intersections) or time (systematic). Examples of systematic grids include square, rectangular, triangular, herringbone and radial grids.

Systematic and grid sampling are used to search for hotspots and to infer means, percentiles or other parameters and are also useful for defining spatial patterns or trends over time. If the property/trend of interest is aligned with the grid, systematic/grid sampling has the potential to introduce bias (over or under representation) to the results.

Even though most contamination is not normally distributed, the data can often be transformed to be approximately normal. Also, if data sets are sufficiently large, statistical inferences can still be made, since in that case the sample mean is approximately normally distributed (Gilbert, 1987).

Where a reliable and full site history is available, judgemental sampling is generally preferred, however, grid sampling may be appropriate where there is an inadequate site history and there is reason to suspect contamination may be present or, there is a large area of contamination that requires characterisation. An example of the latter would be heterogeneous fill suspected or known to contain contaminated materials.

As grid spacing must be small to have a high probability of finding small hotspots, in practice professional judgement is used to locate areas of smaller grid size in areas most likely to contain hotspots and over areas where a higher degree of confidence is desirable. Information on implementing a systematic or grid sampling approach, including applications for soil and groundwater, can be found in US EPA (2006b).

Determining grid size/sampling density from mathematical formulae (for example, Appendix D of Standard AS 4482.1-2005) is not an acceptable approach without consideration of likely contaminant distribution and acceptable hotspot size.

6.2.4 Stratified sampling

In stratified sampling, the assessment area (generally the potentially contaminated area) is separated into non-overlapping sub-areas (or strata) which are known or expected to be more homogeneous than the whole assessment area. Different sampling patterns and densities may be used in the different sub-areas.

The strata may be chosen on the basis of spatial or temporal proximity, or on the basis of pre-existing knowledge (e.g. site history, soil type), or professional judgement. The main advantages of this design are:

- potential for achieving greater precision in estimates of the mean and variance where the measurement of interest is strongly correlated with the variable used to define the strata
- calculation of reliable estimates for subgroups of special interest.

Information on implementing stratified sampling approaches can be found in US EPA (2006b).

6.2.5 Transect sampling

In transect sampling, the samples are collected along a vector (a line of specified bearing, commonly 90˚) across an assessment area. Transect sampling may be appropriate when specific spatial characteristics of the contamination are to be targeted, for example, where there is a predictable contaminant distribution downwind/downgradient from a point source of contamination.

6.2.6 Composite sampling

In contrast to a discrete sample taken from a single location and analysed individually, a composite sample is taken by physically combining a number of subsamples, usually a maximum of four, into a single well-mixed sample for analysis. The subsamples should be preferably composited in the analytical laboratory.

Compositing can be cost-effective where the analysis costs are large relative to sampling costs. However, its use should be considered with caution because of the potential for individual high results to be masked by low results. Composite sampling is not recommended for site-specific health and ecological risk assessments. Its use is also dependent on there being no safety concerns or potential biases (for example, loss of volatile compounds) associated with the compositing process.

The SAQP should clearly state the qualifiers applying to selection of subsamples for a composite sample. Care should be taken to take the subsamples from the same soil horizon or stratum.

Composite sampling is not suitable for clay or fine-grained soils as subsamples are difficult to mix adequately. Consideration should be given to the moisture content of the soils to be sampled as subsamples are mixed without drying whereas laboratory results are reported in terms of dry weight.

Where non-volatile contaminants are present (for example, metals or heavy oils such as heating oils), composite sampling may be adopted as a cost-effective method for achieving low resolution data for screening purposes. Composites may also be useful in conjunction with other sampling designs or when the objective is to estimate the population mean and information on spatial or temporal variability is not needed (e.g. for characterisation of stockpiled materials).

Composite sampling is not suitable for the assessment of pH, volatile substances and semi-volatile substances such as OC/OP pesticides and lower molecular weight PAHs. A good understanding of the site history and the potential contaminants of concern are therefore a necessary precursor to adopting a composite sampling approach.

Where composite sampling has been used, the relevant assessment level should be divided by the number of subsamples in the composite and compared with the laboratory result. Further information may be found in AS 4482.1-2005 and SA EPA (2005).

6.3 Selecting a sampling design

Source: US EPA (2002)

The site should be subdivided into assessment areas based on the information collected in the preliminary site investigation (site history, local geology and hydrogeology and site conditions) and anticipated exposure areas (for example, size and location of proposed residential lots) and the sampling design selected according to the characteristics of the different sub-areas and the DQOs.

In general, when the source of contamination is known or is suspected to be limited to a specific area, sampling points are located relative to the suspected source(s) using judgemental sampling stepping out from the suspected source location, or systematic grid sampling centred on that location.

Specialised professional advice should be sought in developing sampling plans for rock soil mixtures at waste rock dumps, tailings dams, heap leach pads, and other artificial structures associated with mining site contamination.

Table 3 presents examples of example investigation scenarios that may be encountered and suggests sample designs that may be relevant. As indicated below, a more sophisticated sampling design may follow on from a preliminary (screening) investigation.

Table 3. Selecting an appropriate sampling design

If you are…	and there is…	consider using…	in order to…
performing a relatively small scale screening investigation	limited budget and/or schedule	judgemental sampling	assess whether further investigation is warranted
developing an understanding of where contamination is present	adequate budget for the number of samples needed	grid sampling	acquire coverage of the area of concern with a given level of confidence that a hotspot of a given size would be

			detected
estimating a population mean	adequate budget for the number of samples needed	systematic or grid sampling	produce information on spatial or temporal patterns
developing a detailed understanding of where contamination is present and/or estimating a population mean	spatial or temporal information on contaminant patterns	stratified sampling (includes judgemental and grid sampling)	increase the precision of the estimate in key areas of concern

Adapted from US EPA 2002

6.4 Sampling density and depth of sampling

The aims of an SAQP (refer Section 5.3) are to reduce the likelihood of under assessment (that could result in significant adverse effects from unidentified contamination) or over assessment (concluding that a site requires further investigation when in reality it does not) and to enable an appropriate level of remediation of contamination that is sufficient to protect human health and the environment.

The information presented in this section can be applied to both horizontal and vertical sampling.

Consideration of the CSM and DQOs should inform the requirements for sampling density and depth of sampling. The amount of sampling required will depend on an integrated appraisal of factors including:

- the size of contaminated areas to be detected
- the number of stages of sampling considered feasible
- the size of the site and final subdivided lots if the site is to be subdivided
- the distribution of uses on the site and the disposition of structures
- the site history (which may vary across the site).

When developing a sampling program, consideration should be given to numerous factors including, but not limited to:

- the likely heterogeneity of any surface fill and underlying geological units
- whether knowledge of background soil and groundwater quality is required
- the depth and thicknesses of soil/aquifer units
- soil properties that affect contamination migration (e.g. texture, moisture content, clay content)
- physical and chemical nature of the contaminant under investigation (e.g. solubility, volatility and density)
- the nature of the release (e.g. surface spill, leaking underground pipe, buried waste)
- the timing and duration of the release
- the amount of contaminant likely to have been released
- the possible effects of contaminant migration through the unsaturated zone and when and where the contaminant entered the saturated zone

- the effects of potential degradation processes
- the direction and rate of groundwater flow within each aquifer.

If a site is to be subdivided, the size of the subdivided lots should be taken into account when determining the sampling density. While predictions may be made on a 'macro' scale, residents or owners may seek information about their own particular area of land and the risks associated with this land, especially if the potential contamination on the original site was uneven in distribution and type.

The detection of hotspots is an important issue for sites to be used for residential purposes or other sensitive uses where children have regular access to soil or where there is potential groundwater contamination. A greater sampling density is usually required for these sites. The toxicity of the contaminant and the size and magnitude of the potential hotspot(s) needs to be considered in determining the sampling density.

The development of a suitably detailed CSM will inform decisions about the depth of sampling required. For health and ecological risk assessment, the soil strata to which people and other receptors could feasibly be exposed should be adequately sampled. This will result in a weighting towards near-surface sampling unless the history or the nature of the soil and the presence of groundwater suggests it should be otherwise. On residential sites, the maximum excavation depth (such as for a swimming pool) is unlikely to extend beyond three metres, but much deeper soil disturbance may occur on a commercial site.

If dealing with volatile contaminants such as light fraction petroleum hydrocarbons or chlorinated solvents, then vapour transport from depth and through a shallow soil zone may pose a risk. Deeper sampling to determine the nature and extent of the source of the vapours and the risk they represent may be required—refer Section 9.

The risk to groundwater needs to be assessed according to jurisdictional requirements, especially if receptors may be exposed by current or realistic future use of the groundwater resource—refer Schedule B6.

To delineate contamination laterally, typically samples should be taken until either no further contamination is detected or concentrations are below the relevant investigation levels or site-specific risk-based criteria.

The nature and appearance of drill cores will influence sampling at depth. It is essential that samples are taken from within a natural stratum or fill horizon and not across strata.

At the surface, samples at 0–100 mm or 0–150 mm should be taken unless there is evidence of a thin superficial layer of contamination. Where there is good evidence that contamination is restricted to a thin superficial layer, a shorter sampling interval may be appropriate, however, a subset of deeper samples should be analysed to inform/confirm the CSM. At greater depths, the sampled interval should be no more than 500 mm to avoid a compositing effect.

7 Soil assessment

7.1 Introduction

The selection of appropriate site investigation techniques depends on a number of factors including the stage of the investigation (for example, preliminary assessment or detailed delineation, the depth of investigation required, the contaminant type (volatile or non-volatile, bonded or unbonded asbestos-containing-material), the depth and nature of any fill, and whether an undisturbed sample is required.

The most commonly used investigation techniques are test pits, trenching and drilling of shallow boreholes. Samples from shallow depth are generally obtained from test pits and trenches or from augers. Samples from greater depths may be obtained by a range of drilling methods including direct push, hollow stem augers, split spoon, Shelby tube, mud rotary and sonic drilling. Methods capable of providing continuous or near-continuous soil cores, such as direct push, split spoon and sonic drilling, are preferred. Air drilling and solid flight augers provide highly disturbed samples and poor depth control which limits their value for site characterisation purposes. Further information on soil investigation methods can be found in Australian Standards AS 4482.1-2005, AS 4482.2-1999 and AS 1726-1993.

A number of screening tools are also available that can be used to rapidly and cost-effectively identify and delineate VOC and SVOC contamination in both the unsaturated (vadose) and saturated zones. These include soil vapour sampling, and the laser-induced fluorescence (LIF) and membrane interface probe (MIP) tools. LIF and MIP are real-time tools that can provide detailed logs of the sub-surface and can be used in a reactive or adaptive field sampling program, particularly for volatile substances where trial pitting and some coring methods are not as applicable.

Various geophysical techniques can be used for site characterisation purposes including determining depth to bedrock, delineation of groundwater contamination, location of voids, faults or fractures and the presence of buried items such as steel drums and tanks. The information gained can be used for selecting optimal locations for boreholes and test pits as well as to correlate geology between wells. The techniques available include metal detectors, magnetometers, electromagnetic conductivity surveys, electrical resistivity—or electrical impedance tomography—and ground-penetrating radar.

A detailed description of geophysical techniques is beyond the scope of this guideline, however, further information can be found in:

- ASTM D6432-99 (2005) Standard guide for the surface ground penetrating radar method
- ASTM D6429-99 (2006) Standard guide for selecting surface geophysical methods,
- ASTM D5753-05 (2010) Standard guide for planning and conducting borehole geophysical logging
- Clements, et al. 2009, Characterisation of sites impacted by petroleum hydrocarbons: guideline document, CRC CARE Technical Report no. 11, CRC for Contamination Assessment and Remediation of the Environment, Adelaide.
- NJDEP 2005a, Field Sampling Procedures Manual, New Jersey Department of Environmental Protection. (Available online at www.nj.gov/dep/srp/guidance/fspm).

Detailed information on site investigation techniques can be found on the US EPA CLU-IN characterisation and monitoring webpage at www.clu-in.org/characterization/. The advantages and disadvantages of various techniques applicable to petroleum hydrocarbons in soil, soil vapour and groundwater are presented in Clements et al. (2009). NJDEP (2005a) provides detailed field sampling procedures including for soil vapour surveys.

7.2 Soil investigation techniques

7.2.1 Test pits and trenches

Test pits and trenches are generally excavated by hand using a shovel to shallow depths or by machine (backhoe or long-arm excavator) to greater depths. Soil samples may be collected from the walls of a test pit when they are shallow and it is safe to do so in accordance with the site health and safety plan. Only freshly exposed surfaces are suitable for sampling volatile and semi-volatile contaminants. Tests pits and trenches expose a large surface area for visual assessment of soil profiles and potential contamination and generally allow the investigator to gain a better appreciation of soil features and soil heterogeneity than that obtained with an individual borehole.

7.2.2 Intact soil coring

In general undisturbed samples obtained from near-continuous soil cores are preferred to grab samples for inspection and analysis. Intact soil coring is typically conducted by advancing a hollow rod or thin-walled metal tube into the sub-surface by direct push or other method such as sonic drilling. Direct push methods eliminate the need for a drilling fluid and avoid potential interferences from introduced fluids.

Auger and split-spoon samplers fitted with clear acetate sleeve liners may also be used to collect soil samples; however, sample quality is generally not as good as that obtained using direct push or sonic drilling methods.

Once soil cores have been obtained, samples from specific depth intervals can be taken and suitably preserved for laboratory analysis. Where an entire core is to be taken, the soil core tube should be quickly capped, labelled, wrapped and packed (and kept cool using ice bricks or refrigerated to keep the sample in a relatively undisturbed state) and dispatched to the laboratory for analysis.

7.2.3 Cone Penetrometer Testing

Cone penetrometer testing (CPT) is an in situ form of direct push drilling where sensors are mounted in a cone at the tip of the drill rods. As the cone is advanced, the sensors measure the resistance of the soil to the force of the advancing cone and the data is relayed to an on-board computer which interprets the soil stratigraphy and other parameters. A range of additional sensors may be used with CPT for simultaneous measurement of multiple parameters. The range of available sensors includes pressure head transducers (allowing permeability and hydraulic conductivity assessment), conductivity probes (allowing soil types and saturation to be estimated) and nuclear and pH probes.

CPT is a useful tool for providing rapid, continuous profiles of sub-surface stratigraphy and can save considerable time and money, particularly at large sites with complex geology.

7.2.4 Membrane interface probe

The membrane interface probe (MIP) tool consists of a heated probe equipped with a semi-permeable membrane mounted on a direct push or CPT drilling rig. VOCs diffuse across the membrane and enter a carrier gas within the probe. The carrier gas transports the contaminants to a gas chromatograph at the surface which can be equipped with various detectors for measurement of a wide range of VOCs: an electron capture detector (ECD) for chlorinated organics, a photo-ionisation detector (PID) for aromatic hydrocarbons, and a flame ionisation detector (FID) for straight-chained hydrocarbons.

For sites containing light non-aqueous phase liquids (LNAPL), MIP is typically used to locate and delineate dissolved-phase groundwater and soil–vapour plumes, while laser induced fluorescence (LIF) (see below) is used to delineate the LNAPL source zone.

The MIP tool is usually equipped with an electrical conductivity sensor to interpret soil lithologies. The combination of sensors enables an increased understanding of contaminant distribution, particularly in heterogeneous lithologies.

One or more background MIP borings upgradient of each assessment area should be advanced in order to determine the background response. The MIP response can be used to determine concentrations of specific contaminants if it is calibrated with soil and groundwater samples from across the investigation area.

The MIP tool is typically used in the context of an adaptive sampling approach using a dynamic sampling plan and DQOs, that is, the investigation proceeds in a step-wise approach with the location and depth of each subsequent boring being determined in the field based on the results and interpretation of the preceding boreholes using a predetermined decision framework. QA/QC procedures should be developed for MIP surveys as part of the DQO process.

With multiple MIP locations and appropriate data interpolation and visualisation software, MIP data can enable a 3-D depiction of NAPL source zones in both the unsaturated (vadose) and saturated zones.

7.2.5 Laser-induced fluorescence

The laser-induced fluorescence (LIF) tool consists of an ultraviolet (UV) or visible wavelength laser connected to a sapphire window, mounted on the side of a direct push/CPT probe tip.

The LIF laser transmits light through the sapphire window, which is then absorbed by any PAHs in contact with the window causing the material to fluoresce at a characteristic wavelength. The fluorescence emission is recorded continuously by a detection system as the probe is advanced.

LIF tools are available which, depending on the wavelengths monitored, are capable of differentiating different types of product. UV LIF systems are appropriate for light fuels up to mid-range oils, but often fail to adequately respond to heavy fuel oil, heavy crudes, coal tars and creosotes. Visible wavelength systems detect heavy fuel oil, heavy crudes, coal tars, and creosotes but do not respond to light fuels such as petrol and kerosene. If possible, an appropriate NAPL sample should be tested to ensure the appropriate wavelength LIF is used.

One or more background LIF borings upgradient of each assessment area are recommended in order to determine the background LIF response. If NAPL is present at the site, a LIF borehole should be advanced adjacent to a well where NAPL has been measured to calibrate the LIF response to the specific NAPL contamination present at the site. The LIF data should also be validated with soil and groundwater sampling to determine concentrations of specific contaminants throughout the investigation area.

As for MIP, the LIF is combined with an electrical conductivity sensor to interpret lithology and is used in a similar reactive sampling approach. Similarly, with multiple LIF locations and the use of data interpolation and visualisation software, LIF data can enable a 3-D depiction of NAPL source areas in both the unsaturated (vadose) and saturated zones.

7.2.6 Soil vapour surveys

Source: NJDEP (2005a)

Soil vapour surveys may be used to screen sites for VOC and SVOC contamination source areas in the vadose zone and to delineate the extent of contamination. Soil vapour sampling, when applied appropriately, can be used as a screening procedure to assist in locating soil sampling and monitoring well locations.

There are two basic types of soil vapour surveys performed as part of site assessments. The first type is an active soil vapour survey where a volume of soil is pumped out of the vadose zone into a sample container or directly into an analyser. The second type is the passive soil vapour survey where a sorbent material is buried in the vadose zone so that contaminant vapours can be selectively absorbed over time using the ambient flow of vapours through the subsurface. The latter is particularly applicable to low permeability soils where active methods are less effective.

Further information on soil vapour sampling is presented in Section 9.

7.2.7 Ground penetrating radar

Ground penetrating radar is the most commonly used of the geophysical methods and is typically conducted by rolling a radar unit across the site in a grid pattern and recording and processing the data collected to provide a two-dimensional or three–dimensional image of the surveyed area. Metal objects or near–surface features (such as pipes or utilities) can cause noise on the measured signal; if the location of these features is known, their effect can be minimised in the data processing stage. In homogeneous soil profiles, ground penetrating radar surveys may assist in defining the lateral and vertical extent of NAPL plumes in shallow soil or groundwater—see Clements et al. (2009).

7.3 Field description of soils

Accurate documentation and careful consideration of field observations is essential as this can greatly improve understanding of the variability of contaminant distribution across a site.

All boreholes (including groundwater monitoring wells) and test pits should be logged in accordance with AS 1726-1993 and the presence of strata, moisture, seeps or water-bearing zones, elevation of the water level/hydraulic head, imported fill and odorous or stained materials carefully noted. These logs are essential for interpretation of chemical data to establish the extent of contamination and to assist in the design of more detailed investigations. Example logs are included in Appendix D.

A photographic record that is well labelled for date, location and orientation is a valuable reference tool for documenting procedures and for understanding soil/aquifer heterogeneity and variability in laboratory results. Photographs are recommended to be taken of the strata present in test pits and soil cores and the appearance of split samples, particularly to illustrate visible heterogeneity in the field.

Field checklists to aid documentation of essential information are available for download from the EPHC website at www.ephc.gov.au.

7.4 Field testing

A variety of field screening techniques may be used to provide immediate (real-time) information about the concentration and distribution of contaminants on contaminated sites. These tests, by their very nature, are less rigorous and reliable than analytical tests conducted in a laboratory, however, they provide cheaper and quicker results to guide the design of further sampling strategies for site assessment.

The most commonly used field tests include:

- gas detector tubes
- colorimetric test kits
- headspace testing using PIDs and FIDs
- field portable x-ray fluorescence spectrum analysers
- field gas chromatography
- immunoassay test kits.

These techniques can be used to gain a general understanding of the field conditions and the presence of possible contamination and may assist in the selection of samples for laboratory analysis. PID measurements, for example, may be useful as a field guide to indicate areas of volatile compounds. However, their role in providing real-time data needs to be augmented by laboratory chemical analysis.

Their use as the sole source of analytical data in the assessment of potentially contaminated sites is inappropriate as they may give falsely high or low results. For example, naphthalene is commonly reported in petroleum hydrocarbon-impacted soils and will evoke a response from a PID, in contrast to benzo(a)pyrene (a more significant PAH in terms of human health), which will not be detected by a PID. As these measurements do not always correlate well with laboratory results they are generally not suitable for validation sampling.

Prior to use of any field monitoring equipment there should be:

- a determination that they are capable of detecting relevant contaminants
- adequate understanding of the methods of use for the particular instrument, its limitations and site conditions that may affect the results
- appropriate calibration (and recording of the calibration data) for the substances being measured
- an appraisal of site conditions that may affect the results, e.g. high soil moisture may result in artificially high PID results for benzene.

Further information on field characterisation techniques may be found on the US EPA CLU-IN website at www.clu-in.org/characterization/.

7.4.1 Gas detector tubes

Detector tubes have been developed that measure volatile gases including individual compounds, for example, hydrogen sulphide, or groups of compounds, for example, petroleum hydrocarbons. They can provide a direct measure of the analyte in ambient air or an indirect measurement of soil and groundwater contaminant concentration when used in field test kits for measurement of soil vapour and headspace for liquids. The reagents in the tubes may react with compounds of similar chemical properties; consequently, false positives and inaccurate results are possible and should be identified in the DQO process.

7.4.2 Colorimetric test kits

Colorimetric tests rely on the chemical reactions of indicator compounds with individual compounds or classes of compounds. Tests are generally performed by mixing reagents in specified amounts with the soil sample to be tested and comparing the resultant colour change with a colour chart or using a field colorimeter to determine concentration.

Colorimetric tests have been developed for a wide range of substances including BTEX, total PAHs, chlorinated hydrocarbons, PCBs and various individual pesticides and classes of pesticide. The detection limits in soil are generally in the low ppm range (lower detection limits are achievable in water as no extraction stage is necessary). Although these tests are relatively simple to perform, depending on the kit, they can suffer from interferences from other co-contaminants or naturally occurring materials or organic matter. Their usefulness for specific site-characterisation purposes can be evaluated by comparison of field colorimetric results with laboratory results over a range of analyte concentrations.

7.4.3 Headspace testing using photo-ionisation and flame ionisation detectors

Field headspace testing is a commonly used method for screening soil samples for volatile and semi-volatile organic compounds. The procedure involves partially filling an airtight container with a fresh

soil sample and then analysing the headspace vapour using an appropriately calibrated portable instrument, typically a PID or FID.

A FID uses a hydrogen flame to ionise the organic vapours whereas a PID uses an ultraviolet lamp to ionise the vapours. The instrument response is related to the electric current generated by the ionised compounds. FIDs are most sensitive to aliphatic hydrocarbons as these compounds burn more efficiently than aromatic compounds. PID instruments are most sensitive to aromatic hydrocarbons (for example, BTEX compounds) and can measure most VOCs in the range of C_6 equivalent carbon atoms (for example, benzene) to C_{10} (for example, naphthalene). Neither instrument is effective for detecting non-volatile compounds such as highly weathered hydrocarbons. Care should be taken when using PIDs since a positive bias may result from water vapour or moist air and/or dust being drawn into the instrument. FIDs are not sensitive to water vapour.

A standardised field procedure for headspace testing should be followed and the details of the test method documented (size of jar, soil volume, equilibration time and ambient temperature) in the investigation report.

7.4.4 Field portable x-ray fluorescence

X-ray fluorescence (XRF) is a rapid screening tool that can be used to measure metal concentrations in soil. Performance is dependent on the metal, the soil matrix and soil moisture content. Although a range of heavy metals can be simultaneously detected, there are potential interferences that influence the method accuracy and precision. The US EPA has developed a methodology to guide XRF analysis (US EPA 2007b).

The advantages of XRF include real-time results, when used in scanning mode on surface soil, or near real-time results when deeper samples are collected and analysed in the field. The usefulness for specific site-characterisation purposes can be evaluated by comparison of results from split samples analysed by field XRF with laboratory results over a range of analyte concentrations.

7.4.5 Field gas chromatography

Field gas chromatography (GC) may be used for the analysis of volatile and semi-volatile compounds in soil, soil vapour and water. The two main components of a GC are a column to separate the individual constituents and a detector (such as a PID or FID) to measure the signal response of the constituents. The analysis is compound-specific and potentially has the greatest accuracy of all the commonly used field analytical techniques.

7.4.6 Immunoassay test kits

Immunoassay test kits, using antibody-antigen reactions, can be used to measure petroleum hydrocarbons in soil and water. For most kits, the intensity of the colour development is inversely proportional to the amount of substance present. The concentration is determined by comparison with a reference standard or with a portable photometer.

7.5 Stockpile sampling

An in situ soil sampling program informed by site history, inspection and contaminant form is the preferred approach for site assessment. On occasions it is necessary to stockpile soils that have not been assessed or only partially assessed in situ, and to devise a thorough stockpile sampling plan.

7.5.1 Excavation and inspection of the stockpile

Excavation may result in mixing of low-level or uncontaminated soil with smaller quantities of contaminated soil, having the effect of diluting higher concentrations. It is preferable for assessors to supervise excavation and, as far as practicable, segregate stockpiles according to soil and contaminant types and to avoid dilution.

The process of excavating material often results in mixing of strata and different fill and soil types. Stockpiling may cause some segregation of grain sizes particularly on the exterior slopes. Specific grain sizes may contain the contaminant source and concentrate in some stockpile locations; for example, finer material may tend to accumulate at the toe of batters and coarser material towards the crest. Sticky clay material may be distributed into a different part of the stockpile than loose soils. The age and surface condition of the stockpile should be assessed, particularly if it has been weathered and subjected to leaching.

The composition of the stockpile should be documented by inspection of its external appearance and excavations into the stockpile by shovel (for small stockpiles) or excavator bucket where a shovel cannot reach the centre of the stockpile. The stockpile dimensions should be determined noting its regular or irregular shape and a 3-D plan prepared. The volume of material present should be estimated.

7.5.2 Number of samples

Table 4 below provides the minimum number of samples recommended for characterisation of stockpiles up to 200 m^3 comprising similar materials. A greater number of samples may be required when there is a large range in contaminant concentrations or soil types. If only the minimum number of samples is collected and there is a large range in contaminant concentration, then either the maximum concentration should be assumed for disposal purposes or additional samples collected and analysed and the situation re-evaluated. In situ samples taken prior to excavation may be helpful for informing the decision on the number of samples required for adequate characterisation of stockpiles.

Table 4. Minimum number of samples recommended for initial assessment of stockpiles

Stockpile volume, (m^3)	No. of samples
<75	3
75 – <100	4
100 – <125	5
125 – <150	6
150 – <175	7
175 – <200	8

The recommended sampling frequency (Table 4) applies to the characterisation of homogenous soils suspected of contamination. Lower sampling rates may be derived for soil quantities greater than 200 m^3 by applying statistical analysis. Worked examples of applying 95% UCL_{ave} to characterise stockpiles are included in EPA Victoria (2010).

Jurisdictions may have specific requirements where materials are to be recycled, recovered and reused for beneficial purposes.

7.5.3 Sample point distribution

The stockpile should be sectioned into an appropriate distribution of sampling locations based on inspection, site history and other assessment data about the nature of contaminants present. If a section of the stockpile is known to have a higher level of heterogeneity and greater contamination risk and the balance of the stockpile is relatively homogenous with low-level contamination, sampling bias to the more contaminated section may be considered. If this information is not known, a uniform sample point distribution should be used. A plan should be developed of the stockpile sections and the

corresponding sample locations that represent each section. This will allow physical separation of portions of the stockpile for further characterisation, if required, after receipt of the analytical results.

7.5.4 Sampling

Collection of samples from the exterior 300 mm of the stockpile should be avoided due to the higher risk of weathering and grain size grading errors.

Samples for inorganic and non-volatile components should be taken at various depths towards the centre of the stockpile from 300 mm below the stockpile surface. Compositing may improve the reliability of samples for inorganic analysis. Composites should be based on equal quantities of material from 4 random locations and depths in the area of the stockpile allocated to the sample. The trowel should be cleaned after soil collection at each random location and the collected material thoroughly mixed on a clean surface, subsampled and preserved for chemical analysis.

Composites are not suitable for the assessment of pH, volatile substances and semi-volatile substances such as petroleum hydrocarbons, OC/OP pesticides and lower molecular weight PAHs. Samples for volatile and semi-volatile compounds should be taken without delay from a freshly excavated surface 500 mm or greater depth below the stockpile surface.

Systematic sampling directly from excavator buckets during the excavation and stockpile formation process or for appraisal of larger stockpiles using appropriate QA/QC processes is an acceptable strategy in site assessment. Further guidance on stockpile sampling may be obtained from EPA Victoria's *Industrial waste resource guidelines* (2010).

7.6 Assessment of soil leachability to groundwaters and surface waters

7.6.1 Leaching potential to groundwater and surface water

Contaminants in soil can leach to groundwater under certain conditions. For inorganic substances, leachability is particularly affected by soil pH, contaminant solubility and redox (Eh) conditions.

The leachability characteristics of contaminated soil can be used to help assess:

- the impact of soluble soil contaminants on groundwater quality
- the impact of leaving contaminated soil materials on site.

7.6.2 Soil leaching tests

Information on leachability tests applicable to the assessment of site contamination can be found in Sections 2.7 and 12 of Schedule B3.
Samples to be tested should be selected with reference to the CSM and be representative of the impacted materials. Analysis of appropriate background samples should be included for comparative purposes.

7.6.3 Theoretical calculation of porewater concentration

Methodologies are available which aim to predict the impact of leaching soil contaminants on groundwater quality and groundwater resources. An overview of the US EPA (1996) methodology is included in Schedule B5b.

7.6.4 Disposal of contaminated soils

Treatment and disposal of excavated contaminated soils should be in accordance with jurisdictional legislation or guidelines for re-use and/or disposal of contaminated soils.

8 Groundwater assessment

The recommended risk–based approach to the assessment of groundwater contamination is outlined in Schedule B6.

The process involves a staged risk-based approach to delineation of contamination using guidelines such as the *Australian and New Zealand guidelines for fresh and marine water quality (*AWQG) (ANZECC & ARMCANZ 2000), the *Australian drinking water guidelines* (ADWG) *(*NHMRC 2011) and the *Guidelines for managing risk in recreational water* (GMRRW) (NHMRC 2008) as appropriate investigation and response levels. The process may include a detailed assessment of contaminant concentrations over time using fate and transport modelling to predict the current position and future movement of groundwater contaminants to assess potential risk to receptors.

This section deals with the basic requirements for groundwater investigation, including installation of monitoring wells, sampling of groundwater, presentation of data and delineation of groundwater contamination.

Fractured rock aquifers (for example, fractured basalts, bedrock aquifers and limestones) behave fundamentally differently from unconsolidated aquifers such as sands and gravels. Specialist advice should be sought from qualified contaminant hydrogeologists with experience in fractured rock aquifers. Further information on groundwater flow and groundwater sampling in fractured rock aquifers can be found in Cook (2003) and Nielsen (2006) and references therein. The US EPA CLU-IN website also provides useful information on characterising fractured rock aquifers and lists various related resources at: http://www.clu-in.org/contaminantfocus/default.focus/sec/Fractured_Rock/cat/Overview/.
Site assessors should be aware of (and comply with) relevant jurisdictional requirements such as groundwater protection policies and licensing requirements for the construction of monitoring bores and groundwater abstraction.

The collection and assessment of groundwater data and the selection and use of fate and transport models should be undertaken by appropriately qualified and experienced professionals. This is particularly important when applied to fractured and karstic rock environments.

8.1 Groundwater investigation approaches

8.1.1 Introduction

There are several methods for collecting groundwater data. In general, these methods involve collection of:

- in situ measurements to calculate hydraulic head, groundwater flow direction and rate
- in situ measurements of apparent product thickness (NAPL, immiscible with water)
- in situ physical and/or chemical measurements of groundwater quality, e.g. redox potential, electrical conductivity, pH, and dissolved oxygen
- collection of groundwater samples for ex situ measurement/analysis.

The main issues that determine the selection of the appropriate method(s) are:

- the DQOs
- site-specific conditions such as depth to water table, soil/rock competency
- analyte-specific characteristics
- financial and logistical constraints.

Careful consideration and appropriate weighting of each of these issues will assist in determining the most appropriate method(s) of groundwater investigation.

8.1.2 Scope of investigation

The appropriate scope of the investigation is determined through the development of the site CSM and the DQOs. Generally this will include a preliminary site investigation comprising a desktop review of relevant background and historical information and a site visit followed by one or more intrusive field programs to update and refine the CSM until the objectives of the site assessment are met. The results of any earlier investigations, including soil and soil vapour investigations, should be used to refine the CSM and inform the scope of the groundwater investigation.

Typically this will include consideration of the following:

- the nature of the contaminant including its mobility and toxicity characteristics
- the type and location of known and potential contaminant source zones (including off-site) and associated contaminant plumes
- site geological and hydrogeological conditions (lithology, lateral and vertical extent of aquifers, perched water tables, confining layers, aquifer properties, etc.)
- depth to the water table and likely seasonal variation
- potentiometric surface(s)
- upgradient groundwater quality to assist in determining background groundwater quality
- hydrogeochemistry of relevant aquifer units
- direction of groundwater flow and hydraulic gradient
- location of recharge and discharge areas
- location of any abstraction wells
- current and future realistic use(s) of the groundwater resource and nearby surface water resources and water protection zones
- known and/or perceived risks to the environment and/or human health including the presence of potential pathways between contaminant source(s) and potential receptors.

8.1.3 Site-specific conditions

Site-specific conditions that may limit or govern the choice of groundwater investigation techniques include:

- hydrogeological conditions including the depth to groundwater, soil/rock types and the presence of multiple aquifers
- potential risks to uncontaminated aquifers and/or surface water resources
- restrictions with regard to accessibility due to topography, ground bearing capability, site infrastructure or interference with site operations
- risks to the environment and/or public safety
- geotechnical limitations such as soft or saturated ground, cavernous or karstic terrains and stability
- natural events such as flooding and shifting sand dunes.

Any of these conditions may limit the applicability of certain methods of drilling, bore installation and groundwater sampling and make other methods more practical and cost-effective.

Appropriate measures should be taken to minimise the spread of contamination by not creating migration pathways from the surface to groundwater or between different aquifers. For example, where a monitoring well is targeting a deeper aquifer unit and contamination is present in a shallow aquifer unit or overlying fill horizon, this should be cased off so as not to permit cross-contamination between the two units.

8.1.4 Analyte-specific characteristics

The physical and chemical characteristics of contaminants have a profound effect on their sub-surface distribution and/or occurrence in groundwater at a given site. Physical and chemical characteristics that may have an effect on the distribution of contaminants include:

- contaminant solubility
- presence of NAPLs
- relative density (e.g. in the case of NAPLs, LNAPLs such as oils are less dense than water, whereas dense NAPLs (DNAPLs), such as some solvents, are denser than water; for aqueous liquids, relative salinities are important)
- stability (chemically and microbiologically)
- partitioning characteristics (e.g. sorption and volatility)
- aquifer redox conditions.

These characteristics will determine if contaminants are:

- capable of leaching through a soil profile and/or are soluble in the groundwater
- more or less dense than the groundwater, such that there is a likelihood for them to be present close to the water table (e.g. LNAPLs or where low salinity water infiltrates into more saline groundwater) or more extensively throughout the aquifer (e.g. with DNAPLs or where saline water infiltrates through fresh groundwater)
- relatively susceptible to effects of volatilisation, reaction with other chemicals/substances in the sub-surface, biodegradation, or attenuation.

Where there is a potential for contaminants to be present in an aquifer it is important to understand and predict where they are most likely to be concentrated prior to selecting the appropriate groundwater investigation method. Without this consideration, there is the potential for errors, some of which may result in:

- cross-contamination within and/or between aquifers
- non-detection of groundwater contamination
- inaccurate or misleading data
- expenditure of excessive resources where more simple and cost-effective methods could have been used.

8.2 Monitoring well establishment

8.2.1 Introduction

In general, most groundwater investigations in Australia are conducted using information obtained from cased, semi-permanent or temporary groundwater monitoring wells. Monitoring wells are used for a range of applications including:

- groundwater sampling for ex situ analysis
- monitoring and/or profiling in situ groundwater parameters
- monitoring of groundwater level fluctuations
- aquifer testing.

Wells retained as part of a monitoring network should be properly maintained to ensure the integrity of the sample data collected. Well lifespan will depend on the materials used, the standard of installation and whether aggressive ground conditions are present. The monitoring plan should include provisions to inspect and assess monitoring wells for their suitability for monitoring purposes. Damaged or abandoned bores may provide conduits for future contamination unless properly decommissioned. Monitoring wells which are no longer required or are unsuitable for continued monitoring should be decommissioned in accordance with jurisdictional requirements.

The following overview of drilling methods is largely based on information in EPA Victoria (2000) and SA EPA (2007), EA (2006) and *Standard practice for design and installation of groundwater monitoring wells,* ASTM D5092 (2004). Additional information can be found in Aller et al. (1989), Driscoll (1986) and the *Manual of methods, applications and management* produced by the Australian Drilling Industry Training Committee (ADITC 1997).

8.2.2 Logging of boreholes

A careful record of the geology encountered during drilling should be described and classified in accordance with *Geotechnical site investigations,* AS 1726-1993. Example logs are included in Appendix D.

Field check lists are available in the Toolbox at www.scew.gov.au/nepms/assessment-of-site-contamination.html to aid documentation of essential information including the soil profile and well construction.

Monitoring wells may also be logged using various geophysical techniques, for example, to determine aquifer characteristics in more detail and to supplement other methods (for example, geological logging, core analysis, aquifer tests, water sampling and analysis). Further information can be found in ASTM D5753-05 *Guide for planning and conducting borehole geophysical logging* (and related standards) and at the United States Geological Survey website: http://ny.water.usgs.gov/projects/bgag/intro.text.html.

8.2.3 Well construction

Monitoring wells should be constructed to an appropriate standard and from suitable materials to ensure that high quality samples can be collected over the projected lifetime of the well. The assessor should ensure that the drilling technique, depth and diameter of the borehole, screen length, well construction materials (screen, casing, filter pack, seals and grout) and headworks design selected are compatible with the monitoring objectives.

For general guidance on monitoring well installation procedures see ASTM D5092-04.
A decision on the appropriate means of constructing monitoring wells involves consideration of a number of factors including the hydrology, geology and geochemistry of the formation, the nature of the contamination, the chemical resistance and leaching properties of the construction materials, the cost and the necessity to maintain the integrity of samples. Further information may be found in EPA Victoria (2006), SA EPA (2007), EA (2006) and ASTM D 5092-04.

There are several standard drilling methods available including hollow-stem auger, air and mud rotary, cable tool, sonic and direct push. The general suitability of these techniques for a range of ground conditions is discussed in EA (2006). The assessor should consider drilling methods that minimise the introduction of drilling additives wherever possible.

Certain drilling techniques can cause smearing (for example, rotary auger) or compaction (cable tool) of borehole walls and may also promote transport of geological formation materials and drilling fluids into different aquifer zones. In a worst-case scenario, this can result in almost complete blockage of the well screen resulting in non-representative groundwater samples when the boreholes are monitored.

Drilling fluids are used to clean and lubricate the drill bit, to remove rock cuttings from the borehole and to keep the borehole open during drilling. These may include air, water and specific drilling mud formulations or native clay slurries. Drilling fluids can have a range of effects on groundwater quality such as the following:

- air may severely disturb hydrochemical profiles through oxidation processes, e.g. oxidation of ferrous Fe^{2+} to ferric Fe^{3+}
- water may dilute or flush groundwater near the borehole and cause precipitation of minerals, thereby blocking or obstructing groundwater pathways
- mud may invade the permeable formations and block pathways to the well screen.

Care should be taken to avoid contamination of the borehole and surrounding geology during drilling and construction of the well through the inappropriate use of lubricants, oils, grease, solvents, or materials with incompatible coatings. If the groundwater quality is altered, the samples obtained may not be representative, leading to uncertainties and potential errors in the assessment. Some considerations for material selection (EA 2006) include:

- the chemical environment in which the installation is placed – aggressive environments (saline, free-phase, low or high pH) will rapidly degrade or corrode some materials
- effect of contaminants on materials – corrosion, solution, strength, leaching
- effect of materials on groundwater – leaching, oxidation, pH.

As some drilling-related effects are frequently long-lived or even permanent, it is important to record drilling method, materials used and details of bore development on the well logs.

8.2.3.1 Screen depth and length

Groundwater investigations should be designed to target the part, or parts, of the aquifer most likely to be affected by contamination.

Under laminar flow conditions, contaminated groundwater flows in discrete zones controlled by the physical properties of the aquifer and the presence of any preferential pathways such as higher permeability units (such as the cleanest sands in an interbedded sand and silt sequence) and fractures. The location and length of the well screen is therefore critical to obtaining a representative sample of contaminated groundwater.

The selection of screen length depends on the objectives for the monitoring well; however, in general, well screens should be kept as short as possible to avoid potential dilution effects. The interval of aquifer potentially contributing to flow includes the filter pack either side of the well screen as well as the screened interval itself. To minimise the potential for vertical flow between aquifers via the well bore, screens should not be installed across different geological units or water-bearing zones.

Screen design should consider the likely fluctuation in the water table and the well screen should be located such that at least part of the screen remains within the saturated zone throughout the year. Where extreme variations are likely to be present, (e.g. drought and non-drought periods) consideration may need to be given to installing additional monitoring wells.

In the initial phases of investigation, well screen lengths of 3 m or more are common. However, once contamination is suspected or confirmed, shorter screens of the order of 1 m long located specifically within the zone of interest are recommended since small-scale heterogeneities are important in controlling contaminant flowpaths. Where the geological unit of interest exceeds 1–2 m in thickness, multiple wells completed in well nests or vertical groundwater profiles are recommended to evaluate and define the contamination. In thick homogeneous granular aquifers, the benefits of short well screens are more limited, given that mixing and contaminant dilution will occur within the aquifer

itself (EA 2006). However, consideration should still be given to the potential for vertical gradients within thick aquifer systems.

Monitoring dissolved contaminants in plumes requires consideration of the likely plume characteristics and its behaviour in the aquifer. Plumes are typically elongated in the direction of groundwater flow and will undergo longitudinal, lateral and vertical dispersion. Plumes will also tend to sink as additional recharge is added to the aquifer downgradient of the plume source area (EA 2006).

General guidance on the selection of appropriate screen length is given in Table 5 below.

Table 5. General guide to selection of an appropriate monitoring screen length

Aquifer conditions/monitoring objectives	Screen length			
	Multi-level	Very short (<1 m)	Short (1–2 m)	Long (3+ m)
Monitor general water quality (thick aquifer)	✓✓	X	✓✓	✓✓✓
Monitor general water quality in thin or heterogeneous aquifer	X	✓	✓✓✓	✓
Monitor LNAPL (fluctuating water table aquifer)	X	X	✓✓	✓✓✓
Monitor DNAPL	X	✓	✓✓✓	✓✓
Detailed delineation of contamination	✓✓✓	✓✓	✓✓	X

Key:

X Not appropriate

✓ Appropriate but not ideal

✓✓ Appropriate

✓✓✓ Most appropriate

Adapted from EA (2006)

Correct slot size and location of well screens is particularly important when dealing with NAPLs. Representative samples of the dissolved phase can only be obtained if the screened interval is outside the influence of any mobile or residual NAPL. Interface meters and tapes with oil-indicating pastes can be used to confirm the presence and thickness of NAPLs.

Further information on sampling LNAPLs can be found in Clements et al. (2009). Further information on sampling and identification of DNAPLs can be found in Keuper and Davies (2009) and EA (2003).

8.2.3.2 Filter packs and filter socks

A filter or gravel pack is used to minimise the entry of fine-grained material into the well screen. The filter pack should be chemically inert and matched to the aquifer particle size and to the screen slot size. For further information see ASTM D5092-04.

In general, the filter pack should extend no more than 1 m above and below the well screen in the well annulus after settling, taking care not to extend the filter pack across geological units or water-bearing zones.

Geotextile wraps (filter socks) are not recommended for use in bores intended for monitoring groundwater quality. The redox conditions, and therefore biological activity, within the mesh can be

different from that present in the aquifer, which can produce misleading sample results. Where LNAPLs are present, a greater thickness of NAPL would be required to overcome the increased surface tension forces present in the fine mesh compared with the well screen.

8.2.3.3 Sealing and backfilling of boreholes

The annular space from the top of the filter pack to ground level (or next monitoring screen in multi-level monitoring wells) should be backfilled with bentonite or a non-shrinking bentonite-based grout (cement grout or a cement/bentonite mix). A seal should always be placed on top of the filter pack to prevent these materials from entering the well screen.

Incorrect installation of wells can result in costly cross-contamination of aquifers. In installations above the water table, the use of bentonite pellets in isolation to form a seal is discouraged as the bentonite pellets can set dry and crack, resulting in an ineffective seal between the aquifer and contamination near the ground surface. In these circumstances it is preferable to hydrate the bentonite at surface and then install as a slurry. For guidance on appropriate installation procedures see EA (2006) and ASTM D5092-04.

Drill cuttings should be collected in suitable containers and disposed of appropriately. It is not acceptable practice in site assessment to use drill cuttings as backfill in boreholes.

8.2.3.4 Headworks

It is essential to correctly finish all monitoring wells at the surface such as with a suitable bentonite plug and cement seal, to ensure that surface water runoff does not collect at the wellhead and leak down the outside of the casing. The borehole headworks form the interface between the borehole and the surface environment.

In designing headworks a number of issues should be considered (EA, 2006):

- security—to prevent vandalism or malicious actions and to prevent access by animals
- protection—from entry of surface water or other foreign material and from activities at surface (such as vehicle movements)
- accommodation of equipment—storage of equipment such as data loggers and dedicated sampling devices
- visibility—designed to be clearly visible or non-obtrusive depending on location.

In general, an above-ground completion is preferred as this type of design is less likely to suffer from inundation, is easier to find in the field and is more easily secured.

Casing materials such as PVC, ABS, Teflon, etc. which project from the ground can easily be damaged and should be protected by a steel or similar outer protective collar.

8.2.3.5 Well development

All bores intended for monitoring water quality should be developed after drilling to remove fine sand, silt, clay and any drilling mud residues from around the well screen to ensure the hydraulic functioning of the well. Development should be carried out as soon as possible after drilling and installation, however, a minimum of 24 hours should be allowed for bentonite seals to fully hydrate and grout to cure (harden and set). A detailed record should be kept of well development activities and reported in the relevant site assessment report.

Development usually involves agitating the water column in the well bore and pumping the water out until it runs clear. During development, bore yield should be estimated by monitoring the rate of

recovery of water in the bore after pumping. This information can then be used to select suitable methods for subsequent purging and sampling (SA EPA 2007).

Development should continue until a defined endpoint has been reached (EA, 2006), such as:

- chemical indicator stability– using field measuring techniques for pH, EC and dissolved oxygen, development is continued until these parameters stabilise in abstracted water, or
- reduced turbidity – development is continued until the abstracted water is reasonably clear and free of suspended solids.

After development, bores should be left for a period until borewater chemistry can be demonstrated to have stabilised (generally between 24 hours and seven days) before samples are collected. Longer periods are applicable to low permeability aquifers and to reduced groundwater conditions where it may take days to weeks to fully equilibrate, depending on the aquifer properties.

Care should be taken to dispose of any contaminated water responsibly and not to allow it to enter the stormwater drainage network or to impact uncontaminated soils at the site.

8.2.4 Groundwater sampling

It is essential that groundwater sampling methods result in the collection of samples that are representative of aquifer conditions. Management decisions that may involve considerable expenditure and potential inconvenience to the public will be based on these results. In many circumstances, budgeting for additional sampling and analysis costs for site characterisation for definition of groundwater contamination problems could save further assessment expenditure and costly delays to property transactions and site development.

Where possible, established 'standard methods' from recognised sources such as Standards Australia, the United States Environmental Protection Agency (US EPA), the American Public Health Association (APHA) and International Standards Organisation (ISO) should be used for the analysis of groundwaters. The general reference used by laboratories is *Standard methods for the examination of water and wastewater* (APHA et al. 2005).

Overviews of groundwater sampling procedures are readily available; for example, SA EPA (2007) includes information about:

- development of monitoring plans
- pumping and sampling equipment
- sampling methods
  - groundwater level measurement
  - purging
  - sample collection methods
  - filtration
  - NAPL sampling
  - decontamination
- sample identification, transport and storage
  - labelling and identification
  - preservation techniques
  - QA/QC.

Detailed information can be found in Standard AS/NZS 5667.11-1998; MDBC (1997); EPA Victoria (2000); Nielsen (2006) and Nielsen & Nielsen (2005).

An appropriate method of groundwater sampling should be selected in relation to the nature of the target analytes and the hydraulic characteristics of the monitoring well. In general, the use of low-flow submersible pumps or positive-displacement pumps capable of controlling flow rates and minimising purging requirements are the preferred methods of groundwater sampling for site characterisation purposes. A discussion of the benefits and limitations of low-flow purging and sampling can be found in ASTM D6771-02.

No-purge sampling techniques (see below) may also be appropriate, particularly for long-term monitoring applications. A discussion of the applications and the benefits and limitations of passive sampling can be found in ITRC (2005).

Purging and sampling methods using bailers or high speed pumps are not recommended due to the difficulty of obtaining a representative groundwater sample. These methods result in degassing of samples and can also introduce high levels of turbidity. Sampling-induced turbidity may be mitigated by using low-flow purging and sampling techniques (Puls & Barcelona 1996).

Generally, the same methods should be used each time the wells are purged and sampled to avoid introducing sampling method-related uncertainties to the analytical data (SA EPA 2007). Where an improved technique becomes available, it is recommended that it is trialled in combination with the existing sampling method to establish the nature and magnitude of any changes in analytical results as a result of the new sampling method.

Passive sampling devices (for example, passive diffusion bags for VOCs) do not require pumping or purging of groundwater to acquire a sample. These sampling devices are placed at a selected depth in the well and rely on ambient flow through the well screen for sampling.

Three types of passive sampling technologies are available:

- devices that recover a grab sample of groundwater (producing an equilibrated 'snapshot' of groundwater quality)
- devices that rely on diffusion of the analytes for the sample to reach and maintain equilibrium with the sampled medium
- devices that rely on diffusion and sorption to accumulate analytes in the sampler.

Some of these passive sampling devices are applicable to the sampling of surface waters and vapour as well as groundwater. Further information can be found in ITRC (2005) and ITRC (2007a).

The selection of the appropriate equipment for a groundwater investigation should be based on careful consideration of the attributes of the target analytes, the likely contaminant distribution, cost and logistical issues, field filtration requirements, and decontamination requirements.

8.2.4.1 Target analytes

Certain analytes are prone to effects of aeration and agitation and sampling equipment should be selected to cause minimal agitation and chemical alteration of the sample, for example, low-flow techniques are recommended for quantitative assessment of VOCs and SVOCs; bailers are not appropriate.

Information on monitoring and sampling LNAPLs using oil-water interface probes, oil indicator pastes and special bailers can be found in Clements et al. (2009).

Sampling equipment should also have negligible capacity for sorption, precipitation and oxidation of analytes of interest.

8.2.4.2 Contaminant distribution

Due to a range of chemical and/or physical characteristics, contaminants may be concentrated in certain parts of the aquifer under investigation. The sampling equipment should be capable of targeting the depth interval most likely to contain the target analytes. For example, special bailers are available for the sampling of NAPLs, while bottom-loading bailers are available for investigating DNAPLs.

8.2.4.3 Decontamination requirements

All equipment used in the sampling procedure which either enters the well bore or holds the groundwater sample should be decontaminated before and after each sample is collected. Samples of the rinsate should be included in the QA/QC program. Depending on the potential for cross-contamination between wells or within the profile of a single well, certain equipment may be relatively difficult to decontaminate and it may be necessary to opt for more simple sampling systems or to dedicate sampling equipment to a particular well or interval.

In addition to the above decontamination procedures, it is good practice to sample wells with no/minimal contamination first to minimise potential cross-contamination of samples.

8.2.4.4 Field filtration

In surface water bodies, a substantial amount of metals can be transported adsorbed to suspended particles and filtering needs to be undertaken to identify the dissolved component if quantification of dissolved metals is required.

This is much less the case in groundwater systems where particles cannot easily pass through the porous aquifer matrix. Typically filtration with a 0.45 μm filter will remove the majority of suspended particulates, however, it may be necessary to filter samples with a 0.1 μm filter to remove all suspended particulates.

For dissolved metals, in-line disposable filters (or micro-filtration syringes) are recommended to ensure that groundwater samples have minimum exposure to the atmosphere. Micro-filtration syringes are now widely available and present a viable option to filter in the field even for silty aquifers. Filtered samples should be collected in pre-prepared bottles containing sufficient acid to maintain the pH of the sample to < pH 2.

Filtration to remove sediment from groundwaters upon receipt in the laboratory is not recommended for analysis of dissolved metals unless it has been demonstrated that the analytical results are consistent regardless of whether filtering is carried out in the field or the laboratory.

Field filtration is not required for total metal analysis.

Sample filtration devices should be decontaminated between uses or discarded to prevent cross-contamination and to ensure continued effectiveness. Further information may be found in SA EPA (2007) and EPA Victoria (2000).

8.2.5 Monitoring and profiling of groundwater parameters

Some physicochemical parameters cannot be reliably measured in the laboratory as their characteristics change over a very short timescale. Parameters that should be measured in the field include pH, electrical conductivity (EC), temperature, dissolved oxygen (DO) and redox potential (Eh). If ferrous iron is one of the selected analytes, it also is best analysed in the field.

It is recommended that field parameters are measured in a flow-through cell to avoid contact between the groundwater and the atmosphere. A flow-through cell can also enable continuous measurement

and monitoring of key parameters during purging to identify when a representative sample may be obtained.

There is a wide range of equipment available for the measurement and logging of these parameters. It is important that quality assurance protocols are developed and implemented. The procedures should include the use of suitable calibration standards, where the calibration spans the anticipated range of results, and accuracy checks. Where measurements are made over a number of hours, periodic readings of appropriate reference solutions should be incorporated to ensure that the calibration is stable. Calibration procedures vary between meters and between manufacturers so it is important to follow the manufacturer's instructions for correct and accurate operation of each piece of equipment. Further information can be found in SA EPA (2007) and Sundaram et al. (2009).

8.2.6 Groundwater levels and flow direction

Groundwater level measurements are essential to determine groundwater and contaminant flow directions within aquifers and interaction with surface water bodies. These measurements can provide information on lateral and vertical head distribution and hydraulic gradients within individual aquifers and between aquifers in layered aquifer systems (EPA Victoria 2000). Long-term groundwater monitoring data provides information on temporal trends in groundwater levels (and hence flow directions and rates) due to seasonal, climatic and groundwater pumping effects (EPA Victoria 2000).

The groundwater elevation (standing water level) in a monitoring well is an expression of the hydraulic head of the aquifer unit in which the well has been screened. The standing water level should be measured relative to a permanent surveyed reference point (such as the top of the casing) before any purging or sampling takes place using a calibrated pressure transducer and/or purpose-built tape or meter. The data should be reported relative to a common datum, preferably Australian height datum. Bores installed at multiple depths within an aquifer are required to assess vertical groundwater flow direction(s).

Relative groundwater elevations within the same aquifer unit indicate the hydraulic gradient between wells and, given at least three wells spaced roughly equilaterally, a groundwater flow direction may be calculated. Where the wells are completed with long screens and/or at different relative depths within the aquifer, inconsistencies may arise if there are vertical groundwater gradients present. Groundwater flows may vary significantly at a site so it is recommended that groundwater contour maps are based on several bores monitored over a period of time to determine groundwater flow directions and variability across the site over time.

Water level measurements for a given study area preferably should be taken on the same day. High frequency monitoring may be required to quantify groundwater pumping (abstraction) and/or tidal effects. Consideration should be given to use of data loggers to identify fluctuations in groundwater levels depending on the uncertainties identified in the CSM and the assessment objectives.

The use of hydraulic head measurements in groundwater of variable density is more complicated than is the case for constant-density groundwater. Density variations can result from differences in temperature or pressure but more commonly in site assessments, these effects are caused by differences in solute concentration. Variable density is particularly relevant for sites in coastal areas where deeper wells may be screened within a saltwater wedge and shallow wells within freshwater. Water-level data obtained from wells screened within saltwater must be converted to an equivalent freshwater head to enable correct calculation of vertical and horizontal gradients and to interpret groundwater flow. Further information may be found in Post et al (2007) and Serfes (1991).

Where LNAPL is present, it will affect the groundwater elevation measured at a groundwater monitoring well. If significant amounts of LNAPL are present, groundwater level corrections are necessary and are based on the measured thickness and relative density of the product. However, due

to the uncertainties involved, corrected groundwater elevations from wells affected by LNAPL should not be used to definitively determine groundwater flow direction.

The hydraulic heads measured in wells screened in different aquifers should not be used to infer lateral groundwater flow direction at a site; however, they may be used to determine the relative hydraulic head, or potential for vertical flow between aquifers.

If vertical (downward) hydraulic gradients are present, there is the potential for a dissolved-phase contaminant plume to migrate downwards along the flow path resulting in uncontaminated water overlying sections of the contaminant plume (API 2006).

8.2.7 Groundwater velocity and hydraulic conductivity

Knowledge of aquifer hydraulic (hydrogeological) properties is important for:

- the assessment of potential migration of contaminants in groundwater
- calibration and development of numerical models
- determination of applicable groundwater remediation methods.

In particular, knowledge of the rate of groundwater flow or groundwater velocity is essential for determining the timescale in which contamination may migrate off-site or threaten a receptor. Where the nearest receptor lies some distance from the site, screening level estimates may suffice; however, where greater certainty is required (for example, presence of nearby and/or sensitive receptors) then a more precise estimation method will be required.

Groundwater velocity in a porous medium aquifer can be estimated using a modified version of the Darcy equation:

$$v=Ki/n$$

where v is the advective groundwater velocity, K is the hydraulic conductivity, i is the hydraulic gradient and n is the effective porosity of the aquifer unit. The groundwater velocity calculated by this method assumes plug flow of contaminants and ignores dispersion. In reality a proportion of contaminant mass may arrive at a monitoring point (or receptor) much more quickly than is predicted by this method.

Hydraulic gradient is generally calculated based on groundwater elevation data (groundwater flow maps). Effective porosity (the percentage of interconnected pore space) is rarely measured in site contamination assessments and typically falls in a relatively narrow range for defined lithology types, for example, 20–40% for sandstones. Hydraulic conductivity, K, may be estimated with varying accuracy by a variety of methods depending on the level of acceptable uncertainty. Commonly used methods include:

- literature approaches (screening level data only)
  - literature values based on grain size/lithology descriptions
  - hazen formula with grain size analysis
- aquifer tests
  - slug tests that provide an indication of local hydraulic conductivity at the well bore
  - pumping tests that provide information on a much larger volume of aquifer compared with slug tests
  - tracer tests (in which the travel time of a conservative anion such as chloride is monitored between two points over time to directly estimate velocity).

More information on literature-based and aquifer test methods can be found in Fetter (2001) and other standard hydrogeology textbooks.

In general, aquifer testing involves the determination of a range of hydraulic properties within an aquifer. This is accomplished by stressing the aquifer at a test well, either by the addition or removal of water (or an equivalent volume of water using a weight or 'slug') and measurement of the hydraulic response at one or more observation wells within the test area. Depending on the type of aquifer testing carried out, it is possible that groundwater monitoring wells could be used either as test wells and/or observation wells.

However, most aquifer test methods require specific well construction procedures such as screening of the full aquifer thickness. Further information on aquifer testing can be found in Standard AS 2368 (1990) and Kruseman and de Ridder (1994).

Using pumping tests to determine average hydraulic conductivity in an area of severe groundwater contamination can be undesirable (for example, where there is a risk of exacerbating DNAPL contamination) and in these circumstances less intrusive methods should be considered. Alternatively, where aquifer properties do not vary significantly, it may be possible to perform aquifer testing outside the impacted area.

8.3 Delineating groundwater contamination

8.3.1 Lateral delineation of groundwater contamination

The groundwater monitoring bore network should cover an appropriate study area to delineate the lateral extent of the contamination; define background groundwater quality, the groundwater flow system for the geological units of interest; and to assess the risk to relevant receptors. Generally the number of monitoring wells should be sufficient to define, at an appropriate scale, the lateral and vertical extent of the plume exceeding relevant assessment levels (for example, GILs, HILs, HSLs and/or site-specific risk-based criteria) and to understand any seasonal or longer-term variation in groundwater flow direction and rate of plume advance or retreat.

For large and/or complex sites with VOCs and/or SVOCs, consideration may be given to reducing uncertainty in lateral and vertical contaminant distribution by using various screening tools to identify and delineate contamination in both the vadose and saturated zones. These tools include soil vapour sampling (refer Section 9) and the membrane interface probe (MIP) and laser-induced fluorescence (LIF) tools (refer Sections 7.2.4 and 7.2.5 respectively). These tools can also be used in a reactive or adaptive field sampling program.

Although the number of bores, locations, depths and screen intervals are site-specific, groundwater site investigations require as a minimum:

- one upgradient bore to establish the quality of groundwater entering the site (one for each aquifer or geological unit of interest)
- two or three bores to monitor groundwater quality immediately downgradient and also lateral to each contaminant source (for each aquifer or geological unit of interest).

Sites with significant contamination and/or complex hydrogeology will require numerous bores at various depths to assess the lateral and vertical extent of contamination and the nature of any temporal variation.

The initial investigation bores should be:

- close to each potential contamination source
- installed with similar construction techniques to minimise sources of variation and uncertainty in the data

and, where appropriate,

- screened across the water table to locate any LNAPL and to identify contamination derived primarily from surface spills and leaching.

The number, spacing and depth of follow-up wells are site-specific considerations that should be informed by the CSM. The installation of bores without consideration of hydraulic gradient and conductivity values may result in bores being sited at improper spacings.

However, as a general guide for plumes estimated to be <200 m long, well spacing should be of the order of 20–50 m in the direction of groundwater flow and 10–20 m perpendicular to flow. Well spacing should generally be less than 10 m for the delineation of source zones.

Consideration should be given to installing one or more 'sentinel' wells to monitor the migration of an expanding or detached plume or to provide confirmation of the continued absence of contamination at a particular location. For example, in some situations it may not be possible to delineate the position of the contaminant plume front due to logistical constraints such as the presence of buildings. However, in this case an acceptable approach would be to install sentinel wells upgradient of the relevant receptors to provide an early warning of any significant plume advance. The location of the sentinel wells would ideally allow the implementation of management actions to protect the receptor if the plume were to advance significantly.

8.3.2 Vertical delineation of groundwater contamination

Source: Clements et al. (2009)

Delineation of vertical variability in groundwater chemistry is critical for risk assessment and remediation planning and reliance on too few monitoring points can lead to inaccurate estimation of contaminant distribution and behaviour.

Multiple wells may be required to adequately characterise the vertical groundwater profile and contaminant distribution. Samples obtained from short, targeted, multiple screens are more likely to be representative of the maximum concentrations present in the aquifer as they are less likely to be affected by the dilution that may occur with a longer well screen. Multiple monitoring wells should be considered where contaminant distribution is likely to be complex (for example, presence of numerous migration pathways or presence of pooled and residual NAPLs).

There are several methods available for screening multiple depths, including installing multiple wells in a small area, nesting multiple wells in the same borehole, and using a pre-fabricated bundle of multi-level wells. There are cost and technical considerations with each approach. Nested wells are cheaper to install; however, if poorly installed, cross-contamination may occur between screens. Bundled multi-level wells (consisting of multiple small diameter tubes in a bundle) can provide confidence in samples at relatively low cost. Multiple wells are typically more expensive, but provide greater confidence in monitoring results.

Consideration should be given to the potential for a 'diving plume' to develop under the influence of natural or anthropogenic recharge or in response to large scale groundwater abstraction (for example, public supply or industrial process water). The depth to which a plume will downwardly migrate in an unconfined aquifer is dependent on the recharge rate and the groundwater seepage velocity.

Generally, greater recharge rates will result in a greater magnitude of downward migration but the recharge effects will be less at higher seepage velocities (API 2006).The US EPA provides an online tool which can be used to estimate plume diving caused by recharge and assuming simplified flow in a water table aquifer.

See www.epa.gov/athens/learn2model/part-two/onsite/index.html.

8.3.3 Special considerations for DNAPLs

Source: Keuper and Davis (2009) and EA (2003)

DNAPLs are only slightly soluble in water and therefore exist in the sub-surface as a separate phase immiscible with both water and air. Common types of DNAPLs include timber treating oils such as creosote, transformer and insulating oils containing PCBs, coal tar, and a variety of chlorinated solvents such as trichloroethene (TCE) and tetrachloroethene/perchloroethylene (PCE). DNAPLs have the potential to migrate to significant depth below the water table through unconsolidated and consolidated materials and fractured bedrock, where they slowly dissolve into flowing groundwater and give rise to aqueous phase plumes.

Due to their physicochemical properties, DNAPLs migrate through the sub-surface in a very selective and tortuous manner and as a result can be challenging to investigate with traditional drilling techniques. Upon release, DNAPL will move and distribute itself into disconnected blobs and ganglia of liquid (residual DNAPL) and in connected distributions (pooled DNAPL). Residual DNAPL is found both above and below the water table within the migration pathways and typically occupies between 5% and 30% of pore space in porous media and rock fractures. Residual DNAPL is trapped by capillary forces and typically will not enter an adjacent monitoring well, even under the influence of aggressive pumping.

Pooling of DNAPL occurs above capillary barriers, typically layers, and lenses of slightly less permeable materials. Penetration through silts and clays may occur if windows are present within the layers or if the layers are penetrated by preferential pathways, for example, tree roots. The presence of dipping fractures, bedding planes, joints and faults may enable a DNAPL to continue to migrate downwards. Downward migration of chlorinated solvents may cease within a few months to a few years of release in relatively permeable media, compared with many decades for high viscosity DNAPLs such as creosote and coal tar to cease migration.

The DNAPL source zone comprises the overall rock volume of the sub-surface containing residual and/or pooled DNAPL. In addition to the DNAPL, there may be significant amounts of contaminant mass that has diffused into low permeability zones. Back diffusion of sorbed contaminant mass from the aquifer to groundwater may sustain dissolved-phase plumes for significant periods of time (decades to hundreds of years).

Above the water table, volatile DNAPLs can vaporise into air-filled pore spaces and for DNAPLs with significant vapour pressure such as chlorinated solvents, this can lead to expanded vapour-phase plumes in the unsaturated zone. Passive soil vapour surveys may be useful for delineation of DNAPL source zones particularly in situations where groundwater has been impacted by VOCs but the source has not been identified. In warm dry conditions, the persistence of some DNAPLs such as chlorinated solvents can be relatively short (months to a few years) in unsaturated media. The absence of residual and pooled NAPL in the unsaturated zone may not, therefore, be sufficient evidence to conclude that DNAPL has not migrated below the water table at the site of interest.

Determining the presence or absence of a DNAPL is an important consideration for the development of the CSM. If the presence of DNAPL is suspected, care should be taken to avoid drilling through the DNAPL and dragging or spreading pooled DNAPL beyond the current location or creating a pathway for the DNAPL.
It is now commonly accepted that direct visual observation of DNAPL does not occur at most DNAPL sites and instead, the presence of DNAPL is usually inferred from converging lines of evidence. Site-specific considerations will dictate which lines of evidence (see below) should be pursued. Care, however, should be taken to ensure that a negative response to one or more lines of evidence is not simply attributable to inadequate characterisation and an insufficient amount of data.

The site investigation methods and related interpretation techniques (lines of evidence) which can be useful for characterising DNAPL source zones include:

- visual observation in groundwater samples or drill core
- chemical saturations in soil above threshold DNAPL saturation
- chemical concentrations in soil above equilibrium partitioning threshold
- mapping of a vapour-phase plume (based on shallow soil vapour measurements) if present
- hydrophobic dye testing of DNAPL in soil or water samples or using a down-hole ribbon sampler impregnated with dye
- interpretation of groundwater concentration data from locations immediately downgradient of the suspected source zone and trends with depth and over time.

As a general 'rule of thumb', groundwater concentrations in excess of 1% effective solubility may indicate that the groundwater has come into contact with DNAPL. Values of 1% solubility concentration for various chlorinated solvents can be found in Appendix B of US EPA (2009).

Further information may be found in Keuper and Davis (2009) and references therein. A tabulation of parameters and other information that may be needed at various stages of site investigation, risk assessment and selection of management options can be found in EA (2003).

8.3.4 Attenuation of groundwater contaminants

Source: EA (2000b)

Assessors should be aware that dissolved contaminants may move at different rates not only as a result of physical processes, but also because of chemical interactions with soil and aquifer components. Attenuation processes include advection, dilution, dispersion, diffusion, sorption, degradation (biotic and abiotic) and volatilisation.

Consideration should be given to the fate of the contaminant(s) as it moves along the migration pathways. This requires that chemical, physical and biological interactions between sources and sub-surface materials are taken into account. The CSM should describe the processes that control the movement of contaminants in soil and the unsaturated and saturated zones.

Degradation can be a significant process in decreasing contaminant mass. The actual rate of biodegradation varies according to a range of factors including contaminant type, microbial populations, redox conditions, temperature and the chemical composition of aquifer materials and groundwater. Evidence for the occurrence and efficiency of degradation processes should be considered in the development of the CSM. Where modelling is undertaken, care should be taken to ensure that the biodegradation process(es) being modelled is appropriate and that realistic reaction rate constants are used.

Further information on attenuation processes and their effects can be found in EA (2000a, 2000b), ITRC (1999), ITRC (2010) and Beck and Mann (2010).

9 Vapour assessment

9.1 Introduction

This section provides an assessment framework for vapour intrusion (migration of vapours into a building) and basic requirements for measurement of volatile organic compounds (VOCs) in soil vapour, indoor air and outdoor (ambient) air. Primarily, the methods included are applicable to chronic low levels of vapour concentrations as are typically encountered in contaminated site assessments.

Vapours may be generated by biological, chemical and physical decomposition of spilled or dumped wastes. Assessment of ground gases associated with operating or closed landfills ('landfill gas') or buried putrescible wastes is beyond the scope of this guidance. Information on these applications can be found in NJDEP (2005a) and Wilson et al. (2007).

Soil vapour surveys have a wide application in the assessment of volatile contaminants, for example, they may be used when:

- assessing the presence or absence of VOC contamination
- delineating VOC contamination in soil and groundwater
- characterising VOC contamination
- identifying/differentiating between sources of VOC contamination
- assessing VOC migration pathways in groundwater
- monitoring biodegradation of contaminants
- assessing vapour intrusion risk.

An overview of vapour fate and behaviour processes relevant to VOCs can be found in Davis et al. (2004, 2009a and b). Additional information on assessing vapours in the context of human health risk assessment is provided in Schedule B4.

The assessment of vapours should be undertaken by appropriately qualified and experienced professionals.

If vapour intrusion is suspected of posing an existing or imminent threat to human health, including from inhalation exposure or risk of explosion, then immediate mitigation or management strategies should be implemented.

9.2 Vapour intrusion assessment framework

Source: API (2005), ITRC (2007a), Davis et al. (2009a) and ODEQ (2010)

9.2.1 Introduction

For the vapour intrusion pathway to be complete, there must be three components present – a source of sub-surface vapours (in soil and/or groundwater), occupied buildings or the potential for occupied buildings, and a migration route to connect them. Once the pathway is identified as being complete or potentially complete, a staged approach to assessment informed by the iterative development of a CSM is recommended—refer Section 4. For smaller sites, a single phase of work may be adequate to determine vapour intrusion potential, while larger sites can require multiple phases of vapour sampling to fully define the area of concern and accurately characterise the risks.

9.2.2 Preliminary screening

As a preliminary screening measure, the potential for a vapour intrusion risk should be considered where the Henry's law constant for a substance is greater than 10^{-5} atm/m^3/mol and its vapour pressure

is > 1 mm Hg at room temperature. In addition to these measures, a substance should be assessed as volatile if its saturated vapour concentration results in exposure concentrations that are a risk to the exposed population. Some chemicals with low Henry's law constants, or low vapour pressures, are so toxic that even a small amount that moves into the vapour phase could be enough to contribute to a risk. Hence both measures of volatility and toxicity need to be considered (Refer Schedule B4). This includes substances such as petrol, diesel, solvents and certain pesticides and PAHs.

In addition, some sites may be screened out of the assessment by the use of a lateral exclusion distance of 30 m from the sub-surface extent of the vapour source – further information on the rationale for this criterion may be found in Davis et al. (2009a). A shorter exclusion distance may be considered for petroleum hydrocarbons where there are no other volatile contaminants of concern.

To apply this criterion to a groundwater source, there should be a high degree of confidence based on field data that the dissolved phase plume is stable or shrinking in lateral extent and is not continuing to expand.
This exclusion distance is not applicable to soil or groundwater sources where:

- the source is intersected by utilities or other potential preferential pathways
- continuous low permeability cover (for example concrete) is present between the source and the nearest buildings or enclosed spaces which impedes the diffusion of oxygen into the subsurface
- conditions are present that could promote lateral migration (e.g. landfill gas production, highly layered soils).

As an investigation progresses, soil vapour sampling results should be used to inform and establish the site-specific boundaries for the area of potential vapour intrusion concern.

9.2.3 Conceptual site model

A well-developed CSM incorporating vapour risk is essential for understanding current site conditions, determining potential vapour behaviour (including possible variation in soil vapour concentration) and, as part of the DQO process, identifying data gaps and uncertainties and priorities for investigation. The general requirements for the development of CSMs and DQOs are discussed in Sections 4 and 5.

Site-specific data which may be needed for vapour intrusion pathway risk assessment includes measurement of:

- VOCs in soil vapour within the fill and/or native soils below/adjacent to existing buildings
- VOCs in groundwater beneath or adjacent to potentially affected buildings or future buildings
- VOCs in indoor air, outdoor (ambient) air, or soil
- ambient VOCs that may contribute to VOCs measured at the site
- VOCs in preferential migration pathways such as service trenches for utilities
- physical properties, such as soil moisture content, saturation porosity and grain size distribution, relevant to vapour intrusion.

Consideration of preferential vapour migration pathways is an essential part of the development of the CSM. These may intersect vapour sources or soil vapour migration routes, for example, building sumps, drains, or utility and service connections to any buildings. Natural preferential pathways may also occur, for example, tree roots or fractured bedrock where the fractures are interconnected and in

direct contact (including connection by permeable fill) with the building foundation and vapour source.

9.2.4 Multiple-lines-of-evidence approach

For the assessor to conclude that the vapour intrusion/emission pathways are unlikely to be active or to present a significant risk, multiple lines of evidence are required. This requires the assessor to present several reasoned lines of evidence as to why the pathway is considered inactive/unlikely to present a significant risk.

The following are some possible lines of evidence which may be considered (listed in no particular order):

- soil vapour spatial concentrations – sub-slab, near-slab (or crawl space) with some level of vertical profiling if appropriate
- groundwater spatial data with vertical soil vapour profiling if appropriate
- information on background outdoor and indoor sources
- building construction and operating conditions
- indoor air data and concurrent ambient air data
- comparison of vapour constituent ratios in soil vapour with crawl space/indoor air
- biodegradability of vapours and availability of oxygen.

Measurement of indoor or ambient air is the most direct approach to assessment of vapour exposure. However, indoor air sampling can be expensive if many samples over a reasonably long period are needed to obtain representative results. In homes and workplaces, gaining access can be difficult and may lead to unnecessary concern on the part of the occupants. Depending on the volatile compounds considered, ambient and indoor air results may be difficult to interpret since confounding sources of contamination (refer Section 9.3.1) may be present. Where affected by background sources, the collection of indoor or ambient air measurements may not be considered the most appropriate approach.

Soil vapour measurement is the preferred route in most situations where a vapour issue (from a subsurface source) is considered likely to exist.

In the absence of measured soil vapour concentrations, it is also possible to model the generation of vapour from soil, groundwater and non-aqueous phase liquids. This procedure adds another level of uncertainty to the process. The uncertainties associated with the use of a model should be well understood and discussed in relation to the nature of the volatile contaminants assessed. Where unresolved uncertainties or unacceptable risks are predicted by modelling vapour concentrations, direct measurement of soil vapour and/or indoor and ambient air should be obtained.

9.3 Sampling and analysis plan design

Source: API (2005), Davis et al. (2009a), ITRC (2007b), ODEQ (2010) and US EPA (2012a)

When designing an SAQP, consideration should be given to the following:

- confounding sources of VOCs and SVOCs
- degradability of vapours and potential presence of daughter compounds
- land use
- environmental factors including spatial and temporal variability issues

- reliability, representativeness, precision and accuracy of available measurement techniques
- potential for preferential migration pathways.

9.3.1 Confounding sources of VOCs and SVOCs

Indoor air sampling is the most direct method of measuring VOC exposures where the CSM has identified that vapour intrusion is a potentially complete pathway. In circumstances where very high levels of contamination are present or the contamination has a unique character, the data can provide relatively quick confirmation of vapour intrusion impacts. However, for most sites, simply detecting VOCs inside a building is not definitive evidence of vapour intrusion.

Outdoor or ambient air commonly has detectable levels of VOCs, sometimes exceeding ambient air guideline values. The largest sources of these contaminants include vehicle emissions, fuel storage facilities and emissions from commercial/industrial activities (including service stations). As outdoor air typically makes up 99% to 99.99% of indoor air, ambient VOC levels tend to represent the minimum concentrations in indoor air. Buildings can also contain interior sources of VOCs, which include building materials, paints, dry-cleaned clothes and some commercial and household cleaning products. It is therefore advisable to conduct a survey of the building interior in advance of any indoor sampling to identify potential confounding sources and eliminate them as far as practical prior to sampling and to obtain concurrent ambient air samples.

As it is often not possible to remove all interior sources of VOCs prior to sampling, indoor air results should only be used in the context of a multiple-lines-of-evidence approach. To reduce the frequency of false positives, indoor air sampling is not recommended until other information (lines of evidence) indicates a potential vapour intrusion risk.

Further information including detailed protocols for the collection of indoor air data can be found in ITRC (2007b and 2007c), NYSDOH (2006), and NJDEP (2005b).

9.3.2 Biodegradation

The concentration of petroleum hydrocarbon (such as TPH and BTEX) vapours in well-oxygenated, generally near-surface soil can be significantly reduced by biodegradation (Davis et al. 2009a, 2009c). However, this is generally not the case in less oxygenated soil such as under large areas of impermeable hardstanding or building foundations.

For petroleum hydrocarbons, the fundamentals of an approach to include an exposure reduction factor due to aerobic biodegradation are discussed in Davis et al. 2009c and included in Schedule B1. The approach is applicable to vapour sources at depths of 2 m or greater and requires the recovery of a soil vapour sample from a depth of at least 1 m below ground in close proximity to the building (or in a similar nearby soil, soil moisture and soil coverage environment). Where the building slab penetrates the ground by more than 0.3 m, then the additional depth of penetration of the slab below 0.3 m should be added to the depth at which the soil vapour sample is recovered for oxygen analysis. It is noted that the measurement of oxygen in the soil profile can be difficult and care should be taken when using this data to support biodegradation.

Halogenated hydrocarbons can also undergo biodegradation, though the process for most halogenated compounds occurs in anaerobic conditions via a number of steps that can be much slower than for the aerobic degradation of petroleum hydrocarbons. The SAQP should address potential degradation products as appropriate.

9.3.3 Undeveloped land

Assessing the potential for vapour intrusion to a future building on vacant land poses unique challenges. Some of the investigative tools of the vapour intrusion pathway (for example, indoor air

and sub-slab sampling) are not possible when there is no slab or structure present, though others (soil, soil vapour and groundwater sampling) may be able to be used with appropriate precautions or adjustments.

As for existing buildings, a multiplication factor (x10 or x100 as appropriate) due to biodegradation may be able to be applied to relevant HSLs if the proposed maximum building size can be determined with a high degree of certainty, and the exclusion/inclusion criteria listed in Davis et al. (2009c) can be fulfilled (refer Schedule B4).

9.3.4 Preferential migration pathways

If there is significant vapour migration via preferential pathways that connect a contaminant source to a building, then the measurement of contaminant concentrations in soil vapour may not be representative of vapour concentrations that would migrate into the indoor environment. Other investigative techniques (for example, vapour measurements in utilities or indoor air measurements) may provide more representative data for the evaluation of the inhalation exposure pathway in these circumstances (API, 2005).

9.3.5 Environmental factors

VOC concentrations in the environment are highly variable, and collecting sufficient data to thoroughly understand and predict their temporal and spatial distribution can be time-consuming and costly.

VOC levels in ambient air can vary greatly over time. Diurnal fluctuations occur due to changes in vehicle traffic (for example, rush-hour effects), commercial activity, and as a result of atmospheric heating and cooling cycles, air pressure changes and wind speed. These fluctuations and their impact on the data analysis can be reduced by collecting time-integrated samples. Additional information on environmental factors and their effects can be found in Davis et al. (2009a).

To compensate for these inherent uncertainties, consideration should be given to identifying and characterising the main factors that may lead to a reasonable worst-case exposure scenario. The time period selected for sample collection should be appropriate to characterise the site-specific exposure scenario.

Rates of vapour intrusion are affected by both short-term and seasonal changes in weather conditions. Changes in barometric pressure associated with the arrival of weather fronts can move gases into or out of the vadose zone. This phenomenon, known as barometric pumping, increases the rate of vapour emission as low pressure systems arrive and decreases rates when transiting to higher pressure. This effect is only of importance for soil vapour where sampling is shallow (less than 1–2 m). Wind can also enhance vapour intrusion rates by depressurising a building relative to the underlying soil, causing more vapours to enter the building from the sub-surface. Similarly, high volume air conditioning systems in buildings may affect vapour intrusion.

To account for wind and barometric pressure effects, consideration should be given to sampling during stable weather conditions and recording local barometric pressure and wind-speed data over the three days before and during the sampling event.

Variations in soil temperature result in the expansion and contraction of soil air, leading to partial exchange with the atmosphere. Hence vapour measurements may change daily and from season to season. However, temperature effects decrease with depth below ground and typically show minimal variation much below 1 m below ground. Temperature variations are not expected to have a large influence on soil vapour or indoor air concentrations unless the source is very close to the surface.

Soil moisture increases due to rainfall infiltration may inhibit gas exchange processes and, in particular, vapour movement towards the ground surface, and oxygen ingress from the atmosphere. An increase in moisture content decreases the air-filled porosity and results in lower vapour and gas

diffusion rates in the vadose zone. This is likely to be particularly the case for heavier textured (clay) soils (Davis et al. 2009a).

Sampling of soil vapour (particularly from depths shallower than 1 m to 1.5 m) directly after significant rainfall events (greater than 25 mm) should generally be avoided, unless the rainfall is representative of normal conditions. Soil vapour samples collected from depths greater than 1.5 m are unlikely to be significantly affected by rainfall events. No specific guidance on how long to wait before sampling shallow soil vapour (shallower than 1 m to 1.5 m) after a rainfall event is given as it is dependent on the soil type and other climatic conditions.

If uncertainty remains as to the potential for a rainfall event to affect the outcome of a vapour assessment, then consideration should be given to repeat sampling and measurement of soil moisture at the time of vapour sampling.

9.4 Soil vapour sampling

9.4.1 Introduction

This section provides a summary of commonly used methods for sampling and characterising soil vapour at a site and largely has been adapted from information provided in Davis et al. (2009a) and API (2005). The decision on which methods to use will be informed by consideration of the investigation objectives and analytical requirements as documented in the SAQP.

More detailed information can be found in Davis et al. (2009a), Baker et al. (2009), API (2005), NJDEP (2005a and 2005b), NYSDOH (2006), and ITRC (2007b, 2007c). Baker et al. (2009) contains discussion on the advantages and disadvantages of various sampling techniques and sampling equipment.

Sufficient sampling should be carried out to ensure that the results are representative of the site conditions and appropriate for assessing the risk to identified receptors. The following factors should be considered in the design and implementation of a soil vapour sampling program:

- *Location and number of sampling points* — the number of locations will depend on the CSM and access considerations. As a minimum for vapour intrusion assessment, samples should be collected above the maximum source concentration near or under a building (located within the 30 m screening distance – refer Section 9.2.1) and at each corner or along each side of the building (if practical).
- *Depths* — the depth of samples should be based on the CSM and take into consideration the depth of sub-surface sources, the nature of the contamination and the likely migration pathway(s). In most cases sampling should be undertaken at depths >1 m to avoid transient effects. Where shallow sources are present or where deep samples cannot be obtained, the collection of soil vapour from shallow depths (<1 m) may be appropriate; however, sampling from these depths requires justification. When installing sample equipment at (or using data from) shallow depths, the potential for aerobic degradation and potential transient influences should be considered.
- *Frequency* —Multiple sampling events are generally required to characterise and assess vapour risk particularly where (i) the data is close to (compared with the likely variation in soil vapour concentration and the precision and accuracy of the data) or above guideline values, (ii) if samples were collected from shallow depths (<1 m), and/or (iii) seasonal variations in temperature and/or soil moisture and the effects on soil vapour concentrations are not fully understood.

9.4.2 Active soil vapour sampling

Active soil vapour samples may be taken from probes installed in open ground or recovered via access holes drilled through sealed surfaces such as a driveway or parking area ('near-slab') or beneath building foundations ('sub-slab'). Sampling installations may be permanent, semi-permanent or temporary depending on access and the need to re-sample. The basic sampling approaches for soil vapour sampling include:

- point samples at specific depths in one or more lateral locations and
- vertical profiles of samples at two or more depths at one or more lateral locations ('transects').

Additional factors for consideration when designing and implementing an active soil vapour sampling program are discussed in Section 9.4.2.4.

9.4.2.1 Temporary spear probing

Spear probing (driven soil vapour probes) of soil involves driving a spear/rod into the ground to a shallow depth (for example, 1.5 m–2.0 m below ground surface), extracting a soil vapour sample for analysis of the vapours of concern and/or major gases (for example, oxygen), and withdrawal of the spear probe. The reliability of the results may be improved by using bentonite slurry to seal the area around the drive point and conducting a leak test prior to sampling. Samples should be recovered below the zone influenced by transient effects, which is likely to extend to 1 m or greater below the surface. The probe should be decontaminated before using at the next location.

Spear probing is generally used as a screening tool (as it permits a large number of locations to be sampled in a cost-effective manner) to inform a more detailed investigation of identified areas of interest.

The method can be used to collect samples from a vertical profile (from as shallow as 0.3 m) to assist in the identification of various vapour zones and to define the potential aerobic reaction zone. As a quantitative technique, spear probes can be installed and sampled in the same manner as permanent probes/samplers, however, it has the disadvantage that the results are 'snapshots' which cannot be later repeated.

Additional considerations associated with the sampling of soil vapour are noted in Section 9.4.4 in this Schedule.

9.4.2.2 Permanent multi-level probes/samplers

The installation of permanent multi-level probes/samplers for soil vapour measurement permits:

- depth profiling of vapour concentrations through the soil profile from near source to near the ground surface
- repeat sampling and monitoring over time at fixed locations.

Single depth permanent probes can either be installed at depth (close to the source) or in the shallow sub-surface (particularly where the source is shallow). Multiple depth (or multi-level) gas sampling installations may be undertaken by installing multiple sample ports at different depths (separated by a bentonite seal) within the one sampling well (API 2005; Hartman 2002), or installing separate soil vapour probes at different depths (separated by at least 0.6 m) (API 2005; NYSDOH 2006).

There is a range of methods available for installing permanent probes. The probe installation method used should be determined based on site-specific factors such as access and environmental conditions (for example, soil texture or moisture conditions that may limit the use of very narrow tubing).

A log of soil types encountered during drilling should be documented. To assist in the assessment, a soil core may also be recovered and subsampled to determine organic carbon and soil parameters (for

example, bulk density, porosity and soil moisture content) at the depths of the sampling ports, and for analysis of the soil for the chemicals of concern.

Correct sealing of the installations is essential, especially in low permeability soils. Separate installations rather than multi-level samplers may be necessary in low permeability soils to ensure a good seal is able to be achieved or where the upper sample is less than 1 m below the surface.

Sampling of permanent probes and multi-level samplers can be carried out using a range of sampling methods. Typically, permanent probes should be left for a minimum of 24 to 48 hours to equilibrate prior to sampling (DTSC 2009; NYSDOH 2006), depending on the installation method and the site conditions.

9.4.2.3 Online VOC and oxygen probes

Near-continuous measurement of total vapour (or VOCs) and oxygen concentrations is possible using online VOC and oxygen probes (Patterson & Davis 2008; Patterson et al. 1999, 2000). These can be buried at multiple depths to give near-continuous measurements of total vapour and oxygen concentrations for extended periods (months to years). Apart from providing vapour and oxygen depth profiles, the detailed information derived from these probes allows seasonal trends in vapour fluxes and other parameters such as degradation rates to be assessed.

Online VOC probes (at the time of drafting), do not directly monitor individual compounds such as benzene, but can be subsampled to obtain a gas sample which can then be analysed by conventional means for component VOC and major gas concentrations.

9.4.2.4 Factors for consideration when undertaking active soil vapour sampling

The following factors require consideration in the design and implementation of an active soil vapour sampling program:

- *Probe integrity/seal* — soil vapour probes (temporary and permanent) should be installed in a manner that ensures that ambient air is not drawn into the sampling system and that a representative soil vapour sample can be collected. This may require an additional seal around the probe using bentonite slurry and leak testing, even for temporary installations. An effective seal is particularly important in low permeability soils and for shallow probes or sub-slab probes. A number of tracer methods are available to test seal integrity – see API (2005) and ITRC (2007b).
- *Tubing type* — the tubing type should be selected to minimise false positives due to outgassing from the tubing materials. Low sorbent materials such as HDPE and nylon are generally preferred. Further information on material properties may be obtained from suppliers or manufacturers.
- *Sample volume* — sample volumes should be minimised as far as practicable to meet the requirements of the sampling/analytical method selected. A review of available studies on sample volumes by Hartman (2006) suggested that the sample volume is less important for coarse-grained soil, but in finer grained soils large volumes may be difficult to collect due to the creation of a vacuum during sampling. Large sample volumes increase the likelihood that the sample may originate from different depths and locations, hence sample volumes collected should be minimised. Near ground surface, recovering large sample volumes may result in ambient air being drawn from outside the annulus of the shaft of the probe.

- *Purge volumes* — the sample probe, tubing and equipment have an internal volume that must be purged prior to sampling to ensure that only soil vapour is sampled and that the data obtained is representative. Generally, three to four system volumes should be purged where flow rates allow and as long as the purge volume is not large. Whatever calculation is used to estimate the volume purged, this should be consistent for all sample locations. As large purge volumes can result in low pressure/vacuum conditions which may cause contaminant partitioning from the soil to soil vapour, the purge volume should be minimised as far as practicable to ensure that the sample collected is representative. Real-time gas monitoring (using a landfill gas meter to measure oxygen, carbon dioxide and methane or a PID to measure total VOCs) can be useful for assessing the effectiveness of purging (and decrease the purge volume to less than three to four system volumes).
- *Sample flow rates* — to minimise the potential for desorption of contaminants from soil to soil vapour in the sampling zone, the assessor should select a sample flow rate appropriate for the soil type. A low sample flow rate (<0.2 L/min) is important where soil vapour is collected from low permeability soil (McAlary et al. 2009); however, higher flow rates may be applicable for coarse-grained soils. Low permeability or high moisture content can induce greater suction pressures when sampling, which can make samples difficult to recover.
- *Equilibration time after installation* — the equilibration times for soil vapour sampling is highly dependent on the drilling method. Direct push methods cause minimal disturbance to soil vapour profiles and sampling may be carried out after 30 minutes, whereas 48 hours is recommended for augered installations (API, 2005).

9.4.3 Passive soil vapour sampling

Source: Davis et al. (2009a)

'Passive' soil vapour sampling or passive implant sampling refers to the burial or placement of an adsorbent or other material in the ground, which is recovered for analysis after an appropriate period of time (hours to days). It is termed passive because no gas sample is actively recovered from the soil profile. The adsorbed mass cannot be equated to a concentration because the volume of air associated with the adsorbed mass is largely unknown.

The method enables a screening level assessment of the presence of vapours in the vadose zone to identify if the vapour pathway is complete and to identify hotspot areas for further sampling using more quantitative methods. Passive samplers may be of benefit in areas where soil vapour probes cannot be installed, in areas where preferential pathways are suspected (or need to be assessed) or where very low permeability soils limit the practicality and integrity of sampling from soil vapour probes (API, 2005).

Since the sorbent can be deployed for long periods (typically 3 to 14 days), this concentrates the mass of contaminants absorbed to the sampler and enhances sensitivity. Longer exposure time does not improve sensitivity except during prolonged rain events which cause soil saturation and interrupt vapour migration in the subsurface (NJDEP, 2005a).

Passive samplers may desorb soil vapours from fine-grained layers that are otherwise not mobile, thus overestimating the amount of soil vapours that are capable of being transported into overlying zones.

More detailed information can be found in Davis et al. (2009a), Baker et al. (2009), ITRC (2007a and 2007b) and NJDEP (2005a and 2005b).

9.4.4 Flux chamber methods

Source: Davis et al. (2009a)

9.4.4.1 Introduction

A flux chamber (or a flux hood) is a device that is placed on a surface to measure vapour/gas flux (or emission rate) discharging through that surface. The surface may be open ground or be part of a building foundation such as a concrete slab.

Flux chamber methods have generally not been widely used in site assessment or considered a primary vapour intrusion assessment method due to a number of limitations and disadvantages, which are discussed in Davis et al. (2009a) and Baker et al. (2009). However, flux chamber methods may be applicable when a direct measurement of vapour flux is required and as an additional line of evidence in combination with other methods.

The technique enables direct measurement of vapour flux from the surface of the ground or building foundation, thus providing a direct estimate of the parameter of interest (rather than calculating it from sub-surface vapour distributions). Flux methods effectively integrate all sub-surface processes (for example, phase partitioning, biodegradation, preferential pathways, advective and diffusive transport), often close to the point of potential exposure.

There are two primary types of flux chamber methods: a static (closed) chamber method, and a dynamic chamber method.

9.4.4.2 Static chamber

The static chamber method requires the placement of the flux chamber on the surface of the ground or building foundation, excluding passage of air through the chamber. This allows vapours to be trapped and the stagnant chamber vapour concentration to build up over time. Active samples can be collected at discrete intervals through a time period and at the end of a time period.

9.4.4.3 Dynamic chamber

The dynamic chamber method involves the use of an inert sweep gas which is continually introduced into the chamber with an equivalent amount of gas allowed to escape. The system is allowed to reach steady-state, (assumed to be four or five chamber volumes) before the chamber is sampled. The sample can be a discrete sample or monitored continuously.

9.4.4.4 Factors for consideration when using flux methods

When designing a flux chamber sampling program the following should be considered:

- *Coverage of the area of concern* —adequate coverage of possible vapour conduits, areas of maximum source concentrations and consideration of other site-specific building features as required
- *Deployment period* — this should be adequate to address the issues of concern and, where possible, enable temporal variability to be assessed.
- *Basements* — flux chambers may not be suitable for dwellings with basements because of additional potential fluxes from the basement walls to the interior of the dwelling.
- *Sub-surface conditions* — flux monitoring provides little information about the processes that may be occurring within the vadose zone such as oxygen penetration and hydrocarbon degradation. Longer-term controls on emissions and hence potential changes in sub-surface conditions may not be detected with such a device, unless long-term near-continuous emission monitoring is undertaken.

- *Buildings* — because of the usually limited surface area of coverage, flux chambers may not measure the actual flux into a built structure, especially if there is preferential access to the structure. Also, air-movement conditions within the chamber may not reflect natural room conditions in a structure – leading to overestimation or underestimation of fluxes depending on relative pressure differentials inside and outside a chamber.

More detailed information can be found in Davis et al. (2009a), Baker et al. (2009) and Hartman (2003).

9.5 Sample collection and analysis

Source: Davis et al. (2009a)

Samples for analysis may be collected using a range of media which include sorbent tubes (charcoal or multisorbent), Summa canisters, Tedlar® bags, glass vials, and syringes. Standard operating procedures should be developed for sample collection and any variations to the procedure (for site conditions or equipment limitations) fully documented.

It may be possible to carry out a field screening assessment of the contaminants present using a PID, FID or other handheld detector, providing the instrument detection limits are sensitive enough to measure concentrations at levels relevant to health risk assessments. For field screening, soil vapour samples may be collected via a vacuum (evacuation) chamber into a Tedlar® bag or similar and the meter connected directly to the Tedlar® bag.

Commonly used active and passive collection methods are discussed in the following sections, while more detailed information can be found in Davis et al. (2009a).

9.5.1 Active methods

9.5.1.1 Sorbents

Sorbent materials, packed into tubes, typically comprise activated carbon and/or a range of multi-sorbent materials (one or more different sorbent media may be present in each sample tube). Vapour samples are collected by drawing air (using pumps) at a calibrated rate through the tube over a specified period of time. The flow rate and sampling volume are dependent on the sorbent media used, the range of target chemicals and the required limit of reporting. The reporting limit is determined by the volume of air drawn through the sample tube, the adsorbent and analytical method used, and the potential for high concentrations (requiring dilution of the sample during analysis).

Sorbent tubes have a maximum capacity which may be exceeded in circumstances where the source concentrations are high and/or the sample volume drawn through the tube is large. A control section of the tube, analysed separately to the sample section, indicates whether breakthrough has occurred (i.e. whether the capacity of the tube has been exceeded). If the tube capacity is exceeded, the reported concentration will under-represent the actual site conditions.

9.5.1.2 Canisters

Whole air samples can be collected using specially prepared canisters (Summa canisters) which are sent to the field under vacuum and certified clean and leak-free. The canister is fitted with a calibrated regulator that, when opened, allows air to be drawn into the canister over a pre-set time period at a constant flow rate. Initial and final vacuums are recorded for each canister, as well as the vacuum when received at the laboratory.

9.5.1.3 Other methods

Whole air samples can also be collected using Tedlar® bags or syringes and glass vials.

9.5.2 Passive methods

Passive methods can involve the use of a wide range of sorbent materials. These materials are available in a range of forms (badges, canisters, tubes, strips) where the collection of compounds is based on the diffusion of the compound to the surface of the sorbent material. Other samplers/systems are also available and can be used depending on the target analytes, required use and reporting limits.

The range of compounds that are commonly analysed with passive sorbents include petroleum hydrocarbons, chlorinated hydrocarbons, ammonia, aldehydes, phenols and creosols, hydrogen chloride, hydrogen fluoride and hydrogen sulphide. Other compounds can be targeted using passive methods depending on the sorbent materials/housings used, the ability to assess uptake (diffusion) rates, and analysis methods. The limit of reporting varies depending on the sampler (sorbent material) used, the analysis method and the sample time.

Passive methods are generally considered to provide a qualitative measure of concentration; however, quantitative results may be obtained under certain conditions. This is dependent on the concentration present in air, the time sampled and, for some samplers, the movement of air past the sampler. The concentration is calculated based on diffusion principles (uptake rates). The reliability of the results should be assessed as part of the DQO process.

The use of a passive sampling system, selection of appropriate sampler (to adequately address the range of compounds required), sampling time and analysis method should be considered in the design of the sampling plan.

9.5.3 Laboratory analytical methods

The analytical method(s) selected should be considered with respect to the target compounds, DQOs, the availability of analysis, and the advantages/disadvantages of each method.

Ambient air and soil vapour samples are generally analysed using methods sourced from the US EPA's *Compendium of methods for the determination of toxic organic compounds in ambient air* (TO-methods). Site assessors may wish to use alternative methods, in which case the alternative method should be at least as rigorous and reliable as the TO-methods. For further information on reference methods and alternative methods, see Schedule B3.

10 Contaminant fate and transport modelling

Source: EA (2000a)

10.1 Overview of contaminant fate and transport modelling

Risk assessments undertaken when groundwater or soil vapour contamination is present may involve the use of quantitative contaminant fate and transport models. Specific expertise and experience are required to carry out this type of modelling because of the highly complex nature of most contaminant fate and transport problems.

In the context of this guidance, a model is defined as a mathematical representation of reality in the form of equations or computer code and values of parameters. Output from this type of modelling may include travel times to receptors and concentrations of contaminants likely to reach receptors.

A model should only be used when it is clear how and why it is to be used. In deciding whether a modelling approach is appropriate, some of the questions that need to be considered are:

- What is the objective for modelling and what are its benefits? For example, a model may help in the decision-making process by quantifying the potential impact on a receptor and therefore the need to take action to protect the receptor.
- Can a model provide reliable answers? For example, the hydrogeological system may be too complex to be adequately represented by the available modelling resources, in which case the application of a model would serve no purpose.
- Is the hydrogeological system sufficiently understood to warrant the use of a model? A model should not be used as an alternative to collecting further site-specific information; however, it may be used to guide further data collection.

If the decision is taken to use a model, then the limitations and assumptions of the model selected should be assessed to determine whether it is fit for the selected purpose.

Modelling is unlikely to be appropriate where preferential migration pathways are present. These pathways may be natural features; for example, solution channels associated with karst development in limestones, weathered shear zones, and permeable geological faults, or anthropogenic in origin; backfill around foundations, backfill in trenching for buried utilities such as sewer, water, gas and electricity lines, and backfill around buried tanks and associated piping.

The key stages in developing a contaminant fate and transport model are:

- scoping study, comprising a review of existing information and consultation with relevant stakeholders to define the objectives of the study and the scope of work
- development of a CSM of the saturated and unsaturated zones and consideration of how the contaminant fate and transport processes can be represented in a model
- selection of an appropriate model based on the objectives of the study, the CSM and data availability
- construction/application of the model and comparison of model results with field data to assess model validity
- sensitivity analysis to determine which parameters have the most significant influences on the model results
- uncertainty analysis to take account of uncertainty in the conceptual model, parameter measurement and natural variability of parameters
- assessment of results and reporting, including assumptions and limitations.

The development of the CSM is a critical step and should identify and consider the relevant aspects of the flow system and the contaminant transport processes likely to be operating. In constructing the CSM, a number of assumptions regarding the system behaviour will need to be made. The assessment should consider whether the assumptions and uncertainties are important, that is, whether it is possible to adopt a relatively simple mathematical model of contaminant transport or, alternatively, whether understanding and definition of the system behaviour is so poor that development/use of a mathematical model is inappropriate, and that the first priority should be to obtain further site-specific information.

A phased approach to using mathematical models is recommended, moving from simple calculations to analytical models and, finally, to numerical models if appropriate. The quality and quantity of the data available should be taken into account when selecting the mathematical model. Where data is limited, complex models are generally not appropriate. In each case the selection of the modelling approach should be justified and appropriate to the available data and understanding of the system behaviour.

Data collection should be an iterative process and linked to the development and refinement of the CSM and the mathematical model. Site-specific data should be obtained whenever possible and, for certain parameters, site-specific data is essential. Literature values may need to be used for some parameters, and the values selected will need to be justified.

Construction/application of a model using parameter data will generally involve a calibration step whereby the model parameters are adjusted within a credible range to achieve the best fit between model results and field data. If an acceptable fit cannot be obtained in this calibration step, the appropriateness of the model, the need for further site-specific data, and the CSM should be reviewed. When completed, the model should be fully documented, including the objectives of the model, the model code used and its limitations, description of the conceptual model including all parameters used and any assumptions made, how the model was constructed and calibrated, and information on the accuracy of its predictions.

10.2 Data requirements

The quality and reliability of contaminant transport model results are dependent on the data that has been used to develop the conceptual model and to construct and refine the mathematical model. If the data is inadequate, the model results will be unreliable.

Data requirements vary at different stages in the modelling process but are dependent on the objectives, the complexity of the problem and the sophistication of the analysis. The assessor will need to determine the key parameters for which site-specific data is required and those parameters for which literature values will be acceptable. The ease of collection and relative cost of obtaining site-specific values for flow and transport parameters are summarised in Table 6 below.

In some instances, the collection of site-specific hydrogeological data may not be possible; in which case, reasonable default values should be selected based on geological records.

Rigorous scrutiny should be applied to ensure that input parameters are consistent with the geology, hydrogeology and geochemistry of the site or region modelled. Special care should be taken to ensure that values for hydraulic conductivity, contaminant load and degradation rates (if applicable) are appropriate, and that conclusions drawn on the basis of fate and transport modelling are supported by the available monitoring data.

10.3 Limitations of fate and transport modelling

Problems can arise at different stages of contaminant fate and transport modelling due to:

- poor sampling and analysis

- inadequate CSM
- inappropriate model selection
- use of inappropriate data sources (literature)
- (mis)interpretation/use of results.

Further information on generic good practice to avoid these problems can be found in EA (2000a), Middlemis (2000) and Barnett et al. (2012).

10.4 Types of model

The two main types of mathematical model are analytical models and numerical models.

Analytical models use exact solutions to equations that describe the migration of contaminants. In order to produce these exact solutions, the flow/transport equations have to be considerably simplified such that they are typically only applicable to simple flow and contaminant transport systems. Analytical models can be simple formulae, spreadsheets or sequences of calculations packaged up in a piece of software, for example, BIOSCREEN and BIOCHLOR from US EPA.

Numerical models use approximate numerical solutions to the governing equations of groundwater flow and transport. Parameter values are specified at certain points in space and time and provide a more realistic representation of the variation of parameters than is possible with analytical models. Numerical models range from relatively simple one-dimensional steady-state transport models to three-dimensional time-variant models, for example, MODFLOW from the US Geological Survey (USGS), and may consider any or all of advection, dispersion and retardation, biodegradation, multiphase flow and density-driven flow.

A summary of 3-D groundwater modelling codes can be found in Middlemis (2000). When considering using models, advice should be sought from suitably experienced persons in hydrogeology and geochemistry and the application of such models. Comprehensive information and software is available from:

- US EPA Centre for Subsurface Modelling Support
- www.epa.gov/ada/csmos/index.html#download
- USGS water.usgs.gov/software/lists/groundwater/.

A comprehensive software catalogue of a wide range of models, which includes information on the advantages and disadvantages of each type, is maintained by the International Groundwater Modelling Centre at the Colorado School of Mines (www.mines.edu/igwmc/).

Table 6. Summary of site-specific data requirements for contaminant fate and transport modelling

Parameter	Site-specific data essential	Site-specific data useful	Comments on ease and cost of obtaining site-specific data
Aquifer depth/geology	✓		Easy to obtain but data quality, reliability and cost depends on site-investigation techniques used.
Hydraulic conductivity	✓		Relatively easy to obtain but data quality depends on method used. Pump tests provide the best data but can be expensive, particularly

Parameter	Site-specific data essential	Site-specific data useful	Comments on ease and cost of obtaining site-specific data
			where contaminated water needs to be managed.
Hydraulic gradient and direction of groundwater flow/ seasonal variability	✓		Relatively easy to obtain but data quality, reliability and cost depend on number and construction of boreholes and frequency of measurement.
Porosity		✓	Intergranular porosity is inexpensive and easy to measure. Generally difficult to measure in fractured aquifers due to factors such as the presence of preferential flow paths e.g. fractures, joints, faults or caverns.
Transport porosity		✓	Difficult to measure – requires tracer test.
Bulk density		✓	Inexpensive and easy to measure.
Partition coefficient (K_d)	✓ for inorganics		Generally inexpensive and easy to measure but data quality, reliability and cost will depend on methods used.
Cation exchange capacity (CEC)		✓ for in-organics	Inexpensive and easy to measure.
Moisture content of unsaturated zone	✓		Inexpensive and easy to measure.
Total organic carbon (TOC)		✓for organics	Inexpensive and easy to measure. Representative data can be difficult to obtain in low organic carbon aquifers, in which case, reasonable default values should be selected based on geological records.
Infiltration		✓	Meteorological data is easy and relatively inexpensive to obtain.
Degradation	✓ (not for metals)		Relatively difficult and expensive to measure and requires long-term monitoring but is essential to provide confidence in outcomes.
Contaminant concentrations	✓		Cost dependent on analytical suite and number of samples.

Parameter	Site-specific data essential	Site-specific data useful	Comments on ease and cost of obtaining site-specific data
Redox conditions	✓		Cost dependent on analytical suite and number of samples (DO, pH and redox inexpensive and easy to measure).

Adapted from EA (2000a)

11 Assessment of asbestos soil contamination

11.1 Introduction

The recommended general process for assessment of site contamination, including for assessment of asbestos, is shown in Schedule A to the NEPM. The process starts with a Preliminary Site Investigation (PSI), which may lead to a Detailed Site Investigation (DSI) and/or an appropriate management strategy if required. Where remediation is required, appropriate validation should be carried out to verify the effectiveness of the measures undertaken. All soil asbestos investigation and management work should be conducted by a competent person.

A competent person in the context of asbestos and the NEPM is a person who has acquired, through training or experience and qualification, the knowledge and skills to identify, investigate and assess asbestos in the context of an environmental site assessment. This includes identifying the potential for asbestos contamination from site history information.

The site-specific assessment of sites contaminated by asbestos in soil should be aimed at describing the nature and quantity of asbestos present in sufficient detail to enable a site management plan to be developed for the current and/or proposed land use as relevant for the site. The site management plan should consider what action would be necessary in circumstances where asbestos fibres could become airborne and pose a human health risk.

This guidance is designed to be used in combination with the guidance on asbestos in Schedule B1 and with reference to *Guidelines for the Assessment, Remediation and Management of Asbestos-Contaminated Sites in Western Australia* published by the Western Australia Departments of Health in 2009 (WA DoH 2009a). The latter and related publications on asbestos, including a summary of the guidelines, which is updated annually, may be downloaded from http://www.public.health.wa.gov.au/3/1144/2/contaminated_sites.pm.

The types of asbestos referred to in this guidance include:

- bonded ACM
- fibrous asbestos, FA
- asbestos fines, AF.

These terms are defined in Schedule B1 Section 4.4.

11.2 Preliminary site investigation

As for all site assessments, the PSI should include a desktop study (including assessment of site history) and a site inspection. This should be carried out by a qualified and experienced assessor/competent person.

As noted in WA DoH (2009a), asbestos contamination needs to be identified early in the assessment process and properly handled to ensure that disturbance does not result in dissemination of asbestos contamination and hence delays and additional investigation effort.

11.2.1 Site history investigation

The site history investigation should follow the process outlined in Section 3 and include the following asbestos-specific considerations:

- an evaluation of information, including inspection of aerial photographs, to determine the likely presence of asbestos associated with

  - any remaining or demolished structures and buildings (including footprints) particularly of pre-1990 construction
  - possible disposal, burial and dumping activities
- an evaluation of information relating to fill materials on-site, particularly if they may incorporate demolition waste.

11.2.2 Site inspection

The site inspection should include a comprehensive assessment based on a grid-based walkover by a qualified and experienced assessor/competent person to determine whether visual indications of asbestos contamination are present. The assessment report should specifically comment on the presence or absence of asbestos material and the inspection method employed.

The identified areas should be surveyed in more detail (noting condition and distribution) together with any suspect locations identified as a result of the site history investigations. After noting the size and condition of fragments, all visible asbestos should be removed.

Where the site is thickly vegetated, then confidence in the visual inspection results will be lower. Where appropriate, some careful vegetation clearance may clarify the situation.
The default assumption by the assessor should be that any suspect material contains asbestos and further investigation/appropriate management action initiated.

Where confirmation is required regarding the nature of suspect material, laboratory analysis is required. This should be undertaken by a National Association of Testing Authorities Australia (NATA) (or its mutual recognition agreement partners)-accredited laboratory in accordance with Australian Standard AS 4964 – 2004: *Method for the qualitative identification of asbestos in bulk samples.*

Soil contamination by free asbestos fibres may be deduced from the site history and, if suspect material is identified in the site walkover, confirmation of the presence or absence of fibres may be determined according to AS 4964-2004. Where significant amounts of free asbestos fibres may have been exposed over time, the immediate surrounding area should also be considered contaminated. If free fibres are detected, the focus should then be on management, as there is yet no validated method of reliably estimating the concentration of free asbestos fibres in soil.

A video and/or photographic log may assist with site documentation.

11.2.3 Sampling

Sampling during a PSI is generally not recommended, since either an asbestos management strategy may be adequately defined without it or because it is evident that a DSI will be necessary. Limited sampling during the PSI may be appropriate, however, in the following circumstances:

- to confirm that suspect material contains asbestos
- to roughly delineate the extent of bonded asbestos-containing-material (bonded ACM) contamination in surface soil or fill
- to inform the sampling and analysis plan for a DSI
- to inform consideration of appropriate management options.

Issues that should be considered during preliminary assessment are described below.

11.2.3.1 Condition of asbestos materials

Bonded ACM fragments are often present as surface deposits on sites due to poor demolition and building practices. While isolated fragments in good condition across the surface of a site are usually

of low concern, surface material may present an exposure risk to airborne fibres over time from deterioration of the bonding compound through corrosive weathering, abrasion or crushing by vehicle traffic and other activities.

Bonded ACM may be able to be easily broken by hand force and be more readily crumbled when water-saturated or corroded. In a partially crumbled state, bonded ACM may be of greater concern, particularly if it is exposed at the surface and susceptible to abrasion during land use. In particular, roofing material containing asbestos may be heavily weathered, which can corrode the cement matrix and expose fibrous asbestos to the atmosphere.

Bonded ACM that can be easily crushed by hand should be considered friable and assessed for management actions accordingly. Similarly, unbonded asbestos or fibrous asbestos (FA), including loose material such as insulation products and damaged low density board (up to 70% asbestos in calcium silicate), are considered friable.

The condition of asbestos materials should be considered equivalent to the most degraded samples found in the relevant assessment area.

11.2.3.2 Condition of the soil and future uses

Generally accepted guidance for considerations of site setting and characteristics should be applied when developing the scope of an investigation and when developing management strategies and cleanup methods. Any potential for exposure of bonded ACM to an acid generating environment may be a factor that will increase the potential for release of fibres from the bonded matrix. Many Australian soils are weakly acidic, however, some sites may contain acid sulfate soils or other acidic soil conditions that can lead to faster rates of degradation. The clay and moisture content of soils is also a consideration, as these factors tend to inhibit the release of fibres by binding and damping mechanisms.

11.3 Detailed site assessment

A DSI may not be necessary although this will depend on the site-specific circumstances and the proposed remediation approach. Conservative management of presumed asbestos contamination may avoid the need for a DSI. The circumstances where a DSI would be necessary include when:

- the remediation or management approach requires asbestos contamination to be fully delineated and assessed (e.g. asbestos contamination is to be relocated and contained on-site)
- land uses are to be determined and delineated according to the extent and nature of the asbestos contamination.

A DSI may also resolve uncertain findings from the PSI, or assist in assessing the likely effectiveness of alternative remediation and management strategies.

A DSI is not necessary where there is a high degree of confidence that the asbestos contamination is confined to bonded ACM in superficial soil, i.e. the site history can be established with confidence and this clearly indicates that there is no reason to suspect buried asbestos materials and the site inspection confirms that any bonded ACM is in sound condition and only present on the surface/near surface of the site. In these circumstances the assessment can proceed directly to remediation (removal of bonded ACM fragments and ensuring that the soil surface is free of visible asbestos) and validation. However, investigation will be required if the soils at the site have been disturbed and potential asbestos-contaminated-material moved around the site or incorporated into sub-surface soils.

Unnecessary investigations should be avoided, for example, investigation for bonded ACM is not recommended below the proposed deepest excavation level during construction or likely maximum depth of disturbance for the proposed/current land use.

11.3.1 Sampling and analysis

If a DSI is undertaken, a sampling and analysis quality plan (SAQP) is required to support and inform the investigations. A site management plan, including dust management and airborne fibre monitoring, may also be required to protect the public and workers during investigation works and earthmoving/development works. Asbestos fibre and dust (as a surrogate for asbestos fibre) are of particular interest. Dust management measures should be adopted to ensure that airborne fibres remain below 0.01f/ml, which is the practical lower detection limit of the membrane filter method (enHealth 2005). Any dust-related asbestos fibre analysis should be undertaken by a NATA-accredited laboratory. For further information and guidance on dust and airborne fibre monitoring refer to WA DoH (2009a) and enHealth (2005) and relevant work, health and safety guidance such as Section 3.11 of Safe Work Australia (2011b).

The SAQP should include an appropriate CSM and DQOs based on knowledge of the site history and the continuing use and/or future use of the site as relevant. Sampling may include both large area (hand-picking/raking, mechanical screening and tilling) and localised methods (test pits, trenches and boreholes) to delineate lateral and vertical extent (refer Table 7). All methods usually start with handpicking to ensure that the site surface is free from visible asbestos material.

With regards to reliability of findings, test pits and trenches are preferred to boreholes to determine the presence or extent of any asbestos contamination, because a larger area of the subsurface is exposed during assessment and is available for visual inspection. It is therefore recommended that the SAQP places greater reliance on judgmental sampling involving test pits and trenches based on a thorough site history, rather than boreholes. Appropriately designed judgmental sampling plans can help avoid unnecessary broad area sampling. Grid sampling, however, is appropriate when asbestos contamination is widespread or is of unknown extent/location(s) at a site.

The sampling density and field procedures should be sufficient to characterise the nature and extent of contamination and to enable an appropriate management plan to be developed.

Table 7 Sampling methods for evaluating asbestos contamination

Sampling method[1]	Suitable for	Limitations
Hand-picking (emu-bob)/raking • can use rake to sample down to about 10 cm • at least two passes with 90° direction change • hand-picking can be used with care to remove surface FA material (assessment of likely free fibre release associated with it required) • % contamination calculated using 1 cm as soil depth (for hand-picking surveys) or rake teeth length (for raking) as appropriate • Final visual inspection should not detect visible asbestos	• bonded ACM and low levels of FA • surface or near-surface contamination • characterising the extent and level of contamination while reducing bonded ACM impact	• raking may only be effective in sandy soils • reduced confidence for vegetated or debris-covered areas • not suitable for deeper contamination (>10 cm)
Tilling (mechanical turning over of soil) with manual collection • pre-wet soils to control dust • at least two passes with 90° direction change • material should not be further damaged or buried by the process • rotor blade speed should be controlled to allow spotters to hand-pick revealed fragments • conducted across the entire area of suspected impact • % contamination calculated using an estimate of the average tilled depth per grid square • final visual inspection should not detect visible asbestos	• bonded ACM only • contamination to about 30 cm depth depending on rotor blade size • characterising the extent and level of contamination while reducing bonded ACM impact	• not for fibre-generating materials • limited application for deeper contamination (> 30 cm) or areas obscured by surface vegetation or debris • evaluated areas cannot usually be considered representative of other locations

[1] All methods are generally preceded by hand-picking to remove visible asbestos from the site surface. The collected material should be included in any contamination calculations.

Sampling method[1]	Suitable for	Limitations
Mechanical screening • conducted across the entire area of suspected impact, large areas should be sub-divided for assessment • large pieces may be removed by larger mesh sizes prior to final 7 mm effective mesh size • % contamination calculated using weight of bonded ACM found per given volume of soil screened from each strata/location • if percentage of small fragments is high, sampling of resulting screened stockpiles may be appropriate to ensure effectiveness of screening procedure • location of screened material should be carefully documented to permit follow-up sampling or segregation if required • final visual inspection of soil surface and segregated stockpiles should not detect visible asbestos	• minor bonded ACM • larger volumes of reasonably accessible and delineated soil contamination • characterising the extent and level of contamination while reducing bonded ACM impact	• not for fibre-generating materials or high levels of contamination • may not be suitable for compacted soils or soils with high clay content • evaluated areas cannot usually be considered representative of other locations • requires extensive management procedures to monitor and control dust and fibres
Test pits and trenches • sampling to 30 cm below likely limit of contamination or to likely maximum depth of soil disturbance • suspect asbestos materials and construction debris should be targeted	• all types of asbestos contamination but particularly bonded ACM, and FA if fibre disturbance is manageable • asbestos contamination extends below surface soils (about 30 cm) • if contamination is buried and of unknown location and depth	• shoring of exposed faces may be required to protect workers and the public from wall collapse or open excavation hazards and potential fibre release during excavation/sampling

Sampling method[1]	Suitable for	Limitations
Boreholes • sampling to 30 cm below likely limit of contamination or to likely maximum depth of soil disturbance • suspect asbestos materials and construction debris should be targeted • core diameter should be at least 15 cm	• all types of asbestos contamination • if contamination is buried and of unknown location and depth • fibre control may be more successful than for test pits and trenches	• smaller area/volume available for visual examination compared with test pits and trenches • measures may be required to protect workers and the public from potential fibre release during drilling/sampling

Adapted from WA DoH (2009)

Careful documentation of the sampling process and rationale is essential to the assessment of the findings. A summary of the findings annotated on a suitable site inspection plan can be helpful. Documentation should include:

- nature and condition of bonded ACM fragments and whether associated with potential free fibres
- range and average size of fragments in each affected unit
- description of affected fill/soil unit(s).
- location and depth of samples taken for analysis
- location/direction of photographs/videos
- relevant details of equipment/machinery used (e.g. rake teeth length, rotor blade size, screen/sieve size(s).

11.3.2 Assessing concentration and distribution of asbestos in soil

Bonded ACM is the most common and the most readily quantifiable form of asbestos soil contamination due to its ease of visual detection. Gravimetric assessment of bonded ACM is the recommended measure for total asbestos contamination where FA and AF (derived from bonded ACM only) are not likely to be significant as established by the PSI including the site inspection (as a guide, this may be taken to be where FA and AF together are likely to make up less than 10% of the total amount of asbestos present).

For sites contaminated with bonded ACM only (i.e. no insulation materials or other non-bonded asbestos products), assessment for free fibres is only warranted where greater than 10% of the total bonded ACM is significantly damaged i.e. present as small pieces less than 7 mm x 7 mm or can be crushed/crumbled with hand pressure.

Guidance on calculating asbestos concentration in soil is provided in Section 4.10 in Schedule B1.

For large area methods (hand-picking/raking, tilling and mechanical screening), the weight of bonded ACM and its condition should be recorded for each grid area or location evaluated. A grid area of up to 10 m x 10 m is generally reasonable when large surface areas are impacted; however, non-impacted soils should be excluded from calculations to avoid dilution effects.

For localised methods (test pits, trenches and boreholes) samples should be taken from each relevant stratum in one wall of test pits and trenches or each relevant stratum for boreholes at each sampling location, and additional samples from suspect spots.

Bonded ACM and FA samples from test pits, trenches and boreholes

- sampling should be conducted to 30 cm below the likely limit of potential contamination or to the likely maximum depth of disturbance (large sites may be split into sub-areas for sampling purposes, the rationale should be included in the assessment report)
- at least one minimum 10 L sample from each relevant stratum (or per 1 m depth for thick units) for test pits, trenches and core from boreholes and additional samples from any suspect spots
- individual samples (minimum of 10 L) should be manually screened on-site through a 7 mm sieve and the material retained on the sieve examined for any bonded ACM and/or suspect material
- for heavy soils (e.g. clay soils), the samples may need to be gently disaggregated by washing and spread out on a suitable contrasting colour material to determine the amount of bonded ACM present
- if visible FA material is present or suspected, the soil should be wetted to minimise the release of fibres and the sample spread out for inspection on a contrasting colour material, to identify suspect material
- identified bonded ACM and FA (and suspect materials assumed to contain asbestos) should be weighed for each sample and documented to assist with calculating asbestos soil concentration as described in Schedule B1
- if suspect materials are found (which are suspected of containing asbestos), representative samples (e.g. 1 in 10 of similar materials, the number analysed should take into account the variation in appearance and form) should be forwarded for laboratory analysis in accordance with AS 4964-2004. Alternatively the suspect materials may be assumed to contain asbestos.

AF-related sampling

Quantification of bonded ACM may be used as a surrogate measure for AF in certain circumstances, refer to the discussion at the start of this Section.

If sampling for AF is necessary:

- sampling should be conducted to 30 cm below the likely limit of potential contamination or to the likely maximum depth of disturbance (large sites may be split into sub-areas for sampling purposes, the rationale should be included in the assessment report)
- at least one wetted 500 ml sample from each relevant stratum (or per 1 m depth for thick units) for test pits, trenches and core from boreholes and additional samples from suspect spots should be submitted for laboratory analysis. The rationale for this sample size is discussed in Section 4.10 of Schedule B1.

Additional information on recommended practice for carrying out gravimetric analysis can be found in WA DoH (2009a) and the annual summary/update of the guidance document (WA DoH 2012) available from the WA DoH website:
http://www.public.health.wa.gov.au/3/1144/2/contaminated_sites.pm.

12 Assessment of dioxins and dioxin-like compounds

Laboratory analysis for dioxins is only recommended when the site history clearly indicates that dioxins are likely to be present as a by-product resulting from specific manufacturing and industrial activities, or from waste disposal. Dioxin contamination may be present following long-term and large-scale use of a site for the following activities:

- manufacture and waste disposal associated with certain chlorinated compounds, for example, PCBs, phenoxy herbicides, organochlorine pesticides, chlorinated benzenes, chlorinated aliphatic compounds, chlorinated catalysts, and halogenated diphenyl ethers
- bleach pulp and paper mill processes known to produce dioxin
- incineration of substantial chlorinated compounds
- former municipal solid waste incinerators
- hospital waste incinerators
- extensive use of pentachlorophenol (PCP) in timber treatment.

Where dioxins are detected at levels significantly above background, a site-specific assessment will be required to determine the appropriate action (refer to Schedule B4 for further information).

Further background information on dioxins is provided in Appendix E.

13 Data analysis

13.1 Data quality assessment

Prior to carrying out any processing or statistical analysis of the data set, an evaluation of the data quality should be carried out. As a minimum, this should include:

- checks on the completeness of the data as specified in the DQOs (all sample locations, sample depths etc. reported)
- checks on the accuracy of the reported data (all samples are correctly identified by location, depth, type etc.)
- identification of any obviously anomalous results such as elevated levels that are unexpected given the CSM and field notes on sampling (indicating a possible labelling or laboratory error)
- identification of invalid data (for example where the field or laboratory record indicates that sample integrity may have been compromised).

The possible reasons for anomalous data results (also see section 13.2.2) should be investigated and sampling and analysis repeated if appropriate.

Further information is provided in Appendix C and US EPA (2006a).

13.2 Statistical analysis

13.2.1 General

Detailed guidance on statistical procedures is beyond the scope of this guidance but some general considerations are outlined below. It is the responsibility of the site assessor to ensure that appropriate statistical procedures are followed when comparing site data with the investigation and screening levels listed in Schedule B1 and any site-specific assessment levels.

Evaluation of appropriate summary statistics and graphical displays of the sample data set are recommended for developing an improved understanding of contaminant distribution(s) and to determine whether any investigation and/or screening values have been exceeded.

Many spreadsheet and statistical software packages provide graphical methods, for example boxplots and histograms/frequency distributions, which are suitable for displaying site data. These displays can provide insight into the distribution of the data such as multi-modal, normal, log normal or exponential, which is a necessary precursor for selecting an appropriate statistical approach.

Evaluation of graphical displays such as frequency distributions can also assist the assessor in determining whether the data set should be split up into 'domains of interest' in which there is confidence that homogenous populations of data exist (that is, uni-modal and not bi-or multi-modal distribution) and for which sufficient data for meaningful statistical analysis is available.

Given that much of the sampling in contaminated site assessments is judgemental rather than random, caution needs to be taken when applying conventional (parametric) statistical methods which assume a normal (including log normal) distribution. Non-parametric methods (which do not make the assumption that data is normally distributed) provide an alternative approach for data assessment and may be useful in the early stages of a site assessment when typically there is little data available.

Non-parametric methods rely on 'rank' or 'order' statistics that are simply the percentiles of a distribution. Rather than using the mean to describe the centre of the distribution, non-parametric approaches more commonly use the median or 50th percentile. The difference between the upper quartile (75th percentile) and the lower quartile (25th percentile) is called the 'interquartile range' and is

the non-parametric equivalent to the standard deviation for describing the spread of the data about the mean. Boxplots display these key non-parametric statistics and the 'whiskers' show the minimum and maximum of the data. Some software applications also show the position of the arithmetic mean (although this is not a percentile based statistic).

For multiple analytes, the range of concentrations and statistical distribution of results for each assessment area can be presented as in Table 8. Summary statistics should be provided for each soil unit/stratum tested and according to assessment sub-areas or domains of interest, if applicable and where sample size permits.

Table 8: Summary statistics for multiple analytes and assessment areas

Chemical name	**XXX**	
Investigation Level: Number of samples:		
Minimum: Maximum: Inter-quartile (25^{th} – 75^{th} percentile) range:		
Median (50^{th} percentile):		
Arithmetic mean:		
Arithmetic standard deviation:		
Geometric mean:		
Geometric standard deviation:		
95% Upper Confidence level (UCL)		
Frequency distribution[a]	**Number**	**%**
Less than investigation level:		
$\geq$ 1 and < 2 times investigation level:		
$\geq$2 and <5 times investigation level:		
$\geq$5 and < 10 times investigation level:		
$\geq$10 times investigation level:		

a: An arbitrary method used to categorise data.

Maximum observed contaminant concentration—This generally provides a conservative assessment of exposure because if estimated risks from the maximum concentrations are not of concern, then the site should be suitable for the land use scenario(s) considered. However, a maximum concentration may not be representative of the source as a whole and may result in an overestimation or underestimation of risk if the data is extremely limited.

Mean concentration — The mean contaminant concentration can be a suitable metric provided that it can be shown that it adequately represents the source being considered. It is important that small areas of high concentrations or hotspots are not ignored by averaging with lower values from other parts of the site. The mean value may be more representative of the source as a whole than the maximum, and may provide a better estimation of the actual concentration that a population would be exposed to over a period of time.

The *95% upper confidence limit (UCL) of the arithmetic mean* contaminant concentration provides a 95% confidence level that the true population mean will be less than, or equal to this value. The 95% UCL is a useful mechanism to account for uncertainty in whether the data set is large enough for the mean to provide a reliable measure of central tendency. Note that small data sets result in higher 95% UCLs.

The procedure for calculating the 95% UCL of the arithmetic mean and also of the mean for a log normal distribution is provided in NSW EPA (1995).

Further information on understanding data distributions and statistical procedures can be found in Gilbert (1987), US EPA (2006b) and in the ProUCL user guide US EPA (2007a).

13.2.2 Censored data

Source US EPA (2006b)

Data generated from chemical analysis may fall below the limit of detection (LOD) or limit of reporting, (LOR) of the analytical procedure. These measurement data are generally described as 'non-detects' rather than 'zero' or 'not present' and the appropriate limit of detection for the analytical procedure should be reported. Data that includes both numerical data and 'non-detect' results is referred to as censored data in statistical literature.

Where the approximate percentage of non-detects is less than 15% of the relevant data set, then substitution of the LOD/2 or the LOD may be satisfactory, depending on the purpose of the analysis (US EPA 2006b). More detailed adjustments may be appropriate for where more than 15% of the relevant data set is below detection limits. In addition, sample size influences which procedure should be used to evaluate data. For example, the case where 1 sample out of 4 is not detected should be treated differently from the case where 25 samples out of 100 are non-detects. Further information can be found in US EPA (2006b) and US EPA (2007a).

Although substitution methods (such as replacing with the LOD/2 or the LOD) are reported widely in the literature for analysing data with non-detects, these approaches result in bias of the summary statistics calculated from the adjusted data set. The US EPA ProUCL software package provides alternative methods for calculating summary statistics such as the mean which do not rely on substitution methods (US EPA 2007a).

13.2.3 Outliers

Adapted from US EPA (2006b) and BC Environment (2001)

Potential outliers are measurements that are extremely large or small relative to the rest of the data and therefore are suspected of misrepresenting the population from which they were collected (US EPA 2006b). Outliers may result from:

- transcription errors
- data-coding errors
- measurement problems

- true extreme values (hotspots).

Graphical displays of data, for example probability plots (concentration plotted against cumulative frequency), and x-y scatter plots (for example, ratios of contaminants expected to be associated with each other), can assist with identifying outliers. Evaluation of a combination of graphical displays with reference to relevant site layout diagrams is recommended.

It can be tempting to dismiss unexpectedly high values as 'outliers'; however, this is not good practice, as a more thorough examination of the reasons for these unexpected values may lead to new insights into the data (such as the presence of an unsuspected hotspot of contamination) or to reconsideration of underlying assumptions about the data and its distribution.

Potential outliers should be checked for human error due to transcription/data-coding errors and invalid measurements from malfunctioning equipment. The former may be corrected whereas the latter can properly be discarded. Following the procedure outlined in Section 13.1 should minimise the impact of outliers from these causes.

If an outlier is not due to human error, then consider the available qualitative information regarding the data provenance and the site history and discard the outlier only if there is documentation to support the belief that the outlier is not part of the population under study. In all such cases, describe the population that the outlier belongs to and justify why this population is not considered relevant to the study objectives (e.g. elevated PAH due to presence of road bitumen fragments as opposed to contamination in soil derived from fuel leaking from an above-ground storage tank).

Discarding an outlier from a data set should be done with extreme caution as environmental datasets often include legitimate extreme values (US EPA 2006b). The decision taken should be based on scientific reasoning and be fully documented. Repeat sampling close (<1 m) to the original location may provide greater certainty in the decision process.

US EPA (2006b) describes several statistical tests for determining whether or not one or more observations are statistical outliers.

14 Report presentation

14.1 Introduction

An efficient and accurate appraisal of a site requires that the data be collated in a form, or 'model' that facilitates understanding of the location, extent, trends, and likely 'behaviour' of any contamination. An adequate understanding of what is occurring on a site is almost impossible to achieve from pages of raw data, especially where there are abnormal results or more than a handful of results. At its worst, sample identification numbers, sampling points, geotechnical logs, and results for each analyte will be on separate pages.

A uniform approach to the location and presentation of data makes for more rapid and accurate assessments of reports.

The major problems that can occur with data sets and assessments are:

- a failure to collate data and to condense it into logical and comprehensible tables
- cluttered data sets, tables and graphs
- treating the sum of the data as somewhat greater than the sum of its parts.

This is exemplified by:

- over-elaborate contour maps (some can be useful) based on a very limited number of data points which are not annotated on the map
- providing definitive conclusions unsupported by the data
- considering the numbers in isolation from other data important to interpretation, for example, site history and soil characteristics.

14.2 General requirements

Reports should preferably be printed on A4 size paper, with durable covers and binding which allows for easy opening. Photographs and figures should be of high quality and adequately display the points of interest. Tables and figures should be formatted to enable easy reading (font size can be a particular issue when displaying large amounts of data) and printed as foldouts or enclosures where appropriate. Where there is a series of site reports, each succeeding report should summarise the important and relevant portions from the preceding reports. This will assist in the rapid comprehension of new material by all parties involved.

Reports should follow appropriate subject headings and be structured in a logical way.

To support the site history investigation, copies of all current and old site layout plans, diagrams, correspondence, photographs, permits, etc. should be included in appendices. Where the site history is complicated because of numerous past uses and/or occupiers, information may be more effectively presented as a table or time line. An example is provided in Section 3.

A discussion of assumptions made in relation to the assessment, including those related to sampling density, sample locations, choice of analytes, off-site impacts and potential groundwater contamination, should be made.

Reports should also include the assessor’s opinion and conclusions relating to the environmental condition of the site, as well as recommendations for any further assessment of site contamination or site work the assessor considers necessary.

14.3 Graphics overview

For all but the most simple of sites, some form of graphical representation is imperative for the assessor and other relevant parties to accurately visualise the site. Without such representations, inaccurate (and probably costly) decisions may be made. For large and complex sites, 3-D visualisation software may also be useful to illustrate the distribution of contamination etc. Graphics should be well designed to promote understanding of the data. Some basic principles of graphic representation are given in Table 9.

Example graphics can be found in Appendix D.

Table 9. Helpful vs unhelpful graphics

Helpful	Unhelpful
No cryptic abbreviations No elaborate encoding	Numerous abbreviations requiring searching the text for explanation
Words run in natural left to right direction	Words run vertically or in several directions
Brief text messages explain data	Understanding graphic requires repeated references to text
No elaborate shading, cross hatching or overpowering colouring	Elaborate or obscurely coded patterns requiring continual return to legend or key
Simple, upper-and-lower case font	Multiple overbearing fonts
Clearly printed	Murky and clotted printing
Enlightens and arouses curiosity	Graphic repels interest and obscures meaning

(adapted from Langley 1993 and Tufte 1983)

14.4 Site plans

Site plans should be drawn to a scale appropriate to the size of the project and the level of detail required. Drawings on A3 or larger paper as foldouts or enclosures may be necessary. Plans should show:

- a north-facing arrow
- scale
- lot boundaries
- location of present and former infrastructure and site activities
- distribution of fill types
- locations of affected vegetation, stains, odours, chemical containers, etc.
- direction of surface run-off and drainage
- presence of above and below ground services
- areas covered by an impermeable seal (e.g. concrete, bitumen and buildings).

In some situations, it may be necessary to show previous site layouts as overlays over the current layout and perhaps have another overlay of sample locations or show sample excavation boundaries (see Appendix D).

Figures showing topographical contours in relation to site features and sample locations can assist with the assessment of sites with varied topography/changes of level.

14.5 Presentation of contamination data

Sample locations, identification numbers, results and depths should be plotted on one or more site layout figures. Sites with a large number of sample locations and numerous elevated results can be difficult to fully comprehend and time-consuming to assess. Therefore, to minimise assessment times and to allow, at a glance, a clear representation of contamination issues associated with the site, site plans should be used to display sample results. For large and complex sites, 3-D visualisation software may also be useful.

Contoured figures and/or maps can be useful for illustrating the distribution and trends of contamination, however, the interpolation methods used, for example, kriging, regression, minimum curvature, etc. can influence the results. For this reason, contours should be interpreted with caution and figures should include labelled data points for clarity.

If there is 'too much' data available, this may be addressed by displaying only significant results on the map. However, this should be done cautiously as censoring some of the data can obscure trends. 'Normal' results can be important if elevated results were anticipated and may need to be displayed.

An alternative method is to display a subset of the data e.g. separate figures for metals and petroleum hydrocarbons or provide some form of surrogate measure of where contamination may occur on a site. A series of figures, each with a different analyte, can be useful in this situation.

The following techniques may be useful to clearly display results:

- a separate site plan for each elevated analyte, which displays sample locations, sample identification numbers and depths, and shows different concentration ranges in different colours
- a separate site plan displaying analyte results (including locations, identification numbers and depths) for each elevated analyte, highlighting any exceedences of the guidelines by concentration range.
- a site plan displaying all analytes tested at each depth at each location and highlighting all results above environmental investigation thresholds in one colour and all results above health investigation thresholds in another colour (same colour regardless of analyte)
- a site plan displaying all results at each depth at each location in a specific colour for each analyte
- concentration contours, for each specific sample depth, to show plumes from a point source. Care should be taken when using this technique because inferred areas may be misleading if only a small number of sample locations are used
- cross-sections (noting vertical exaggeration) to illustrate the distribution and concentrations of contaminants and to display complex local geology
- statistical diagrams such as histograms and side-by-side boxplots.

It may be necessary to provide separate site plans for various depth ranges if plots are cluttered.

A particular technique will not be suitable in every situation. For example, choosing the third point above would not be useful if the majority of sampling results were above investigation levels. In this situation, a technique which showed concentration ranges in different colours would be more applicable.

To assist report assessors to review a site, a blank site plan which shows only sample locations, identification numbers and depths should also be provided. Examples of appropriate data presentation on site drawings are shown in Appendix D.

A separate site plan for validation samples should always be provided which clearly displays locations, depths and results of all relevant samples, including samples from Stage 1 and 2 reports.

14.6 Presentation of tabulated laboratory analytical results

Summary tables should show at least the essential details of sample locations and depths against the laboratory results. Results exceeding investigation threshold levels should be highlighted. For ease of reference, the addition of information such as date sampled, date received at laboratory, date analysed, and soil profile data to the summary table can expedite assessments by reducing cross-referencing. Examples are shown in Table 10 to Table 14.

Copies of the analytical results as originally received from the laboratory should be included as an appendix to the report together with details of relevant QA protocols, and QC results and chain-of-custody documentation. Further information is provided in Appendix C.

14.7 Presentation of bore logs

Bore logs and test pit logs are necessary to provide accurate descriptions of soil types encountered throughout the profile and should clearly distinguish natural soils from fill. Sample locations and perched water and groundwater levels should be shown. If rubble or rubbish is encountered, the percentages of each type of foreign matter should be estimated. Soil profile information may be presented as an appendix or used to construct cross-sectional drawings of the site. Presentation of the locations of odours, stains and field test measurements on the logs would assist with the site assessment. Bore logs are also to be used to represent the construction of monitoring wells. Refer also to Section 7.3 and Section 8.2.2.

Examples of bore, monitoring well and test pit logs are shown in Appendix D.

14.8 Photography

A photographic record that is well labelled for date, location and orientation is a valuable reference tool for items such as the site inspection (for example, topography, soil staining, state of underground storage tanks when removed, visual signs of plant toxicity), and the strata present in test pits and soil cores. It may also be useful for recording the appearance of split samples, particularly any visible heterogeneity in the field.

14.9 QA/QC documentation

The following QA/QC documentation should be included (but not limited to):

- disposal dockets and receipts issued when contaminated soil and fuel tanks or other structures are removed from the site
- validation of any 'clean fill' used at the site
- certificates of clearance for asbestos removal or remediation clearance
- QA/QC protocols for field and laboratory work
- calibration reports for all field monitoring equipment

- chain-of-custody documents for all soil, vapour, groundwater and surface water samples and laboratory receipt notices.

Further information is provided in Appendix C.

14.10 Electronic data

Consultants, assessors and government agencies should have access to electronic data (such as site data in spreadsheet form) as it avoids a further source of transcription error and facilitates the further analysis of data using other software packages.

Users of data should be aware of copyright, data protection and data integrity issues.

Table 10. Example report structure for soil analytical results

Soil Bore	Depth	As	Cd	Co	Cu	Hg (inorganic)	Ni	Pb	Zn
	mm	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
A7	0-50	3	<0.05	4	28	0.25	14	200	210
A7	150-300	3	<0.05	6	100	0.25	15	170	220
A7	300-450	<0.05	<0.05	8	20	<0.05	20	10	34
A8	0-50	2	<0.05	18	8.0	0.50	75	36	24
A8	150-300	2	<0.05	12	28	0.05	28	<0.05	46
A9	0-50	3	<0.05	4	50	0.55	15	250	310
A9	150-300	2	<0.05	5	60	0.40	13	160	240
A10	0-50	<0.05	<0.05	<0.05	1.0	<0.05	8	10	16
A10	150-300	5	1	5	1.8	0.05	13	24	34
A10	300-450	3	1	7	1.8	<0.05	15	12	30
A10	750-900	<0.05	1	6	1.5	<0.05	14	4	22
A11	0-50	5	<0.05	4	24	0.10	11	290	540
A11	150-300	10	<0.05	5	1750	0.70	15	**450**	760
A11	300-450	5	<0.05	9	1.9	0.05	17	90	30
A12	0-50	3	2	6	28	0.25	15	100	80
A12	150-300	5	<0.05	7	60	2.70	18	**940**	190
A12	300-450	1	<0.05	12	26	0.20	24	46	46
	HIL A	**100**	**20**	**100**	**6000**	**40**	**400**	**300**	**7400**

HIL A = Health investigation level for standard residential use

BOLD font indicates result exceeds the relevant HIL

Table 11. Example tabulation of analytical results against geological profiles *to illustrate* correlation between contamination and particular fill types

Bore/ test pit	Depth (m)	Description	Sample depth (m)	Analysis results in mg/kg																	Sample date	Date to lab	Analysis date (organic)	Analysis Date (inorganic)
				C_6-C_9	C_{10}-C_{14}	C_{15}-C_{28}	C_{29}-C_{36}	B	T	E	X	Total PAH	As	Cd	Cr	Cu	Pb	Zn	Ni	Hg				
TP1/1	0.0-0.1	Silty sand, brown, damp, loose, fine sand	0.0-0.2	1500	2240	1200	<100	<1	<1	<1	<1	<5	66	<1	8	312	209	310	97	<0.05	27/05/97	28/05/97	28/05/97	29/05/97
/2	0.1-3.55	Gravelly silt sand, dark grey red, loose, fine to coarse sand, ASH FILL	0.3-0.5	1000	1900	1100	<100	<1	<1	<1	<1	11	45	4	8	269	307	274	85	<0.05	27/05/97	28/05/97	28/05/97	29/05/97
/3		bricks and steel throughout	0.85-1.05	700	59	900	<100	<1	<1	<1	<1	8	32	5	5	211	253	213	69	<0.05	16/09/97	17/09/97	18/09/97	18/09/97
/4	3.55-3.75	Clay, olive grey, moist, soft, plastic	3.55-3.75	50	<20	200	<100	<1	<1	<1	<1	<5	1	<1	1	82	21	20	62	<0.05	16/09/97	17/09/97	18/09/97	18/09/97
TP2/1	0.0-0.3	Sandy silt, brown, dry, loose, soft, non-plastic	0.0-0.2	60	130	1200	1500	9	5	8	11	30	22	<1	64	100	541	450	27	0.05	27/05/97	28/05/97	28/05/97	28/05/97
/2	0.3-0.5	Silty sand, black, dry, loose, fine to coarse sand, ASH FILL	0.3-0.5	<20	110	700	<100	3	2	<1	5	22	34	3	4	184	400	533	22	<0.05	27/05/97	28/05/97	28/05/97	28/05/97
/3	0.5-1.0	Clay, brown, dry, hard, plastic	0.5-1.0	<20	<20	<50	<100	<1	<1	<1	2	7	<1	<1	<5	52	30	142	23	<0.05	27/05/97	28/05/97	28/05/97	28/05/97
TP3/1	0.0-0.3	Gravelly silty sand, black, loose, damp, fine to coarse sand, ASH FILL	0.0-0.3	<20	<20	<50	<100	<1	<1	<1	<1	9	17	6	1	115	218	264	23	<0.05	27/05/97	28/05/97	28/05/97	29/05/97
/2			0.3-0.5	<20	<20	<50	<100	<1	<1	<1	<1	<5	12	2	15	88	123	425	23	<0.05	27/05/97	28/05/97	28/05/97	29/05/97
/3	0.3-1.0	Silty clay, brown, damp, soft, non-plastic clay and silt	0.5-1.0	<20	<20	<50	<100	<1	<1	<1	<1	<5	1	<1	16	35	25	166	19	<0.05	16/09/97	17/09/97	19/09/97	18/09/97
TP4/1	0.0-0.5	Silty sand, brown, dry, loose, fine sand	0.0-0.2	1200	224	1200	1000	27	15	17	25	<5	15	2	12	45	900	540	15	<0.05	16/09/97	17/09/97	19/09/97	22/09/97
/2	0.5-2.2	Gravelly silty sand, grey, dry, loose, fine to coarse sand, ASH FILL	0.2-0.5	600	220	1300	900	19	9	12	19	13	23	<1	75	209	1000	560	13	<0.05	16/09/97	17/09/97	19/09/97	22/09/97
/3			0.5-1.0	300	230	1350	875	11	4	8	13	<5	34	5	92	75	1200	230	14	<0.05	16/09/97	17/09/97	19/09/97	22/09/97
/4	2.3+	Clay, brown, damp, moderately soft, plastic	2.3-2.5	105	127	760	716	<1	<1	<1	2	<5	18	<1	65	38	45	150	11	<0.05	16/09/97	17/09/97	19/09/97	22/09/97
/5			2.5-3.0	<20	<20	<50	<100	<1	<1	<1	2	<5	4	<1	34	19	36	68	5	<0.05	3/11/97	4/11/97	5/11/97	5/11/97
TP5/1	0.0-0.2	Gravelly silty sand, black, dry, loose, fine to coarse sand, ASH FILL	0.0-0.2	110	95	500	1400	2	1	<1	3	26	18	4	75	187	640	150	43	<0.05	27/05/97	28/05/97	29/05/97	29/05/97
/2			0.2-0.5	105	71	<50	400	1	1	1	2	19	1	5	46	95	500	199	29	<0.05	27/05/97	28/05/97	29/05/97	29/05/97
/3	1.2+	Clay brown/reddish brown, damp, soft, plastic IN SITU	1.2-1.5	<20	<20	<50	<100	<1	<1	<1	<1	<5	<1	8	87	25	23	35	35	<0.05	27/05/97	28/05/97	29/05/97	29/05/97
BH1/1	0.0-0.2	Silty sand, brown, damp, loose, fine sand	0.0-0.2	<20	<20	<50	<100	<1	<1	<1	<1	<5	43	2	25	15	125	55	16	<0.05	16/09/97	17/09/97	19/09/97	22/09/97

Bore/ test pit	Depth (m)	Description	Sample depth (m)	Analysis results in mg/kg																	Sample date	Date to lab	Analysis date (organic)	Analysis Date (inorganic)
/2	0.2-0.45	Silty sand, black, dry, loose, fine to coarse sand, ASH FILL	0.2-0.45	<20	<20	<50	<100	<1	<1	<1	<1	<5	25	3	4	62	119	171	89	<0.05	16/09/97	17/09/97	19/09/97	22/09/97
/3	0.45-1.0	Silty clay, brown, damp, soft, non-plastic clay and silt	0.45-1.0	<20	<20	<50	<100	<1	<1	<1	<1	<5	7	<1	8	19	104	25	15	<0.05	16/09/97	17/09/97	19/09/97	22/09/97
/4	1.0-1.3	Clay, brown, dry, hard, plastic	1.0-1.3	<20	<20	<50	<100	<1	<1	<1	<1	<5	6	<1	18	15	31	32	25	<0.05	16/09/97	17/09/97	19/09/97	22/09/97

Table 12. Example tabulation of field observations against soil profiles

Bore / test pit	Location	Depth (m)	Description	Remarks	PID Readings	Sample depth (m)
TP1/1	Bowser	0.0-0.1	Silty sand, brown, damp, loose, fine sand	surface staining	100	0.0-0.2
/2		0.1-0.65	Gravelly silt sand, dark grey red, loose, fine to coarse sand, FILL	no odour	<5	0.2-0.5
/3		0.65-1.0	Clay, medium brown, soft, plastic	slight odour	10	0.5-0.7
TP2/1	Triple interceptor tank	0.0-0.3	Gravelly silty sand, black, loose, damp, fine to coarse sand, FILL	surface staining	30	0.0-0.3
/2		0.3-1.0	Silty clay, brown, damp, soft, non-plastic	no odour	25	0.3-0.5
/3					10	0.5-1.0
TP3/1	Tank pit east	0.0-0.5	Silty sand, brown, dry, loose, fine sand	surface staining	250	0.0-0.2
/2		0.5-2.8	Gravelly silty sand, grey, dry, loose, fine to coarse sand, FILL	no odour	50	1.8-2.0
/3		2.9	Clay, brown, damp, moderately soft, plastic	no odour	25	2.9-3.2
TP4/1	Tank pit west	0.0-0.2	Gravelly silty sand, black, dry, loose, fine to coarse sand, FILL	surface staining	10	0.0-0.2
/2		0.2-3.2	Sandy silt, red brown, loose, coarse FILL	no odour	10	1.8-2.3
/3		3.3	Clay brown / reddish brown, damp, soft, plastic	no odour	5	3.3-3.5
TP5/1	Tank pit south	0.0-0.35	Gravelly silty sand, dark reddish brown, loose, fine to coarse sand, FILL	surface staining	10	0.0-0.35
/2		0.35-2.5	Gravelly silty sand, brown, loose, fine to coarse, FILL	slight odour	40	0.35-0.5
/3				moderate odour	135	2.0-2.5
/4		2.5-3.3	Clay, medium brown, wet, soft, plastic	slight odour & heavy stains	800	2.5-3.0
/5		3.4	Clay, brown, dry, hard, plastic	faint HC odour	65	3.4-3.7
BH1/1	Tank pit southeast	0.0-0.2	Silty sand, brown, damp, loose, fine sand	surface staining	80	0.0-0.2
/2		0.2-0.45	Silty sand, black, dry, loose, fine to coarse sand, FILL	faint HC odour	60	0.2-0.45
/3		0.5-2.9	Gravelly sand, brown, loose, coarse, FILL	faint HC odour	25	1.5-2.0
/4				moderate odour	100	2.5-2.8
/5		3.0-3.5	Clay, brown ,dry, hard, plastic	strong HC odour & heavy stains	420	3.0-3.5
/6		3.5-4.0	Clay, brown, dry, hard, plastic	strong HC odour	230	3.5-4.0

Adapted from Queensland Department of Environment, 1998

Table 13. Example statistical analysis of results for a particular sampling event

Sample no.	Arsenic	Cadmium	Cobalt	Chromium	Copper	Nickel	Lead	Zinc
A1	12	1	27	256	51	69	116	398
A2	9	3	12	316	131	36	47	105
A3	8	1	26	294	236	82	25	73
A4	7	1	5	15	1290	19	154	1660
A5	8	1	34	132	403	166	99	105
A6	4	1	20	39	333	130	11	64
A7	12	1	43	300	546	84	58	128
A8	10	2	11	231	766	45	117	159
A9	6	1	52	304	642	62	57	131
A10	36	1	7	254	836	34	95	571
A11	8	1	22	255	33	92	19	46
A12	7	5	27	225	541	63	140	1380
A13	4	1	24	365	321	87	42	150
A14	3	0.5	83	257	453	71	22	30
A15	4	4	57	235	678	84	111	261
A16	3	1	22	223	165	59	385	584
A17	5	2	58	277	207	92	840	1740
A18	7	2	45	330	105	86	1870	649
A19	5	0.5	62	503	26	65	80	94
A20	6	1	46	400	345	65	217	4310
A21	12	1	30	273	16	81	180	458
A22	12	1	27	256	789	69	116	398
A23	15	1	15	254	345	44	117	218
A24	9	3	12	316	16	36	47	105
A25	34	1	29	169	342	100	43	135
A26	8	1	26	294	132	82	25	73
A27	12	1	32	215	107	104	272	360
A28	7	1	5	15	1290	19	154	1660
A29	14	2	51	266	119	112	383	852
A30	6	1	77	365	74	91	23	64
A31	14	1	53	205	33	101	34	39
A32	8	1	34	132	40	166	99	105
A33	17	1	43	291	32	74	58	112
A34	4	1	20	39	357	130	11	64
A35	12	1	31	285	1260	79	66	139
A36	12	1	43	300	345	84	58	128
A37	8	2	121	236	156	148	32	94
A38	9	2	53	454	435	79	10	19

Sample no.	Arsenic	Cadmium	Cobalt	Chromium	Copper	Nickel	Lead	Zinc
A39	6	1	32	207	534	81	15	37
A40	8	1	46	240	39	102	84	165
A41	8	1	15	269	30	48	59	88
A42	10	2	11	231	66	45	117	159
A43	9	2	44	250	42	88	92	155
A44	6	1	52	304	42	62	57	131
A45	5	2	35	412	615	62	25	982
A46	36	1	7	254	55	34	95	571
A47	6	1	39	221	453	59	11	30
A48	8	1	22	255	65	92	19	46
A49	7	1	55	278	34	87	28	64
A50	5	1	34	239	66	87	21	67
A51	9	1	79	300	75	103	57	142
A52	8	2	29	188	67	83	312	643
A53	9	2	34	227	34	72	86	164
A54	4	1	57	153	42	204	33	80
A55	7	1	48	259	50	101	204	251
A56	16	4	24	143	169	79	1310	10 900
A57	8	1	45	207	36	191	30	122
A58	5	1	34	239	1185	87	21	67
A59	8	2	29	188	1034	83	312	643
A60	4	1	57	153	442	204	33	80
A61	16	4	24	143	116	79	1310	10 900
A62	5	1	40	147	47	199	10	100
A63	6	1	28	177	231	106	54	110
A64	2	1	16	107	184	35	79	366
A65	9	1	48	206	395	98	33	166
A66	11	1	26	156	845	54	216	251
A67	6	1	13	287	25	70	46	71
Arithmetic mean	9	1	36	239	314	86	164	675
Standard deviation	7	1	21	92	346	41	322	1913
Geometric mean	8	1	30	210	158	77	70	193
Minimum	2	0.5	5	15	16	19	10	19
Maximum	36	5	121	503	1290	204	1870	10900
Median	8	1	32	250	165	82	58	135
90 percentile	14	2	57	322	808	137	312	1141

Sample no.	Arsenic	Cadmium	Cobalt	Chromium	Copper	Nickel	Lead	Zinc
95 percentile	17	4	72	390	1140	184	703	1716
Number of data points	67	67	67	67	67	67	67	67

Adapted from Queensland Department of Environment 1998

Table 14. Example frequency distribution data for copper data listed in Table 13

Concentration range (ppm)	Frequency	Cumulative %
0–60	20	**30**
60–200	16	**54**
200–400	11	70
400–600	8	82
600–800	5	90
800–1000	2	93
1000–1200	2	96
1200–1400	3	100

Adapted from Queensland Department of Environment 1998

15 Protection of the environment during site assessment

15.1 General considerations

Assessment of site contamination, or potential contamination, can present risks to the environment as well as to site personnel and local residents. This guidance provides the minimum measures that should be adopted to ensure protection of the environment during site assessment. Site-specific environmental management measures must ensure compliance with environmental management and protection legislation applying in each jurisdiction.

All states and territories have work health and safety legislative requirements. Plans developed under such legislation should address all relevant exposure pathways for site-specific contaminants of concern. Site assessment activities should comply with relevant work health and safety guidance and legislation applying in each jurisdiction.

15.1.1 Core environmental protection elements

Environmental protection plans should address the following issues:

- management of dust emissions and on-site and off-site odours
- protection of groundwater resources
- prevention of migration of contamination to adjacent sites or uncontaminated areas within the site
- prevention of contaminated run-off water reaching stormwater systems or local surface water environments
- prevention of initiation or spread of fire, either underground or above ground
- collection and disposal of excavation spoil
- collection and disposal of contaminated groundwater.

15.1.2 Less obvious concerns

Less obvious assessment issues that need to be addressed include:

- extending contamination or assisting contaminant migration during site investigation works by, for example, drilling through a contaminated aquifer into an uncontaminated lower aquifer thereby creating a conduit through which contamination may migrate
- introducing contamination to an otherwise clean soil stratum by backfilling a test pit found to be contaminated at surface level but clean at depth using the contaminated soil. It is always preferable to temporarily stockpile test pit spoil in excavation sequence so that it may be returned to the pit to roughly the same depth from which it was excavated
- initiating or extending underground fire by the introduction of oxygen
- enhancing acid run-off by enabling oxidation of in situ materials through exposure to atmosphere
- destabilising an otherwise stable embankment by introducing water.

15.2 Addressing environmental protection issues

The following elements of environmental protection should be considered prior to site assessment and be incorporated into the site assessment plan for each site. In particular, site contamination that is likely to cause public concern by the scale of operations, the nature of the site contamination or the potential for emission of noxious or offensive odours should indicate the commencement of public consultation and community engagement (refer Schedule B8) well before the commencement of site assessment works.

15.2.1 Management of dust and offensive and noxious odours

Environmental concerns regularly encountered on site assessments are dust and odour emissions which may be wind-blown and aggravated by the actions of trucks or other plant on the site. When warranted by the scale of site assessment and specific site conditions, area/boundary monitoring for dust deposition, inspirable and respirable dust and respective contaminants should be undertaken. Protection measures are important to ensure that dust inhalation or noxious or offensive odours do not pose a health risk for site operatives, nor a health risk or nuisance to local residents or passers-by and that concentrations of chemical substances do not exceed any relevant state or territory guidelines.

The traditional methods of dust and odour control include:

- application of a water spray with the objective to dampen the soil and not to saturate it, as potentially contaminated run-off from saturated soils entering adjacent sites, stormwater systems, or local waterways must be avoided (note: care should be taken when applying water onto soil that has recently been contaminated with volatiles or semi-volatiles, as this can result in a large increase in contaminant emissions from the soil)
- covering exposed faces with barriers (e.g. synthetic barriers, mulch) to prevent the emission of odours and dust
- minimising traffic and its speed on exposed contaminated soils
- the use of ground covers
- installation of screens to act as windbreaks.

Many sites, particularly those with petroleum hydrocarbons, organic contamination or putrescible wastes, may generate offensive odours or noxious vapours. In such cases, intensive odour control measures should be considered including minimising the exposed surface of the odorous materials at all times, timing excavation activities to minimise off-site nuisance, and by re-covering exposed faces overnight or during periods of low excavation activity. Such odorous materials should not be stockpiled unless closely contained or covered.

When dealing with volatile pollutants an assessment should be made of the need for the regular analysis of atmospheric levels of pollutants on site and at site boundaries to ensure that workers and residents are not being exposed to unacceptable levels of substances (for example, benzene) that may give rise to adverse health effects.

In addition, site boundary and competent community monitoring of offensive odours should be regularly undertaken during assessment of problematic sites. Site work practices relating to odour-generating activities should be promptly amended or stopped and reassessed in response to the results of boundary and community monitoring.

The social impact from the excavation of odorous or noxious materials can often be mitigated by excavating only when the wind direction is such that there will be the minimum possible effect upon neighbouring populations.

Where excavation of odorous or noxious material is expected or planned as part of an assessment process, the local population and other stakeholders should:

- be advised of the expected duration of the operation
- be advised that the operation will last for a limited time only
- be advised whether or not the odours may pose any potential health risk
- be given reassurance with regard to mitigation measures being undertaken.

An effective risk communication and community engagement program is an essential consideration for sites that pose a risk of offensive or noxious odours. Information on development and implementation of community engagement and risk communication programs is provided in Schedule B8.

15.2.2 Protection of groundwater resources

Before commencement of any drilling work, sufficient research should be undertaken to establish how much information is available regarding the geology and hydrogeology of the area to be investigated. If groundwater contamination is suspected there should be an audit of local bores. If more than one aquifer is expected, care should be taken to ensure that the potential for cross-contamination is minimised. Bores should be constructed so that different aquifers are isolated.

Licensing of monitoring bores may be a statutory requirement in some states and territories. There may also be state or territory guidelines that apply to minimum bore requirements and their decommissioning.

15.2.3 Site run-off, drainage and sedimentation

Care must be taken to avoid surface run-off from assessment activity impacting on adjacent sites, wetlands, water courses or stormwater drainage systems. The site assessor should be aware of the topography and geology of the site under assessment, and the possibility of migration of contaminants within the site or to adjacent sites, whether wind-blown, adhering to vehicles, plant and equipment, as free-flowing liquids, as surface run-off, or in groundwater flow. Stockpiled, excavated materials awaiting removal from site may create a particular risk to the environment.

Mitigation measures may include the use of temporary (waterproof) covers, excavation of drainage or run-off water diversion trenches, collection or absorption pits, or installation of temporary barriers in the form of hay bales, geofabrics or similar materials. Temporary bunding around stockpiles, or location of stockpiles on waterproof surfaces such as asphalt or concrete, or under cover where available, should be considered. Designation of an area within which all run-off and infiltration is to be controlled in accordance with strict performance objectives (for example, zero uncontrolled run-off) should also be considered. Disposal of any run-off should be carried out in accordance with relevant state or territory legislation.

Following rainfall it may be necessary to retrieve any sediment which has been carried in run-off or drainage water and manage this material appropriately. Respraying contaminated water onto stockpiles of contaminated soil as a means of effectively managing the water is also a possibility depending on jurisdictional guidelines and the nature of the contamination.

Treatment and disposal of collected contaminated run-off water should be appropriate to the contamination expected. If water treatment facilities are not immediately available, following consultation with local waste water authorities, diversion to sewer should be considered. Removal to landfill (not permitted in certain states) or treatment facility by means of road tanker is an expensive final option.

15.2.4 Contamination carry-over to public roads and highways

Potential carry-over of contamination to public roads and highways is an issue where excavation plant is operating on a site. Care must be taken to ensure that potentially contaminated material is not transported off site. Vehicle washing systems with facilities for handling the wash water and the installation of 'rumble strips' to help dislodge dust and mud, should be considered for installation at exits from sites where potential carry-over is perceived to be a problem. Procedures should be set in place for the handling and disposal of potentially contaminated water arising from wheel-wash operations.

15.2.5 Collection and disposal of contaminated water

Sample pits should be backfilled soon after sampling and sampling should not take place during rain. Contaminated water may be encountered where sample pits have been left open, and in boreholes. Care should be taken in disposing of contaminated flush water from borehole purging to ensure that contamination is not spread on the site. Gross contamination from borehole purging should be collected in drums or other suitable container for approved off-site disposal.

After excavation test pits may fill with rain or groundwater. Care should be taken to ensure that backfilling of the test pit does not rapidly displace this water, causing it to flow over the site. If necessary, the test pit should be part-backfilled and then bailed out to a suitable storage to enable full backfilling with spoil. Contaminated water should be disposed as appropriate.

All containers remaining temporarily on-site, and containing potentially contaminated materials, should be labelled with appropriate hazard warnings and waste producer contact details.

15.2.6 Collection and disposal of excavation spoil

It is normal practice to return excavation spoil from test pits to the excavation from which it came. However, care should be taken to ensure that materials are replaced in soil horizon order and that contaminated materials are not returned to a pit where they could contaminate unaffected strata or groundwater. Due to practical difficulties in compaction of excavation spoil there will inevitably be excess spoil after backfilling of a test pit. Care should be taken to ensure that contaminated spoil does not become spread across an otherwise uncontaminated surface. Drilling cuttings should not be returned to a bore.

Excess spoil should be stored in a lined skip or lined drums brought to site or placed on an impermeable surface such as concrete, asphalt, polyethylene sheeting or similar until analytical results can be assessed to enable cost-effective and safe methods of disposal. Where excess spoil is stored on site, and is not stored within a container, bunding should occur around the area to contain potential run-off. If contaminated materials are to be drummed for disposal or for treatment, the contents should be analysed, and management decisions made, based on the analytical results. All containers remaining temporarily on-site, and containing potentially contaminated materials, should be labelled with appropriate hazard warnings and waste producer contact details.

Allowances should be made within site assessment budgets for any necessary safe removal of a quantity of soil/fill from the site to an appropriate waste disposal or treatment facility. Transport and disposal of contaminated soil should be carried out in accordance with relevant state or territory legislation.

15.2.7 Noise and vibration

Noise can be a health risk to workers and is often a nuisance to those in the vicinity of a site. The potential for noise arising from site assessment activities should be evaluated and appropriate control measures put in place to reduce unacceptable noise (for example, by installing screens or noise baffles). Noise should not be a nuisance to people living or working around the site. Activities with

potential for noise generation should be carried out in accordance with relevant state or territory legislation.

Similarly, vibration from excavation and drilling, from plant, or from the movement of heavily laden trucks can sometimes result in damage to foundations of adjacent structures or to underground services or utilities. This possibility should be addressed and any risks assessed prior to choice of excavation or drilling method.

15.2.8 Acid sulfate soil

Acid sulfate soils (ASS) are naturally occurring soils, sediments and peats which contain iron sulfides. In an anoxic state, these materials are benign and do not pose a significant risk to human health or the environment. However, the disturbance of ASS and exposure to oxygen has the potential to cause significant environmental and economic impacts including fish kills and loss of biodiversity in wetlands and waterways and contamination of groundwater resources by acid and metals (WA DEC 2009). Activities that have the potential to disturb ASS, either directly or by affecting the elevation of the water table, need to be managed appropriately.

Where ASS is identified as a potential hazard, investigation and management of ASS should be carried out in accordance with relevant state or territory requirements. Jurisdictions should be consulted for advice on appropriate control measures to apply on the management of ASS prior to any dewatering or excavation activities taking place. Further technical information is provided in WA DEC (2011).

15.2.9 Heritage sites

Special care should be taken to ensure that any assessment works or activities on or adjacent to sites of cultural or natural heritage significance will not have an adverse impact. Heritage places may include buildings, structures, archaeological remains, or landscaped or natural areas of aesthetic, historic, scientific or social value. Where appropriate, advice should be sought from the local representatives of the National Congress of Australia's First Peoples, the Australian Heritage Council, and state or territory heritage bodies and local councils.

15.2.10 Rare habitats or endangered species

Special care should be taken to ensure that any assessment works or activities will not impact upon rare natural habitats or any endangered species. Advice may be sought from the relevant jurisdiction to ensure that site environmental protection plans are sufficiently protective.

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17 Appendix A: Possible analytes for soil contamination

This list is indicative only and analytes for analysis should be selected based on site history.

Inorganic contaminants

Analysis name	
Metals	*Where a general purpose screen for metal contamination in soils is indicated, it may include:* Arsenic, Cadmium, Chromium Copper, Lead, Manganese Mercury, Nickel, Zinc *If more detailed investigation is indicated, soil may be examined for:* Aluminium, Antimony, Barium Beryllium, Boron, Calcium Cobalt, Iron, Magnesium Molybdenum, Potassium, Selenium Silver, Strontium, Thallium Tin, Vanadium
Anions	*Where a general purpose screen for anion contamination in soils is undertaken, it may include:* Bromide, Iodide, Sulfate Chloride, Nitrate and Nitrite, Sulfide Cyanide, Phosphate, Fluoride

Organic contaminants

Analysis name	
Monocyclic aromatic hydrocarbons (MAHs)	*Where a general purpose screen for MAH contamination in soils is undertaken, it may include:* Benzene Toluene ortho-Xylene meta- Xylene (para- Xylene) Ethyl benzene Styrene (vinyl benzene) Cumene (isopropylbenzene) 1,3,5 Trimethylbenzene 1,2,4-Trimethylbenzene 1-Methyl-4-isopropylbenzene n-Propylbenzene n-Butylbenzene

Analysis name	
	iso-Butylbenzene tert-Butylbenzene sec-Butylbenzene *If more detailed investigation is indicated, the following analytes may be included:* Chlorobenzene 1,2-Dichlorobenzene 1,3-Dichlorobenzene 1,4-Dichlorobenzene Nitrobenzene Dinitrobenzenes Nitrotoluene Dinitrotoluenes Trinitrotoluenes
Polycyclic aromatic hydrocarbons (PAHs)	*Where a general purpose screen for PAH contamination in soils is undertaken, it may include:* Naphthalene Benzo(a) anthracene Acenaphthylene Chrysene Acenaphthene Benzo(b) fluoranthene Fluorene Benzo(k) fluoranthene Phenanthrene Benzo(a) pyrene Anthracene Dibenz (a,h)anthracene Fluoranthene Benzo(ghi) perylene Pyrene Indeno(123-cd) pyrene
Phenols	*Where a general purpose screen for phenols contamination in soils is undertaken, it may include:* Phenol o-Cresol p-Cresol 2,3-Dimethylphenol 2,4-Dimethylphenol 2,5-Dimethylphenol 2,6-Dimethylphenol 3,4-Dimethylphenol 3,5-Dimethylphenol 2,3,5-Trimethylphenol 2,3,6-Trimethylphenol 2,4,6-Trimethylphenol

Analysis name	
	If more detailed investigation is indicated, the following may be included: 2-Nitrophenol 4-Nitrophenol 2,4-Dinitrophenol
Chlorinated phenols	*Where a general purpose screen for chlorinated phenols contamination in soils is undertaken, it may always include:* 2-Chlorophenol 3-Chlorophenol 4-Chlorophenol 2,4 –Dichlorophenol 2,6 –Dichlorophenol 2,4,5 –Trichlorphenol 2,4,6 –Trichlorphenol 2,3,4,5-Tetrachlororphenol 2,3,4,6-Tetrachlororphenol 2,3,5,6 –Tetrachlororphenol Pentachlorophenol
Chlorinated benzenes	*Where a general purpose screen for chlorinated benzenes contamination in soils is undertaken, it may include:* Chlorobenzene 1,2-Dichlorobenzene 1,3-Dichlorobenzene 1,4-Dichlorobenzene 1,2,3-Trichlorobenzene 1,2,4-Trichlorobenzene 1,2,4,5 Tetrachlorobenzene Pentachlorobenzene Hexachlorobenzene
Organochlorines (OCs)	*Where a general purpose screen for OCs contamination in soils is undertaken it may include:* Aldrin HCB alpha-HCH, beta-HCH gamma-HCH (lindane), delta-HCH Chlordane DDD, DDE, DDT Dieldrin Endrin Endosulfan (alpha-, beta- and sulfate)

Analysis name	
	Heptachlor, Heptachlor epoxide Methoxychlor Toxaphene (chlorcam, campheclor) *Where site history indicates possible PCB contamination, the following may be included:* PCB (Aroclors 1016, 1221, 1232, 1242, 1248, 1254 and 1260)
Organophosphorus insecticides (OPs)	*Where a general purpose screen for OP contamination in soils is undertaken it may include:* Chlorpyrifos Coumaphos Diazinon Dichlorvos Dimethoate Ethion Fenthion Malathion Parathion methyl Parathion ethyl *If more detailed investigation is indicated, the following may be included:* Azinphos methyl Sulprofos Demeton-s-methyl Disulfoton Ethoprophos Mevinphos Monocrotophos Naled Phorate Prothiophos Tetrachlorvinphos A Nitrogen/Phosporus Detector (NPD) or flame photometric detector (FPD) or GC/MS should be employed for screening purposes.
Acid/phenoxy herbicides	*Where a general purpose screen for acid herbicides contamination in soils is undertaken, it may include:* 2,4-D 2,4-DB 2,4,5-T 2,4,5-TP (Silvex) Dicamba and 5-Hydroxydicamba

Analysis name	
	MCPA MCPP 4-Nitrophenol *If more detailed investigation is indicated, the following may be included:* Acifluoren Bentazon Dichlorprop Dalapon Picloram
Triazine herbicides	*Where a general purpose screen for triazine herbicide contamination in soils is undertaken, it may include:* Atrazine Ametryn Prometryn Simazine Hexazinone
Phthalate esters	*Where a general purpose screen for phthalate contamination in soils is undertaken, it may include:* Bis (2-ethylhexyl) phthalate Butyl benzyl phthalate Di-n-butyl phthalate Dicyclohexyl phthalate Diethyl phthalate Dihexyl phthalate Diisobutyl phthalate Dimethyl phthalate Dinonyl phthalate Di-n-octyl phthalate *If more detailed investigation is indicated, the following may be included:* Bis (2-n-butoxyethyl) phthalate Bis (2-ethoxyethyl) phthalate Bis (2-methoxyethyl) phthalate Bis (4-methyl-2-pentyl) phthalate Diamyl phthalate Hexyl 2-ethylhexyl phthalate

18 Appendix B: Data quality objectives (DQO) process

18.1 Introduction

The DQO process is a seven-step iterative planning approach that is used to define the type, quantity and quality of data needed to inform decisions relating to the environmental condition of a site. The summary of the process below is adapted from US EPA (2006a) and NSW DEC (2006).

The DQO process should commence before any investigative work starts, with the timing for various stages of the project being clearly understood by all parties. It is useful to apply the process initially at a project level to determine the overall project requirements and then modified as required for specific investigation activities.

The seven steps in the DQO process are:

Step 1: State the problem

Step 2: Identify the decision/goal of the study

Step 3: Identify the information inputs

Step 4: Define the boundaries of the study

Step 5: Develop the analytical approach

Step 6: Specify performance or acceptance criteria

Step 7: Develop the plan for obtaining data

18.2 The seven-step DQO process

18.2.1 Step 1: State the problem

The first step involves summarising the contamination problem that will require new environmental data and identifying the resources available to resolve the problem. A preliminary CSM will be required to complete this step. The matters to consider at this stage include:

- the objective of the proposed investigation, noting that the ability to meet objectives may be limited by constraints such as time, resources, climatic conditions and access restrictions
- the possible content of a problem statement that gives a brief summary of the contamination issue(s) at the site that is to be addressed in the project
- the reason the project is being undertaken
- identification of the project team and technical support experts, such as field manager/site supervisor, field personnel, toxicologists, risk assessors and statisticians
- budget and community concern issues that may also be factors in designing and carrying out the environmental assessment
- identification of the regulatory authority(ies) and the local government area.

Step 1 of the DQO process should assist in developing the following:

- a concise description of the problem
- a list of the planning team members and identification of decision-maker
- a summary of available resources and relevant deadlines for the study
- a preliminary conceptual model of the site, based on available information prior to the commencement of the site investigation, covering:
  - previous investigations
  - present and historical use(s) of the site and adjacent sites
  - geology, hydrogeology
  - potential contaminants of concern
  - potential contaminant migration pathways both to and from the site (such as waterways, drains, service conduits)
  - areas of environmental concern (drawings showing chemical storage, use, disposal)
  - media in which potential contaminants of concern may be present and through which they may migrate (habitat(s) of contamination, lateral and depth extent, temporal and climatic variability)
  - potential exposure pathways to human and/or environmental receptors
  - future land uses.

The conceptual model of contamination of the site that is produced at this early point can be progressively refined through subsequent stages of the assessment.

18.2.2 Step 2: Identify the decisions/goal of the study

The second step involves identifying the decisions that need to be made about the contamination problem and the new environmental data required to make them.

The objective(s) of the data collection part of the investigation is project-specific and may be identified by:

- referring to the history of use of the site, chemicals of concern and likely concentration range(s), media that may be impacted and likely migration routes, such as groundwater, surface water flow, wind, and service trenches
- considering relevant site criteria for each medium (fill, soil, sediment, groundwater, surface water, air)
- making a series of decision statements that need to be addressed (e.g. a decision statement could consider whether parts of the site would be suitable for a proposed use if the 95% UCL on the mean concentrations for all chemicals of potential concern were less than the appropriate site criteria).

Step 2 of the DQO process should assist in developing a decision statement linking the principal project objective(s) to the possible actions that will address the problem.

The existing conceptual model can then be reviewed to determine whether existing data is satisfactory to complete the investigation or whether data gaps or an unacceptable level of uncertainty exists.

18.2.3 Step 3: Identify information inputs

The third step involves identifying the information needed to support any decision and whether new environmental data will be needed.

Decisions made during this step are of a draft or preliminary nature and are reviewed in Step 7 to develop the sampling analytical and quality plan (SAQP).

Step 3 of the DQO process should assist decision-makers to resolve decision statements and make informed, defensible decisions by identifying:

- the media that needs to be collected, such as fill, soil, groundwater, sediments, surface water and air
- the environmental parameters that will be measured for each media
- site criteria for each medium of concern
- analytical methods that are required for chemicals of potential concern so that assessment can be made relative to the site criteria
- the basis for any decisions that are to be made from field screening, such as from PID data, and what action is to be taken if a defined concentration is attained
- any additional information required to make the required decisions.

18.2.4 Step 4: Define the study boundaries

The fourth step involves specifying the spatial and temporal aspects of the environmental media that the data must represent to support decision(s). The matters to consider at this stage include:

- the geographical extent of the proposed investigation
- time and budget constraints
- spatial extent (property boundaries, accessibility constraints to parts of the site, potential exposure areas)
- temporal boundaries (the time frame of the investigation, taking into account seasonal conditions, presence of near-surface groundwater or surface water and discharges, access restrictions, availability of key personnel)
- for large sites, the boundaries of each segment to be investigated (based on proposed use of each area of the site, which will influence the required sample density, appropriate regulatory guidance)
- the lateral and vertical intervals in which contamination distribution is believed to be uniformly distributed
- the scale of decisions required: site-wide, each residential lot, etc.
- the presence of any heterogeneous materials that may require specific sampling methods
- potential constraints to carrying out the investigation, such as access, presence of infrastructure, health and safety issues.

Step 4 of the DQO process should assist in developing:

- a detailed description of the spatial and temporal boundaries of the problem
- an understanding of any practical constraints that may interfere with the assessment.

18.2.5 Step 5: Develop the analytical approach (or decision rule)

The fifth step involves defining the parameter of interest, specifying the action level, and integrating information from Steps 1–4 into a single statement that gives a logical basis for choosing between alternative actions. Acceptable limits should be defined for the following:

- chemicals of concern detected in field blanks, rinsate blanks, volatile-spiked trip samples, laboratory method blanks
- recovery of matrix spike additions, surrogate spike additions, laboratory control samples
- relative percent differences (RPDs) of matrix spike and matrix spike duplicates.

Step 5 of the DQO process should assist in producing:

- the statistical parameter (the parameter of interest) that characterises the population
- confirmation that the action level exceeds measurement detection limits
- an 'if …, then …' statement that defines the conditions that would cause a decision-maker to choose from alternative actions.

18.2.6 Step 6: Specify the performance or acceptance criteria

The sixth step involves specifying the decision-maker's acceptable limits on decision errors, which are used to establish performance goals for limiting uncertainties in the data. (For more information about decision errors and decision-making, see notes at the end of this Appendix). Some of the matters to consider at this stage include:

- determination of the possible range of the parameter of interest
- identification of decision errors and formulation of the null hypothesis
- specification of a range of possible parameter values where the consequences of decision errors are relatively minor (grey region)
- assignation of probability values to points above and below the action level that reflect the tolerable probability for the occurrence of decision errors.

Step 6 of the DQO process should assist in calculating the decision-maker's tolerable decision error rates based on a consideration of the consequences of making an incorrect decision.

18.2.7 Step 7: Optimise the design for obtaining data

The seventh step involves identifying the most resource-effective sampling and analysis design for generating the data that is required to satisfy the DQOs.

Step 7 of the DQO process should assist in developing:

- the most resource-effective design for the study that is expected to achieve the DQOs
- the optimum manner in which to collect the data required to meet the objectives for the assessment and which will meet the project DQOs
- the SAQP.

18.3 Notes about decision errors and decision-making

Decision errors are incorrect decisions caused by using data that is not representative of site conditions due to sampling or analytical error. As a result, a decision may be made that site clean-up is not needed when really it is, or vice versa.

There are two types of decision error:

- sampling errors occur when the sampling program does not adequately detect the variability of a contaminant from point to point across the site. That is, the samples collected are not representative of the site conditions (e.g. an appropriate number of representative samples have not been collected from each stratum to account for estimated variability)
- measurement errors occur during sample collection, handling, preparation, analysis and data reduction.

The combination of the above errors is referred to as 'total study error'. This directly affects the probability of making decision errors. Study error is managed through the correct choice of sample design and measurement systems. Note that the attainment of a nominated probability generally requires use of a statistically based sampling plan.

The possibility of making a decision error, although small, is undesirable because of the adverse consequences arising from that incorrect decision. Decision error can be controlled through the use of hypothesis testing. This test can be used to show either that the baseline condition is false (and therefore the alternative condition is true) or that there is insufficient evidence to indicate that the baseline condition is false (and therefore the site assessor decides by default that the baseline condition is true).

The burden of proof is placed on rejecting the baseline condition, because the test hypothesis structure maintains the baseline condition as being true until overwhelming evidence is presented to indicate that the baseline condition is not true.

The null hypothesis is an assumption assumed to be true in the absence of contrary evidence, for example, that the site is contaminated unless proved to be clean.

If we reject a hypothesis when it should be accepted, we say that a type I error has been made. If, on the other hand, we accept a hypothesis when it should be rejected, we say that a type II error has been made. In either case, a wrong decision or error in judgment has occurred:

- type I error (false positive decision error) — rejecting the hypothesis as false when it is really true
- type II error (false negative decision error) — accepting the hypothesis as true when it is really false.

In order for decision rules (or tests of hypotheses) to be sound, they must be designed to minimise decision errors. This is not always simple, as for any given sample size, an attempt to decrease one type of error is generally accompanied by an increase in the other type of error. The only way to reduce both types of error is to increase the sample size, which may or may not be always possible. In testing a given hypothesis, the maximum probability with which we would be willing to accept a type I error is referred to as the 'level of significance' or significance level of the test. A significance level of 0.05 or 0.01 is commonly adopted, although other values are used.

If for example the 0.05 (or 5%) significance level is selected for a decision rule, then we are accepting that there is a 1 in 20 (that is, 5 chances in 100) chance that we would reject the hypothesis when it should be accepted; that is, we are about 95% confident that we have made the right decision. In this case we say that the hypothesis has been rejected at the 0.05 significance level, which means that the hypothesis has a 0.05 probability of being wrong.

19 Appendix C: Assessment of data quality

19.1 Assessment of reliability of field procedures and laboratory results

Source: NSW DEC, 2006.

Contaminated site practitioners should undertake an assessment of the reliability of field procedures and analytical results using the data quality indicators (DQI) of precision, accuracy, representativeness, completeness and comparability. DQI should be used to document and quantify compliance or otherwise with the requirements of the project SAQP.

19.2 QA/QC analytical methods

The DQI for chemical data will differ depending on which analytical methods have been used in a site assessment. These fall into three main categories:

- field methods
- laboratory screening methods
- methods specific for contaminants that are known or expected to be present at a site.

19.3 Field methods

The following issues should be documented and discussed in assessment reports:

- the applicability and limitations of field methodologies where used
- instrument calibration and validation of field measurements, and comparison with laboratory results
- the significance of the results of field screening methods compared with the results of laboratory analyses, for example, that the results reported for field screening using a photo-ionisation detector are compatible with the results reported by the laboratory for volatile organic compounds. Where not compatible, an adequate explanation should be provided.

19.4 Laboratory screening methods

Laboratory screening methods are used to determine the type of contamination present and the constituents of a sample that might cause interferences in specific methods. Assessment reports should include appropriate discussion of the applicability and limitations of any screening methodologies used.

DQI for screening methods may be less rigorous than for specific analytical methods. Nevertheless, screening method performance should be known and should be expressed as a multiple of specific analytical method performance.

19.5 Methods specific for contaminants

Site assessors should ensure that appropriate discussion and documentation about the following issues is included in the assessment report:

- that the analytical methods used for site validation are of appropriate precision and accuracy, and that the sensitivity and selectivity of the analytical methods are appropriate for the assessment of the risk
- that the precision and accuracy criteria set out in the QA/QC plan, for a given method and matrix, meet the performance expected of the reference method

- that the quality of data supplied by the analytical laboratory meets the objectives of the testing laboratory's quality plan for at least 95% of test results for blanks, spikes, control samples, duplicates and holding times. (Note that these DQOs do not refer to field duplicate reproducibility or other measures of sampling variance. Sampling variance should be addressed in the choice of sampling method.)

19.6 Data quality indicators (DQIs)

Contaminated site practitioners should undertake an assessment of the DQIs that relate to both field and laboratory procedures, and provide appropriate documentation in the assessment report.

Completeness A measure of the amount of usable data (expressed as %) from a data collection activity		
Field considerations	**Laboratory considerations**	**Comments**
All critical locations sampled All samples collected (from grid and at depth) Standard operating practices (SOPs) appropriate and complied with Experienced sampler Documentation correct	All critical samples analysed according to SAQP All analytes analysed according to SAQP Appropriate methods and PQLs Sample documentation complete Sample holding times complied with	The required percentage completeness should be specified in the SAQP All required data must be obtained for critical samples and chemicals of concern Incompleteness is influenced by: • field performance problems (access problems, difficulties on site, damage…) • laboratory performance problems (matrix interference, invalid holding times…) • matrix problems
Comparability The confidence (expressed qualitatively) that data may be considered to be equivalent for each sampling and analytical event		
Field considerations	**Laboratory considerations**	**Comments**
Same SOPs used on each occasion Experienced sampler Climatic conditions (temperature, rainfall, wind…) Same types of samples collected (filtered, size fractions...)	Sample analytical methods used (including clean-up) Sample PQLs (justify/quantify if different) Same laboratories (justify/quantify if different) Same units (justify/quantify if different)	Same approach to sampling (SOPs, holding times…) Quantify influence from climatic or physical conditions Samples collected, preserved, handled in same manner (filtered, same containers)

Representativeness The confidence (expressed qualitatively) that data is representative of each medium present on the site		
Field considerations	**Laboratory considerations**	**Comments**
Appropriate media sampled according to SAQP All media identified in SAQP sampled	All samples analysed according to SAQP	Samples must be collected to reflect the characteristics of each medium Sample analyses must reflect properties of field samples Homogeneity of the samples Appropriate collection, handling, storage and preservation Detection of laboratory artefacts, e.g. contamination blanks
Precision A quantitative measure of the variability (or reproducibility) of data		
Field considerations	**Laboratory considerations**	**Comments**
SOPs appropriate and complied with	Analysis of: • laboratory and inter-laboratory duplicates • field duplicates • laboratory-prepared volatile trip spikes	Measured by the coefficient of variance or standard deviation of the mean or by RPDs Field duplicates measure field and laboratory precision Laboratory duplicates measure analytical precision*
Accuracy (bias) A quantitative measure of the closeness of reported data to the true value		
Field considerations	**Laboratory considerations**	**Comments**
SOP appropriate and complied with	Analysis of: • field blanks • rinsate blank • reagent blank • method blank • matrix spike • surrogate spike • reference material • laboratory control sample • laboratory-prepared spikes	Bias introduced: • by chemicals during handling or transport • from contaminated equipment • from contaminated reagent • during laboratory analysis • during laboratory preparation and analysis (may be high or low) • precision of preparation and analytical method • during laboratory analysis • during collection/ transport (may be high or low)

* Laboratory duplicates measure analytical precision when the sample is totally homogenous. When sample heterogeneity exists, laboratory duplicates (and intralaboratory splits) measure the sum of laboratory precision plus sample heterogeneity. High sample heterogeneity impacts confidence in data and may warrant additional sampling to increase confidence or detect hotspots.

19.7 Field QA/QC

Environmental practitioners should ensure that the following issues are addressed in the field QA/QC program and that appropriate documentation is included in the assessment report:

- replicate samples are split in the field and submitted to two separate laboratories in accordance with the requirements of Schedule B3
- the sampling program includes assessment of all relevant environmental media, including soil, dust, surface water, groundwater, air, sediments and biota as appropriate
- the sampling strategy is appropriate for the conditions at the site and the nature of the contamination, with the rationale for the strategy described in the assessment report and the sampling locations shown on a scaled site sampling plan
- sample collection, handling and transportation procedures are documented and appropriate to meet the project DQOs
- sampling is representative of site conditions, based on the selection of appropriate numbers of sampling points and of samples from each relevant strata and material types stated in a site sampling plan to meet the project DQOs
- the field QA/QC plan includes details of:
    - the sampling team
    - sampling method(s), including the actual methods employed for obtaining samples, type(s) of sample containers, order and degree of filling, preservation, labelling, logging, custody
    - evidence of appropriate decontamination procedures carried out between sampling events
    - completed logs for each sample collected, showing time, location, initials of sampler, duplicate locations, duplicate type, chemical analyses to be performed, site observations and weather conditions
    - completed chain-of-custody documentation, identifying for each sample the name of the sampler, the nature of the sample, collection date, analyses to be performed, sample preservation method, departure time from the site and dispatch courier(s) and condition of samples at dispatch
    - sample splitting techniques
    - a statement of duplicate frequency for intra-laboratory and inter-laboratory duplicate samples and duplicate sample results
    - field blank results
    - background sample results
    - rinsate sample results
    - laboratory-prepared trip spike results for volatile analytes
    - trip blank results
    - field instrument calibration for instruments used on site.

19.8 Laboratory QA/QC

Environmental practitioners should ensure that the following issues are addressed in the laboratory QA/QC program and that appropriate documentation is included in the assessment report:

- sample analyses use appropriate methodologies for each potential contaminant in the matrix in laboratories accredited for those analyses by the National Association of Testing Authorities (NATA) or an equivalent government-endorsed provider of accreditation for laboratories
- appropriate practical quantitation limits (PQLs) for the chemicals of concern for use in the assessment of risk
- a laboratory QA/QC plan with the following information:
    - a copy of signed chain-of-custody forms acknowledging receipt date and time, conditions of samples on receipt and identity of samples included in shipments
    - record of holding times and a comparison with method specifications
    - analytical methods used
    - laboratory accreditation for analytical methods used
    - laboratory performance in inter-laboratory trials for the analytical methods used, where available
    - the results for blind duplicate samples collected from the field.

19.9 QA/QC documentation

The site assessment reports should include documentation of QA/QC procedures including all information relevant to the site assessment:

- the QA/QC checklist items (see Section 19.10), related to field quality assurance and quality control, laboratory QA/QC and data evaluation QA/QC
- the names of the accredited laboratories used and relevant details of their accreditation for each analytical method
- the limits of reporting (ensuring that appropriate assessment can be made according to site criteria as stated in the DQOs for relevant media)
- the acceptance limit(s) for each QC test, such as duplicate RPDs and recoveries for laboratory quality control analyses
- where used, the origin of certified reference material (CRM), its batch number and the concentrations of the chemicals of potential concern
- the QC results relevant to the sample analysis
- for each sample, the highest measurement result wherever replicate measurements are taken (or all measurement results for each sample)
- results for all data tabulated separately according to each type of soil, fill, groundwaters, surface waters and sediments, with appropriate statistical analysis
- the laboratory specifying compliance with the requirements of Schedule B3 and equivalence with the reference method or non-standard methods.

19.10 Quality assurance and quality control checklist

Field quality assurance and quality control

- ☐ details of sampling team
- ☐ decontamination procedures carried out between sampling events
- ☐ field logs for samples collected — including time, location, initials of sampler, duplicate locations, duplicate type, chemical analyses to be performed, site observations and weather conditions
- ☐ chain-of-custody fully identifying (for each sample) the sampler, nature of the sample, collection date, analyses to be performed, sample preservation method, departure time from the site and dispatch courier(s)
- ☐ sample splitting techniques
- ☐ statement of duplicate frequency
- ☐ field blank results
- ☐ background sample results
- ☐ rinsate sample results
- ☐ laboratory-prepared trip spike results for volatile analytes
- ☐ trip blank results
- ☐ field instrument calibrations (when used)

Laboratory QA/QC

- ☐ a copy of the signed chain-of-custody forms acknowledging receipt date and time, and identity of samples included in shipments
- ☐ record of holding times and a comparison with method specifications
- ☐ analytical methods used
- ☐ laboratory accreditation for analytical methods used
- ☐ laboratory performance in inter-laboratory trials for the analytical methods used, where available
- ☐ description of surrogates and spikes used
- ☐ percent recoveries of spikes and surrogates
- ☐ instrument detection limit
- ☐ method detection limits
- ☐ matrix or practical quantification limits
- ☐ standard solution results
- ☐ reference sample results
- ☐ reference check sample results
- ☐ daily check sample results

- ☐ laboratory duplicate results
- ☐ laboratory blank results
- ☐ laboratory standard charts

QA/QC data evaluation

- ☐ evaluation of all QA/QC information listed above against the stated DQOs including a discussion of:
  - documentation completeness
  - data completeness
  - data comparability
  - data representativeness
  - precision and accuracy for both sampling and analysis for each analyte in each environmental matrix informing data users of the level of reliability or qualitative value of the data
- ☐ results of data comparability checks to assess bias that may arise from various sources, including:
  - collection and analysis of samples by different personnel
  - use of different methodologies
  - collection and analysis by the same personnel using the same methods but at different times
  - spatial and temporal changes (because of environmental dynamics)
- ☐ relative percent differences for intra- and inter-laboratory duplicates.

20 Appendix D: Example data presentation on scale drawings and borehole logs

Figure 2. Example site layout overlay

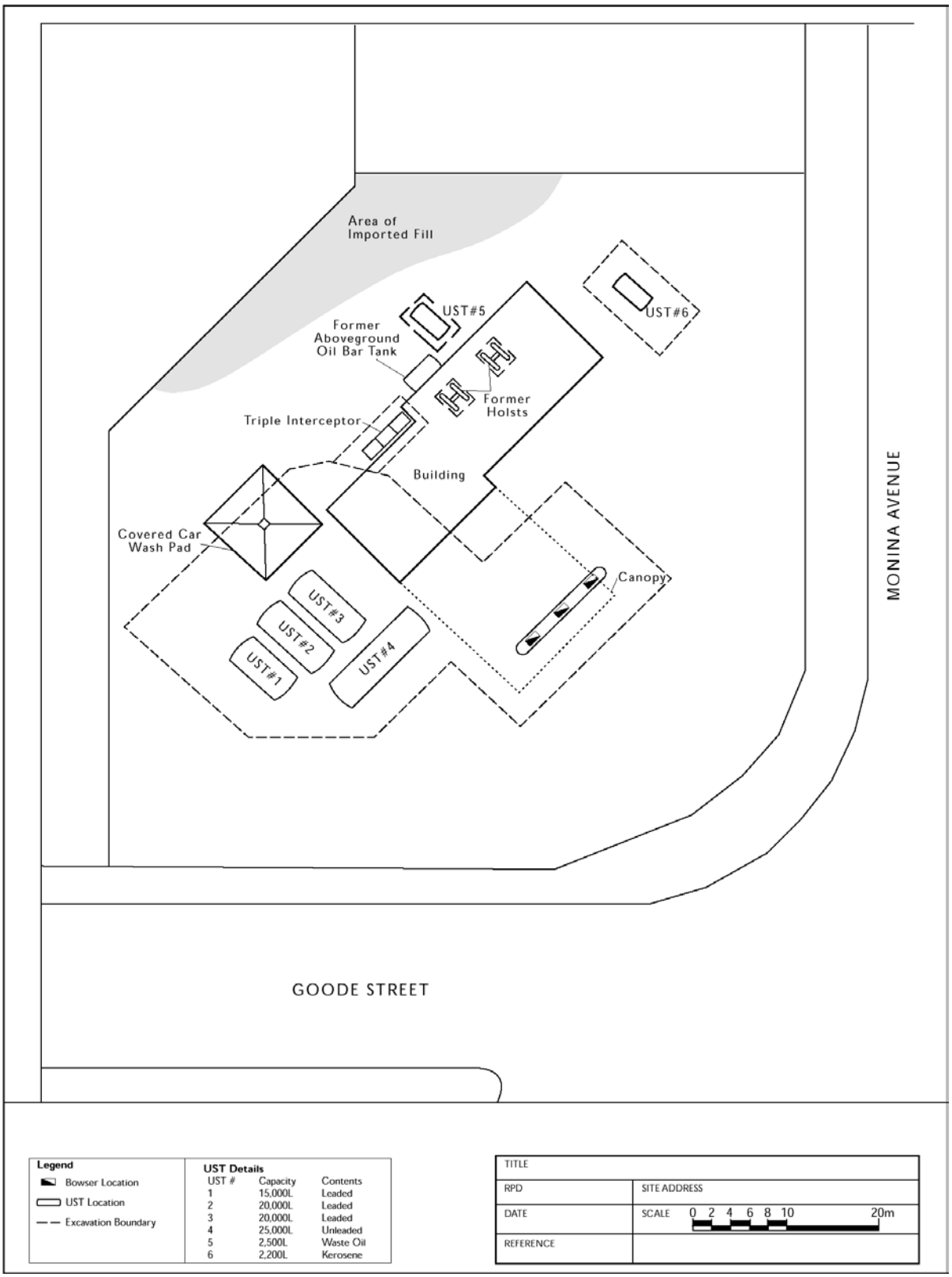


Figure 3. Example Results – v – Site Features

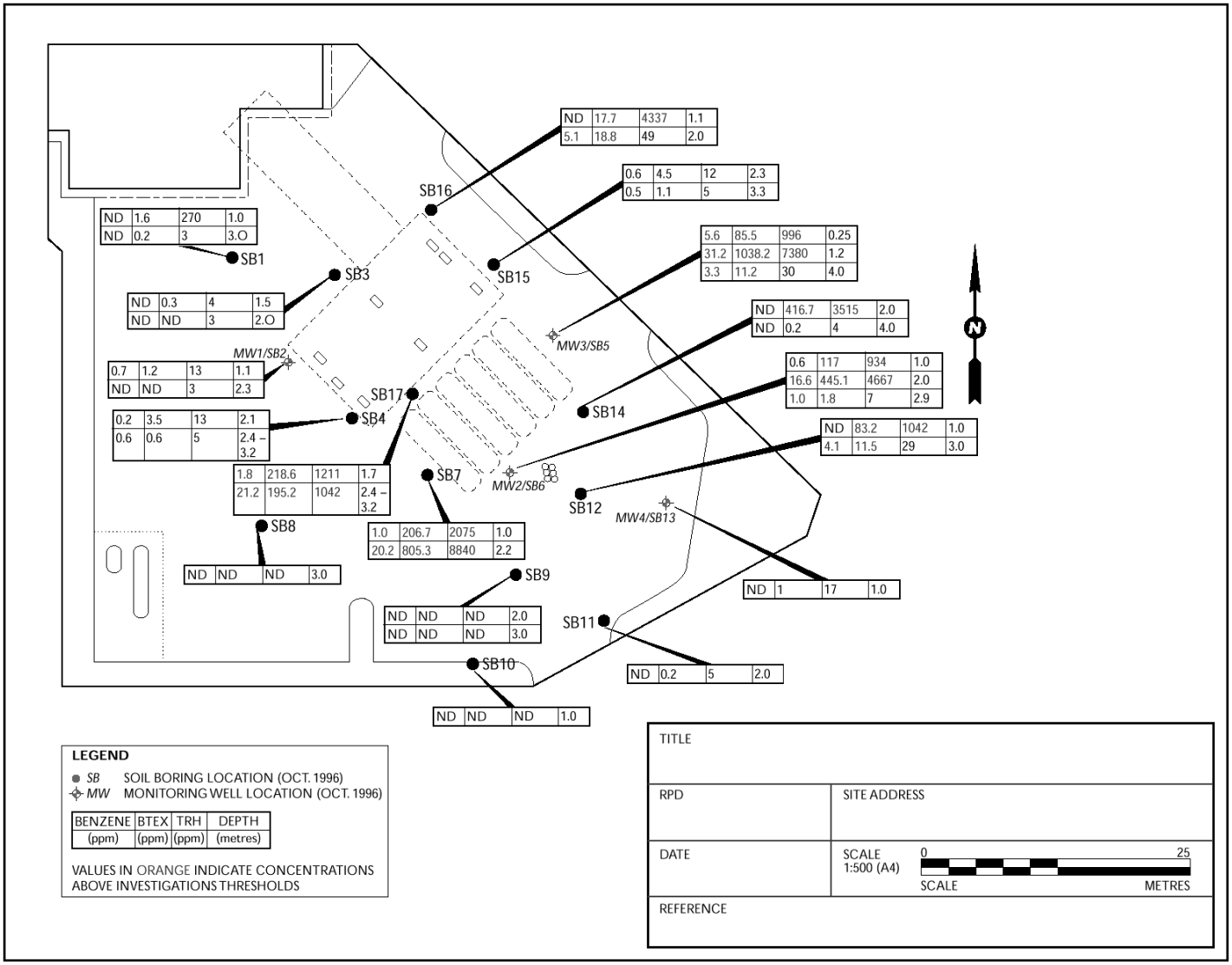


Figure 4. Example cross-section showing contaminant concentrations through soil profile

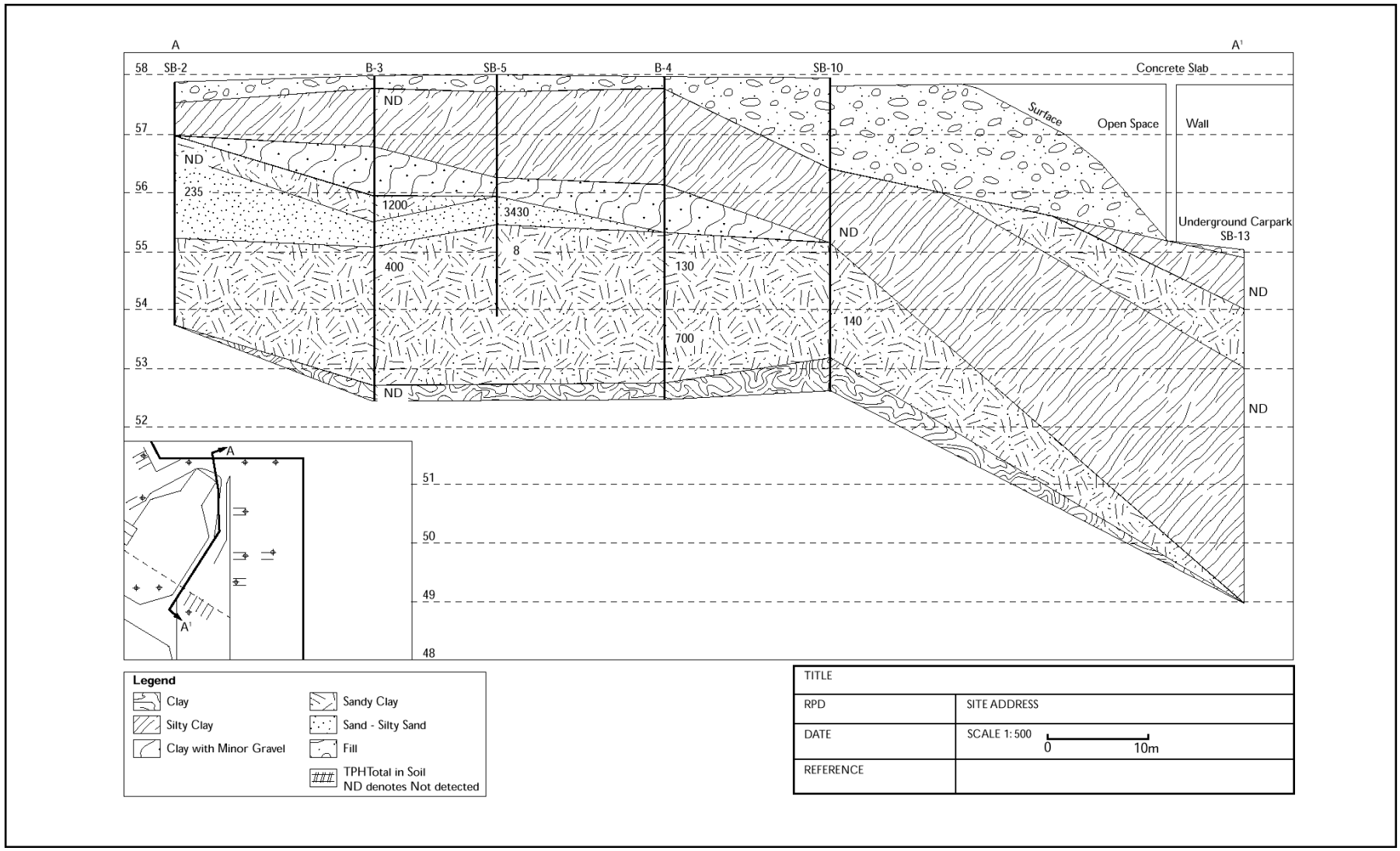


Figure 5. Example results from excavation assessment

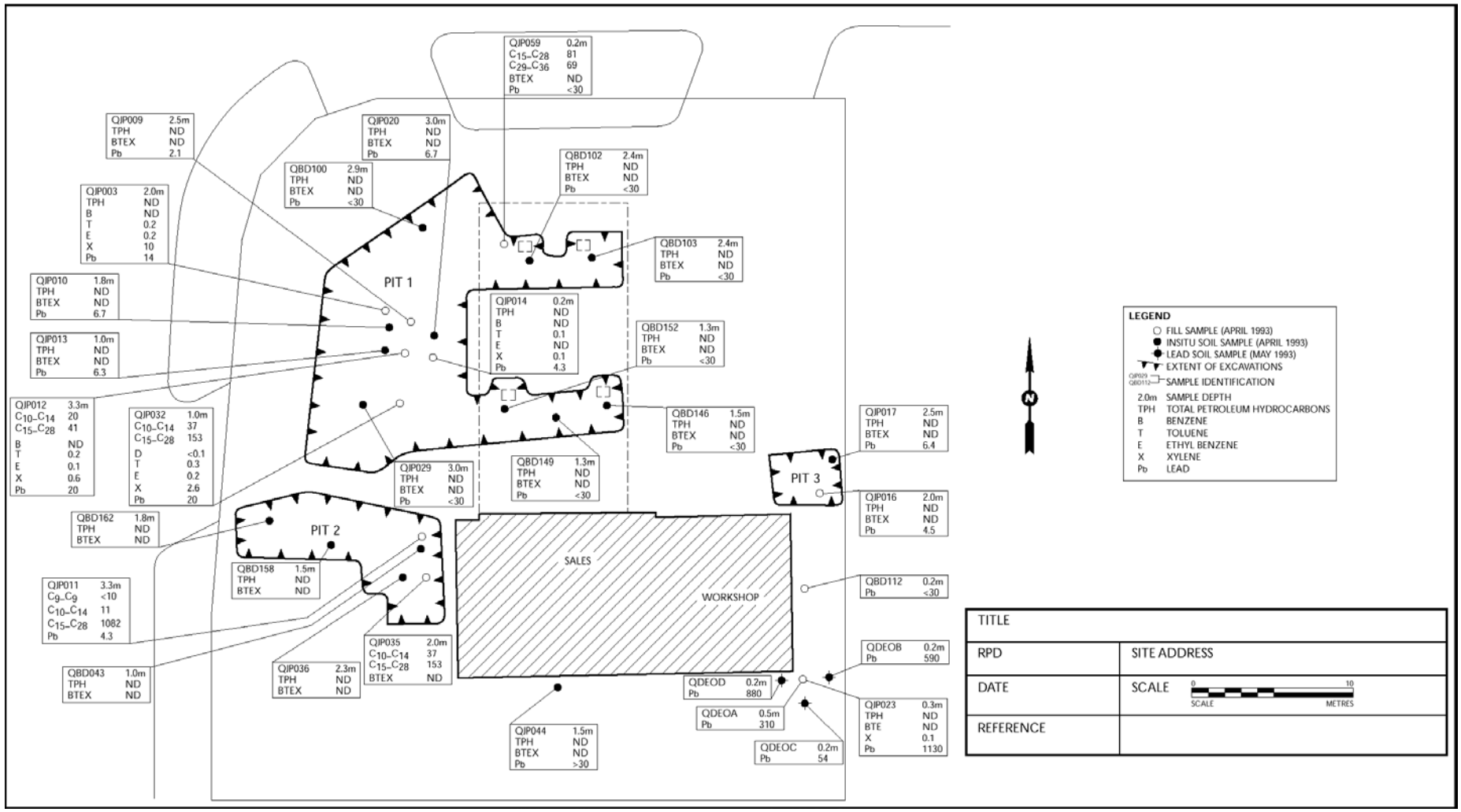


Figure 6. Example site plan with analyte concentration contours

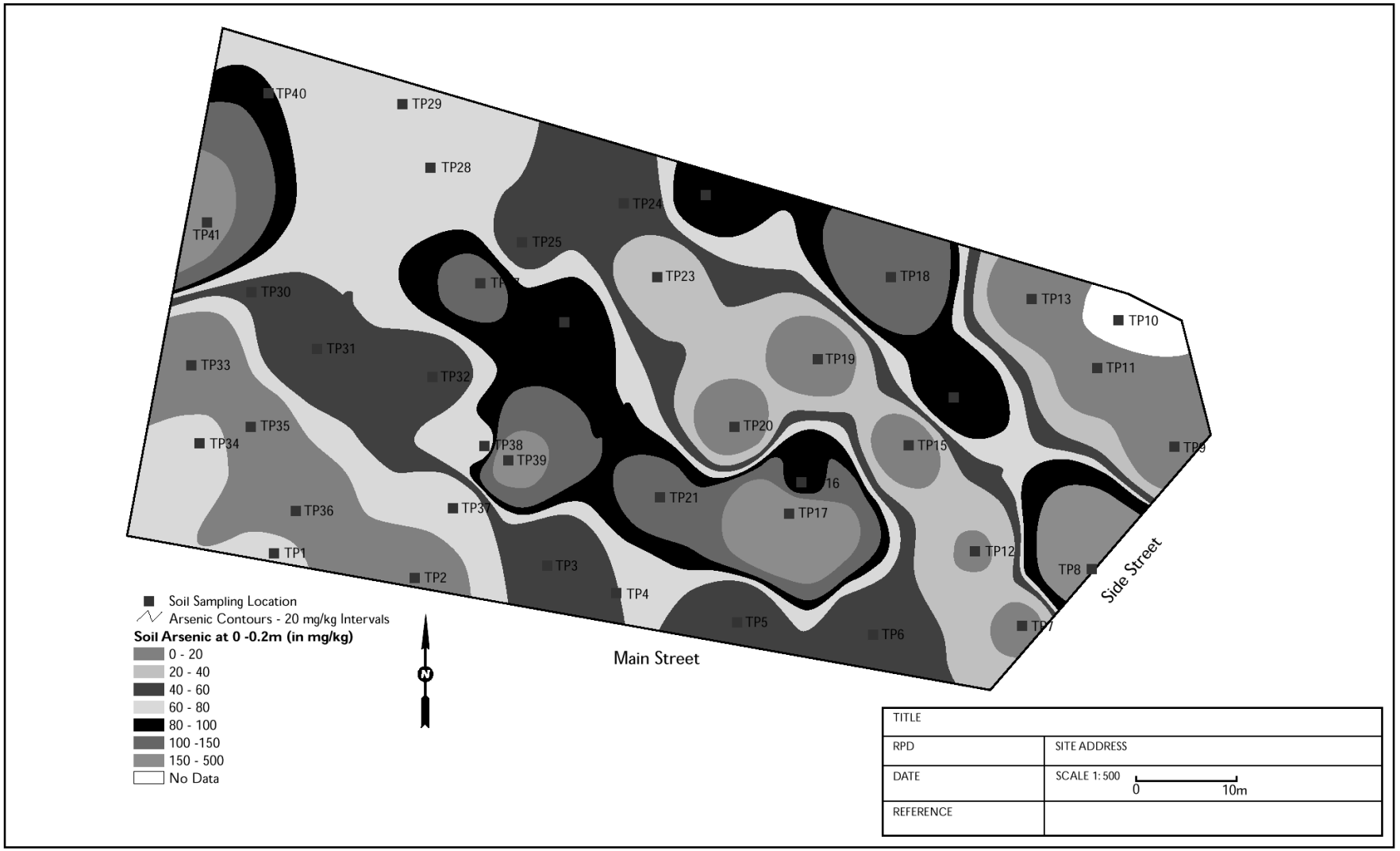


Figure 7. Example Borehole Log – B68

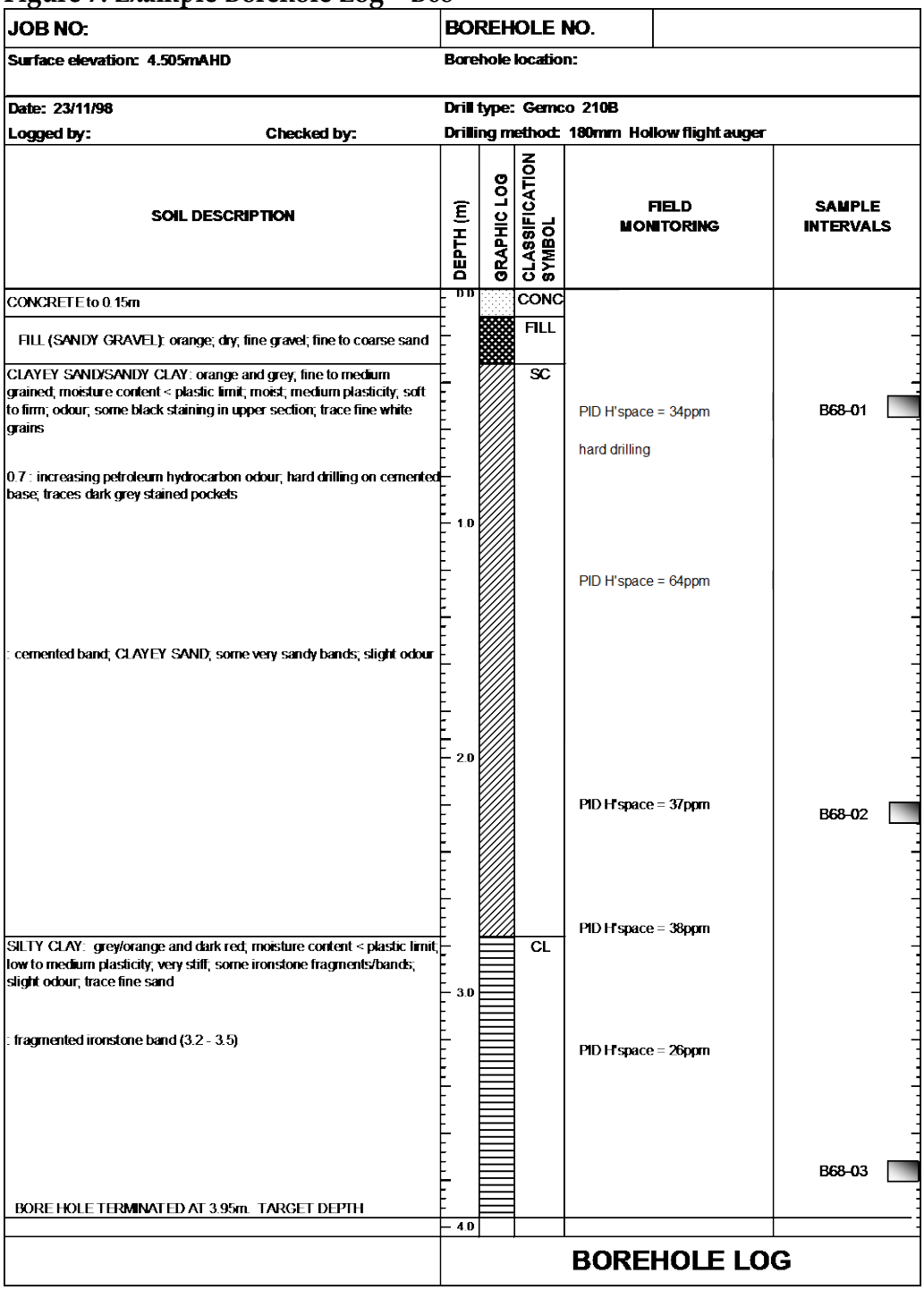


Figure 8. Example Borehole Log – B69

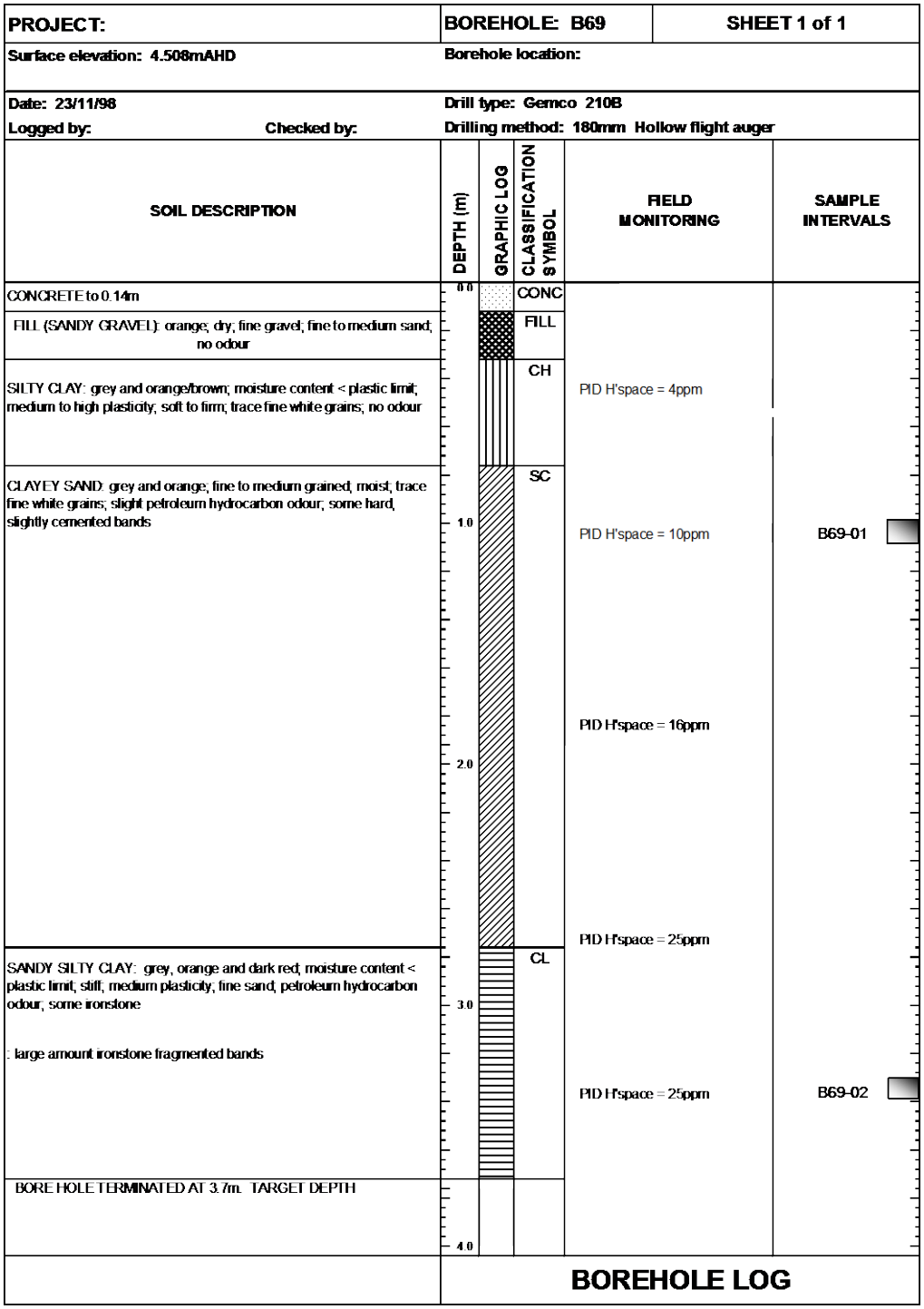


Figure 9. Example Borehole Log – W60 (sheet 1 of 2)

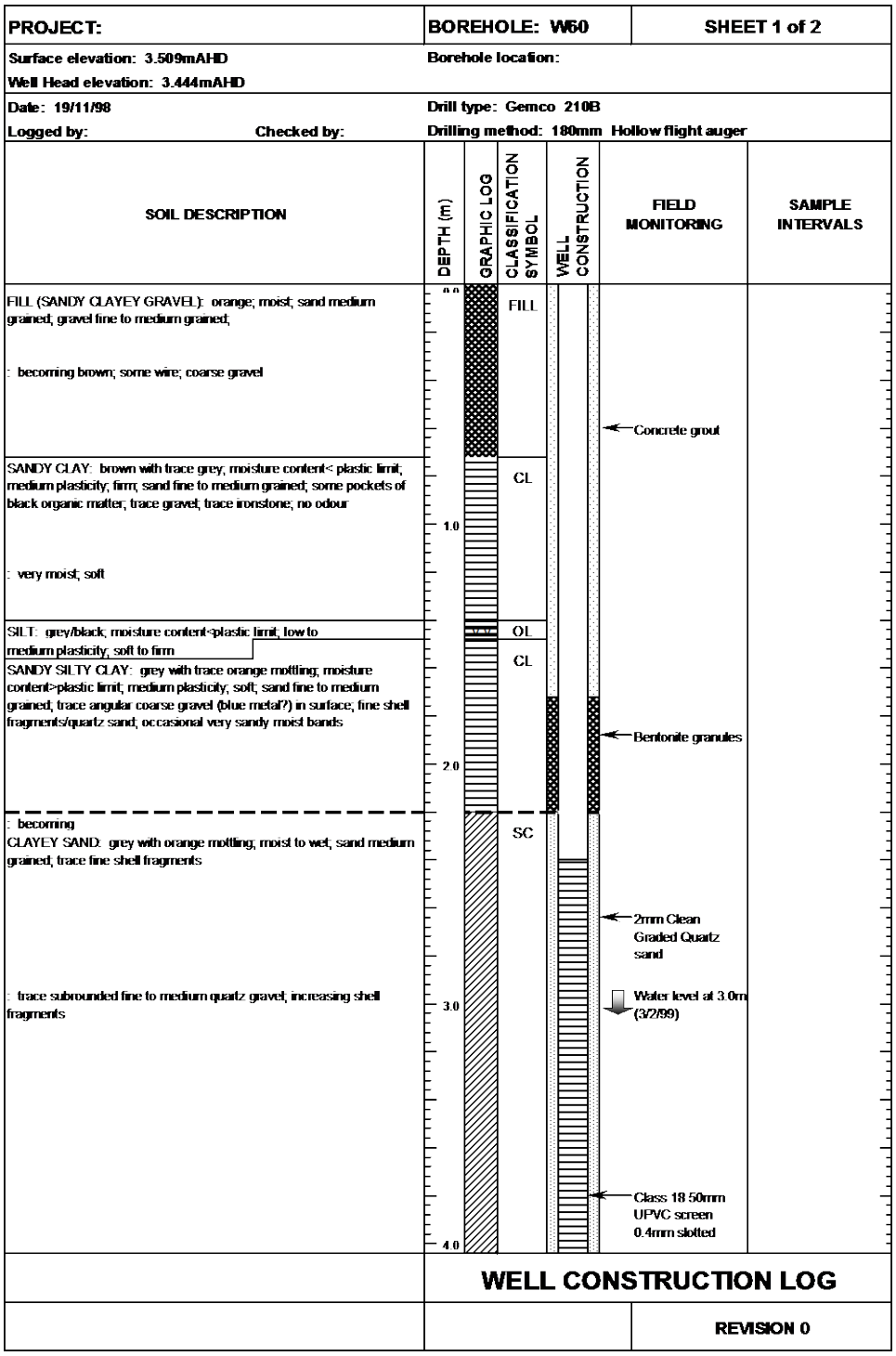
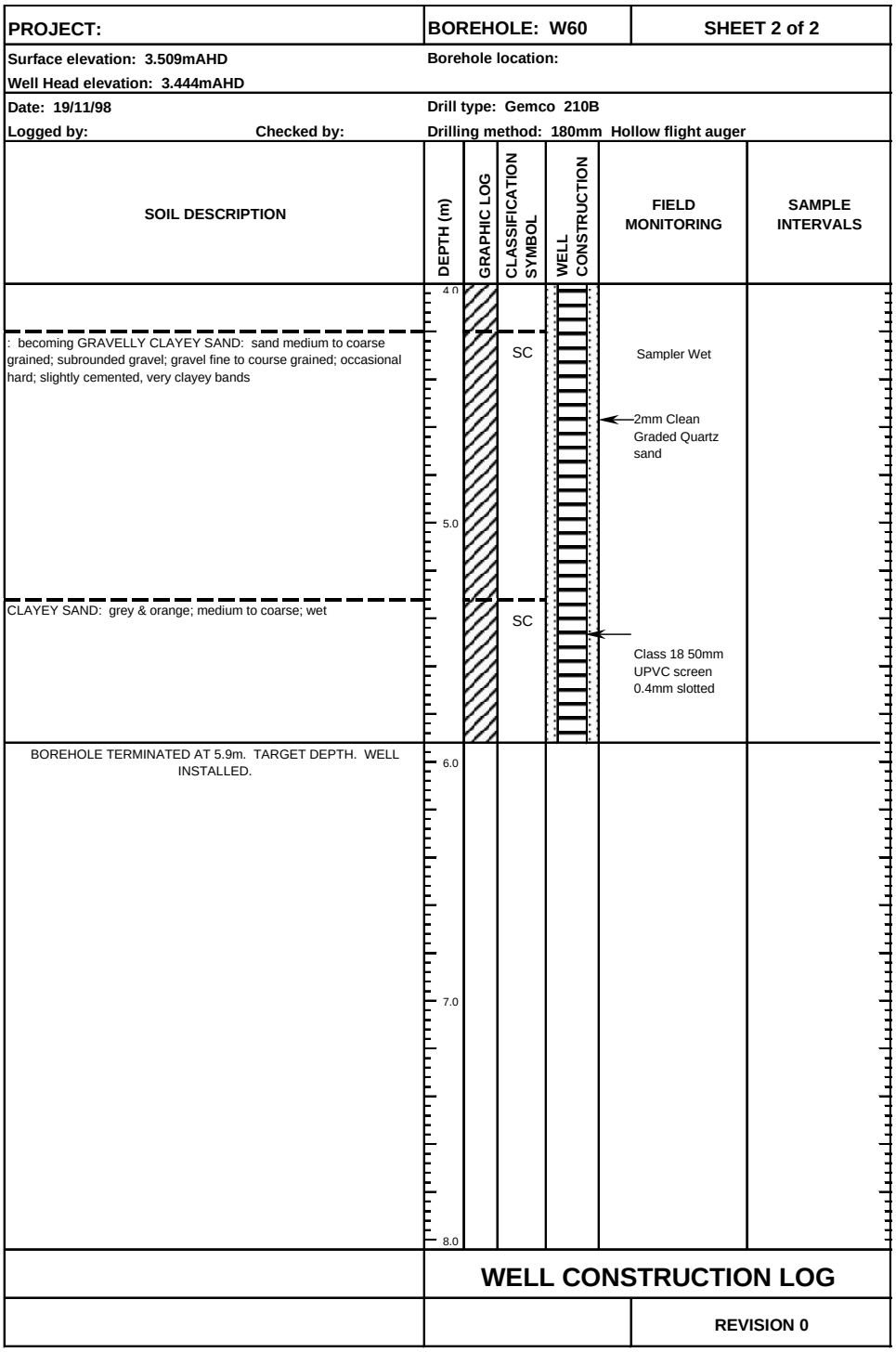


Figure 10. Example Borehole Log – W60 (sheet 2 of 2)



21 Appendix E: Dioxins and dioxin-like compounds

21.1 Background

Dioxins and dioxin-like compounds are chlorinated organic pollutants formed as trace amounts of undesired impurities or by-products in the manufacture of other chemicals such as chlorinated phenols and their derivatives, chlorinated diphenyl ethers, and PCBs (WHO 1989) and combustion of chlorine-containing materials under some conditions. These compounds are one class of persistent organic pollutants (POPs).

The dioxins group comprises 75 polychlorinated dibenzo-p-dioxin (PCDD) congeners and 135 polychlorinated dibenzofuran (PCDF) congeners. There are no known technical uses for PCDD and PCDF (WHO 1989).

Some PCBs also have dioxin-like properties and are included as part of dioxin and dioxin-like compounds. PCBs are a class of organic compounds with 1 to 10 chlorine atoms attached to the biphenyl molecule. There are 209 possible PCB congeners although only 130 were found in commercial PCB mixtures.

The World Health Organization (Van den Berg et al. 2006) identified 29 dioxins and dioxin-like compounds of environmental concern based on similar toxicological profiles. These include 7 PCDD, 10 PCDF and 12 co-planar 'dioxin-like' PCBs. While these substances have similar toxicological profiles, they have differing toxicological potencies. Thus, their concentrations in environmental and biological media are reported using toxicity equivalence (TEQ) relative to a reference compound, which in this case is 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD). The relative toxicity of each compound is expressed as a toxicity equivalency factor (TEF) and the product of the concentration and the TEF for each substance in the mixture results in a TEQ concentration relative to 2,3,7,8-TCDD. The sum of the resultant TEQ for each substance yields a single concentration for the TEQ of the mixture.

The history of TEQ systems is as follows:

- the international TEQ (I-TEQ) was developed largely by the United States Environmental Protection Agency (US EPA) in 1990
- the WHO modified the I-TEQ in 1998 by incorporating 'dioxin-like' PCBs; this was known as the WHO98 TEQ
- in 2005 the WHO98 TEQ system was updated to WHO05 TEQ (Van den Berg et al. 2006).

The WHO 2005 TEQ values are recommended for use in site assessment work involving dioxins and dioxin-like compounds. Further information on the TEF approach and the necessary adjustments required to normalise historical data to WHO 2005 TEQ can be found in enHealth (2012).

21.2 Occurrence of dioxins and dioxin-like compounds

The major causes of soil contamination by dioxin and dioxin-like compounds are from accidental or incidental spillages in the manufacture, transport, storage and use of various chlorinated compounds and past disposal of these compounds. Land uses associated with waste disposal, pulp and paper mills and chemical manufacturing may have resulted in soil contamination by these compounds.

Other industrial sources of dioxin and dioxin-like compounds such as thermal or combustion sources and reservoir sources such as sludges may be less significant as contaminant sources for soil.

Dioxins and dioxin-like compounds also occur naturally and are released into the atmosphere from creation or entrainment during bush fires and from volcanic activity.

21.3 Results from the National Dioxins Program (May 2004)

As part of the National Dioxins Program (NDP), soils from around Australia were collected and analysed for dioxins. Dioxin-like chemicals were found in all but one of the 114 Australian soils sampled, with concentrations ranging from the limit of detection (0.05 pg TEQ g-1 dwt) to 43 pg TEQ g-1 dwt. Note the results of the study are reported based on WHO98 TEQs.

The greatest concentrations of dioxin-like chemicals were found in soils collected near centres of population within the south-east coastal area of Australia, whereas concentrations were consistently low in soils collected from locations in Western Australia and inland areas. Data from the study showed that levels of dioxin-like chemicals in soils from urban and industrial locations were substantially higher relative to agricultural land use and remote locations. This pattern was consistent regardless of whether levels were expressed as toxic equivalents or as concentrations.

Homologue and congener profiles for the PCDD/Fs were strongly dominated by octachlorodibenzo-p-dioxin (OCDD). Similarly, the tetra-heptachlorinated 2,3,7,8-chlorine substituted profiles are dominated by the highest chlorinated PCDD, 1,2,3,4,6,7,8-heptachloro dibenzodioxin. The source or formation processes by which dominance of higher chlorinated congeners could occur remains unresolved despite intensive studies. With regards to the TEQs, on average, more than 80% of the toxic equivalency across soil samples was attributed to 2,3,7,8-PCDD/Fs.

There is no Australian guideline threshold for dioxin-like chemicals in soils. Comparison of concentrations of dioxin-like chemicals in the NDP soil samples against a categorisation derived from German thresholds showed that only 15% of the Australian samples (all but one of which were from urban or industrial locations) exceeded the German derived target value of < 5 pg TEQ g-1 dwt and only one sample exceeded the guideline threshold of acceptability for specific agricultural uses of soil. Australian jurisdictions do not have a generic action or response level for dioxin-like compounds, but may adopt a site-specific investigation and/or response level for dioxins following a site-specific risk assessment.

The concentrations of dioxin-like chemicals in urban and industrial locations sampled as part of the NDP were similar to those reported in previous Australian studies and in the New Zealand Organochlorine Program. On the basis of toxic equivalents, concentrations of dioxin-like chemicals are on average much lower than those reported from many industrial sites internationally and, globally, can be considered among the lowest background concentrations reported in soil from any industrialised nation.

22 Shortened forms

AHD	Australian Height Datum
APHA	American Public Health Association
ASS	acid sulfate soil
Bonded ACM	bonded asbestos-containing materials
BTEX	benzene, toluene, ethylbenzene, and xylenes
CPT	cone penetrometer testing
CRM	Certified reference material
CSM	conceptual site model
DNAPL	dense non-aqueous-phase liquid
DO	dissolved oxygen
DQI	data quality indicator
DQO	data quality objectives
DSI	detailed site investigation
EC	electrical conductivity
ECD	electron capture detector
Eh	Redox potential
FID	flame ionisation detector
FPD	flame photometric detector
GC	gas chromatography
ISO	International Standards Organisation
LIF	laser-induced fluorescence
LNAPL	light non-aqueous-phase-liquid
LOD	limit of detection
LOR	limit of reporting
MAH	monocyclic aromatic hydrocarbon
MIP	membrane interface probe

MSDS	Material Safety Data Sheet
NAPL	non-aqueous-phase liquid
NATA	National Association of Testing Authorities Australia
NPD	nitrogen/phosphorus detector
OC/OP	organochlorine/organophosphorus (pesticide)
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCE	perchloroethene
PCP	pentachlorophenol
PID	photo-ionisation detector
PQL	practical quantitation limit
PSI	preliminary site investigation
QA	quality assurance
QC	quality control
RPD	relative percentage difference
SAQP	sampling and analysis quality plan
SOP	standard operating procedure
SVOC	semi-volatile organic compounds
TCE	trichloroethene
TEF	toxicity equivalence factor
TEQ	toxicity equivalence
TPH	total petroleum hydrocarbon
UCL	upper confidence limit
USGS	United States Geological Survey
UST	underground storage tank
VOC	volatile organic compound
XRF	X-ray fluorescence

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

Compilation start date:	16 May 2013
Includes amendments up to:	*National Environment Protection (Assessment of Site Contamination) Amendment Measure 2013 (No. 1)*

This compilation has been split into 22 volumes

Volume 1: sections 1-6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

Schedule B3

GUIDELINE ON

Laboratory Analysis of Potentially Contaminated Soils

PRECAUTIONARY CAVEAT

This guideline refers to methods of analysis that may require the use of hazardous materials, operations and equipment. It does not, however, address all of the associated real or potential safety problems. It is the responsibility of the user of these guidelines to establish adequate health and safety practices such as those outlined in AS 2243 *Safety in laboratories,* Parts 1–10 as amended (available online at http://www.standards.com.au), and to ensure that any person involved in performing any relevant procedures is adequately trained and experienced.

DISCLAIMER

Any equipment or materials that meet stated specifications and result in satisfactory method performance may be used to carry out the methods referred to in this guideline. Mention of specific trade names, products or suppliers does not constitute endorsement by NEPC of those items, materials, or suppliers over other suitable products or sources. Rather, it is intended to provide users with examples of suitable products and information on those sources that are known to NEPC.

Explanatory note

The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

It aims to ensure accuracy and precision in analytical results from the laboratory analysis of potentially contaminated soils. It should be read in conjunction with Schedule B2 of the NEPM.

The original Schedule B3 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document.

The National Environment Protection Council (NEPC) acknowledges the contribution of a number of individuals and organisations towards the development of these guidelines. In particular, these include Environment Protection Authority (EPA) Victoria (principal author), members of the Environmental Laboratories Industry Group (ELIG), other individual staff members of commercial and government laboratories, members of the Australian Contaminated Land Consultants Association (ACLCA) and individual contaminated site consultants, environmental auditors, officers of the NSW Environment Protection Authority and CRC CARE.

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Laboratory analysis of contaminated soil

1 Introduction

This guideline is applicable to laboratory analysis of contaminated soils for assessment of site contamination and disposal of contaminated soil. It also contains information on the collection of contaminated soil, including storage and handling considerations to enable valid analysis.

Rigorous characterisation and quantification of soil contaminants helps to ensure valid assessments of site contamination. Consistency in analysis and assessment can only be achieved if there is uniformity in procedures including sample collection, storage and handling, pre-treatment, extraction, analytical methodology and data analysis. This document gives guidance on quality control, quality assurance and techniques for sample preparation, extraction and analytical methods.

1.1 Audience

This guideline should be used by people undertaking sampling and analysis of potentially contaminated soils. Its main audience includes but is not limited to:

- laboratory staff
- environmental consultants, site assessors
- regulatory licence holders (e.g. for waste management or other statutory processes)
- custodians of waste/sites containing waste.

1.2 Exclusions

Groundwater analyses are beyond the scope of this Schedule.

1.3 Schedule structure

The Schedule provides guidelines on laboratory analysis of potentially contaminated soils, including:

- the philosophy behind the methods selected
- guidance on quality assurance procedures
- techniques for sample preparation designed to provide confidence and comparability of analytical results.

The Schedule provides analytical methods for potentially contaminated soils and, in particular, a list of methods for analysis of physicochemical properties of inorganic and organic chemicals in soil.

2 Laboratory analysis of potentially contaminated soil

This Schedule provides guidance on analysis of physicochemical properties of soil, including inorganic and organic analytes commonly found in contaminated soils, and on procedures for sample preparation and for quality assurance.

Where possible, the Schedule adopts established 'standard methods' from recognised sources such as Standards Australia, the United States Environmental Protection Agency (US EPA), the American Public Health Association (APHA), the American Society for Testing and Materials (ASTM) and the International Standards Organisation (ISO). When analysis is required for contaminants not included in this guideline, analysts should seek comparable established standard methods. Laboratories should ensure any such methods are validated prior to use.

2.1 Scope

Types of soil analyses for assessment of contaminated sites can fall into three broad categories:

- field measurements that can be performed on-site when collecting samples
- laboratory-based screening tests to determine type of contamination present
- quantitative methods specific to known or expected soil contaminants.

This guideline provides detailed guidance for the third category only. The principal objective is to foster greater standardisation of the test methods most likely to be used in the final assessment of a site. General guidance on the first two categories listed above is available in Section 2.5.

Whenever possible, accreditation to ISO 17025 should be obtained for all analytical procedures and matrices for the analytes of concern, from the National Association of Testing Authorities (NATA) or one of its mutual recognition agreement partners.

2.2 Determinative methods

This guideline specifies procedures for extraction and digestion of common contaminants. The inclusion of determinative procedures for identification and quantification of contaminant concentrations in sample extracts and digests for every analyte is outside the scope.

Descriptions of determinative methods are available for analytes in a range of reference documents including Standards Australia and International standards (US EPA SW-846, APHA 2005, ASTM 2008). In selecting an appropriate method for a particular analyte, the analyst needs to consider the chemical characteristics of the final extract and analyte, and the specificity of the detector.

2.3 Philosophy of methods selected

Soil samples from contaminated sites may be submitted for analysis for various reasons, including to assess:

- potential risks to human and environmental health
- legal/financial risks to individuals and organisations.

These circumstances require highly reliable analyses, with analytical data representative of site condition.

In addition, large numbers of samples from a site may be required to be analysed within a short time; the sooner results are available, the sooner decisions can be made about the need for site remediation or protection of the public and environment from further contamination.

To meet these competing demands for speed and reliability, the extraction/digestion and analytical methods should:

1. be simple—procedures should be easy to follow and to perform, using equipment and reagents generally available in most environmental laboratories.
2. be rapid ideally, extraction/digestion and analysis should be sufficiently rapid and non-labour-intensive to enable a large number of samples to be processed within acceptable turnaround times. This should not be at the expense of meaningful analytical results.
3. be accurate and precise—the test methods listed in these guidelines are regarded as 'reference' procedures, mostly derived from authoritative Australian references or internationally recognised authorities such as US EPA or APHA.
4. They are considered to be sufficiently rigorous and reliable for the assessment of contaminated sites, by virtue of their measured accuracy and precision in validation studies and/or their usage and acceptance as rigorous techniques by the scientific community.
5. be capable of batch or automated analysis—samples should be able to be processed in large batches without being cumbersome; automated analyses are often preferred.
6. be capable of simultaneous analysis—procedures should allow a variety of chemical components to be analysed using aliquots of a single extract per sample. This minimises sample processing time and cost and maximises sample throughput.
7. have an appropriate limit of reporting (LOR)—the selected method should have a limit of reporting, where practicable, no greater than 20% of the relevant soil criteria and validated for a variety of soil matrices, including sand, clay and loams.
8. be safe—safety should never be compromised, especially when undertaking large batch processing and handling soils from contaminated sites.

The analytical methods referenced in this guideline have been selected on the basis of having reliability and where possible, ease of use and efficient data turnaround. The methods primarily measure the potentially mobile or bioavailable fraction of contaminants in soil (not necessarily the total residual contaminant concentrations) because many such residual components (for example, those bound to a silicate matrix) pose little immediate threat to human health or the environment.

2.4 Referenced methods and use of alternative methods

Analysis for regulatory or statutory purposes, or conducted under the principles of this Schedule, should be undertaken by either:

- the methods specified in this guideline (as updated over time)

or

- a method verified to be equivalent in outcome to the relevant referenced method.

Other extraction and determinative methods may be at least as efficient, accurate and precise (as well as possibly faster and less expensive) than those recommended here, including specially designed commercial systems, for example, digestion units, distillation units and auto analysers. However, it is beyond the scope of this guideline to evaluate all possible alternatives.

Where such alternative methods are used, (that is, any methods apart from those specified in this guideline), the user should ensure that the alternative method is at least as rigorous and reliable as the reference method, and either that:

- it has been validated against an appropriate certified reference material (CRM) on the range of soil types and concentrations most likely to be analysed. This requires adequate recovery of analytes using CRMs during method validation, as well as regular participation in national proficiency trials by bodies such as the National Measurement Institute (NMI) or Proficiency Testing Australia (PTA) or other accredited provider

and/or

- it has been verified against quantitative data generated by a laboratory that is accredited for the reference method to ISO 17025 by NATA or one of its mutual recognition agreement partners.

The laboratory should document the method performance verification and make the data available for independent audit.

See Section 3.2 for more guidance on method validation.

2.5 Screening tests

Some screening tests in common usage—including laboratory screening tests and field tests, (for example, field chemical test kits and field analysers)—may be fast and cheap but, by their nature, are less rigorous and reliable than the analytical methods described here. They may be suitable for less exact tasks such as preliminary assessments, mapping contaminant distribution at known contaminated sites or monitoring the progress of site clean-up or remediation programs (refer Schedule B2, Section 7.4).

Data from screening tests is not suitable for detailed assessment of contaminated sites or for validating clean-up. These tasks require a high degree of accuracy and reliability and data should be based upon results from one of the validated analytical tests referenced here, or other methods that have been shown to be at least as rigorous and reliable for the soil matrix in question.

The accuracy and precision of any analysis should be sufficient for the intended purpose. Therefore screening methods should be evaluated for appropriate analyte specificity, repeatability and reproducibility prior to use.

2.6 Confirmation of organic compounds (for non-specific techniques)

Where non-specific analytical techniques are used, (e.g. gas chromatography (GC) or high performance liquid chromatography (HPLC)), the identity of organic compounds should be confirmed by one of the methods detailed in the NATA Field Application Document ISO/IEC 17025 (NATA 2011). These include mass spectrometric detection, variation of the test procedure (e.g. different column stationary phase), another test procedure (e.g. alternative detector) or conversion of the analyte to another compound (e.g. derivation technique).

A mass spectral library match alone is only sufficient for tentative identification. Confirmation is achieved (i.e. no additional confirmatory analysis is required) if GC/MS or HPLC/MS methods are employed *and* standards of the compound are analysed under identical conditions (US EPA SW-846, Method 8000B). A compound identity is then confirmed if *all* of the following criteria (US EPA SW-846: Method 8260B, Method 8270D) are met:

- the intensities of the characteristic ions of the compound in the sample should maximise in the same scan, or within one scan, as that of the reference calibration check standard
- the relative retention time (RRT) of the sample component is within ±0.06 of the RRT of the reference calibration check standard
- the relative intensities of the characteristic ions (see note immediately below) in the sample check standard.

Note: The characteristic ions are generally defined as the three ions of greatest intensity in the preceding calibration check standard.

2.7 Leachability and bioavailability

Some methods for assessing mobility and availability of soil constituents are based on methods designed for agronomic studies and land surveys (e.g. metal availability, as part of soil nutrient assessment) and hence are only applicable to soils expected to have relatively low contaminant concentrations (e.g. background samples or natural soil).

Such methods should be used with caution on contaminated soils, as the high concentrations of analytes in contaminated soil may exhaust the exchangeable capacity of the reagents and lead to false results. These tests have not yet been shown to apply to contaminated soils, and meaningful results can only be obtained from natural soils or background samples.

This Schedule describes two leachability methods for assessing the mobility of common metal contaminants in contaminated site assessments. Other methods available to study mobility of metal ions and nutrients for agronomic reasons are highly specific to the soil type, chemical species, and biota (usually plants) being studied, and are not recommended for generic studies of contaminated soils.

See Section 12 for more discussion of methods to assess leachability of soil contaminants.

2.8 Use of laboratory results

Effective site assessment is dependent on a partnership between the site assessor and the laboratory, to ensure that:

- samples are collected, transported and received by the laboratory in a condition suitable for analysis
- the laboratory understands the information required by the site assessor
- the analyst communicates all relevant information to the site assessor in a timely manner
- the assessor appreciates the uncertainties and limitations associated with the analytical data.

When using the results of laboratory analysis, the site assessor should be aware of the relationship between the property measured by the method (e.g. total or leachable concentrations), the measurement uncertainty and the basis for the derivation of any investigation level or response level with which it is compared.

3 Quality assurance and quality control

3.1 Definitions

The terms ‘quality assurance’ and ‘quality control’ are often misinterpreted. This guideline defines them as follows (ISO 8402–1994):

‘Quality assurance (QA) is all the planned and systematic activities implemented within the quality system and demonstrated as needed to provide adequate confidence that an entity will fulfil requirements for quality.’

This encompasses all actions, procedures, checks and decisions undertaken to ensure the accuracy and reliability of analysis results. It includes the application of routine documented procedures to ensure proper sample control, data transfer, instrument calibration, the decisions required to select and properly train all staff, select and maintain equipment, select analytical methods, and the regular scrutiny of all laboratory systems and corrective actions applied forthwith.

Quality control (QC) is ’the operational techniques and activities that are used to fulfil the requirements for quality’.

These are the QA components that serve to monitor and measure the effectiveness of other QA procedures by comparing them with previously decided objectives. They include measurement of reagent quality, apparatus cleanliness, accuracy and precision of methods and instrumentation, and reliability of all of these factors as implemented in a given laboratory from day to day.

A complete discussion of either of these terms or the steps for implementing them is beyond the scope of this guideline; suffice to say, sound laboratory QA systems and QC procedures are essential. In brief, laboratories should incorporate quality laboratory management systems and participate in accreditation and/or self-audit systems, to ensure reliable results are produced by trained analysts, using validated methods and suitably calibrated equipment, and to maintain proper sample management and recordkeeping systems.

For more information on good laboratory practice and QA procedures, refer to guidance from NATA (Cook 2002) and Standards Australia (AS 2830.1–1985).

3.2 Method validation

This is the process of obtaining data on a method in order to determine its characteristic performance and to establish confidence in the use of that method to provide reliable results. Method validation needs to be performed by each laboratory before that method is adopted and applied to the analysis of actual samples.

It is difficult to obtain complete validation data for all analytes covered in these guidelines due to large variations in soil types and physicochemical properties, and lack of suitable or reliable reference standard materials. For some analytes (e.g. soil pH), conventional validation data has no bearing on method performance between one soil sample and the next; for such analyses, better performance indicators may be obtained through inter-laboratory comparisons.

This guideline recommends certain extraction procedures or, in some cases, complete methods—each laboratory should fully validate each method used (from extraction through to the determinative step) following the principles for quality assurance and method validation described in this Section and other relevant references (US EPA SW-846, APHA 2005-1040B method validation, NATA Technical Note 23, NATA Technical Note 17).

Validation should be performed on the range of soil types most likely to be analysed, or on the most complex soil type likely to be analysed (e.g. clay instead of sand).

All validation steps pertaining to the method should be recorded and retained while the method is being used.

Method performance should be based on extraction of a CRM and/or spiked samples (NATA Technical note 17) or compared with a more rigorous method.

The minimum validation data required are:

- Accuracy Precision
- Limit of detection (LOD)and limit of reporting (LOR)
- Linearity (range over which accurate quantification is expected)
- Uncertainty of measurement (MU).

3.2.1 Accuracy

Accuracy is a measure of the closeness of the analytical result to the 'true' value (NATA Technical note 17). When low analyte concentrations are present the results of a reference method may differ by as much as ±30 % of:

- the expected value of a certified reference material (CRM) of similar matrix; or
- the value obtained by another currently-accepted and separately validated quantitative method for the sample matrix.

This is a particular issue when analyte concentrations are less than 10 times the minimum detectable concentration. Apparent lower recoveries than those specified for the method will occasionally be obtained for CRMs which have been assessed by more rigorous methods involving matrix dissolution. The specific analyte cited in the CRM certificate should match that being determined under this Schedule. For example, if the certified reference values are obtained using aqua regia digest, only the aqua regia method should be applied for comparison with this CRM. Otherwise, an alternative CRM should be used.

3.2.1.1 Percent recovery

This is the most realistic and useful component of the daily quality control performance (APHA 2005), and describes the capability of the method to recover a known amount of analyte added to a sample (in the form of either a laboratory control sample (LCS), matrix spike or surrogate compound spike).

The sample is spiked with a known quantity of the analyte, such that the total of the suspected natural concentration of the analyte plus the spike is within the working range of the method. For compliance monitoring, the spike level should be at or near the regulatory limit, or in the range of 1–5 times the background concentration.

If the background concentration is not known, the spike level may be at the equivalent concentration to the midpoint of the calibration range, or approximately 10 times the LOR in the matrix of interest (US EPA SW-846, Method 3500C).

The longer the spiked analyte can remain in the sample before extraction or digestion, the closer is the simulation to recovering the analyte from the natural sample (except for volatile organics).

Percent recovery is calculated as follows:

$$\text{Per cent recovery} = \frac{c - a}{b} \times 100$$

where:

a = measured concentration of the unspiked sample aliquot
b = nominal (theoretical) concentration increase that results from spiking the sample
c = measured concentration of the spiked sample aliquot

Note: If ‘a‘ is known beforehand, then ‘b‘ should be approximately equal to ‘a‘, and ‘c‘ should be approximately twice that of ‘a‘, for 100% recovery.

In general, at least 70% recovery should be achievable from a reference method; some standard methods state that recoveries for *validated* methods can be lower.

’Recovery of the analyte need not be 100%, but the extent of the recovery of the analyte and internal standard should be consistent, precise, and reproducible’ (FDA 2001).

Further information may be obtained from *General requirements for the competence of testing and calibration laboratories* (ISO 17025, 2005) and *Uncertainty of measurement—Part 3: Guide to the expression of uncertainty in measurement* (ISO/IEC Guide 98-3:2008).

3.2.2 Precision

Precision is a measure of the variation in the method results. It is a combination of two components, repeatability and reproducibility, and is expressed in terms of standard deviation (SD) or relative standard deviation (RSD) of replicate results (APHA 2005).

3.2.2.1 Repeatability

This is a measure of the variation in the method results produced by the same analyst in the same laboratory using the same equipment under similar conditions and within a short time interval (Eaton et al. 2005).

3.2.2.2 Reproducibility

This is a measure of the variation in the method results for the same sample(s) produced by different analysts in different laboratories under different conditions and using different equipment. It measures the 'ruggedness' of the method. Reproducibility data should be obtained as part of the method validation procedure, and are best obtained through inter-laboratory comparisons and proficiency studies.

3.2.2.3 Confidence limit and confidence interval

When results are qualified with standard deviations (SD) or their multiples (for example, ‘x ± SD‘), these are taken to be their confidence limits. This means that a result of 10±4 mg/kg would have confidence limits (CLs) of 6 and 14 mg/kg and a confidence interval (CI) from 6 to 14 mg/kg (APHA 2005). In a normal distribution, 95% of results are found within approximately twice the standard deviation of the mean (e.g. ‘95% CI = x ± 2SD‘). Further clarification of these terms may be found in standard statistics texts.

3.2.3 Limits of detection and reporting

3.2.3.1 Method detection limit

The method detection limit (MDL) is the concentration of analyte which, when the sample is processed through the complete method, produces a response with a 99% probability that it is different from the blank (NATA Technical Note 17). It is derived by:

- analysing at least 7 replicates of a sample with a concentration close to the estimated MDL, and determining the standard deviation
- calculating the MDL as follows

MDL = t * Std Deviation, using a one-sided t distribution where, for 7 replicates, t= 3.14 for 99% confidence levels.

3.2.3.2 Limit of Reporting

The limit of reporting (LOR) is the practical quantification limit (PQL), and is the lowest concentration of an analyte that can be determined with acceptable precision (repeatability) and accuracy under the stated conditions of a test (NATA Technical Note 17). It is calculated as follows (APHA 2005):

LOR = PQL = 5 x MDL

The LOR should be at or below the relevant HIL, HSL or EIL and should be equal to the lowest calibration standard (as expressed in units of mg/kg of soil sample).

3.3 Laboratory Batch QC procedures

The laboratory should adopt, at a minimum, the QC concepts and procedures described below and be able to demonstrate:

- method proficiency within the laboratory
- conformance to the performance characteristics expected of the method
- confidence in the results produced.

Recommended QC procedures for all soil analyses are described in US EPA SW-846 Chapter 1: 'Quality Control'.

3.3.1 Process batch and QC interval

For the purposes of QC requirements and QC monitoring intervals, a laboratory process batch is deemed to consist of up to 20 samples that are similar in terms of matrix and test procedure, and are processed as one unit for QC purposes. If more than 20 samples are being processed, they are considered as more than one batch.

3.3.2 Method blank

This refers to the component of the analytical signal that is not derived from the sample but from reagents, glassware, analytical instruments, etc. It can be determined by processing solvents and reagents in exactly the same manner as for samples. When laboratories report method blanks, the uncorrected result and the method blank should be reported in the same units of measurement.

There should be at least one method blank per process batch.

Method blank data is reported with the primary sample data, thus enabling the site assessor to assess potential method bias for the relevant analytes.

3.3.3 Laboratory duplicate analysis

This is the analysis of a duplicate sample from the same process batch. If possible, the sample selected for duplicate analysis should have an easily measurable analyte concentration. The variation between duplicate analyses should be recorded for each process batch, to provide an estimate of the method precision and sample heterogeneity.

Samples reasonably perceived to contain target analytes should be chosen for the duplicate analyses, though samples with obviously high concentrations of interferents—which will likely require subsequent dilution of sample extracts and raised LORs—should not be used for duplicate analysis. There should be at least one duplicate per process batch, or two duplicates if the process batch exceeds 10 samples.

If results show greater than 30% difference, the analyst should review the appropriateness of the method being used.

Duplicate analysis data is reported with the primary sample data, thus enabling the site assessor to assess method precision for the relevant analytes.

3.3.4 Laboratory control sample

A laboratory control sample (LCS) comprises a standard reference material, or a matrix of proven known concentration or a control matrix spiked with all analytes representative of the analyte class. Representative samples of either material should be spiked at concentrations equivalent to the midpoint of the preceding linear calibration or continuing calibration check, upon which sample quantification will be based. Thus the concentrations should be easily quantified and be within the range of concentrations expected for real samples.

The LCS should be from an independent source to the calibration standard, unless an ICV (independent calibration verification) is used to confirm the validity of the primary calibration.

There should be at least one LCS per process batch.

LCS percent recovery data is reported with the primary sample data, thus enabling the site assessor to assess method accuracy for all targeted analytes, as distinct from method accuracy for site-specific soil samples (see Section 3.3.5 Matrix spikes below). The laboratory should use statistically derived quality control limits from ongoing LCS percent recovery data, for all target analytes, and report such QC limits with the sample data.

3.3.5 Matrix spikes

A matrix is the component or substrate (e.g. water, soil) that contains the analyte of interest. A matrix spike is an aliquot of sample spiked with a known concentration of target analyte. A matrix spike documents the effect (bias) of matrix on method performance.

Matrix spikes should be added to the analysis portion before extraction or digestion and, in most cases, added at a concentration as close as practicable to the corresponding regulatory level (e.g. the relevant HIL or EIL). If the analyte concentration is less than half the regulatory level, the spike concentration may be as low as half the analyte concentration but not less than the LOR.

To avoid differences in matrix effects between sample and spiked sample, the matrix spikes should be added to the same nominal mass of soil sample as that which was analysed for the unspiked sample.

There should be one matrix spike per soil type per process batch.

If the percent recovery of the matrix spike is below the expected analytical method performance, the laboratory should investigate the likely cause and, where a suitable amount of soil mass remains, re-extract and analyse another spiked soil. It may be necessary to use other internal calibration methods (for example, isotope dilution, a modification of the analytical method or alternative analytical methods) to accurately measure the analyte concentration in the extract.

If, after investigation, the matrix spike percent recovery is still below method QC limits then this failed recovery should be reported to the client with an explanation to show the limitations of the method for that particular matrix. An acceptable LCS result may indicate that it is the matrix, not the method, that may be the issue but it is not acceptable to assign poor recovery to matrix effects, without a reasonable investigation.

3.3.6 Surrogate spikes (where appropriate)

Surrogate spikes are known additions to each sample, blank, matrix spike or reference sample, of compounds that are similar to the analytes of interest in terms of:

- extraction efficiency
- recovery through clean-up procedures

- response to chromatography or other determination
- instrumental detector response

but which:

- are not expected to be found in real samples
- will not interfere with quantification of any analyte of interest
- may be separately and independently quantified by virtue of, e.g. chromatographic separation or production of different mass ions in a GC/MS system.

Surrogates provide a means of checking that no gross errors have occurred at any stage of the procedure and which may cause significant analyte losses.

Surrogate spikes are only appropriate for organic analyses, for example, chromatographic methods. Where they are used, they should be added to all samples being analysed and are added to the analysis portion before extraction. Surrogate spike compounds may be deuterated, alkylated or halogenated analogues, or structural isomers of analyte compounds. Surrogate compounds used in analytical methods, normally three per method, should be chosen to monitor the variable method performance of the entire target analyte list.

3.3.7 Internal standards (where appropriate)

Use of internal standards is highly recommended for chromatographic analysis of organics and some inorganic analyses, to check the consistency of the analytical step (e.g. injection volumes, detector response and retention times for chromatographic systems). Internal standards provide a reference against which quantitative data may be corrected for sample-specific variation in instrumental response (for organics analysis only).

For organics analysis, internal standards are normally synthetic deuterated compounds (isotopic analogues) of target compounds. Internal standards are added to each final extract solution after all extraction, clean-up and concentration steps. The addition is a constant amount of one or more compounds with qualities like those listed for surrogates, i.e. a similar instrumental response to that of the target compounds, etc.

Adjustments for variations in injection volume and instrument sensitivity are made by quantifying against the ratio of:

(peak height or area for analyte) : (peak height or area for the referenced internal standard) X (a response factor determined from a preceding calibration standard)

Methods should define specific QC criteria for internal standard response and procedures for analyte quantification where response is observed outside of predefined limits.

3.4 Documentation of validation and QC procedures

All method validation steps (including raw data and data validation assessment) should be recorded and retained while the method is in use. Results of validation procedures should be retained to enable monitoring of method reliability, confidence intervals for analysis results and trends in precision and accuracy over time, or with variation of equipment, source of calibration or analyst.

After completion of analysis of each sample process batch, all documentation relating to the samples and their analysis (including raw data and supporting QC data) should be retained for at least three years (NATA 2011, Section 4.13) so that all relevant information may be easily retrieved. This helps establish chain-of-custody of the sample and traceability of all data, and enables reviewing the analysis during an audit or investigation of a questionable result.

This data retention requirement applies to both hard copy data and data in electronic formats. Laboratories should ensure adequate electronic data storage and backup to ensure data and documentation relating to analyses can be retained.

3.5 Field duplicate and secondary duplicate (split) samples

These field QC processes are implemented by the site assessor rather than the laboratory though laboratories and sample collectors should both be aware of the requirement and purpose.

3.5.1 Field duplicate

Field Duplicate: a blind field replicate sample submitted to the laboratory to provide a check of the precision (repeatability) of the laboratory‘s analysis.

At least 5% of samples (i.e. 1 in 20 samples) should include a larger than normal quantity of soil collected from the same sampling point, removed from the ground in a single action if possible, and mixed as thoroughly as practicable and divided into two vessels. These samples should be submitted to the laboratory as two individual samples and coded separately to avoid identification of their common source.

A similar test of analysis repeatability is provided by re-submission of previously analysed samples, provided the stability of analyte is adequate under the storage conditions used between the two submission dates.

Data for primary and duplicate is collated and reported as a relative percent difference (RPD) of the mean concentration of both samples. If results show greater than 30% difference, a review should be conducted of the cause (e.g. instrument calibration, extraction efficiency, appropriateness of the method used, etc.).

3.5.2 Secondary duplicate

Secondary Duplicate: a blind field replicate sample submitted to a secondary laboratory (inter-laboratory check sample) to provide a check of the analytical performance of the primary laboratory and specifically, the reproducibility of primary laboratory data.

At least 5% of samples from a site should be homogenised and split, with one duplicate sample set submitted to a secondary laboratory (independently accredited for ISO 17025, by NATA or one of its mutual recognition agreement partners) and the remaining samples submitted to the primary laboratory. The duplicate sample should be submitted independently and coded to avoid identity as a duplicate sample. The client should stipulate that each laboratory analyses the split samples for the same analytes using, as far as possible, the same methods recommended in these guidelines.

For comparability of data, there should be minimal delay in sample submission to each laboratory to allow minimum time difference between analyses, especially for analysis of volatile analytes. It is best practice to submit the secondary duplicate (‘check sample‘) directly to the secondary laboratory to minimise time in transit.

Data for primary and duplicate is collated as a relative percent difference (RPD) of the mean concentration determined by both laboratories. Higher variations can be expected for organic analyses compared to inorganic analyses, and for samples with low analyte concentrations or non-homogeneous samples.

If results show greater than 30% difference, a review should be conducted of both laboratories and of the appropriateness of the methods being used.

3.5.3 Replicates for volatile organic compound analysis

For analysis of volatile organic compounds (VOCs), field duplicate and secondary duplicate samples should be created as rapidly as possible by halving the sample and placing each half in a smaller container, compacting and topping up to achieve zero headspace in each, attempting to minimise volatile losses. They should be submitted as soon as possible to the laboratory/ies to prevent loss while in storage or transit.

4 Sample control, preparation and storage

The laboratory should maintain rigorous procedures and documentation for sample control, *from the time the sample is received*. This includes the entire process from registration of the sample through to pre-treatment and sample analysis, sample storage and disposal. Unique identification of each and all portions of every sample is mandatory. Sample integrity should be maintained as far as possible, even after completion of analysis; samples should be stored in controlled refrigeration for at least two weeks after issue of analytical data, to enable repeat analysis in case any anomalous results are observed by the laboratory or the site assessor, subsequent to reporting analytical data.

4.1 Sample preparation – general principles

To obtain reproducible results it is essential that laboratories use standardised procedures when preparing samples. These procedures will not necessarily be the same for each sample but will comprise various combinations of the following treatments:

- separation and removal of extraneous components
- homogenising
- drying
- hand grinding
- sieving
- partitioning (to obtain representative portions).

The combination of treatments applied to any sample will depend primarily on the nature of the analytes of interest. These can be split into three broad categories:

1. non-volatile compounds (including most metals, inorganics and some heavy organics)
2. semi-volatile compounds (many organics, some metals and other inorganics subject to evaporative losses)
3. volatile compounds (such as organic solvents and inorganic gases).

The following sections discuss the individual steps in sample preparation for these three categories.

Throughout the sample preparation step, the analyst should be aware of the potential for any bias to be introduced, and report any bias noted in the results.

WARNING: *Handling potentially contaminated soil and fine dust may present a health hazard. All preparations described in this section should be performed in accordance with work health and safety requirements.*

Asbestos or acid sulfate soils: This Section does not apply to the sampling and handling of soil containing asbestos or acid sulfate materials. For guidance consult *Analysis of acid sulfate soil—dried samples—methods of test* (AS 4969.0-14-2008/2009) and the *Method for the qualitative identification of asbestos in bulk samples* (AS 4964-2004).

4.2 Sample preparation: non-volatiles and semi-volatiles

4.2.1 Separation and removal of extraneous (non-soil) components

Prior to processing the sample (e.g. drying, grinding or mixing), remove any vegetation and other non-soil material (including rocks, gravel, concrete, particles naturally greater than 5 mm) by hand or by sieving, except for samples to be analysed for volatile components, since this process may lead to significant analyte losses. The analyst should confirm with the site assessor or client whether any fraction of the removed material is to be analysed.

Also take a separate weighed portion of the sample to determine moisture content (see Analytical Methods, Section 5 in this Schedule). Report moisture content with the analytical result so that analyte concentrations may be estimated on a 'dry-weight' basis.

As stated previously, the analytes of concern should be the 'available' contaminants, which generally reside on the surface of the soil particles. It is likely that larger particles and rocks will contain, on a weight basis, considerably less contaminant than the smaller particles. In certain circumstances, however, it will be prudent to also analyse the larger particles, preferably separately. The reverse will be likely if contamination of a site has arisen by importation of contaminated screenings or other large particles.

Any material removed for analysis should be weighed and its proportion relative to the entire sample, and its description, recorded. If required, this mass and the description may be included in the analytical report. The significance of the analyte concentration in the soil or fraction of removed material can then be assessed relative to the entire sample composition.

The removed material (including the materials retained on the sieve) should be labelled and retained for possible future analysis.

4.2.2 Homogenising (for non-volatile constituents)

Note: This section only applies to non-volatile samples; samples of volatile contaminants should not be homogenised by stirring, grinding or sieving. Procedures for volatile analytes are described in Section 4.3 below.

Most analytical methods require analysis of only a portion of the sample, sufficient to provide a quantifiable response. The amount of sample received by the laboratory is usually larger than required for a single determination and any additional analyses for QA purposes.

Depending on the analyses required (excluding volatile analysis), a homogeneous test sample is prepared from either the field-moist (i.e. 'as received') or dried sample. The analysis portions are then taken from this test sample.

The sub-sample taken should comprise at least 25% by weight or 200 g of the sample received by the laboratory (laboratory sample), whichever is the smaller, or some other sub-sample that can provide a well-mixed portion representative of the whole sample. It should be thoroughly disaggregated and mixed using a mortar and pestle, or other appropriate method. If no test requiring the original untreated sample will be needed in future, the entire sample may be homogenised; however, it is advisable to keep a portion in the 'as received' state to check, if necessary, that no contamination has occurred during the homogenising process. Described below are the pre-treatment procedures to obtain homogenised field-moist and dry analysis portions.

4.2.3 Preparation of field-moist ('as received') analysis portions

In general, soils to be tested for organic analytes, especially rapidly degradable or otherwise labile contaminants, should not be dried but should be analysed in a field-moist state. If an excess of moisture would affect the extraction efficiency, the sample may be 'dried' by mixing the analysis portion with anhydrous sodium sulfate or magnesium sulfate prior to extraction (US EPA SW-846, Method 3540C).

Field-moist samples will often not be amenable to mechanical grinding or sieving. For those samples that are suitable, the process involves taking at least 25% by weight or 200 g of the laboratory sample, whichever is the smaller (or other sub-sample that can provide a well-mixed portion representative of the whole sample), and thoroughly grinding and mixing by hand in a mortar and pestle, or using other

appropriate techniques, to obtain a homogeneous sub-sample. Equipment should be thoroughly cleaned between samples, or other systems put in place to ensure no cross-contamination.

For most metals and inorganics, better analytical reproducibility is obtained using air-dried soil (see Section 4.2.4 below). However, if the sample is to be analysed for these analytes in the field-moist state and if it is amenable to sieving (for example, sandy loam), it should be passed through a 2 mm plastic sieve to remove large soil particles and other extraneous particles—ensure that the sample contains no solid particles distinctly different from the soil, such as fragments of metal or other unusual particles.

Note: Do not grind samples being analysed for metal contaminants, as this can release natural metals from the interior of soil grains that are not normally available.
Store the treated sample in a suitable container.

Clean all equipment to minimise sample cross-contamination; this can be confirmed by analysing equipment rinsates and/or control samples.

4.2.4 Preparation of dry analysis portions (non-volatiles only)

Air-drying helps to give a representative analysis portion by producing samples amenable to grinding, sieving and splitting. However, air-drying may modify the chemical form of some species and hence affect the results obtained (Adam & Anderson 1983, Bartlett & James 1980, Harry & Alston 1981, Khan & Soltanpour 1978, Leggett & Argyle 1985, Specklin & Baliteau 1989).

The effect of air-drying temperature on analyte modification is not completely understood but in some cases it seems to change the bioavailability or extractability of the analyte. The impact of air-drying on analysis may be more pronounced in certain soil types and in sediments. Therefore, air-drying is only applicable to some methods of soil analysis.

Soils for most metals and some other inorganic analytes can be air-dried, and then sieved. However, the procedure described below is not applicable to analysis of volatile constituents—including volatile metallics such as metallic mercury, methyl mercury or tetraethyl lead—or where analytical methods specifically forbid such preparation (e.g. certain leaching tests). Samples for volatile metallics should be homogenised and sub-sampled in the field-moist state.

Note: Grinding samples will increase surface area and may give higher results.

4.2.4.1 Sample drying

Dry at least 25% by weight or 200 g of the sample, whichever is the smaller, by spreading the soil on a shallow tray of a suitable non-contaminating material, such as plastic or stainless steel. If necessary, break up large clods with a spatula to speed up the drying process. Allow the soils to dry in the air (at <40°C), ideally with the trays placed in a clean air chamber, or a non-contaminating oven at 40 ± 3°C. The relative humidity should be less than 70% to achieve drying within a reasonable time. The sample is dry when the loss in mass of the soil is not greater than 5% per 24 hours (AS 4479.1-1997).

4.2.4.2 Grinding of dry sample

Note: Grinding increases the surface area and can give higher results.

Grinding is not recommended for analysing ‘available‘ metal contaminants, as it can release natural metals inside the soil particles that are not normally available.

Where necessary, crush the dry sample in a mortar and pestle of appropriate material (glass, agate or porcelain) or other suitable grinding apparatus to achieve a particle size appropriate to the analysis. Mix the sample as thoroughly as possible.

Take care to avoid contamination during the grinding process, and clean equipment between each sample to prevent cross-contamination. See below. To evaluate decontamination efficiency, the final wash solution should be sampled and analysed (Barth & Mason 1984); one final wash sample per process batch or 1 in every 10 samples ground, whichever is the smaller. Alternatively, treat a well-characterised control soil sample similarly. If there is significant carry-over due to the grinding process, the results from that process batch may have to be rejected.

WARNING: *Grinding of soils can produce fine dust particles that may present a health hazard if inhaled. Sample grinding, and subsequent handling, should be performed in accordance with work health and safety requirements.*

4.2.4.3 Sieving

Unless impracticable or not recommended for a specific method, the sample portion for analysis should be of a size to pass a 2.0 mm aperture sieve. This may be achieved by grinding, if appropriate.

If small analysis portions (<10 g) are required, or smaller sieve sizes, grind at least 10 g of the <2 mm fraction to pass through smaller mesh sieves (0.15, 0.5 or 1.0 mm sieve size for sample sizes of <1 g, <2 g and 2–9 g respectively).

If another particle size is chosen, this should be consistently used within an analysis regime and reported with analytical results.

4.2.4.4 Partitioning of dry samples to obtain representative analysis portions

The analysis portion of the dry sample should be a representative sample. For sufficiently dry samples, use of a chute splitter (riffler) is recommended, or the entire sample should be thoroughly mixed and divided using the 'cone-and-quarter' technique or by any other suitable sampling apparatus. This equipment should be made of appropriate material (e.g. stainless steel) to avoid contamination.

Cone and quarter technique:

a. Spread soil into thin even layer
b. Divide into four quadrants
c. Combine and mix soil from two opposite quadrants.

Repeat steps a. to c. until required quantity of soil is obtained for analysis (including any replicate analyses and extra portions required for quality assurance purposes).

If using mechanical sample divider, use in accord with the manufacturer's instructions.
Store the remaining homogenised dry sample separately in a glass screw-cap jar or other appropriate vessel.

Note: Mechanical grinding of dry soil, for example, in a ring mill, will mix the sample but use of the cone-and-quarter technique or a mechanical sample divider is preferred, to avoid sub-sampling only the larger particles.

4.2.4.5 Equipment cleaning during sample preparation (including grinding, sieving and homogenising procedures)

Cleaning procedures will vary according to the analyte/s being determined. Minimum procedures include detergent washing followed by rinsing with deionised water and then oven drying. For trace metal analysis, it may be necessary to incorporate soaking in dilute acid followed by deionised water

rinsing. For analysis of organics, equipment will normally need solvent rinsing followed by air drying, prior to homogenising samples.
For quality control, the final wash solution should be sampled and analysed to evaluate the decontamination efficiency (Barth & Mason 1984); one final wash sample per process batch or 1 in every 10 samples ground/sieved/processed, whichever is the smaller. Alternatively, treat a well-characterised control soil sample similarly. If there is significant carry-over due to the grinding/sieving process, the results from that process batch may have to be rejected.

4.2.5 Sample Preparation Summary – Non-volatiles and semi-volatiles

Note: Analysis of volatile contaminants such as C_6–C_{10} fractions should be undertaken prior to any other analysis required from that sample. Sampling and sub-sampling for volatiles should be undertaken as described in Section 4.3 below.

All samples (non-volatile and semi-volatile)

1. Remove vegetation and large stones and other particles (>5 mm) unless they are to be included for bulk analysis. Record proportion by weight with a description of each fraction of material removed.
2. Select at least 25% by weight or 200 g of the laboratory sample, whichever is the smaller, including sufficient amounts for repeat analyses or other analysis on this same sample including moisture content (using field-moist sample).

Field-moist sample analysis e.g. semi-volatiles, analytes for which drying may lead to losses (*Details in S.4.2.3)*	**Dried sample analysis** non-volatiles (*Details in S.4.2.4)*
3. (Intentionally left blank)	3. Dry in oven or air chamber (40±3°C) Sample is dry when the loss in soil mass is not greater than 5% per 24 hours.
4. Grind in clean mortar and pestle to disaggregate soil particles and to produce a homogeneous test sample. – *Where suitable (e.g. for non-volatiles)*	4. Where appropriate *(usually organics, not metals)*, grind to disaggregate the soil particles, using a clean mortar and pestle or using other appropriate techniques, to obtain a homogeneous sub-sample.
5. For 'field-moist'metal samples or other inorganics or non-volatiles that are amenable to sieving (e.g. sandy loam), pass through a 2 mm plastic sieve. Ensure no extraneous particles in sample, otherwise analyse in air dried state.	5. Pass through a mesh sieve (2 mm).
6. Dry a separate sub-sample to determine moisture content (see method in Section 6). Report moisture content with analytical result so that analyte concentrations may be estimated on a 'dry-weight' basis.	6. Weigh the particles >2 mm diameter and set aside for later analysis if required (and to examine for large particles of solid contaminant if necessary).
	7. Partition the (<2 mm diameter) fraction with sample divider (e.g. riffler) or 'cone & quarter' or alternate comparable method. Ensure sufficient soil is obtained to cover all analyses, including repeats and QA. *(See S 4.2.4.4)*
	8. If small analysis portions (<10 g) are required, or smaller sieve sizes, grind at least 10 g of the <2 mm fraction to pass through smaller mesh sieves (0.15, 0.5 or 1.0 mm sieve size for sample sizes of <1 g, <2 g and 2–9 g respectively).

4.3 Volatile analytes – sample collection and preparation

These guidelines generally do not include instructions for sample collection, with the exception of samples collected for volatile analytes, as the sampling method has a direct bearing on the analysis method and reliability of the results. The site assessor may request the laboratory to advise on relevant collection techniques or to supply appropriate equipment.

For samples requiring analysis of volatiles as well as non-volatiles and/or semi-volatiles, it is recommended that additional, separate samples are taken for the various types of analysis, to allow for volatile analysis to be completed and repeated if necessary on samples which have not been homogenised or otherwise inappropriately treated.

4.3.1 Sample collection

Samples should be collected with minimal sample disturbance and handling to avoid evaporative losses, as detailed in AS 4482.2-1999. Ideally, sampling is carried out using a coring device; however if this is not available, an alternative device such as a trowel may be used. In all cases, the sample-taker should ensure that the sample remains intact and the container is filled as full as possible to ensure minimal headspace and void space and evaporation potential. In many cases, taking duplicate samples is recommended to allow sample re-analysis if required (e.g. if contaminant levels are over range).

Since volatiles are easily lost from the ground‘s surface, sampling soil for volatile analysis should not be carried out from the surface layer unless a very recent chemical spill is being investigated.

Where the sample container will be subsequently opened to obtain a sub-sample for analysis, the dimensions of the original sample core taken should be such as to leave a minimum of void space (headspace, and between core and container walls) in the vessel. Where the whole sample is to be purged or extracted without prior opening, this need not apply.

If soils are granular and easily sampled, place sample cores immediately into:

- two or more pre-weighed 40 mL glass volatile organic analysis (VOA) vials with PTFE-lined pierceable silicone septum caps

or

- one or more wide-mouth glass jars (usually 125 mL or 250 mL) with PTFE-lined lid (see Table 4-1, Chapter 4 in SW-846 revision 4, 2007), and sub-sample according to the procedures given below.

If soils are difficult to sample, (for example, highly compacted or hard clays), it is recommended that a minimum of three core samples be placed into pre-weighed 40 mL glass VOA vials marked at a level corresponding to the required sample weight for analysis. One sample may be used for preliminary screening analysis if desired, the others for analysis by purge and trap.

Once samples are taken, ensure that jar or vial closures are free of soil particles before capping. Samples should be sealed and transported to the laboratory as soon as practicable, under suitable cooling aids (preferably ice bricks or in a refrigerated container) to ensure samples start cooling as soon as possible, and they should be stored in a refrigerator (≤6°C) until analysis.

Note 1: The 40 mL VOA vials are particularly effective in conjunction with modified closures (US EPA SW-846, Method 5035), or suitably designed purge and trap instruments, which allow the vial to function as a sparge vessel for purge and trap analysis. This means there may be no need to open the vial to prepare an analysis sample.

Note 2: Using larger containers may be more convenient and possibly result in fewer analyte losses where removal of test sub-samples is required (Ilias & Jaeger 1993).

Note 3: While immersion of samples into methanol on-site is effective in preserving volatile organics (Lewis et al. 1991), such a practice may not be practicable or permissible according to local laws. Handling volatile chemicals in the field, and transporting them, can have work health and safety implications and is not generally recommended unless so advised by the analyst to meet a specific requirement.

4.3.2 Preliminary screening analysis

Laboratories may perform a preliminary screen analysis of soils to prevent contamination of purge and trap equipment by samples with a high contaminant load. This is done by:

- methanol extraction of a core sample in a 40 mL VOA vial. (Methanol is added with a syringe through the septum cap. A portion of the methanol extract is analysed by purge and trap or other method.)

or

- headspace analysis (US EPA SW-846, Method 5021)

or

- hexadecane extraction (US EPA SW-846, Method 3820)

or

- rapidly removing a core sample from a chilled 125 mL/250 mL jar sample and transferring to a vial for analysis as in methanol extraction or headspace analysis above.

After sub-sampling, immediately reseal jar and return to refrigerator storage (≤6ºC).

If analysing whole 40 mL vial samples, note pre-sample weight beforehand and subtract vial weight to determine sample mass.

If screening results indicate a low analyte level suitable for purge and trap analysis, perform this using a second 40 mL vial sample (preferably using the sample vial as the sparge vessel), or take one or more fresh core samples from the larger jar sample.

If screening results indicate a high analyte level, use the data to predict the required sample mass or methanolic extract dilution needed to achieve sample extract concentration at or near the midpoint of the method calibration range. Note that high concentrations, far exceeding the linear range of the method will normally underestimate true sample concentration.

4.4 Sample storage

To maintain sample integrity, samples should be collected and kept in a container that will not increase or reduce the analyte concentration in the sample (i.e. will not add contaminants or leach them). The sooner the sample is analysed after collection, the more closely the analytical result will reflect the condition of the sample at the time of sampling.

Table 1 below lists the recommended containers, maximum holding times and soil conditions for the analytes included in these guidelines. State regulatory agencies may specify different holding times or container types; in which case the jurisdictional requirements should be followed.

Long-term storage of field-moist samples has the disadvantage of allowing faster degradation of analytes via microbial activity, particularly if samples are stored at ambient temperatures. Moist samples should be stored at low temperature (≤6°C) and analysed as quickly as possible.

Air-dried or oven-dried samples can easily absorb moisture in storage. Immediately after homogenising and partitioning, the prepared samples should be transferred into clearly labelled and sealed containers and stored under dry, relatively cool (<18°C) and low light conditions while awaiting analysis.

All unanalysed portions of the sample should be retained for a reasonable amount of time after the dispatch of the analytical report (i.e. at least two months) or until agreed to or advised by the client that they may be discarded.

4.4.1 Holding Times

The holding times in Table 1 are the recommended maximum times before sample extraction. They are taken from a number of sources, and are a guideline only; the integrity of the sample and reliability of results will depend not only on the length of time the sample has been stored, but also on the conditions of sample handling and storage. The effects of storage on sample integrity will be based on the concentration of analyte in the sample, sample temperature, reactions with other compounds that may be present, degradation by microbiological factors, etc. Analytes such as metals and some semi-volatile organics (including PCBs, PAHs) are persistent in the environment and are not likely to change significantly after sampling; analysis slightly outside of these holding times is not likely to cause significant variation in results if samples have been handled and stored correctly. However, all tests should be carried out as soon as practicable after sampling, and according to any jurisdictional requirements.

Table 1. Recommended sample containers, holding times[a] and condition of soil for analysis[b].

Analyte	Container[c]	Maximum holding time	Sample condition
Moisture content - Moisture content only - Moisture correction	 - P, PTFE or G - As for analyte of interest	 - 14 days - As for analyte of interest	 Field-moist Field-moist
pH	P, PTFE or G	24 hours recommended; 7 days allowed	Air-dry or field-moist, depending on analyte of interest
Electrical conductivity	P or G	7 days	Air-dry or field-moist
Organic carbon	G with PTFE-lined cap[d]	28 days	Air-dry or field-moist
Metals (except Mercury & Chromium VI)	P, PTFE or G	6 months	Air-dry or field-moist
Mercury & Chromium VI	P (AW)[d]	28 days. For Cr VI, can hold up to 7 days *post-extraction*	Field-moist
Cation exchange capacity, exchangeable cations	P (AW)	28 days	Air-dry or field-moist
Chloride (water-soluble)	P, PTFE or G	28 days	Air-dry or field-moist
Bromide (water-soluble)	P, PTFE or G	28 days	Air-dry or field-moist
Cyanide	P, PTFE or G[d]	14 days	Field-moist
Fluoride	P or G	28 days	Air-dry or field-moist
Sulfur – total	P, PTFE or G	7 days	Air-dry or field-moist
Sulfate	P, PTFE or G	28 days	Air-dry or field-moist
Sulfide	P or G[e]	7 days	Field-moist
Volatile Organics, except for vinyl chloride, styrene, or 2-chloroethyl vinyl ether	G with PTFE-lined lid/septum[f]	14 days	Field-moist
Vinyl chloride, styrene, 2-chloroethyl vinyl ether	G with PTFE-lined lid/septum[f]	7 days	
Semi-volatile organics, except PCBs, dioxins & furans	G with PTFE-lined lid/septum[g]	14 days[h]	Field-moist
PCBs, dioxins & furans	G with PTFE-lined lid/septum[g]	28 days[h]	Field-moist

Notes

a – Recommended maximum time until sample extraction.

b – Sourced from various references including US EPA SW-846 and Australian and international standards

c – Minimum volume of 250 mL. Containers should be free from contamination, either washed as appropriate or use clean food-grade containers.

P = Plastic G = Glass PTFE= polytetrafluoroethylene AW = Acid-washed SR = Solvent rinsed.

d – Store in the dark.

e – Add sufficient 2M zinc acetate to fully cover surface of solid with minimal headspace; refrigerate (<6°C) (see SW-846 Method 5021, Method 9030B).

f – The vials and septa should be washed with soap and water and rinsed with distilled deionised water. After thoroughly cleaning the vials and septa, they should be placed in an oven and dried at 100°C for approximately one hour. Food-grade containers may also be used without the need for cleaning. Containers should be free from contamination.

g – Containers used to collect samples for the determination of semi-volatile organic compounds should be washed with soap and water then rinsed with methanol (or isopropanol) (see US EPA SW846 Chapter 4 Section 4.1.4 for specific instructions on glassware cleaning). Food-grade containers may also be used without the need for cleaning. Containers should be free from contamination.
h – Once the SVOC is extracted, the extract can be held for 40 days.

4.5 Documentation and reporting

4.5.1 Sample receipt report

Upon receipt of sample, laboratories should issue a Sample Receipt Report detailing the condition of samples, including temperature upon receipt (recorded and reported per individual sample delivery container) and sample preservation status, and chain-of-custody details. As well as commencing a record for the future analytical report, this provides an opportunity for the analyst and sample submitter/site investigator to confirm their requirements.

4.5.2 Analytical report

The analytical report should describe all information and data relevant to the analysis of the sample. This includes:

(a) Requirements for AS ISO/IEC 17025–2005:

- a title
- the name and address of the analytical laboratory (including accreditation details from NATA or one of its mutual recognition agreement partners)
- the analytical report number (a unique identification)
- sample identification (a unique identification for each sample)
- the identity of the test method and any deviations from it analytical results
- a statement of uncertainty where relevant to the validity or application of results or where uncertainty affects compliance to a specification limit, or where requested by the client. (The statement of uncertainty may be implicit in the results presented, e.g. a result may be rounded to the nearest 100 or 1000 indicating an uncertainty of 50 or 500 respectively.)
- any other information specified by the test method or statutory regulation
- a statement of conditions pertaining to reproduction of the report
- the name(s), function(s) and signature(s) or equivalent identification of person(s) authorising the test report
- the date of analytical report issue.

Plus

(b) Other relevant information including:

- the date the sample was received
- the name of the person receiving the sample
- a description of the sample
- the sample condition upon receipt; including temperature upon receipt, any broken or leaking containers, inappropriate containers for the analyte, incorrect storage conditions during transit (e.g. sample temperature control)

- brief description of analytical method and equipment used, including pre-treatment procedures and test conditions where appropriate (e.g. whether the sample was homogenised, ground or sieved)
- confidence interval, QC data and LOR
- any bias noted during the analysis or information on the analysis that may affect the interpretation of the result
- the date/s on which sample analysis was commenced and finalised, and whether extraction and/or analysis was conducted within relevant holding times
- information on all laboratories performing analyses (identify any subcontracted samples).

Where laboratories are required to report analysis blanks, the uncorrected result and the method blank should be reported.

The analytical report should be checked for transcription errors, accuracy in the calculation and expression of results, description of the sample, and whether the QC data meets the acceptable limits for the method. These are all components of the laboratory QA processes.

5 Analytical methods

The following Sections describe the methods recommended to analyse soil from a contaminated site.

It sets out methods for:

physicochemical analyses:
soil moisture
pH
electrical conductivity
cation exchange capacity
water soluble chloride
organic carbon

inorganic contaminants:
metals – including separate methods for mercury, chromium VI
halides – bromides, fluoride
non-metals – cyanide, sulfur compounds

organic contaminants:
volatile organics – including MAHs, VHCs, and vTRHs
semi-volatile organics – including PAHs, PCBs, pesticides (OPPs, OCPs, chlorinated herbicides), phenols, phthalate esters, dioxins and furans, TRH and TRH – silica.

leachability

5.1 Method selection

For some analyte groups, two or more alternative procedures are suggested, which differ in extraction method, clean-up (or lack of), the final determinative step, or a combination of these. The preferred technique will incorporate mass-selective detection and will have more favourable detector selectivity or clean-up steps employed. These methods are less likely to be subject to errors due to interference from co-extracted, non-target compounds. The alternative techniques are known to be useful but would normally require additional independent verification of analyte identity and concentration.

The preferred method is denoted by ‘P‘.

6 Physicochemical analyses

6.1 Soil moisture content

6.1.1 Scope and application

This method (AS 1289.2.1.1-2005) measures the amount of water lost after drying a soil sample (field-moist or air-dried) in an oven (105–110ºC) to constant mass. This allows a correction factor to be obtained to then express chemical concentrations on a dry weight basis.

This drying method will not remove all the water of crystallisation that may be associated with minerals.

The oven-dried moisture content is always determined on a separate representative sub-sample of the soil; the oven-dried sample should not be used for other chemical or physical tests as the drying step may affect results of other tests.

6.2 Soil pH

6.2.1 Scope and application

This method (AS 1289.4.3.1-1997) measures the hydrogen-ion concentration in a soil-water or soil-aqueous calcium chloride suspension and is expressed in pH units.

It is recommended that soil pH be measured whenever other chemical constituents, particularly metals, are to be evaluated, as the pH may have a profound effect on the form and behaviour of chemicals in the soil.

The use of 0.01 M calcium chloride extract is recommended where the soil salt content may influence the pH value (Rayment & Higginson 1992, p. 17). Generally, the pH of the calcium chloride extract is about 0.5 to 1.0 pH units lower than the water extract and gives more accurate values.

The same 1:5 soil–water suspension for electrical conductivity determination may be used for measuring pH but to avoid contamination of the suspension from KCl in the pH probe, electrical conductivity should be analysed first.

When assessing acid sulfate soils, consult *Analysis of acid sulfate soil—dried samples—methods of test— determination of pH_{KCl} and titratable actual acidity (TAA)* (AS 4969.2-2008) and *Analysis of acid sulfate soil—dried samples—methods of test—determination of peroxide pH (pHOX), titratable peroxide acidity (TPA) and excess acid neutralising capacity (ANCE)* (AS 4969.3-2008).

6.2.2 Principle

Soil pH is measured electrometrically on a 1:5 soil–water suspension at approximately 25°C. A 1:5 soil – calcium chloride extract is also provided as an option. The analytical report should state which method was used.

6.3 Electrical conductivity

6.3.1 Scope and application

This method measures the electrical conductivity (EC) of a 1:5 soil–water suspension. Electrical conductivity of the soil is sometimes used to estimate the soluble salt content of a sample (Rayment & Higginson 1992, p.17). A high soluble salt content may have physical detrimental effects on a soil, compromising its agronomic and structural attributes, for example, increasing potential for corrosion of below-ground structures.

The same 1:5 soil–water suspension for pH determination may be used for measuring the electrical conductivity but to avoid contamination, electrical conductivity should be analysed first.

6.3.2 Principle

The electrical conductivity is measured on the aqueous extract of a 1:5 soil–water suspension and recorded in dS/m at 25°C.

6.4 Cation exchange capacity and exchangeable cations

6.4.1 Scope and application

Methods in the following table measure the cation exchange capacity (CEC) of major exchangeable cations/'bases' (Ca^{2+}, Mg^{2+}, Na^{+} and K^{+}) of near-neutral and alkaline soils.

Soil type	pH	Extractant	Salt content*	Method **	Comments
Non-calcareous & non-gypsiferous soils	7.0	1M ammonium chloride	EC< 0.3 dS/m EC> 0.3 dS/m * Based on EC determined on a 1:5 soil–water extract.	15B1 15B2 15B3 ** Soil Chemical Methods	No pre-treatment for soluble salts Pre-treatment: soluble salts are removed using aqueous ethanol and aqueous glycerol. Adjustment: corrected for soluble Na^{+} when NaCl is the dominant soluble salt.

Limitation: These methods are designed to assess the ion-exchange characteristics of soils for land surveys or soil fertility studies, not contaminated soil; they should only be used with natural soils or background samples to give supporting information about the extent of contamination. In other samples the methods are qualitative and the results will be indicators only. Soils heavily contaminated with soluble metals may saturate an extractant's exchangeable sites and may not, by itself, provide a true indication of the soil's exchangeable capacity.

US EPA Method 9081 (US EPA SW-846) can be used on most soils (calcareous and non-calcareous) to measure the total amount of displaced ions from exchangeable sites in soil, compared with the summation of individual ions to express the soil's CEC.

6.4.2 Principle

The soil is shaken with an appropriate extractant under certain conditions to exchange cations in the soil with the chosen extracting ions. The processed extract is then analysed for exchangeable cations including Na^+, K^+, Ca^{2+} and Mg^{2+}, or total CEC.

6.5 Water-soluble chloride

6.5.1 Scope and application

This method measures water-soluble chloride in soil water extracts (1:5 soil–water) (Rayment & Higginson 1992, p.24–25).

6.5.2 Principle

Chloride in soil is extracted in deionised water and the chloride concentration determined by colorimetric analysis or potentiometric titration.

6.5.3 Interferences

Water-soluble colour in the soil may mask the colour change at the endpoint of the titration. If this occurs, the colour can be removed by adding an aluminium hydroxide suspension (APHA Method 4500-Cl). Alternatively, chloride in the water extract can be determined using an ion-selective electrode or ion-chromatography.

6.6 Organic carbon

6.6.1 Scope and application

This determination (Rayment & Higginson 1992, p. 29), also known as the Walkley & Black method, measures the oxidisable organic carbon content of soils and may also be used to estimate their total organic carbon (TOC) content.

Soil organic carbon comprises a variety of carbonaceous materials including humus, plant and animal residues, microorganisms, coal, charcoal and graphite. It does not include carbonate minerals such as calcite or dolomite. Australian soils generally contain less than 5% organic carbon, with higher levels common in surface soils (Rayment & Higginson 1992, p. 29 and p. 32).

The first method listed in Rayment gives poor recoveries of carbonised materials such as graphite, coal, coke and similar coal derivatives. If such materials make up the bulk of the carbon in the sample or if the total organic carbon content is required, an alternative method, which makes use of an external heat source, is recommended (Rayment & Higginson 1992, p. 32).

For organic carbon analysis in acid sulfate soils, consult the Australian standard for the *Analysis of acid sulfate soil—dried samples—methods of test—introduction and definitions, symbols and acronyms*, (AS 4969.0-2008) for relevant definitions and recommended analytical procedures.

6.6.2 Interferences

Overestimation of organic carbon may occur due to large amounts of chloride or metallic or ferrous iron in the sample. Underestimation may result when large amounts of higher oxides of manganese are present. These interferences are common in Australian soils. The potential interferences should be taken into account, particularly when analysing some types of poorly aerated soils.

Since the first method recovers variable proportions of organic carbon actually present in a soil sample (recoveries typically in the range of 65–85%), a correction factor is usually needed. In the absence of a specific correction factor for the soil being tested, a correction factor of 1.3 is commonly used such that:

Total organic carbon (%) = Oxidisable organic carbon (%) x 1.3

7 Metals

7.1 Aqua regia digestible metals

7.1.1 Scope and application

Method AS 4479.2-1997 may be used to obtain extracts from soils for the analysis of most metals and metalloids. Extracts obtained here are not suitable for speciation studies, and analysis of the extracts does not necessarily result in total or bioavailable heavy metal levels in a soil.

Metals extractable by this digestion include metallic components adsorbed on soil particles, complexed by and adsorbed on organic matter, and soluble metal salts. Complete decomposition of the soil is not possible using aqua regia; therefore metals bound within part or most of the silicate matrix may not be fully recovered by this method.

Samples extracted by this method can be analysed for metals by a suitable spectrophotometric method, while accounting for likely interferences, for example, chlorides.

US EPA SW-846 Method 3050B, SW-846 Method 3051A (microwave-assisted digestion) or Method 200.2 may be used as alternatives to this method.

7.1.2 Principle

Boiling aqua regia (3:1 hydrochloric/nitric acid) is used to extract metals from soil. This concentrated acid mixture can extract inorganic metals as well as those bound in organic or sulfide forms.

7.2 Acid digestible metals in sediments, sludges and soils

7.2.1 Scope and application

This method (US EPA SW-846, Method 3050B) may be used to prepare extracts from sediments, sludges and soils for the analysis of metals by various common spectrophotometric techniques.

It can be used to determine the following extracted metals:

FAAS/ICP-AES		GFAAS/ICP-MS
Aluminium	Magnesium	Arsenic
Antimony	Manganese	Beryllium
Barium	Molybdenum	Cadmium
Beryllium	Nickel	Chromium
Cadmium	Potassium	Cobalt
Calcium	Silver	Iron
Chromium	Sodium	Lead
Cobalt	Thallium	Molybdenum
Copper	Vanadium	Selenium
Iron	Zinc	Thallium
Lead		

FAAS = Flame atomic absorption spectroscopy
GFAAS = Graphite furnace atomic absorption spectroscopy
ICP–AES = Inductively coupled plasma atomic emission spectroscopy
ICP–MS = Inductively coupled plasma mass spectrometry

7.2.2 Principle

Two separate digestion procedures, whose extracts are not interchangeable for each other‘s determinations, are provided for determination of the above elements.

7.2.2.1 For FAAS and ICP–AES

The field-moist or dry sample is digested at 95°C in nitric acid and hydrogen peroxide until the volume is reduced, or heated for two hours. Hydrochloric acid is then added and the mixture digested further at heat.
For improved solubility and recovery of antimony, barium, lead and silver, an optional nitric acid/hydrochloric acid digestion step may be used when necessary.

7.2.2.2 For GFAAS and ICP–MS

The field-moist or dry sample is digested at 95°C in nitric acid and hydrogen peroxide until the volume is reduced, or heated for two hours.

7.3 Metals by microwave assisted acid digestion of sediments, sludges, soils and oils

7.3.1 Scope and application

This method (US EPA SW-846, Method 3051A) describes a rapid acid-assisted microwave procedure for digesting sediments, sludges, soils and oils for the analysis of most metals, some metalloids and some non-metals, including (but not limited to):

Aluminium	Cadmium	Iron	Molybdenum	Sodium
Antimony	Calcium	Lead	Nickel	Strontium
Arsenic	Chromium	Magnesium	Potassium	Thallium
Barium	Cobalt	Manganese	Selenium	Vanadium
Boron	Copper	Mercury	Silver	Zinc
Beryllium				

7.3.2 Principle

The sample is digested in concentrated nitric acid, or a mixture of nitric and hydrochloric acids, using microwave heating in a sealed Teflon™ vessel at elevated temperature and pressure. The final digest can be analysed for the element by various common spectrophotometric methods, as described in US EPA Method 3051A.

7.4 Mercury

7.4.1 Scope and application

This method (US EPA SW-846, Method 7471B) may be used as an alternative to methods described in this Schedule for mercury. It uses strong acid digestion (aqua regia) to determine total mercury (inorganic and organic) in soils, sediments, bottom deposits and sludge-type materials.

7.4.2 Principle

Mercury is digested with aqua regia (1:3 nitric acid/hydrochloric acid) at 95°C in the presence of a strong oxidant (potassium permanganate). The digest is then analysed by cold-vapour atomic absorption spectrometry.

CAUTION: Mercury vapour is highly toxic. Use appropriate safety precautions ensuring the mercury vapour is vented into an appropriate exhaust hood or, preferably, trapped in an absorbing medium (e.g. potassium permanganate/sulfuric acid solution).

Note: US EPA Method 1630 may be used for methyl mercury.

7.5 Hexavalent Chromium

7.5.1 Scope and application

This method (US EPA SW-846, Method 3060A) is an alkaline digestion procedure for extracting hexavalent chromium [Cr (VI)] from soluble, adsorbed and precipitated forms of chromium compounds in soils, sludges, sediments and similar waste materials.

7.5.2 Principle

The method uses an alkaline digestion to solubilise both water-soluble and water-insoluble Cr(VI) compounds. The pH should be carefully monitored during digestion to prevent reduction of Cr(VI) or oxidation of native Cr(III).

Cr(VI) in the digest can then be determined colourimetrically by UV visible spectrophotometry (US EPA SW-846, Method 7196), ion chromatography (US EPA SW-846, Method 7199) or other suitable validated methods.

CAUTION: Cr(VI) is highly toxic. Use appropriate safety precautions when handling and disposing of waste.

8 Halides

8.1 Bromide

8.1.1 Scope and application

This method (Adriano & Diner 1982, p. 449) is applicable to the determination of water-soluble bromides in soils, sediments and other solids.

8.1.2 Principle

Most bromides in soils are considerably soluble and can be readily leached using water. In this method, bromide in the sample is extracted into water with a suitable soil:water ratio, which will depend on the bromide species and concentration present. Determination is by suitable APHA methods (APHA Methods 4500-Br and 4110).

8.2 Fluoride

8.2.1 Scope and application

This method is applicable to the determination of total fluoride in plants, soils, sediments and other solids (ASTM D3269-96 (2001), McQuaker & Gurney 1977, ASTM D3270-00 (2006)).

8.2.2 Principle

The sample is fused with sodium hydroxide at 600°C and a solution of the melt is analysed for fluoride.

Note 1: To avoid fluoride losses, do not use glassware to hold sample extracts for long periods; use plasticware as far as possible.

Note 2: This method is not appropriate for samples with high aluminium concentrations, which can cause negative interferences.

9 Non-metals (cyanide and sulfur)

9.1 Cyanide (weak acid dissociable)

9.1.1 Scope and application

Free cyanide (defined as the cyanide ion (CN^-) or hydrogen cyanide (HCN)) is only formed in environments that are dominated by weak cyanide–metal complexes (for example, silver cyanide) and dissolved cyanide complexes. The presence of free cyanide in soil and the potential for formation of HCN is complex and depends on the soil pH, ionic strength and complexation.

The HIL has been derived on the basis of free cyanide and it is recognised that the measurement of free cyanide in soil is difficult, due to instability of free cyanide and also the instability of cyanide metal complexes that can produce free cyanide. A cautious approach, (Department of Resources, Energy and Tourism 2008 and ICMI 2009), is to measure not only the free cyanide but also to measure several other dissociable cyanide species that could furnish free cyanide either by dilution or by other natural processes (refer to US EPA method 9016).

The US EPA Weak Acid Dissociable Cyanide (WAD) method is a surrogate (and conservative) measure of free cyanide, due to the difficulty in measuring free CN.

9.1.2 Principle

The US EPA Weak Acid Dissociable Cyanide (WAD) method measures free cyanide plus the cyanide associated with most unstable metal cyanide complexes. The WAD cyanide refers to any species where cyanide is liberated at pH of 4.5. Such species include HCN (aq) and CN^-, the majority of Cu, Cd, Ni, Zn and Ag complexes. If the WAD result conforms to the HIL then the free cyanide level is also in compliance with the HIL.

9.2 Total sulfur

9.2.1 Scope and application

This method (Tabatabai et al. 1988, Tabatabai 1982) is applicable to the determination of total sulfur in soil, sediment, plants and other solids.

9.2.2 Principle

Sulfur is oxidised to the sulfate form by fusion. The sample is ignited with sodium bicarbonate and silver oxide at 550°C for three hours and the melt is dissolved in acetic acid. The resultant solution is analysed for total sulfur as sulfate (SO_4^{2-}) using a validated method, for example, ion chromatography (APHA Method 4110).

Other decomposition methods for total sulfur analysis, for example, high temperature furnace combustion method, may be used if they can be demonstrated to be at least as rigorous as this method or validated against a CRM (Peverill et al. 2001). Examples include nitric/perchloric acid digestion (Tabatabai & Bremner 1970), sodium hypobromide digestion (Tabatabai & Bremner 1970) and sodium carbonate/sodium peroxide fusion (AOAC 1980).

9.3 Sulfate

9.3.1 Scope and application

These methods are applicable to the determination of soluble and adsorbed inorganic sulfate in soils, sediments and other solids (AS 1289.4.2.1-1997, Rayment & Higginson 1992, ASTM C1580-09, Tabatabai 1982).

9.3.2 Principle

The sample is shaken in a 1:5 soil:water extract, or in some cases a calcium phosphate solution (500 mg phosphorus/L) (Tabatabai 1982) and the resulting extractant subsequently analysed (APHA Method 4110). In the latter, phosphate ions displace adsorbed sulfate while calcium ions depress extraction of soil organic matter and thus eliminate interference from extractable organic sulfur.

9.4 Sulfide

9.4.1 Scope and application

This method (US EPA SW-846, Method 9030B) is suitable for soil samples containing 0.2–50 mg/kg of sulfide. It measures 'total' sulfide, usually defined as acid-soluble sulfide. For soils with significant metal sulfides, total sulfide is defined as both the acid-soluble and acid-insoluble fractions, and both procedures should be employed.

9.4.2 Principle

For acid-soluble sulfides, sulfide is separated out by adding sulfuric acid to a heated sample. For acid-insoluble sulfides (for example, metal sulfides such as CuS, SnS_2) sulfide is separated by suspending the sample in concentrated hydrochloric acid with vigorous agitation.

10 Organics

The table below lists the US EPA SW-846 methods specified for organics analysis. Use the current or most recent version of the method.

Code	Method Title
3540 C	Soxhlet extraction
3541	Soxhlet extraction (automated)
3545 A	Pressurised fluid extraction (accelerated solvent extraction)
3546	Microwave extraction
3550 C	Ultrasonic extraction
3561	Supercritical fluid extraction (of PAHs)
3620C	Florisil® clean-up
3630 C	Silica gel clean-up
3640A	Gel-permeation clean-up
3650B	Acid-base partition clean-up
3660B	Sulfur clean-up
3665A	Sulfuric acid/ permanganate clean-up
3820	Hexadecane extraction and screening for purgeable organics
5021	Volatile organic compounds in soils and other solid matrices using equilibrium headspace
5030B	Purge and trap
5035	Closed-system purge-and-trap and extraction for volatile organics in soil and solid wastes
8015C	Non-halogenated organics by GC
8021B	Aromatic and halogenated volatiles by GC using photo-ionisation and electrolytic conductivity detectors
8041A	Phenols by GC
8061A	Phthalate esters by GC with electron capture detection
8081B	Organochlorine pesticides by GC
8082A	Polychlorinated biphenyls (PCBs) by GC
8121	Chlorinated hydrocarbons by GC: capillary column technique
8141B	Organophosphorous compounds by GC
8151A	Chlorinated herbicides by GC using methylation or pentafluorobenzylation derivation
8260B	Volatile organic compounds by GC/MS
8270 D	Semi-volatile organic compounds by GC/MS
8280 B	Polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) by high-res GC/low-res MS
8290 A	PCDDs and PCDFs by high-res GC/MS
8310	Polynuclear aromatic hydrocarbons (HPLC)
8440	TRPs by infrared spectrophotometry

10.1 Volatile organics

10.1.1 Scope and application

Unless indicated otherwise, the methods described in this section are contained in SW-846. This section lists methods for the following classes of volatile compounds:

MAH
VHC
miscellaneous volatile organic compounds
volatile TRH.

10.1.2 Monocyclic aromatic hydrocarbons

This method is applicable to most volatile compounds with boiling points less than 200°C and which are insoluble or only slightly soluble in water, including (but not limited to):

benzene	ethyl benzene
toluene	xylenes
styrene (vinyl benzene, ethenylbenzene)	propyl benzene
trimethylbenzenes	cumene

10.1.2.1 Preliminary screening

Preliminary screening by headspace analysis (Method 5021) or hexadecane extraction (Method 3820) is appropriate for samples that may contain high concentrations.

Note 1: Headspace analysis may not be as rigorous or reliable as purge and trap (Method 5035) though it is suitable as a 'screening analysis'.

Note 2: Flame ionisation detection (FID) may be substituted for MS or PI detection, for screening purposes but FID is more susceptible to interference and erroneous quantification due to its non-specific response. Accordingly, residues should be confirmed by chromatography on a stationary phase of different polarity or by measurement using MS or PI detector.

10.1.2.2 Sample extraction

Low concentration: (approx <200 μg/kg, for individual compounds)

- purge and trap technique (Method 5035, Method 5030B)

Analysts should determine an appropriate concentration limit and ensure that quantitative results are based on sample concentrations that do not exceed the instrumental range.

High concentration: (≥200 μg/kg, for individual compounds)

- methanol extraction followed by purge and trap technique (Method 5035 or 5030B).

10.1.2.3 Sample clean-up

Not applicable.

10.1.2.4 Sample analysis

The table below lists the US EPA SW-846 methods specified for MAHs.

8021B	GC/PID
8260B	GC/MS

10.1.3 Volatile halogenated compounds (VHC)

This method (Method 5035) is applicable but not limited to analysis of the following volatile halogenated hydrocarbons.

Allyl chloride	Chloromethane	Epichlorhydrin
Benzyl chloride	Chloroprene	Ethylene dibromide
Bis(2-chloroethy)sulphide	1,2-Dibromo-3-chloropropane	Hexachlorobutadiene
Bromoacetone	1,2-Dibromomethane	Hexachloroethane
Bromochloromethane	Dibromomethane	Iodomethane
Bromodichloromethane	Dichlorobenzenes	Pentachloroethane
Bromoform	1,4-Dichloro-2-butene	Tetrachloroethanes

Bromomethane	Dichlorodifluoromethane	Tetrachloroethene
Carbon tetrachloride	Dichlorethanes	Trichlorobenzenes
Chlorobenzene	Dichlorethene	Trichloroethanes
Chlorodibromomethane	Dichloromethane (methylene chloride)	Trichloroethene
Chloroethane	1,2-Dichloropropane	Trichlorofluoromethane
2-Chloroethanol	1,3-Dichloro-2-propanol	Trichloropropanes
2-Chloroethyl vinyl ether	1,3-Dichloropropene	Vinyl chloride
Chloroform		

10.1.3.1 Sample extraction

Low concentration (<200 µg/kg, for individual compounds):

- purge and trap technique (Method 5035, Method 5030B)

Analysts should determine an appropriate concentration limit and ensure that results are based on sample concentrations that do not exceed the instrument range.

High concentration (≥200 µg/kg, for individual compounds):

- methanol extraction followed by purge and trap technique (Method 5035 or 5030B).

10.1.3.2 Sample clean-up

Not applicable.

10.1.3.3 Sample analysis

The table below lists the US EPA SW-846 methods specified for volatile halogenated compounds.

8021B	GC/ELCD
8260B	GC/MS

Note: Preliminary screening by headspace analysis (Method 5021) or hexadecane extraction (Method 3820) is appropriate for samples that may contain high concentrations.

10.1.4 Miscellaneous volatile organic compounds

The following volatile compounds do not fall into the aromatic or chlorinated categories detailed in the sections above, and may be analysed using the methods below.

10.1.4.1 Scope

Analysis of other volatile organics by these methods is not precluded. These methods could also be appropriate for volatile petroleum products (hydrocarbon fuels and solvents).

Acetone	Ethyl methacrylate
Acetonitrile	2-Hexanone
Acrolein	2-Hydroxypropionitrile
Acrylonitrile	Isobutyl alcohol
Allyl alcohol	Light alkanes (e.g. as in petrol)
2-Butanone (MEK)	Malononitrile
t-Butyl alcohol	Methacrylonitrile
Carbon disulfide	Methyl methacrylate
Chloral hydrate	4-Methyl-2-pentanone (MIBK)
bis-(2-Chloroethyl) sulphide	2-Picoline
2-Chloroethyl vinyl ether	Propargyl alcohol
1,2:3,4-Diepoxybutane	b-Propiolactone

Diethyl ether	Propionitrile
1,4-Dioxane	n-Propylamine
Ethanol	Pyridine
Ethylene oxide	Vinyl acetate

10.1.4.2 Sample extraction

Low concentration (<200 µg /kg, for individual compounds):

- purge and trap technique (Method 5035)

Analysts should determine an appropriate concentration limit and ensure that results are based on sample concentrations that do not exceed the instrumental range.

High concentration (≥200 µg/kg, individual compounds):

- methanol extraction followed by purge and trap technique.

10.1.4.3 Sample clean-up

Not applicable.

10.1.4.4 Sample analysis

The table below lists the specified US EPA SW-846 method.

8260B	GC/MS

10.1.5 Total recoverable hydrocarbons - volatile

The term 'TRH' (total recoverable hydrocarbons) is equivalent to the previously used term 'TPH' (total petroleum hydrocarbons), and represents extracted biogenic and petrogenic (petroleum) hydrocarbons by selected solvents. The new terminology has been chosen to avoid confusion with past practices.

TRH fractions are based on newly derived health screening levels (HSL) for petroleum hydrocarbon products.

The vTRH method is applicable but not limited to analysis of volatile hydrocarbons which may be constituents or residues present in or from materials such as the following:

petrol
dry cleaning liquids
industrial solvents
paints, thinners and strippers.

10.1.5.1 Scope

This method, which is a modified version of the 'closed-system purge and trap and extraction for volatile organics in soil and waste samples method' (Method 5035), is applicable to hydrocarbons eluting between nC_6 and nC_{10}. A clean-up procedure is not applicable here since only the volatile components are being investigated.

10.1.5.2 Sample extraction

The table below lists the specified US EPA SW-846 method.

5035	Purge and trap extraction using methanol

10.1.5.3 Extract clean-up

Not required/applicable.

10.1.5.4 Extract analysis

The table below lists the specified US EPA SW-846 method.

8260B	GC/MS or GC/FID. Volatile TRH fraction is specified as nC_6–nC_{10}. Details of GC conditions, standards, and procedure for quantification of fractions as suggested by CRC CARE are listed in Appendix 1.

10.2 Semi-volatile organics

10.2.1 Scope and application

This section lists methods for the following classes of non-volatile compounds:

non-volatile chlorinated hydrocarbons
PAHs by solvent extraction
PAHs by supercritical fluid extraction
organochlorine pesticides (OCPs) and PCBs
OPPs
total recoverable hydrocarbons – non-volatile
phenols
chlorinated herbicides
phthalate esters
dioxins and furans.

Note: Many of these methods use ultrasonic extraction. When this method is used, ensure samples do not overheat; consider putting ice packs into the ultrasonic bath.

This method should not be used for volatile contaminants.

10.2.2 Semi-volatile chlorinated hydrocarbons

This method is applicable but not limited to the analysis of the following semi-volatile chlorinated hydrocarbons.

Benzal chloride
Benzotrichloride
Benzyl chloride
2-Chloronaphthalene
Dichlorobenzenes
Trichlorobenzenes
Tetrachlorobenzenes
Pentachlorobenzenes
Hexachlorobenzene
Hexachlorobutadiene
Hexachlorcyclopentadiene
Hexachloroethane
Hexachlorocyclohexane (alpha-HCH)
Hexachlorocyclohexane (beta-HCH)
Hexachlorocyclohexane (gamma-HCH or Lindane)
Hexachlorocyclohexane (delta-HCH)

10.2.2.1 Sample extraction

The table below lists the specified US EPA SW-846 methods.

3540C	Soxhlet extraction using: acetone/hexane (1:1) or dichloromethane/acetone (1:1)
3550C	Ultrasonic extraction* using: **a.** for low concentration (individual compounds <20 mg/kg): dichloromethane or dichloromethane/acetone (1:1) or hexane/acetone (1:1) or methyl tertiary-butyl ether or methyl tertiary-butyl ether/methanol (2:1). The solvent system chosen should be shown to give optimum, reproducible recovery of analytes spiked into the particular matrix (soil type) under test. Analysts should determine an appropriate concentration limit and ensure that quantitative results are based on sample concentrations that do not exceed the instrument range. **b.** for high concentration (individual compounds >20 mg/kg): dichloromethane or hexane

* Ensure samples do not overheat.

3545A	Pressurised fluid extraction
CRC CARE TPH TECHNICAL WORKING GROUP	End-over-end tumbling/shaking

10.2.2.2 Extract clean-up

3620C	Florisil® column clean-up or
3640A	Gel permeation column clean-up and
3660B	Sulfur clean-up if necessary.

10.2.2.3 Extract analysis

	8121	GC/ECD
(P)	**8270D**	GC/MS

10.2.3 Polycyclic aromatic hydrocarbons by solvent extraction

10.2.3.1 Scope and application

This method is applicable but not limited to analysis of the following polycyclic aromatic hydrocarbons (PAHs):

Naphthalene	Anthracene	Benzo(k)fluoranthene
Acenaphthylene	Fluoranthene	Benzo(a)pyrene
Acenaphthene	Pyrene	Dibenz(a,h)anthracene
Fluorene	Benzo(a)anthracene	Benzo(ghi)perylene
Phenanthrene	Chrysene	Indeno(123-cd)pyrene
Benzo(b)fluoranthene		

10.2.3.2 Sample extraction

The tables below list the specified US EPA SW-846 methods.

3540 C	Soxhlet extraction using: acetone/hexane (1:1) or dichloromethane/acetone (1:1)
3550 C	Ultrasonic extraction* using: **a.** for low concentration (individual compounds <20 mg/kg): dichloromethane or dichloromethane/acetone (1:1) or hexane/acetone (1:1) or methyl tertiary-butyl ether or methyl tertiary-butyl ether/methanol (2:1). The solvent system chosen should be shown to give satisfactory, reproducible recovery of analytes spiked into the particular matrix (soil type) under test. Analysts should determine an appropriate concentration limit and ensure that results are based on sample concentrations that do not exceed the instrument range. **b.** for high concentration (individual compounds >20 mg/kg: dichloromethane.

* Ensure samples do not overheat.

3545A	Pressurised fluid extraction using dichloromethane/acetone (1:1).
CRC CARE TPH TECHNICAL WORKING GROUP	End-over-end tumbling/shaking

10.2.3.3 Sample clean-up

3630C	Silica gel column clean-up

The extract should be concentrated using a Kuderna Danish (KD) evaporator or other suitable method and solvent exchanged to cyclohexane, prior to clean-up.

10.2.3.4 Extract analysis

(P)	**8270D**	GC/MS (capillary column)
	8310	HPLC with UV* and fluorescence* detectors

*Due to the high probability of interferences using these less specific detectors, clean-up of extracts using Method 3630C will normally be necessary. Protocols for verification of analyte identities should be developed when Method 8310 is used.

10.2.4 Polycyclic aromatic hydrocarbons by supercritical fluid extraction

PAHs / supercritical fluid extraction (SFE)

3561	SFE of PAHs

10.2.4.1 Sample extraction

The tables below list the specified US EPA SW-846 methods. The extraction is a three-step process using:

supercritical CO_2
supercritical CO_2 plus water and methanol modifiers
supercritical CO_2 (to purge system of modifiers).

Collection of SFE extract:
either

octadecylsilyl (ODS) trap with elution of trap using:

a. acetonitrile/tetrahydrofuran (50/50) for HPLC determination, or
b. DCM (dichloromethane)/isooctane (75/25)

or

solvent trapping in solvent system (a) or (b) above, or another system validated by the laboratory.

10.2.4.2 Extract clean-up

The table below lists the specified US EPA SW-846 methods.

3620C	Florisil® column clean-up or
3640A	gel permeation column clean-up and
3660B	sulfur clean-up if necessary

10.2.4.3 Extract analysis

The table below lists the specified US EPA SW-846 methods.

(P)	**8270D**	GC/MS
	8310	HPLC with UV and Fluorescence detectors

10.2.5 Organochlorine pesticides and polychlorinated biphenyls

10.2.5.1 Scope and application

This method is applicable but not limited to analysis of the following organochlorine pesticides: (OCPs) and polychlorinated biphenyls (PCBs):

Aldrin	Endrin
HCB	Endosulfan (alpha-, beta- and sulfate)
alpha-HCH, beta-HCH	Heptachlor, Heptachlor epoxide
gamma-HCH (lindane), delta-HCH	Mirex
Chlordane (alpha, beta chlordane and oxychlordane)	Methoxychlor
DDD, DDE, DDT	Toxaphene
Dieldrin	PCBs (Aroclor 1016, 1221, 1232, 1242, 1248, 1254, 1260, 1262).

10.2.5.2 Sample extraction

The table below lists the specified US EPA SW-846 methods.

3540C	Soxhlet extraction using: acetone/hexane (1:1) or dichloromethane/acetone (1:1).
3550C	Ultrasonic extraction* using: **a.** for low concentration (individual compounds <20 mg/kg): dichloromethane or dichloromethane/acetone (1:1) or hexane/acetone (1:1) or methyl tertiary-butyl ether or methyl tertiary-butyl ether/methanol (2:1). The solvent system should be chosen to give optimum reproducible recovery of analytes spiked into the matrix (soil type) under test. Analysts should determine an appropriate concentration limit and ensure that quantitative results are based on sample concentrations that do not exceed the instrumental range. **b.** for high concentration (individual compounds >20 mg/kg): dichloromethane or hexane
CRC CARE TPH TECHNICAL WORKING GROUP	End-over-end tumbling/shaking

* Ensure samples do not overheat.

Note: Extract clean-up. Methods for the clean-up of some co-extracts/analytes are suggested below. The tables below list the specified US EPA SW-846 methods.

For samples of biological origin or containing high molecular weight materials:

3640A	Gel permeation column clean-up

If only PCBs are to be determined:

3665A	sulfuric acid/permanganate clean-up followed by:
3620C	Florisil® column clean-up or
3630C	silica gel fractionation.

If both PCBs and pesticides are to be measured:

3630C	silica gel fractionation

If only pesticides are to be determined:

3620C	Florisil® column clean-up and
3660B	sulfur clean-up.

Elemental sulfur may interfere with determination of pesticide and PCBs. This should be removed using Method 3660B: sulfur clean-up, which uses reaction with reactive copper.

10.2.5.3 Extract analysis

The table below lists the specified US EPA SW-846 methods.

8081B	GC/ECD (capillary column)
8082A	GC/ECD or GC/ ELCD
8270D	GC/MS (capillary column)

10.2.6 Organophosphorus pesticides

10.2.6.1 Scope and application

This method is applicable but not limited to the analysis of the following organophosphorus pesticides (OPPs):

Atrazine
Azinphos methyl
Bolstar (Sulprophos)
Chlorpyriphos
Coumaphos
Demeton, O and S
Diazinon
Dichlorvos
Dimethoate
Disulfoton.
EPN
Ethoprop
Fensulfothion
Fenthion
Malathion
Merphos
Mevinphos
Monocrotophos
Naled
Parathion ethyl
Parathion methyl
Phorate
Ronnel
Sulfotep
TEPP
Stirophos (Tetrachlorvinphos)
Tokuthion (Protothiophos)
Trichloronate

10.2.6.2 Sample extraction

The table below lists the specified US EPA SW-846 methods.

3540C	Soxhlet extraction using: acetone/hexane (1:1) or

	dichloromethane/acetone (1:1).
3550C	Ultrasonic extraction* using: **a.** for low concentration (individual compounds <20 mg/kg): dichloromethane or dichloromethane/acetone (1:1) or hexane/acetone (1:1) or methyl tertiary-butyl ether or methyl tertiary-butyl ether/methanol (2:1). The solvent system chosen should be shown to give satisfactory, reproducible recovery of analytes spiked into the particular matrix (soil type) under test. Analysts should determine an appropriate concentration limit and ensure that quantitative results are based on sample concentrations that do not exceed the instrumental range. **b.** for high concentration (individual compounds >20 mg/kg): dichloromethane or hexane.
CRC CARE TPH TECHNICAL WORKING GROUP	End-over-end tumbling/shaking

* Ensure samples do not overheat

10.2.6.3 Extract clean-up

This step is not usually necessary. The tables below list the specified US EPA SW-846 methods.

3620C	Florisil® column clean-up. (Analyst should verify the use of this step for the pesticide of interest, as low recoveries have been reported for certain OPPs.)
3660B	Sulfur clean-up

10.2.6.4 Sample Analysis

8141B	GC/ FPD or GC/ NPD
8270D	GC/MS

10.2.7 Total recoverable hydrocarbons

The term total recoverable hydrocarbons (TRH) is equivalent to the previously used total petroleum hydrocarbons (TPH), and represents extracted biogenic (biological) and petrogenic (petroleum) hydrocarbons by selected solvents. The term has been chosen to avoid confusion with past practices. Where significant levels of non-petroleum hydrocarbon interferences are suspected, a silica gel clean-up is recommended, in which case the analytical report should include a clear statement about this and any relevant interpretation of the chromatogram; the analysis should be referred to as ‘TRH–silica‘. See Section 11.2.8.1.

When soil contains high levels of non-petroleum-based hydrocarbons (e.g. from heavy manure, compost additions or polymeric materials), inspection of the TRH–silica chromatogram may reveal that the silica gel clean-up was not sufficient to remove the non-petroleum-based hydrocarbons from the sample and resolve interferences. This can result in false positive results for petroleum-based hydrocarbon determination. In these cases it is recommended that GC–MS—or other appropriate analytical method, e.g. nuclear magnetic resonance (NMR)—is applied to the extract or a silica gel cleaned sample to improve accuracy.

The analyst should discuss any unusual profiles—and the possibility of interferences from high biogenic hydrocarbon—with the site assessor, before issuing the report.

Where it can be determined that compounds in the sample are of non-petroleum origin, the results should be adjusted as far as practicable to finalise the level of petroleum-based hydrocarbon in the sample.

TRH fractions are based on those used to derive the Health Screening Levels (HSLs) for petroleum hydrocarbon compounds (See Schedule B1).

The TRH method is applicable but not limited to the analysis of hydrocarbons that may be constituents or residues present in or from materials such as the following:

kerosene
diesel
aviation fuel
lubricating oil
heating oil/marine fuel
dry cleaning liquids
tars
gasworks wastes
industrial solvents
paints, thinners and strippers.

10.2.8 Total recoverable hydrocarbons by solvent extraction

10.2.8.1 Scope

This method is for the determination of semi-volatile TRH in soil by gas chromatography applicable to hydrocarbons eluting between $>nC_{10}$ and nC_{40}. The method extracts major hydrocarbons such as aliphatic linear, branched and cyclic hydrocarbons, PAHs, and other compounds in the boiling point range up to nC_{40}. If PAHs are suspected of being present in a sample, target analysis techniques are preferred for risk assessments.

Hydrocarbons with boiling points less than nC_{10} (volatiles) or greater than nC_{40} (heavy petroleum compounds) will not be quantitatively determined using this method.

TRH can be defined as those compounds that are extractable into the solvent and elute from a GC column under the conditions specified in the test method. Hydrocarbon interferences such as vegetable and animal oils and greases, organic acids, chlorinated hydrocarbons, phenols and phthalate esters will also be measured. The presence of petroleum hydrocarbons in TRH may be confirmed by clean-up of the extract with silica gel. However, silica gel clean-up may not completely remove non-petroleum hydrocarbon interferences of biological origin.

10.2.8.2 Sample Extraction

The table below lists the specified US EPA SW-846 methods.

3540C	Soxhlet extraction using: dichloromethane/acetone (1:1).
3550C	Ultrasonic extraction* using: dichloromethane/acetone (1:1)
3545A	Pressurised fluid extraction (PFE) using: dichloromethane/acetone (1:1) or hexane/acetone (1:1).
CRC CARE TPH TECHNICAL WORKING GROUP	End-over-end tumbling/shaking using: dichloromethane/acetone (1:1) This procedure, specified for TRH, has evolved from work carried out by CRC CARE (2009). Although all components of it are in common use, no validation data are currently available for the entire method.

* Ensure samples do not overheat.

The solvent system chosen should be shown to give optimum, reproducible recovery of analytes spiked into the particular matrix (soil type) under test.

10.2.8.3 Extract clean-up

(Recommended when there is significant amount of non-petroleum hydrocarbon interferences, to avoid reporting false positive results.)

The table below lists the specified US EPA SW-846 methods.

3630C	Silica gel clean-up
	Clean-up is necessary if the extract contains interfering quantities of polar non-petroleum compounds evidenced by a GC/FID profile or GC/MS analysis uncharacteristic of petroleum hydrocarbons. Clean-up may be achieved after solvent exchange to hexane or other suitable solvent. Clean-up can be either carried out using a silica gel column or by shaking a solvent extract with loose silica gel. Silica gel activity may have to be adjusted by water addition for optimum retention of PAHs and TRH in the extract. US EPA Method 3630C gives conditions for silica gel clean-up of PAHs.

10.2.8.4 Extract Analysis

The table below lists the specified US EPA SW-846 methods.

8015B	Specifies GC/FID conditions up to nC28 alkanes
	GC/FID conditions for >nC28 alkanes can be obtained from 8270D or in Appendix 1 (CRC CARE method). Due to the non-specific response of GC/FID, identities of unusual mixtures and predominant individual compounds should be confirmed using GC/MS. TRH fractions are specified as $>C_{10}–C_{16}$, $>C_{16}–C_{34}$ and $>C_{34}–C_{40}$.

	Details of GC conditions, standards, and procedure for quantification of fractions are listed in Appendix 1. Where clean-up with silica gel has occurred it should be clearly stated on the report. The result will be reported as TRH–silica.

10.2.9 Phenols

10.2.9.1 Scope and application

This method is applicable but not limited to the analysis of the following phenolic compounds:

Phenols
Chlorophenols, Dichlorophenols, Trichlorophenols
Tetrachlorophenols, Pentachorophenol
Cresols (methyl phenols)
Nitrophenols, Dinitrophenols

10.2.9.2 Sample extraction

The table below lists the specified US EPA SW-846 methods.

3540C	Soxhlet extraction using: acetone/hexane (1:1) or dichloromethane/acetone (1:1) plus exchange solvent (2-propanol).
3545A **3550C**	Pressurised fluid extraction (PFE) Ultrasonic extraction* using: **a.** for low concentration (individual compounds <20 mg/kg): dichloromethane or dichloromethane/acetone (1:1) or hexane/acetone (1:1) or methyl tertiary-butyl ether or methyl tertiary-butyl ether/methanol (2:1) and exchange solvent (2-propanol). The solvent system chosen should be shown to give satisfactory, reproducible recovery of analytes spiked into the particular matrix (soil type) under test. Analysts should determine an appropriate concentration limit and ensure that quantitative results are based on sample concentrations that do not exceed the instrumental range. **b.** for high concentration (individual compounds >20 mg/kg): dichloromethane.

* Ensure samples do not overheat.

CRC CARE TPH TECHNICAL WORKING GROUP	End-over-end tumbling/shaking.

10.2.9.3 Extract clean-up

The tables below list the specified US EPA SW-846 methods.

3630C	Silica gel column clean-up (for samples derived for GC/ ECD determination).
3640A	Gel permeation clean-up
3650B	Acid/base partition extraction (it is recommended that all extracts undergo this clean-up): pentafluorobenzyl bromide derivatisation (for GC/ECD analysis) phenols by GC/capillary column technique

Extract Analysis

	8041A	GC/FID GC/ECD (after derivatisation, if interferences prohibit proper analysis by GC/FID)
(P)	**8270D**	GC/MS

Note: GC analysis of some un-derived phenols is difficult (e.g. chlorinated and nitro compounds). The GC injector port should be clean and adequately silanised.

10.2.10 Chlorinated herbicides

10.2.10.1 Scope and application

The method described below for chlorinated herbicides (by gas chromatography) is applicable but not limited to the determination of:

2,4-D	DCPA diacid	5-Hydroxydicamba
2,4-DB	Dalapon	MCPA
2,4,5-T	Dicamba	MCPP (mecoprop)
2,4,5-TP (Silvex)	3,5-Dichlorobenzoic acid	Pentachlorophenol
Acifluoren	Dichlorprop	Picloram
Chloramben	Dinoseb	

10.2.10.2 Sample extraction

The tables below list the specified US EPA SW-846 methods.

8151A	The soil is extracted and may be derived with diazomethane or 2,3,4,5,6-pentafluorobenzyl bromide.
3545A	Pressurised fluid extraction (PFE)

10.2.10.3 Extract clean-up

3650B	Acid/base partitioning step if required

10.2.10.4 Extract analysis

8151A	GC/ECD
8270D	GC/MS

10.2.10.5 Extract analysis

8151A	GC/ECD
8270D	GC/MS

10.2.11 Phthalate esters

10.2.11.1 Scope and application

This method is applicable but not limited to analysis of the following phthalate esters:

Bis (2-n-butoxyethyl) phthalate
Bis (2-ethoxyethyl) phthalate
Bis (2-ethylhexyl) phthalate
Bis (2-methoxyethyl) phthalate
Bis (4-methyl-2-pentyl) phthalate
Butyl benzyl phthalate
Diamyl phthalate
Di-n-butyl phthalate
Dicyclohexyl phthalate
Diethyl phthalate
Dihexyl phthalate
Diisobutyl phthalate
Dimethyl phthalate
Dinonyl phthalate
Di-n-octyl phthalate
Hexyl 2-ethylhexyl phthalate

10.2.11.2 Sample extraction

The table below lists the specified US EPA SW-846 methods.

3545A	Pressurised fluid extraction (PFE)
3540C	Soxhlet extraction using: acetone/hexane (1:1) or dichloromethane/acetone (1:1).
3550C	Ultrasonic extraction* using: a. for low concentration (individual compounds <20 mg/kg): dichloromethane or dichloromethane/acetone (1:1) or hexane/acetone (1:1) or methyl tertiary-butyl ether or methyl tertiary-butyl ether/methanol (2:1). The solvent system chosen should be shown to give satisfactory, reproducible recovery of analytes spiked into the particular matrix (soil type) under test. Analysts should determine an appropriate concentration limit and ensure that results are based on sample concentrations that do not exceed the

	instrumental range. b. for high concentration (individual compounds >20 mg/kg): dichloromethane or hexane.

* Ensure samples do not overheat.

CRC CARE TPH TECHNICAL WORKING GROUP	End-over-end tumbling/shaking

10.2.11.3 Extract clean-up

Note: The analyst should verify that quantitative recovery of phthalates is achieved for whichever clean-up procedure used.

The tables below list the specified US EPA SW-846 methods.

3620C	Florisil® column clean-up
3640A	Gel-permeation clean-up

10.2.11.4 Extract analysis

8061A	GC/ECD
8270D	GC/MS

10.2.12 Dioxins and furans

10.2.12.1 Scope and application

This method is applicable but not limited to the analysis of the following PCDDs and PCDFs by high resolution gas chromatography/low resolution mass spectrometry (HRGC/LRMS), or HRGC/high resolution mass spectrometry (HRMS):

2,3,7,8 tetrachloro dibenzo-p-dioxin
2,3,7,8 tetrachloro dibenzofuran.

10.2.12.2 Sample extraction

The tables below list the specified US EPA SW-846 methods.

3545A	Pressurised fluid extraction (PFE)
3546	Microwave extraction using hexane: acetone (1:1)
8290A	Soxhlet and Dean-Stark separator extraction using toluene **(a)** for low concentration (individual compounds (<1 µg/kg): toluene
8280B	Soxhlet and Dean-Stark separator extraction using toluene **(b)** for high concentration (individual compounds (>1µg/kg): toluene

10.2.12.3 Extract clean-up

Methods for the clean-up of some co-extracts/analytes are suggested below.

8280B	Acid/base clean-up followed by: silica gel column clean-up alumina clean-up carbon clean-up.

Note: Acid/base clean-up may not be necessary for uncoloured extracts.

10.2.12.4 Extract analysis

8280B	PCDDs and PCDFs by HRGC/LRMS. This method applies to reporting of total concentration of TCDD/PCDF in a given level of chlorination. Complete chromatographic separation of all 210 isomers is not possible under stated instrumental conditions. Quantification limits are greater than 1 µg/kg of solid (parts per billion).
8290A	PCDDs and PCDFs by HRGC/HRMS. This method applies to reporting individual concentration of tetra- through to octa-chlorinated TCDD/PCDF homologues. Quantification limits are less than 1 µg/kg of solid (parts per billion). Sensitivity of method is dependent on level of interference in matrix.
1613B	Isotope dilution. High resolution GC/MS.

11 Leachable contaminants

11.1 Scope and application

The leachability characteristics of a contaminant can be used to help predict the likely impact it will have if the soil is left on site, proposed for re-use or intended for disposal.

Contaminants in soil can leach into groundwater under certain conditions, depending on the local chemistry and geology of a site—leachability is particularly affected by soil pH, contaminant solubility and Redox conditions. These parameters are not controlled in leaching tests but should be recorded from field tests, and other laboratory tests, to ensure that leachability test results can be evaluated accordingly.

A variety of leaching tests are available, and it is important to specifically test leachability in soil under conditions approximating those found in the field or the proposed end-use environment.

Leachability testing can be of two types:

- batch leaching (or static extraction tests) – equilibrium based
- dynamic leaching – column and diffusion tests.

Generally, batch tests have a much shorter duration than dynamic tests though the latter may give a better representation of contaminant leaching. Batch extraction protocols assume that a steady-state condition is achieved by the end of the test.

All methods are designed to simulate leaching conditions in the environment and thus estimate the likely availability of contaminants. The choice of leaching reagent should be based on the environmental conditions to which the soil or wastes are likely to be exposed — ideally using actual surface and groundwater from the relevant site.

The two most relevant leaching tests for Australian conditions are:

- Australian standard leaching procedure (ASLP) as per Australian standards 4439.1 (AS4439.1-1999), 4439.2 (AS 4439.2-1997) and 4439.3 (AS 4439.3-1997)
- toxicity characteristic leaching procedure (TCLP) as per US EPA method 1311, (US EPA SW846, Method 1311).

The ASLP allows a wide range of leaching reagents to be used and is generally the most appropriate leach test to cover a range of conditions encountered in contaminated site management in Australia, whether soil is to remain on site or be moved.

The exception is where contaminated soil is to be disposed of at a municipal landfill and mixed with municipal solid waste (MSW), in which case TCLP is more appropriate.

The TCLP was designed to simulate conditions in a MSW landfill. It is not suitable for soil that is NOT intended to be mixed with MSW.

Leachable organics (volatile and semi-volatile), metals and anions (except cyanide) may be determined using ASLP (or TCLP if permitted by local regulatory guidelines). The zero headspace methods for ASLP (AS 4439.2-1997) and TCLP (US EPA SW-846, Method 1311) list the volatile compounds of concern. The ASLP procedure lists an informative group of volatile compounds, but does not preclude others. The TCLP (US EPA SW-846, Method 1311) lists benzene, carbon tetrachloride, chlorobenzene, chloroform, 1,2-dichloroethane, 1,1- dichloroethylene, methyl ethyl ketone, tetrachloroethylene and vinyl chloride as toxicity characteristic constituents at a contaminated site.

Leachable cyanide may be determined by the synthetic precipitation leaching procedure (US EPA SW-846, Method 1312) using deionised water leach fluid or by the ASLP methods described in AS 4439.2-1997, also using distilled or deionised water as the leach fluid.

Leachates collected from the leaching procedures should be analysed using methods listed for waters and wastewaters.

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13 Appendix 1: Determination of total recoverable hydrocarbons (TRH) in soil

This material has been adapted from procedures developed by the CRC CARE TPH Technical Working Group, convened by CRC CARE in 2009. References used include:

- CRC CARE 2009, *Health screening levels for petroleum hydrocarbons in soil and groundwater*, CRC CARE TPH Technical Working Group, Cooperative Research Centre for Contamination Assessment & Remediation of Environment, Adelaide, Australia.
- US EPA 1999, Method 1664: *n-Hexane Extractable Material (HEM; Oil and Grease) and Silica Gel Treated n-Hexane Extractable Material (SGTHEM; Non-polar Material) by Extraction and Gravimetry,* Revision A, US EPA Office of Water, United States Environment Protection Authority.

13.1 Volatile (C_6 – C_{10}) and semi-volatile (>C_{10}–C_{40}) TRH

These methods can be used to determine TRHs in soil by gas chromatography with an appropriate detector. The term 'TRH' is equivalent to the historically reported 'TPH'.

Method A1 can determine volatile TRH (vTRH) and can be used to investigate sites contaminated with petrol, other light fuels and petroleum-based solvents.

Method A2 can determine semi-volatile TRH and can be used to investigate sites contaminated with diesel, other petroleum fuels, mineral oil and petroleum-based solvents.

The methods are performance-based and designed to be rapid and economical. To obtain consistent and reliable results, they should be carried out by experienced analysts trained in the operation, maintenance and troubleshooting of GC instrumentation and in interpretation of gas chromatograms.

This section describes the general principles common to both methods, including quality control and method validation procedures.

The term 'TRH–total recoverable hydrocarbons' should be used when referring to data generated using these test methods where no clean-up is employed.

If silica clean-up is employed, the results should be qualified as 'TRH–silica'.

13.1.1 Quality control considerations

Standard quality controls are required to ensure the correct performance of these methods (see Section 4). Quality control measures should include a calibration verification standard (CVS)—consisting of a hydrocarbon product mix—and a laboratory control sample (LCS)—consisting of a suitable hydrocarbon product mix. Ideally, the LCS should be spiked with hydrocarbons that test all fractions reported.

Calibration verification standard (CVS) – A known quantity of hydrocarbon product(s) is/are dissolved in extraction solvent. This standard should contain hydrocarbons covering the required hydrocarbon fractions being analysed and serves as a check on the GC system and quantification procedure. The CVS should be between 80 and 120% of the expected concentration in the sample. This can be run once per sequence or 24 hour period.

Laboratory control sample (LCS) – As a minimum, a laboratory control sample should be run with each batch of 20 samples. This quality control sample should be processed through the entire analytical method and reported with the data. The LCS is a clean soil fortified with the same hydrocarbon product mix as used for the CVS, or a reference sample with a consensus hydrocarbon value. Recovery of product should be checked by analysing either ethanol-free petrol or any other

suitable product with predominant hydrocarbons in the nC_6– nC_{10} range. The calculated LCS concentration should be between 70 and 130% of the expected concentration or a recovery range established by ongoing quality control charts.

13.1.2 Method validation

The methods should be validated by each laboratory using them, in accord with this Schedule. Some method validation parameters require particular attention, as below.

13.1.2.1 Hydrocarbon product linearity

Establish linearity of the detector response using hydrocarbon products that cover the particular hydrocarbon fraction (for example, ethanol-free petrol for Method A1 (analysis of volatiles), or a mix of diesel and motor oil for Method A2, (analysis of semi-volatiles). Linearity should be within 15% in each of the calibrated carbon ranges. As a general principle, the peak height of the largest product component in a fraction should not exceed the peak height of the single n-alkane in the highest level calibration standard.

13.1.2.2 Product standard reference materials

A reference hydrocarbon product(s) should be prepared and analysed. The products(s) should cover the range of hydrocarbon fractions specified in this method. The product or products should be well characterised, such that the quantitative composition of the relevant fractions is known. This allows the assignment of a portion of a known quantity of this product to a particular fraction. This solution can then be ideally used as the CVS for ongoing quality control.

Accuracy of the method should be established by obtaining acceptable recoveries for hydrocarbons from a certified reference material (i.e. soil contaminated with hydrocarbons).

13.1.2.3 Proficiency studies

Ongoing participation in relevant proficiency studies is required to validate this method.

13.2 Method A1: Determination of volatile TRH: TRH C_6 – C_{10}

13.2.1 Scope and application

This method is applicable to the determination of hydrocarbons eluting between nC_6 and nC_{10} alkanes, inclusive of BTEX. Target compound analysis can occur simultaneously when running this method, provided that suitable specific detectors are employed, e.g. PID for aromatic compounds, or MS.

Note: Semi-volatile hydrocarbons with higher boiling points should be analysed by the TRH semi-volatile method (see Method A2 below (Section 14.3) and Section 11.13).

13.2.2 Limitations

- This method does not distinguish between petrogenic and biogenic compounds or synthetic compounds, such as chlorinated solvents; it measures the *total* recoverable hydrocarbons present, hence it is designated TRH.
- Excess moisture in sample: the method requires extraction of the sample with methanol, which is soluble in water. Excess moisture can dilute the extraction solvent, increasing the solvent volume thus diluting the extract.
- High organic carbon content in sample: methanol is a relatively weak solvent for non-polar compounds. Volatile analytes may be retained by matrices containing high organic carbon levels. Surrogates added to extractions may preferably partition onto the carbon matrix.

13.2.3 Interferences

The method is subject to certain interferences including:

- highly contaminated samples may cause a carry-over on the instrument
- laboratory background, including ambient air, carry-over and contaminated soils.

13.2.4 Principle

A soil sample (>5 g) is extracted with a sufficient volume of methanol, then the methanol is separated from the soil and added to a purging vessel or other equivalent apparatus for determination of volatile compounds, using FID or MS in scan mode.

13.2.5 Method

13.2.5.1 Apparatus

A gas chromatograph with appropriate detector for hydrocarbon determination. Columns suitable for volatiles, as specified in US EPA Method 8260B (latest version).

13.2.5.2 Reagents and standards

Reagents

Unless otherwise specified, all reagents shall be of analytical grade (AR) and all solvents of chromatography grade. Chromatography grade methanol and organic-free water are recommended, and ultra-pure carrier gas for gas chromatography.

Standards

Internal standard

This solution comprises a suitable compound dissolved in methanol to a suggested concentration of 10 mg/L and should be stored at 4°C. Suitable compounds are specified in US EPA Method 8260B.

Surrogate standard

This standard comprises a methanol solution containing at least one surrogate compound. Suitable compounds include 4-bromofluorobenzene, dibromofluoromethane, toluene-d_8. It should be stored at 4°C.

Calibration standard solutions

nC_6–nC_{10} TRH Standard (standards for mass selective detector or flame ionisation detector).

Owing to the differential responses of mass spectrometric detectors towards aliphatic and aromatic compounds, it is essential that the standard contains representatives of both groups.

This standard should therefore consist of about 40% aromatic and 60% aliphatic target analytes, in order to be representative of a typical Australian fuel. The aromatic compounds shall comprise the components of BTEX. The aliphatics shall comprise equal proportions of all n-alkanes in the C_6–C_{10} range.

These solutions are stable for 6 months when stored at ≤6°C with minimum headspace and away from all possible sources of contamination.

Note: If a different fraction split is requested, the relevant compounds shall be represented in the calibration standard solution.

While it may be possible to store and use the stock solutions for longer than 12 months after preparation, the laboratory should assure itself of the stability of the solution by carrying out regular checks of the concentration of the analyte. The laboratory should retain records to confirm the stability of the solutions.

Calibration verification standard solution
Calibration performance should be assessed against ethanol-free petrol or any other suitable product with predominant hydrocarbons in the nC_6–nC_{10} range used to check validity of the calibration curve.

The product should be well characterised, such that the quantitative composition of the relevant fractions is known. This allows the assignment of a portion of a known quantity of this product to a particular fraction.

Calibration standards

Initial calibration

This involves analysis of at least five different concentrations covering the working range of the instrument used. Extrapolation of the response curve above the highest calibration level is not recommended. Initial calibration is run at the beginning of each analytical sequence.

13.2.5.3 Procedure

1. Open the sample jar quickly, scrape off the top 1 cm of sample and discard. Remove all extraneous material (grass, pebbles, etc.) from the sample. Obtain the subsample by driving an inert coring device (PTFE or stainless steel spatula) into the sample and rapidly transfer a minimum of 5 g into a tared extraction vessel. Record the weight.
2. Add methanol (at a minimum ratio of 1:2 sample:solvent) and an appropriate amount of surrogate standard solution in order to produce a final surrogate concentration at about the midpoint of the calibration range, taking further dilutions into consideration.
3. Shake extract for about 30 minutes using end-over-end tumbler, orbital shaker or ultrasonic bath. Allow to settle. Clay samples should be completely disintegrated before an aliquot is taken for analysis. Samples should be maintained in a cool environment to ensure they do not overheat.
4. Analyse an aliquot of methanol extract using an appropriate instrument for hydrocarbon analysis. If an internal standard is used, it should be included with the methanol extract transfer. Alternatively, the internal standard may be added automatically by instruments having this capability.

13.2.6 GC Analysis

13.2.6.1 Calibration

At least five calibration standards should be prepared from the relevant calibration standard solution.

- The calibration curve should have a linear regression of >0.99
- At a minimum, run a daily check of the lowest calibration standard and the midpoint calibration standard to confirm stability of the calibration curve. Rerun the calibration curve if the low standard deviates by more than 30% from the curve or if the midpoint calibration standard deviates by more than 20% from the curve.
- A CVS is run to check the validity of the calibration curve against a characterised hydrocarbon product.

13.2.6.2 Measurement of test sample

After calibration, carry out the determination on the test samples (field or laboratory methanol extracts). Where the analyst has some prior knowledge regarding the relative concentration of analytes in the samples, the run should be arranged in order of increasing concentration. In the absence of such information and if samples with high concentration of analytes occur in the middle of a run, the analyst should examine the analytical run for possible carry-over, and re-analyse affected samples, if required.

13.2.7 Calculations

13.2.7.1 Integration of peaks

All peaks in a chromatogram should be integrated and included in the calculation of results. The total area contributed by the surrogate and internal standards should be excluded from the calculation of the final result.

13.2.7.2 Calculation of vTRH (C_6 – C_{10}) content

Integrate the appropriate chromatogram.

The C_6–C_{10} fraction is integrated from the peak start of the nC_6 peak to the time corresponding to the end of the nC_{10} peak.

The vTRH content is calculated according to the following formula:

$$C = \text{Area of C in sample} \times \frac{I_{STD} \times \text{conc. of standard} \times VF \times ME \times 100}{I_{SAM} \times \text{Area of standard} \times MA \times W \times (100 - \%\ \text{moisture})}$$

where:

C	=	*vTRH in soil (mg/kg)*
VF	=	*Volume of water–methanol extract as analysed by purge and trap (L)*
MA	=	*Volume of methanol extract transferred into reagent water (L)*
ME	=	*Volume of methanol added to soil/sediment (L)*
W	=	*Weight of soil/sediment analysed (kg)*
I_{STD}	=	*Peak area or height produced by internal standard in calibration chromatogram*
I_{SAM}	=	*Peak area or height produced by internal standard in sample chromatogram*
% Moisture	=	*Moisture content of original soil/sediment expressed as % w/w*

The method blank should contain no detectable levels of analytes of interest and results of the method blank should not be subtracted from sample results.

13.3 Method A2: Determination of semi-volatile TRH: TRH >C_{10} – C_{40}

13.3.1 Scope and application

The method is applicable to the determination of hydrocarbons eluting between >nC_{10} and nC_{40} alkanes. The method extracts target component hydrocarbons such as PAHs. If the presence of PAHs is suspected, target analysis techniques are preferred for risk assessments. Volatile hydrocarbons with lower boiling points than nC_{10} or heavy petroleum products (boiling points >nC_{40}) will not be quantitatively determined using this method.

Where significant levels of non-TPH interferences are suspected, a silica gel clean-up procedure is included as an optional but recommended clean-up step (with the results qualified as 'TRH-silica').

13.3.2 Limitations

The method cannot be used to provide quantitative data for the nC_6 to nC_{10} hydrocarbon range, as it allows loss of the most volatile components in the sample, mainly during the weighing and chemical drying steps. For quantitative analysis of nC_6 to nC_{10} hydrocarbons, refer to Method A1 in this Schedule.

13.3.3 Interferences

Interferences may be caused by any organic compounds that are soluble in the extracting solvent and that elute from the GC under the conditions used. These may include vegetable and animal oils and fats, chlorinated and other solvents, plasticisers, etc. The use of silica to adsorb polar compounds may reduce these interferences.

Impurities in the extracting solvent, drying agents and silica will interfere, and can be reduced by the use of high purity solvents. Laboratory blanks should be analysed with each batch of samples.

Carry-over from previous highly contaminated samples extracted in the same glassware may cause spurious elevated results, which can be minimised through efficient cleaning of all glassware, syringes, etc.

13.3.4 Principle

A soil sample (>10 g) is treated with anhydrous sodium sulfate then extracted into a minimum of 20 mL 1:1 DCM:acetone. The sample is extracted by mechanical end-over-end shaking for a minimum of 1 hour or other suitably validated extraction techniques (ASE©, horn probe ultrasonication, mechanical wrist action shaker or soxhlet extraction). Where non-TPH interferences are suspected, a silica gel treatment step is recommended.

The extract is analysed with a phenyl polymethylsiloxane phase column containing up to 5% polymethylsiloxane using a GC equipped with an FID. The results are reported as the amount of hydrocarbon in three defined fractions – $>nC_{10}–nC_{16}$, $>nC_{16}–nC_{34}$ and $>nC_{34}–nC_{40}$.

13.3.5 Method

13.3.5.1 Apparatus

- Gas chromatograph with FID
- Column: non-polar or semi-polar bonded phase capillary column is strongly recommended (polymethylsiloxane up to 5% phenyl polymethylsiloxane)
- Integrator or computer and integration software
- Volumetric pipettes and glassware—they should all be regularly calibrated and a calibration record maintained.

13.3.5.2 Reagents and standards

Reagents

All reagents used in this method should be reagent grade or higher.

Dichloromethane (DCM) and acetone should be high purity and give no interference peaks by GC-FID.

Anhydrous sodium sulfate may contain plasticisers leached from plastic storage containers;

each batch should be checked before use. A suggested clean-up method is as follows:

1. Spread the sodium sulfate on a metal tray to a depth of <2 cm.
2. Ignite in a muffle furnace at 600°C for 1 hour.
3. Cool and store in a sealed metal or glass container.

Silica (e.g. Merck, Silica Gel 60, 70–230 mesh, methods may require a specific mesh size)

Should be appropriately activated to meet the performance requirements of the method. For example, dry at 200–250ºC for 24 hours minimum and store in a desiccator or tightly sealed container. Deactivate by adding an appropriate weight of reagent grade water and mix thoroughly.

Note: degree of deactivation depends on the constitution of the solvent extract to be cleaned up.

Calibration standards

- The fraction definition standards for this method—and the calibration standards used to quantify the fractions—are nC_{10}, nC_{16}, nC_{34} and nC_{40}.
- A calibration verification standard consists of hydrocarbon product dissolved in extraction solvent. Products used as calibration verification mixes should cover the applicable carbon ranges of the method.
- Freshly made calibration standards should be checked by GC–FID against the calibration standards currently being used in the TRH method as a check for any gross error in their preparation.

13.3.5.3 Procedure

- Weigh a minimum of 10 g of sample into a tared vessel.
- Add sufficient amount of anhydrous sodium sulfate to permit drying of sample.
- Add a minimum of 20 mL DCM:acetone (1:1) and extract by end-over-end tumbler for a minimum of 1 hour. Alternative extraction solvent mixes or extraction procedures can be used if results meet method performance criteria.

13.3.5.4 Silica gel clean-up

Quantities of silica gel used will vary with the volume of extract and the suspected concentration of polar substances. The choice of solvent and suitably deactivated silica gel should demonstrate a quantitative recovery of aliphatic and aromatic hydrocarbons of between 70 and 130%. When validating a particular procedure, this should be demonstrated to quantitatively remove a typical surrogate polar compound, for example, palmitic or stearic acid.

The procedure described below is for a dispersive sorbent clean-up. Mini-columns or commercial silica solid phase cartridges (SPC) may also be used if comparable method performance criteria can be met.

- Exchange an aliquot of sample extract into a suitable solvent for clean-up. For example, a 1:1 DCM:acetone extract should be exchanged into a solvent other than acetone, to allow for removal of polar substances.
- To the solvent-exchanged extract add an appropriate weight of silica gel. If an empirical determination of bulk density has been made, the weight may be replaced with an appropriate volume.
- Mix the extract and silica gel thoroughly (e.g. witha vortex mixer) and allow the sorbent to settle before removing a portion of the extract for analysis.

US EPA 3630C silica clean-up method gives information about clean-up of PAHs, PCBs, OCs and phenols but not specifically for hydrocarbons. On the other hand, US EPA Method 1664 gives silica gel clean-up information specifically for hydrocarbons.

Limitations

1. Silica gel has a capacity to adsorb polar compounds, at approximately 30 mg per gram of material. Silica may become overloaded if too much polar material is present beyond the capacity of silica gel used. In such cases, multiple clean-up steps may be required.
2. Waste sludges containing paint can give anomalous results due to clean-up procedures being unable to remove all such unwanted material. Such non-polar polymeric materials remaining in a solvent extract can then degrade in the high temperature GC injector, producing smaller hydrocarbon molecules recorded as petroleum hydrocarbons. In such situations, alternate clean-up procedures should be investigated, for example, gel permeation chromatography (GPC).
3. Soils high in organic matter may also give false positive results.

13.3.6 GC analysis

The sample should be analysed using a gas chromatograph fitted with an FID.

13.3.6.1 GC conditions

The exact conditions used will vary from laboratory to laboratory.

Injector: a split/splitless injector at >250°C is recommended. The injection liner should be checked and replaced regularly.

Oven: the oven ramp should be a single linear ramp. The final temperature of the oven program should be as high as possible to ensure maximum removal of the higher molecular weight hydrocarbons from the column prior to the next analysis.

Column: the capillary column should be a non-polar to semipolar phase—such as a bonded phase of polydimethylsiloxane containing up to 5% phenyl polydimethylsiloxane.

13.3.6.2 Chromatographic integration

The sample sequence should have adequate solvent blanks run to monitor baseline drift. Samples are integrated by taking a horizontal line from a baseline point after the elution of nC_{10}. The fraction areas are calculated by the software and concentrations determined according to the 'Calculations' section below.

13.3.6.3 GC calibration

Perform calibration and retention time marking for the nC_{10} to nC_{40} hydrocarbons using approximately equal weights of nC_{10}, nC_{16}, nC_{34} and nC_{40} hydrocarbons dissolved in hexane (toluene can be added to assist dissolution).

- At a minimum, run a 5-point calibration curve using the nC_{14}, nC_{24} and nC_{36} hydrocarbons and a blank before analysis begins. Linearity should have a linear regression of >0.99.
- At a minimum, run a daily check of the lowest calibration standard and the midpoint calibration standard to confirm stability of the calibration curve. Rerun the calibration curve if the low standard deviates by more than 30% from the curve or if the midpoint calibration standard deviates by more than 20% from the curve.

13.3.7 Calculations

Calculation of TRH fractions in a sample:

$$>C_{10}–C_{16}\ \text{hydrocarbons (mg/kg)} = A_{>C10\text{-}C16} \times \frac{C_{14\ conc}}{A_{C14}} \times \frac{Vol_{ext} \times F \times 100}{W \quad \%DW}$$

$$>C_{16}–C_{34}\ \text{hydrocarbons (mg/kg)} = A_{>C16\text{-}C34} \times \frac{C_{24\ conc} \times Vol_{ext} \times F \times 100}{A_{C24} \quad W \quad \%DW}$$

$$>C_{34}–C_{40}\ \text{hydrocarbons (mg/kg)} = A_{>C34\text{-}C40} \times \frac{C_{36\ conc} \times Vol_{ext} \times F \times 100}{A_{C36} \quad W \quad \%DW}$$

where:

$A_{>C10–C16}$ = *the integration of all area counts from the end of the* nC_{10} *to the end of the* nC_{16} *peak*
$A_{>C16–C34}$ = *the integration of all area counts from the end of the* nC_{16} *to the end of the* nC_{34} *peak*
$A_{>C34–C40}$ = *the integration of all area counts from the end of the* nC_{34} *to the end of the* nC_{40} *peak*
C_{14} = *concentration of* C_{14} *standard (mg/litre)*
C_{24} = *concentration of* C_{24} *standard(mg/litre)*
C_{36} = *concentration of* C_{36} *standard (mg/litre)*
Vol_{ext} = *Final volume of sample extract (litre)*
F = *Dilution factor applied to bring the samples and standards into appropriate peak height range*
W = *weight of sample taken (kg)*
% DW = *% Dry weight*

14 Shortened forms

ABC	ambient background concentration
ACL	added contaminant limits
ADWG	*Australian drinking water guidelines*
AM	arithmetic mean
ANCE	excess acid neutralizing capacity
APHA	American Public Health Association
AS	Australian Standard
ASE©	accelerated solvent extractor
ASLP	Australian standard leaching procedure
ASTM	American Society for Testing & Materials
AWQG	*Australian and New Zealand guidelines for fresh and marine water quality*
BTEX	benzene, toluene, ethylbenzene and xylenes
CEC	cation exchange capacity
CI	confidence interval
CL	confidence limit
CRC CARE	Cooperative Research Centre for Contamination Assessment and Remediation of the Environment
CRM	certified reference material
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CVS	calibration verification standard
CWS PHC	*Canada Wide Standard for Petroleum Hydrocarbons (PHCs) in Soil*
DQO	data quality objective
EIL	ecological investigation level
ESL	ecological screening level
FA	fibrous asbestos
FID	flame ionisation detector
GC	gas chromatography
GC/ECD	GC/electron capture detector
GC/ELCD	GC/ electrolytic conductivity detector
GC/FID	GC/flame-ionisation detector
GC/FPD	GC/flame photometric detector
GC/MCD	GC/microcoulometric detector
GC/MS	GC/mass spectrometry
GC/NPD	GC/nitrogen-phosphorus (thermionic) detector
GC/PID	GC/photo-ionisation detector
GIL	groundwater investigation level
GM	geometric mean
GMRRW	*Guidelines for managing risk in recreational water*
HEM	n-Hexane extractable material
HIL	health investigation level
HPLC	high-performance liquid chromatography
HPLC/ECD	HPLC/electrochemical detector
HPLC/F	HPLC/fluorescence detector
HPLC/MS	HPLC/mass spectrometry
HPLC/UV	HPLC/ ultraviolet detector
HRGC/HRMS	high-resolution gas chromatography/high-resolution mass

	spectrometry
HRGC/LRMS	high-resolution gas chromatography/low-resolution mass spectrometry
HSL	health screening level
ICV	independent calibration verification
IEUBK	Integrated exposure uptake biokinetic model (for lead)
ISO	International Standards Organisation
ISQG	*Interim sediment quality guideline*
KD	Kuderna-Danish evaporator
LCS	Laboratory Control Sample
LNAPL	light non-aqueous phase liquid
LOD	limit of detection
LOEC	lowest observed effect concentration
LOR	limit of reporting
MAH	monocyclic aromatic hydrocarbon
MDL	method detection limit
MS	mass spectrometry
MSW	municipal solid waste
MU	Uncertainty of Measurement
NATA	National Association of Testing Authorities, Australia
NL	non limiting
NMI	National Measurement Institute
NMR	nuclear magnetic resonance
OCP	organochlorine pesticides
OPP	organophosphorus pesticides
(P)	preferred method
PAHs	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyl compounds
PFE	pressurised fluid extraction
pHox	peroxide pH
PID	photo ionisation detector
PQL	practical quantification limit
PTA	Proficiency Testing Australia
PTFE	polytetrafluoroethylene
QA	quality assurance
QC	quality control
RPD	relative percent difference
RRT	relative retention time
RSD	relative standard deviation
RT	retention time
SD	standard deviation
SFE	supercritical fluid extraction
SGT-HEM	silica gel treated n-hexane extractable material
SPC	solid phase cartridge
SRM	standard reference material
SVOC	semi-volatile organic compounds
TAA	titratable actual acidity
TCLP	toxicity characteristic leaching procedure
TDS	total dissolved solids
TEF	toxicity equivalence factor
TEQ	toxicity equivalent quotient

TOC	total organic carbon
TPA	titratable peroxide acidity
TPH	total petroleum hydrocarbons
TRH	total recoverable hydrocarbons
TRH-silica	total recoverable hydrocarbons - silica gel clean-up employed
UCL	upper confidence limit
US EPA	United States Environmental Protection Agency
VHC	volatile hydrocarbons
VOA	volatile organic analysis
VOCC	volatile organic chlorinated compound
vTRH	volatile total recoverable hydrocarbons
WAD	weak acid dissociable cyanide
WHO	World Health Organization

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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This compilation has been split into 22 volumes

Volume 1: sections 1-6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

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Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of

Schedule B5a

GUIDELINE ON

Ecological Risk Assessment

Explanatory note
The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

The original Schedule B5 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document, together with Schedule B5b and Schedule B5c. The National Environment Protection Council (NEPC) acknowledges the contribution of the Commonwealth Scientific and Industrial Research Organisation (CSIRO), the NSW Environment Protection Authority and the NSW Environmental Trust to the development of this Measure.

Contents
Ecological risk assessment

1 Background

The framework for conducting ecological risk assessment (ERA) was first set out nationally in the *Australian and New Zealand guidelines for the assessment and management of contaminated sites* (ANZECC & NHMRC 1992). It is based on the US EPA model and consists of four main phases: data collection and evaluation, toxicity assessment, exposure assessment and risk characterisation (US EPA 1989).

The National Environment Protection (Assessment of Site Contamination) Measure 1999 (the NEPM) refined and expanded upon this model. The tiered approach outlined in the 1999 Measure consisted of three levels of assessment:

Level 1 – a comparison of measured concentrations to the ecological investigation levels (EILs)

Level 2 – a desktop study where site-specific factors were used to modify the EILs, which were then compared to the measured concentrations

Level 3 – a detailed, site-specific, probabilistic ERA.

Each level consisted largely of the same basic four considerations but incorporated an increasing degree of complexity from Level 1 to Level 3.

The development of ERAs in Australia was further enhanced by the risk-based hierarchical approach adopted in the *National water quality management strategy – Australian and New Zealand guidelines for fresh and marine water quality* (ANZECC & ARMCANZ 2000).

2 Introduction

It is now well recognised that a risk assessment provides information to distinguish between important and trivial contamination issues. When coupled with political, social, cultural, economic and engineering considerations, it enables decisions about the need and methods to be used to reduce risk. ERA is this approach applied to ecological situations.

Inherent in an ERA is the need to recognise the following principles:

- It needs to be focused on maintaining ecosystem structure and function, which are both vital to maintaining healthy and sustainable ecosystems.
- It must recognise that all aspects of the environment are interdependent and cannot be considered in isolation, thus leading to a holistic approach.
- Its objectives must recognise the sustainable use of resources in an environmental, economic, social and cultural context. It is imperative that the environmental values to be protected are the driving force for the assessment, noting that the values of sites with different land uses (for example, land used for industrial purposes or for a national park) may be different. The existing or proposed land use of a site assessed for contamination will influence the selection of ecological values.
- An ERA requires an integrated approach, using multiple lines of evidence gathered from physical, chemical and biological data combined with site-specific data about exposure, toxicological and chemical parameters and the consideration of properties of soil, sediments and water relevant to the site, in order to estimate the level of effects. The movement of contaminants from soil to other environmental media (that is, air, water or sediment) and subsequent exposure to biota should be included in the ERA.
- Communication strategies are integral to the success of any ERA, so the process requires a cooperative approach to encourage effective communication among industry, government and communities.

The ERA process described in this guideline assesses the risk posed to terrestrial ecosystems (including soil processes, soil flora and fauna, and terrestrial invertebrates and vertebrates) from the adverse effects of chemical contaminants in soil. Section 2.4.1.4 of Schedule B5b provides information and limitations of the equilibrium partitioning method (EqP), which is used to predict the toxicity of a contaminant in soils based on aquatic toxicity data. Examples of how to derive EILs that consider off-site aquatic effects are provided in Schedule B5c (Sections 3.6.2 and 4.6.2). Further guidance for assessing risks to aquatic ecosystems is available from *National water quality management strategy – Australian and New Zealand guidelines for fresh and marine water quality* (ANZECC & ARMCANZ 2000).

This risk-based process is inextricably linked to the principles of ecologically sustainable development (ESD). ESD aims to protect biodiversity and maintain ecological processes and functions and it is a central paradigm to both Australian and international environmental regulations and policies. However, it is also acknowledged that all human activity impacts on the environment and hence it is not possible to protect all species, processes and functions. Rather, it is necessary to manage the risks associated with various human activities in order to achieve the goals of ESD.

In this way, we recognise that we aim towards protecting the vast majority of, but not all, species from the harmful effect of contaminants. The assumption here is that protecting the majority of species (the structure of ecosystems) will enable the functions conducted by the ecosystems (for example, nutrient cycling, leaf litter degradation) to be maintained. The actual percentage of species that are protected is a policy decision. Human health risk assessment uses a similar approach as it aims to protect not every human, but the vast majority.

3 The ecological risk assessment framework

The methodology in Schedule B5b provides the means for deriving ecological investigation levels (EILs) used within the ERA framework. In developing the EIL derivation methodology, the approaches used by other entities (such as the USA, the Netherlands, Canada, the EU and the UK, Germany and New Zealand) were considered. A summary of these approaches is presented in an appendix of Schedule B5b.

This risk-based methodology incorporates the latest scientific findings in the areas of ecotoxicology, soil science and geochemistry. It enables:

- protection of introduced and native animals, plants, microorganisms and microbial processes (including nutrient cycling)
- setting levels of protection based on land use
- accounting for background concentration of contaminants
- accounting for changes in bioavailability of contaminants over time and in different soils
- accounting for contaminants that biomagnify.

The EILs are calculated using a species sensitivity distribution (SSD) method that permits the EILs to be set to protect any selected percentage of species (for example, for urban residential, it is 80%). They are derived based on the LOEC (lowest observed effect concentrations) and EC_{30} (30% effect concentration) toxicity data. Further information is provided below but full details of the EIL derivation methodology can be found in Schedule B5b and the derivation of the EILs can be found in Schedule B5c. In addition, an EIL calculation spreadsheet can be found in the ASC NEPM Toolbox on the EPHC website, which provides step-by-step guidance on deriving EILs specific to the site, with consideration of certain physicochemical properties of soils.

The toxicity of some contaminants is affected by physicochemical properties of the soils in which the contaminant is located. When empirical relationships able to model the effect of soil properties on toxicity are established, then soil-specific EILs can be developed. The EILs take into account the biological availability of the element in different soils and separate naturally occurring concentrations of a contaminant and the added contaminant in deriving EILs which are based on the 'added risk approach' (Struijs et al. 1997; Crommentuijn et al. 1997). This approach assumes that the availability of the ambient background concentration (ABC) of a contaminant is zero or sufficiently close that it makes no practical difference. More importantly, it assumes that the background 'has resulted in the biodiversity of ecosystems or serves to fulfil the needs for micronutrients for the organisms in the environment' (Traas 2001). Therefore, the approach views only the effect of added contaminants to the environment as adverse (for further information refer to Section 2.4, Schedule B5b). Thus, rather than having a single numerical limit for a contaminant, different soils will have different limits. The EIL derivation methodology generates, wherever possible, soil-specific EILs. However, in developing this ERA framework, it was not possible to derive soil-specific EILs for all contaminants so the EILs for some contaminants are soil-specific while for others they are generic.

In addition, most of the available toxicity data for contaminants in soil was obtained in laboratories where the contaminant is added to the soil immediately prior to commencing the test. However, it is known that some contaminants become less bioavailable in the field and over time (they age). Thus, laboratory-based experiments may overestimate toxicity in the field.

Also, laboratory experiments that use soils spiked with soluble metal salts overestimate toxicity compared to equivalent field soils, due to a lack of leaching of soluble salts that affect metal sorption. These factors have been addressed in recent EU risk assessments for metals in soils using 'ageing/leaching' factors.

Therefore, whenever ageing/leaching factors were available, they were used to correct the laboratory-based toxicity data (see Schedule B5c).

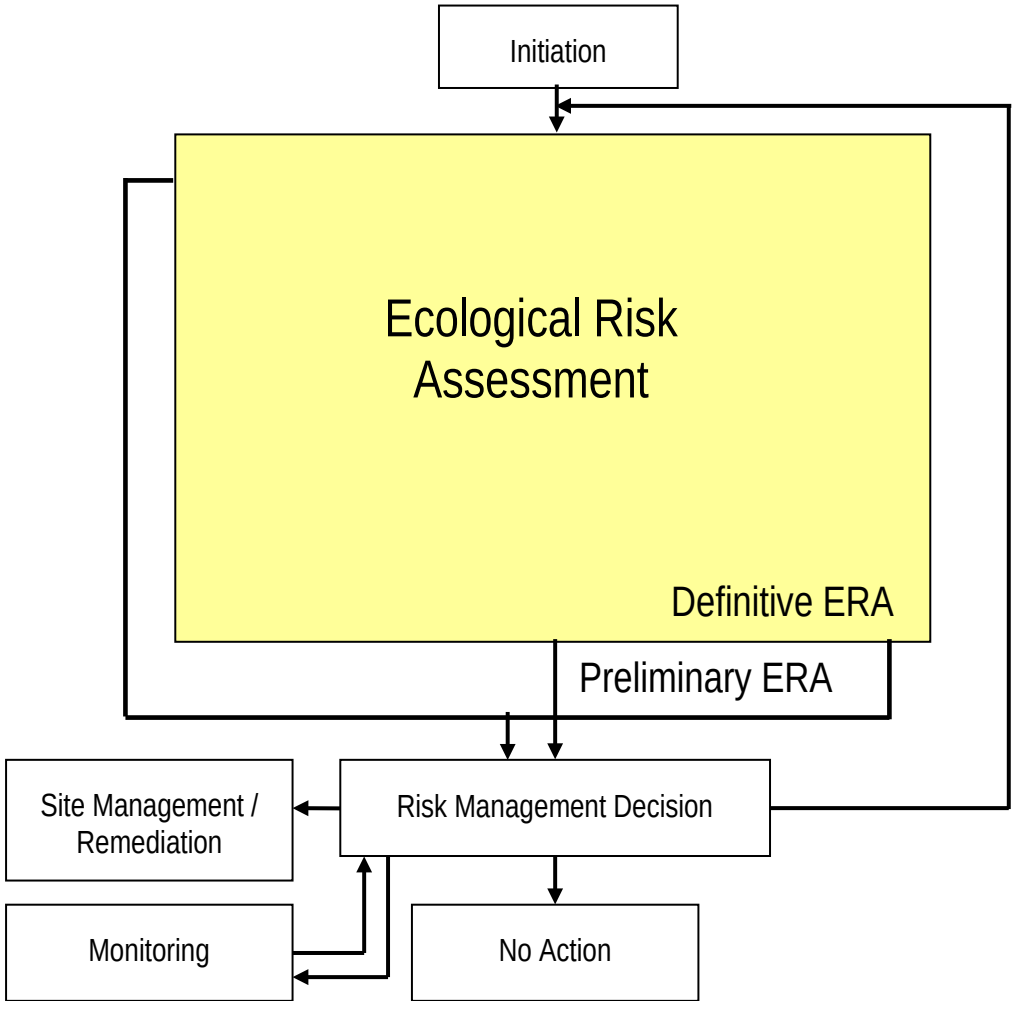
Where sufficient data permitted, EILs were derived for sites with fresh (<2 years) and aged (≥2 years) contamination. For the contaminants with generic EILs, there is a single value for each combination of land use and age of the contamination. For the contaminants with soil-specific EILs, a suite of values was derived (based on the soil physicochemical properties that control the toxicity) for each combination of land use and age of contamination.

Soil-specific physicochemical properties and ageing are two characteristics that would have been considered in Level 2 ERAs in the previous Measure (NEPC 1999).

By deriving EILs that account for soil-specific properties and ageing, the first ERA component is, in effect, a combination of Level 1 and Level 2 of the previous ERA framework (NEPM 1999). In summary, the framework for conducting ERAs has been simplified and now consists of two levels: a Preliminary ERA and a Definitive ERA (see Figure 1).

A summary of the EILs for eight chemicals (arsenic, copper, chromium (III), DDT, lead, naphthalene, nickel and zinc) is provided in Appendix 1. More details on the methodology and the data used in the derivation of these EILs can be found in Schedules B5b and B5c.

Figure 1. The framework for conducting ecological risk assessments



It is important to note that the EILs only apply to soil down to a depth of two metres[1] below the current soil surface, which corresponds to the root zone and habitation zone of many species.

The tiered ERA approach used in this guideline permits:

- identification of the ecological receptors of concern
- estimation of the concentration of a contaminant of concern to which the ecological receptors are exposed
- consideration of the toxicity-modifying or toxicity-enhancing capacity of the receiving environment (whether that be soil, sediment or water)
- determination of whether the ecological receptors and ecological values may be at risk
- application of a multiple-lines-of-evidence approach to assess risks.

This tiered approach relatively quickly and cheaply screens out those sites where the environmental risk is minimal. It thus focuses resources on those sites that pose the greatest potential risk. It should be emphasised that the majority of sites will only require a Preliminary ERA.

3.1 Preliminary ERA

Generally the first step in the ERA process is to decide whether a Preliminary ERA is necessary for the site in question. In some jurisdictions, at least some level of ERA is mandatory. Reasons for initiating a Preliminary ERA should be clearly stated in all ERA reports. ERAs are conducted using conservative assumptions (that is, they tend to favour protecting the environment). Thus, if a Preliminary ERA indicates the site faces a low risk from the contaminants, then there can be confidence that this is the case.

3.2 Definitive ERA

A Definitive ERA is required only in a situation where the concentration of the contaminant(s) is sufficiently high that it may pose a risk. A Definitive ERA requires greater data collection, uses more complex and environmentally realistic methods and reduces the uncertainty in the outcome of the ERA compared to the Preliminary ERA. As a result, Definitive ERAs are considerably more time-consuming and costly than ERAs.

3.3 Components of an ecological risk assessment

Both Preliminary and Definitive ERAs consist of the same five basic components:

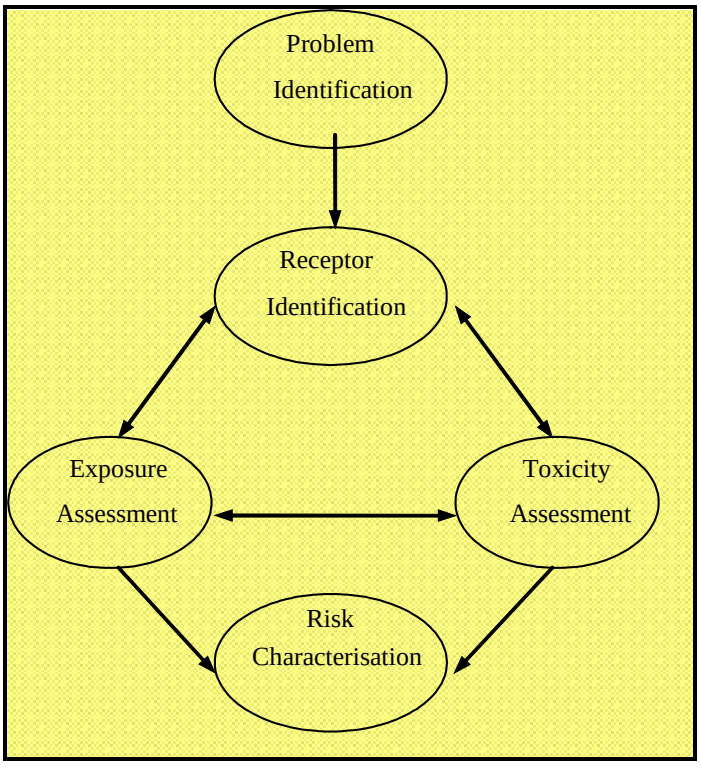
1. **Problem identification** is a scoping phase that establishes the objectives of the ERA and identifies the data required to achieve those objectives. It is essential that engagement with various stakeholders is undertaken early in this phase to provide opportunities for their input.
2. **Receptor identification** focuses on 'what species may be at risk?' and 'what do we want to protect?'. Of importance in this phase is the need to introduce the concept of what is acceptable risk in the context of the ecological values that need to be protected. This requires the identification of local species, communities and ecological processes that are of ecological value based on the relevance and significance of societal, cultural, ecological, and economic factors.

[1] On a site-specific basis, gradation of EILs to higher values may be permitted at depths greater than 2 m provided that there is sufficient assessment of risk from issues such as actual land use, proposed development basement levels, leachate characteristics, potential impacts on ground and surface water quality, vertical migration and the potential for further excavation and surface exposure of deeper contamination.

3. **Exposure assessment** characterises the site, identifies potential exposure pathways and estimates exposure duration, concentrations and intakes.
4. **Toxicity assessment** involves estimating the concentration of contaminants at which species and ecological functions experience no harmful effects and those at which toxic effects are caused. This data is in turn used to determine the concentration of contaminants that an ecosystem can be exposed to without adverse effect or with adverse effects of a certain magnitude (that is, EILs).
5. **Risk characterisation** involves combining data and information from the exposure and toxicity assessments to determine the risk that ecosystems at the site face from the contaminants. This is usually done by comparing the measured contaminant concentrations with the EILs.

The relationships between the five components are shown in Figure 2 below. Receptor identification, exposure assessment and toxicity assessment components are interrelated, as the assessment of any of these components is dependent upon the characteristics of the other two. Risk characterisation includes the combination of information gained in the exposure and toxicity assessments. The types and amount of information available to a risk assessor are always limited—whether it is information about the chemical levels at the site or the potential effects on an organism that a chemical could cause—so all ERAs are estimates of what the risks might be. Hence, it is important that the objectives (developed in the problem formulation stage) are re-set taking into account any additional information gleaned at every phase. Any assumptions or extrapolations made in an ERA should be highlighted where they occur. Uncertainty is discussed further in a later section of this Schedule.

Figure 2. Components of an ERA



The objectives and what is done in each component varies depending on whether the components are being conducted as part of a Preliminary ERA or a Definitive ERA.
A detailed discussion of what should be done in each component of Preliminary and Definitive ERAs is presented in later Sections of this Measure and examples on the application of EILs can be found in Schedule B1.

3.4 Risk management decision

At the conclusion of an ERA, a risk management decision needs to be made (as depicted in later discussions of Preliminary and Definitive ERAs). This decision is based on both the risk characterisation component of ERA and risk management considerations (such as economic, social, cultural and engineering matters) and should be made by the decision manager, in compliance with jurisdictional requirements.

This step ensures that both risk assessment and risk management considerations (including conflicting results and uncertainty in any part of the ERA) are reviewed prior to the outcome being determined. It also ensures that risk assessors and risk managers are each aware of the objectives of the other.

The risk management decision determines the outcome of the assessment. There are four potential outcomes:

1. to take no action
2. to monitor the site
3. to remediate or actively manage the site
4. to proceed from a Preliminary ERA to a Definitive ERA.

Additional information on each of these potential outcomes is provided in the following sections.

3.4.1 No action

The 'no action' outcome implies that no site management or remediation, monitoring or further assessment is required at the site. It reflects a high degree of confidence that the ecological values of the site are adequately protected from the effects of the contamination based on the relevance and re-setting of objectives and taking into consideration multiple lines of evidence. This outcome ends the ERA process.

It is also possible that this could be the outcome even if there was some level of risk estimated, depending on the use of the site and the technological options available.

3.4.2 Monitoring

Biological and/or chemical monitoring may be considered where there remains uncertainty if an impact has occurred, is occurring, or may occur at some time in the future or if there are data gaps. Biological monitoring may focus on individual species, selected biota in a given environment, or communities and ecosystems for signs of chemical impact or exposure. Examples of parameters that may be monitored with regard to individual species or selected biota include chemical or enzyme concentrations in tissues to assess exposure, or histopathological examination and behavioural change to assess impact. Typical parameters monitored when examining populations and communities may include species number, population number, number of offspring and biomass. Chemical monitoring can also be conducted, but its aim is to identify and quantify the chemical present in the various exposure media (for example, soil, surface water, groundwater, air, dust or food).

Ecological systems are stochastic (chaotic) and thus slight variations in initial conditions can make a big difference to the outcome. Therefore, monitoring is also often undertaken to demonstrate that the actual remediation or management process is not impacting on-site or off-site ecological values. Post-management/remediation monitoring may also be used to demonstrate the effectiveness of site management or remediation.

Monitoring may include chemical monitoring to demonstrate that the level of exposure continues to be acceptable, or biological monitoring to demonstrate that exposure continues to be acceptable and/or that residing species and populations are not being affected or that key species are returning to the site.

Results from this monitoring process feed back into the risk management decision-making process to determine further outcomes.

3.4.3 Site management/remediation

Site management/remediation is one of two potential outcomes when the on-site soil concentration of contaminants, including mixtures of the contaminants, exceeds the EIL or $EIL_{mixture}$. Site management includes any active control at the site that reduces the ecological impact to an acceptable level.

This may include reducing the exposure of biota to the contaminants by reducing their exposure to the site (for example, fencing), maintaining a physical condition of the soil that reduces the contaminants' availability/mobility, immobilising the soil contaminants or removing the soil contaminants (that is, remediation). Monitoring is an essential part of any site management/remediation program to assess the effectiveness of the program in reducing ecological impact.

3.4.4 Proceeding from a Preliminary ERA to a Definitive ERA

Alternately, where there is reasonable certainty that an impact has occurred, is occurring or may occur at some time in the future, the decision may be made to move from a Preliminary ERA to a Definitive ERA.

3.5 Ecological values

An important part of assessing a contaminated site is identifying what ecological values are present at the site or nearby and which are to be protected. Ecological values are flora, fauna and supporting ecological processes (that is, factors that influence a species' ability to grow, survive, develop and reproduce, and remain viable) that are associated with a defined piece of land and are considered to have societal, cultural, ecological and/or economic significance.

Ecological values naturally vary from site to site according to variation in the natural habitat, the degree to which humans have physically altered the natural environment and the expectations of society. Ecological values can be established for any environment being assessed. There are two types of ecological values—generic and site-specific. Both are discussed below.

3.5.1 Generic ecological values

The aim of the EILs is that varying levels of protection will be provided to the following ecological receptors at all sites:

- biota supporting ecological processes, including microorganisms and soil invertebrates
- native flora and fauna
- introduced flora and fauna
- transitory or permanent wildlife.

Hereafter, the above list of protected organisms will be referred to as 'species and supporting ecological processes'.

The level of protection provided to species and supporting ecological processes varies depending on the land use and whether the contaminant in question biomagnifies. Differing levels of protection are provided by protecting differing percentages of species and supporting ecological processes (see Table 1).

By using SSD methods to derive the EILs and having different levels of protection for different land uses, it is assumed that not every individual organism or species can be or needs to be protected.

Due to the fact that the concentration of biomagnifying chemicals increases as food webs are ascended (for example, higher trophic level organisms such as eagles have higher tissue concentrations than lower trophic organisms such as algae), a high level of protection is warranted for such chemicals. Refer to section 2.3.2 of Schedule B5b for further information about biomagnification. The levels of protection provided for biomagnifying chemicals in the three land uses are presented in Table 1.

Table 1. Percentage of species and soil processes to be protected for different land uses depending on whether the contaminant is classed as a non-biomagnifying or biomagnifying chemical.

Land use	Standard % protection	Biomagnification[a] % protection
Areas of ecological significance	99	99
Urban residential and public open space	80	85[b]
Commercial and industrial	60	65[c]

[a] if a contaminant has a logarithm of the octanol–water partition coefficient (K_{ow}) of equal to or greater than 4. Refer to glossary for K_{ow} and biomagnification.

[b] if surface area exceeds 250 m^2

[c] if surface area exceeds 1000 m^2

As the types of organisms being protected by the EILs do not change, irrespective of the land use, they are based on a generic set of ecological values. Generic ecological values are conservative in that they protect all biota considered of value within the land use regardless of whether or not they occur at the contaminated site. It is also possible to derive generic ecological values for biota that inhabit a state, region or local area regardless of land use.

EILs have been developed for three land uses: areas of ecological significance, urban residential and public open space, and commercial and industrial. The land uses are defined below:

An area of ecological significance is one where the planning provisions or land use designation is for the primary intention of conserving and protecting the natural environment. This would include national parks, state parks, and wilderness areas and designated conservation areas. These reserves are generally considered to be of high ecological value and quality and worthy of maintaining at as close to a pristine state as possible.

Urban residential and public open space is land where the primary activity is (a) human residency, such as at separate dwellings and townhouses, and is usually associated with an area of exposed soil or garden that is used for recreational purposes although some is used for vegetable and other consumables production, and (b) reserves, sporting grounds, parks, golf courses and other areas used for recreation and which are located in an urbanised area. Urban parklands may include urban land adjacent to waterways and rivers. In most circumstances, hospitals, day care centres, pre-schools, primary schools and secondary schools belong to this land use.

Commercial and industrial land is land where the primary activity is related to (a) commercial operations and occupancy (for example, service stations, railways, roads, warehouses/distribution depots, convenience shops, shopping complexes and the main streets of towns), and (b) the production, manufacture or construction of goods (for example, manufacturing factories, warehouses, transport depots, refineries and timber treatment plants).

Commercial and industrial land, particularly in long-established industrial areas, is often heavily contaminated by past activities or fill materials used to level the area. In these cases, jurisdictions may determine that HILs are the most appropriate soil quality criteria and that EILs are not applicable. In many cases, the only generic ecological value for this land use will be 'transitory wildlife'.

In cases of a site having a mixed land use (for example, an industrial site with a nature reserve), it is necessary to either apply the appropriate EILs to each land use or to apply the EILs for the most sensitive land use to the entire site.

In cases where land is to be converted from one land use type to a more sensitive land use, the ecological values identified for the more sensitive land use should be applied to the entire site.

3.5.2 Site-specific ecological values

Site-specific ecological values are those ecological values that are specific to the site under investigation. Identifying site-specific ecological values involves knowledge of the biota and supporting ecological functions that are expected to inhabit or visit the site. It also requires identification of stressors that may be present in the locality as well as an in-depth understanding of the relevance of the species.

Site-specific ecological values would be identified during a Definitive ERA, in conjunction with relevant stakeholders including appropriate government agencies, local government, and community groups and/or by conducting a biological survey of the site.

Site managers and consultants should carry out appropriate community engagement and consult with the site auditor/third party reviewer and/or relevant jurisdictional agency before finalising site ecological values. Further information can be found in Suter (1993) and in Schedule B8.

4 Preliminary ecological risk assessment

This section provides guidance for conducting a Preliminary ERA. A Preliminary ERA is a screening level assessment of generic situations and should protect a selected percentage of all biota and supporting ecological processes that are likely to inhabit soils with specific land uses.

ERAs may be undertaken for a variety of reasons. The main reasons are listed below:

- A previous assessment of soil contamination at a site identifies significant areas where contaminant concentrations are above background levels.
- Site history suggests that chemicals may be present that may create an adverse environmental effect.
- There are knowledge gaps in the soil contamination assessment that may be potentially important.
- There are ecological values that are important at the site or nearby (e.g. rare and/or endangered species or habitats).
- As part of due diligence investigations, an owner or occupier of a site may voluntarily conduct an ERA. Such risk assessments may also be conducted as part of environmental reporting requirements.
- An assessment of the suitability of land for its existing or proposed use has identified contaminants at concentrations above the background concentration.

The main question that a Preliminary ERA seeks to answer is whether the generic ecological values used to derive the EILs, and that therefore should be protected, are adversely affected by on-site contamination. This enables an informed risk management decision to be made.

A Preliminary ERA should:

- set clear objectives, taking into consideration the issues of concern, conceptual site model (CSM) and data quality objectives
- identify the ecological values relevant for the site
- determine if the ecological values used to derive the EILs are consistent with those identified for the site
- identify contaminants of concern
- establish the extent and degree of contamination on the site
- assess the linkages between causes and effects of the contamination on the site
- identify the most appropriate EILs for the soil contaminants
- determine whether the identified EILs are exceeded
- identify elements of uncertainty (including an assessment of the appropriateness of all the scientific tools used in the ERA (e.g. criteria, benchmarks, data evaluation and relevance of objectives) and data gaps
- provide justification for the conclusion of the Preliminary ERA or for proceeding on to conducting a Definitive ERA.

The various components that comprise a Preliminary ERA, the order in which they are conducted, and the interrelationships between each component are presented in Figure 1 above. A summary of the types of data and other information needed for each component of a Preliminary ERA is set out in Table 2.

Table 2. Information that may be collected for each component of a Preliminary ERA

ERA component	Indicative requirements for a Preliminary ERA
Problem identification	Clear objectives Site history Extent and degree of on-site soil contamination and development of a CSM Appropriate EILs Identification of stakeholders and implementation of communication strategies
Receptor identification	Identification of information required to set the most appropriate EILs The components of the ecosystem that constitute the ecological value of the site, including threatened and endangered species
Exposure assessment	Exposure pathways used to calculate the most appropriate EILs Exposure pathways relevant to the site
Risk characterisation	On-site soil concentrations of contaminants of concern, The most appropriate EILs
Toxicity assessment	Justification in the Preliminary ERA report for why a Definitive ERA, including toxicity assessment, is required.
An assessment of the appropriateness of the requirements for each component should be part of an uncertainty analysis	

4.1 Problem identification

The Preliminary ERA begins with problem identification to assist in the development of a CSM that summarises all that is known about the site. Where there is potential for off-site migration of contamination from a contaminated site to surrounding areas or groundwater, this should be identified and included in the site model. The model is then used to establish the objectives of the Preliminary ERA that are to be addressed. Once the objectives have been identified, the data and other information requirements of the ERA are determined. Problem identification is critical to ensure that the degree of assessment is appropriate for the problem. If there is the potential for off-site migration of contamination, a qualitative evaluation of the risk this poses should form part of the Preliminary ERA.

Depending on the data quality objectives (DQOs), in some cases the extent and degree of site contamination and the contaminants present at a site will already have been established by the existence of a soil contamination assessment. Where an ERA has been initiated in the absence of on-site soil contamination data, a soil contamination assessment should be undertaken.

This assessment should include information such as site history, site conditions, proposed land use and relevant environmental policies or regulations that may affect the site or actions to be taken. Sampling and analysis of contaminated soil should be undertaken in accordance with guidance contained in Schedule B2 and Schedule B3.
The preceding work identifies both the extent and degree of on-site contamination and the contaminants of concern. At this point in the ERA framework, contaminants of concern are those

chemicals that have concentrations above the background concentrations or those that may have concentrations above the background based on the site history.

The selection of the most appropriate EILs to apply for the contaminants of concern is dependent on whether soil-specific EILs are available for the appropriate land uses(s). If soil-specific EILs are available, then the decision should be based on the physicochemical properties of the soil at the site. Otherwise, the selection will be based on land use. Schedule B1 provides examples on the application of EILs.

4.2 Receptor identification

In a Preliminary ERA, it is assumed that all biota and supporting ecological processes that are of ecological value to the land use (that is, areas of high ecological value, urban residential and open public space, commercial and industrial) are of ecological value to the site. However, where a particular species (for example, giant Gippsland earthworm) or type of organism (for example, soil microbial processes) that is an important part of the ecological value[2] at a site was not considered in the derivation of the most appropriate EILs (see Section 5.5), the EIL may not provide adequate protection and a Definitive ERA should be undertaken. The basis for such a decision should be clearly presented in the Preliminary ERA report.

4.3 Exposure assessment

In a Preliminary ERA, it is assumed that all exposure pathways considered in the derivation of the EILs are applicable. The physical setting of the site significantly influences exposure, since features such as soil type, soil organic matter content, paving and buildings can impact upon exposure pathways and contaminant availability. Exposure is also influenced by physical and chemical properties of the contaminants (for example, solubility in water, n-octanol/water partition coefficient (K_{ow}), soil/water partition coefficient and volatility). Each of these parameters may be evaluated to take account of site conditions, therefore providing a more site-specific estimate of the amount of a chemical an organism or a population may receive. If the results of the above analysis indicate that exposure pathways that are thought to be significant have not been considered, or that the magnitude of an exposure pathway is suspected to be underestimated in the derivation of EILs, a Definitive ERA should be undertaken. The basis for a decision to proceed or not to a Definitive ERA should be clearly presented with justifications in the Preliminary ERA report.

4.4 Toxicity assessment

In a Preliminary ERA, it is assumed that the toxicity data and methods used to calculate the endorsed EILs are sufficiently protective of the general ecological system and biota at the site. However, where it is suspected that this is not the case (for example, certain threatened or endangered species need to be protected), a Definitive ERA should be undertaken. The basis for such a decision should be clearly presented with justifications in the Preliminary ERA report.

4.5 Risk characterisation

In a Preliminary ERA, risk characterisation consists of the comparison of on-site soil contaminant concentrations with the most appropriate EILs for the contaminants of concern.

If the on-site soil concentration of any contaminant of concern is equal to or less than the most appropriate EIL, then the site contamination is considered unlikely to be having an adverse impact on ecological values.

[2] The species or organism type not included must be important to the ecological value of the site because the method used to calculate the EILs uses all the existing high quality toxicity data as surrogates to represent the sensitivity of all organisms at the site.

If the on-site soil concentration of any contaminant of concern is greater than the most appropriate EIL, the site contamination may be having an adverse impact on ecological values. Due to the general nature of data collected and the methods used to calculate EILs, the EILs are generally conservative. Therefore, levels of contamination above an EIL should not automatically necessitate remedial or clean-up action, but rather they trigger further evaluation.

The uncertainty associated with on-site soil concentrations (due to spatial heterogeneity both horizontally and vertically) and EILs and any conflicting results should be highlighted and discussed in the Preliminary ERA report.

If there is more than one contaminant of concern at the site then the risk posed by the combined effects of the contaminants should be assessed using the method set out in Appendix 2 of this Schedule.

It is important to consider the background concentration of contaminants of concern at the site or in sites with similar soil. If the most appropriate EIL for a contaminant of concern is lower than the background concentration, the background concentration becomes the EIL. It should be noted that this could only occur for EILs that are based on total concentrations rather than added concentrations[3].

4.6 Risk management decision and ERA outcomes

After risk characterisation, a risk management decision is necessary. This decision weighs up the findings of the Preliminary ERA against risk management considerations.
Factors that may influence a risk management decision (and therefore determine ERA outcomes) are generally based on economic, ecological or societal considerations as well as the scientific information and results generated within the Preliminary ERA. Examples include:

- the size of the site, land value, and cost of remediation (economic)
- the type of contaminants present, current and potential site land use, surrounding land use (societal)
- the ecological significance of the values identified in the receptor identification component of the Preliminary ERA that are to be protected (e.g. a rare and endangered species or a species that supports a valued ecological process or a sensitive introduced species of low ecological significance, e.g. a rabbit).

The risk management decision may also be determined or affected by the need to refine the uncertainty of the information gathered and/or to fill data gaps. Where the risk assessor has identified a high level of uncertainty in the risk characterisation (for example, because there was limited data from a site characterisation or because there was limited toxicity information for particular chemicals) then a decision manager may decide to either:

- develop and implement a site management/remediation program

or

- undertake further assessment and proceed to a Definitive ERA.

If the Preliminary ERA finds that the decisions on exposure and ecological values that were made in deriving the EILs were appropriate for the site and the risk characterisation suggests that there is unlikely to be an adverse impact on ecological values, the risk manager must decide to either:

- adopt the 'no action' outcome

or

[3] Wherever possible, the EILs were derived by expressing the toxicity data in terms of added concentrations (e.g. mg Cu added/kg soil). Then an added contaminant limit (ACL), the amount of a contaminant that can be added to a soil, was determined. To derive the EIL, the ambient background concentration was added to the ACL. Therefore, where the EIL is expressed in terms of added contaminant concentration, it is not possible for the EIL to be less than the background concentration (Heemsbergen et al. 2009).

- adopt the 'monitoring' outcome.

If however, the Preliminary ERA raises concerns about the suitability of decisions made in applying the EILs to the site and/or the risk characterisation suggests that there may be an adverse impact to ecological values, the risk manager must decide to either:

- develop and implement a site management/remediation program

or

- proceed to the Definitive ERA.

The decision that is taken depends on the level of estimated risk and the social, cultural economic and engineering considerations relevant to the site. Proceeding to a Definitive ERA may not be cost-effective where the cost of managing a site is relatively low. Risk reduction measures rather than further investigations can follow a Preliminary ERA if that is considered appropriate—this would be considered in consultation with the decision-maker.

Where there is no suitable EIL[4] for a contaminant of concern and the on-site concentrations of the contaminant are above background concentrations, the risk manager must decide to either proceed to a Definitive ERA or develop and implement a site management/remediation program. The decision should be based on a multiple-lines-of-evidence approach.

The expected output from a Preliminary ERA is a report that highlights the extent and degree of the on-site soil contamination and justifies the use and selection of the most appropriate EILs. An analysis of uncertainty in all the data used should also be included. Uncertainty and reporting are discussed later in this Schedule. The rationale for the final risk management decision should be explained in detail.

Risk managers may find it useful to consider the DQO approach as described in Schedule B2, which emphasises the importance of ensuring data collected for use in decision-making regarding a site is of an appropriate quality. A DQO approach should be adopted early in the assessment process in relation to data used in risk assessment and in making risk-management decisions based on estimates of risk.

[4] If available, EILs should always be used, but if they are not, then assessment levels from other jurisdictions can be adopted. However, it is important that any assessment levels adopted are calculated using a comparable method (preference to be given to SSD methodologies) and provide a comparable level of protection. A full justification for any limit adopted from another jurisdiction must be included in the Preliminary ERA report.

5 Definitive ecological risk assessment

Generally, a Definitive ERA is only commenced once a Preliminary ERA has been conducted and has demonstrated that the contaminants present at the site pose a potential ecological threat. This iterative procedure allows each tier of ERA to be reviewed to determine whether the assessment is meeting the objectives set and to establish what the next phase should be.

This section provides guidance on how to conduct a Definitive ERA (see Figure 4 below). In a Definitive ERA, the focus is on quantifying exposure levels through field studies and the use of sophisticated computer models. Emphasis is placed on gathering detailed, site-specific information as part of the receptor identification, exposure assessment and toxicity assessment. A summary of data that may be collected as part of a Definitive ERA is included in Table 3.

Based on site-specific information, site-specific EILs for soil are derived. The comparison of the on-site soil concentrations of contaminants of concern against the site-specific EILs characterises the ecological risk at the site and influences any outcomes.

Figure 4. Definitive ERA

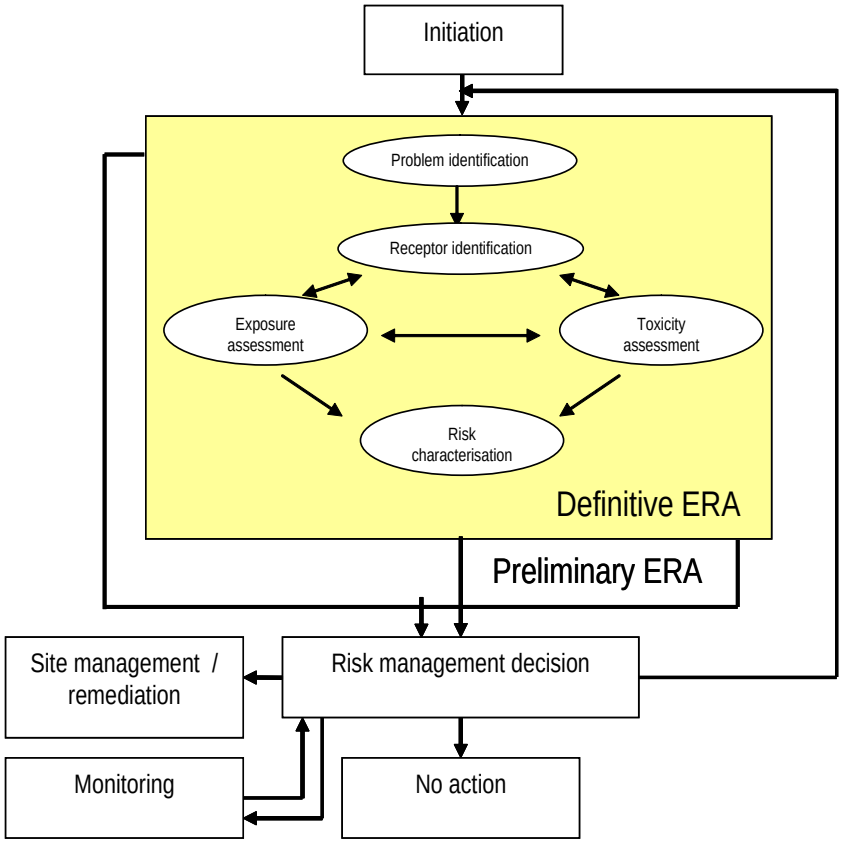


Table 3. Information that may be collected for each component of a Definitive ERA.

ERA component	Indicative requirements for a Definitive ERA
Problem identification	Refined objectives and updated CSM based on information in the Preliminary ERA Identification of contaminants of concern (including mixtures and contaminant form) that exceed EILs Formulation of the assessment end point, e.g. will the assessment end point be based on species abundance; growth rates, frequency of chlorosis or necrosis in plants; or failure to develop?
Receptor identification	Flora and fauna surveys of the site and surrounding area Identification of species of concern, including threatened and endangered species Ecosystem function and ecosystem interaction established Confident that the interface between biological monitoring plans and previous risk assessment is sufficiently robust to improve the risk assessment?
Exposure assessment	Fate and transport modelling of contaminants of concern Species-specific inhalation, ingestion and absorption rates Identification of on-site soil properties that affect contaminant mobility/availability (e.g. organic carbon content, pH, bulk density, porosity, soil moisture) Bioavailability factors Sampling and analysis of food, water and air for effects of contamination Information on biota behaviour relevant to assessing exposure
Toxicity assessment	Detailed literature review of relevant toxicological studies since the EILs were derived Results of in situ field or laboratory toxicity tests
Risk characterisation	Information on chemical mixtures, concentration of contaminants of concern (derived from problem identification)

5.1 Problem identification

When commencing a Definitive ERA, it is important to reconsider the objectives that were used for the Preliminary ERA, taking into account the results of the Preliminary ERA. If appropriate, new objectives should be identified.

The main objectives for a Definitive ERA should be to:

- identify contaminants of concern (including mixtures and contaminant form, such as metal valency state, e.g. As^{3+})
- produce clearly defined quantitative predictions regarding the current and future risks to site-specific ecological values due to contaminants at the site
- determine site-specific EILs that take into account the ecological values at the site.

The objectives of this stage may need to be revised from time to time and should always be informed by the outcomes of the preliminary ERA.

5.2 Receptor identification

In a Definitive ERA, a biological survey of the site and surrounding areas that may be affected by off-site migration of the contaminants of concern (and/or public consultation on both areas) may be conducted. The objective of this is to identify the key ecosystems, processes and species that may be adversely affected by the contamination. Assumptions made linking site ecological values to receptors should be documented in the ERA report. If any ecological values that were identified are not to be protected then the basis of this decision should also be reported.

5.3 Exposure assessment

Advanced models may be used to describe present and future transport, transformation and environmental partitioning of the contaminants of concern. These models will need to be refined and calibrated using actual field data to enhance the level of assurance of the model predictions. Such fate and transport models should examine the partitioning of the contaminants of concern between the environmental compartments (for example, water, soil, sediment, biota and air) that are relevant for the site and areas that may receive off-site migration.

In addition to transport models, specific information regarding food, soil, water, ingestion rates and inhalation rates may be estimated from site-specific field data, providing a specific exposure assessment for each biota.

The sampling and analysis of other environmental media for contamination such as food, air and water supplies may also provide specific exposure information.

Other techniques of exposure assessment may include biopsy analysis of tissues, body fluids or excreta of biota from the site.

Detailed analysis of the uncertainty of the exposure assessment should also be conducted to define the boundaries of the risk posed by the uncertainty levels in the exposure assessment. Various statistical techniques are available to determine the level of uncertainty and also to identify the most sensitive exposure assessment parameters.

This may guide further studies and field activities to reduce the uncertainty.

5.4 Toxicity assessment

As part of the Definitive ERA, it may be useful to review the currency of the toxicity data used in the derivation of the generic EILs. A detailed review of the literature since the EILs were derived should be conducted to update the toxicological profile of each contaminant of concern and mixtures of the contaminants. If there is additional data then it should have its quality and appropriateness assessed using the data quality assessment method in Schedules B5b and B5c. The acceptable quality data should then be added to the toxicity data used to derive the current EILs and new generic or soil-specific EILs derived using the method in Schedules B5b and B5c.

Alternatively, or in addition, the toxicity of each contaminant of concern and mixtures of the contaminants of concern may be measured directly. Such toxicity testing can be particularly useful where a site is contaminated by numerous contaminants and assessing the impact of the mixture from individual EILs is not straightforward, or where a site is contaminated by chemicals for which EILs do not exist, although in this case appropriately adapted data from similar studies may also be used.

Toxicity tests for a range of soil and terrestrial species have been developed by various regulatory and international agencies, for example, the American Society for Testing and Materials (ASTM), the International Standards Organisation (ISO 1993, 1995), the Organisation of Economic Cooperation and Development (OECD 1984a, 1984b), Environment Canada (EC 2004, 2005) and the United States Environmental Protection Agency (US EPA). Such standardised methods are generally preferred though at some sites it may be more appropriate to use endemic species for which there are no standardised toxicity test methods. The use of such tests is appropriate providing the methods used are based on standardised toxicity tests that have been modified to suit the test species and/or site conditions. The species to be used in site-specific toxicity tests and the experimental design should be based on information provided by the problem identification, receptor identification and exposure assessment components of the Definitive ERA.

Where toxicity testing is undertaken as part of a toxicity assessment, it is crucial that the end points measured are ecologically relevant. This includes tests with end points such as growth and reproduction rather than just biochemical changes that may or may not be adverse. The suitability of such non-standardised tests can be determined using the method in Schedules B5b and B5c, which assesses the quality of terrestrial toxicity data in terms of experimental design, analytical and statistical techniques used, and whether appropriate quality assurance and quality procedure measures were in place.

The toxicity tests can be conducted using artificial soils or soil from the site. They can also be conducted in the field or in the laboratory. The most environmentally relevant toxicity tests are those that expose species that occur (or would be expected to occur if the contamination was not present, based on known distributions), excluding threatened or endangered species, at the site or surrounding areas to the contaminants of concern in soil from the site. In addition, toxicity tests could be conducted using (1) uncontaminated soil from the site or similar sites that is spiked with increasing concentrations of the contaminants of concern, or (2) contaminated soil from the site diluted using an appropriate soil.

Toxicity tests that expose the test organisms for long periods of time—generally, greater than two weeks (that is, chronic tests)—are preferred for the derivation of EILs rather than those with short exposure durations (that is, acute tests). In order to derive site-specific EILs, toxicity data for certain minimum numbers of species that belong to a minimum number of taxonomic groups are required (Heemsbergen et al. 2009). It is strongly advised that the advice of appropriately qualified and experienced ecotoxicologists is sought before commencing any toxicity testing, in order to conduct toxicity tests that will be useable in deriving site-specific EILs.

A detailed analysis of the uncertainty, strength and relevance of the toxicity data that has been collated from the literature or generated through conducting toxicity tests should be reported.

The methodology for deriving soil-specific EILs is provided in Schedule B5b. Worked examples of the EIL derivation methodology can be found in Schedule B5c and details on how to derive relationships between soil physicochemical properties and toxicity are provided in Warne et al. (2008a, 2008b).

5.5 Risk characterisation

Data gained during the exposure and toxicity assessment phases are used to modify the assumptions underlying the EILs and to calculate site-specific EILs. The site-specific EILs should be calculated using the methodology described in Schedule B5b. The on-site concentrations of each contaminant of concern should then be compared to the site-specific EILs[5].

If the on-site soil concentration of contaminants is equal to or less than the site-specific EILs for each contaminant and the toxicity of the mixture of contaminants does not exceed the $EIL_{mixture}$ (see Appendix 2), the site contamination is considered unlikely to pose an adverse ecological impact.

If the on-site soil concentration of any contaminant of concern is greater than the corresponding site-specific EIL or the toxicity of the mixture exceeds the $EIL_{mixture}$ (see Appendix 2), the site contamination is considered to pose an adverse ecological impact.

5.6 Risk management decision and ERA outcomes

After risk characterisation, a risk management decision is necessary. If the risk characterisation suggests that there is unlikely to be an adverse impact to ecological values of the site (that is, on-site soil concentrations are equal to or less than the most appropriate site-specific **EIL**), the risk manager should decide between the 'no action' or 'monitoring' outcomes.

If the risk characterisation suggests that there may be an adverse impact to ecological values of the site (that is, on-site soil concentrations are greater than the most appropriate site-specific **EIL**), the risk manager should develop and implement a site management/remediation program.

Figure 4 above shows an arrow leading from the risk management decision back into the ERA process. This loop has been designed to allow for the further refinement of the characterisation of ecological risk. It uses a predictive approach based on monitoring undertaken as part, or as a result, of site management/remediation.

Expected outputs from a Definitive ERA include a report that extends the problem identification of the Preliminary ERA, provides detailed exposure and toxicity assessments for the contaminants as well as conclusions and recommendations. The report should detail the derivation of any modified site-specific **EILs** for the contaminants and describe the uncertainties in the field data (that is, contaminant levels and distribution) as well as in the modified **EILs**.

[5] If a site-specific EIL for a contaminant is lower than the ambient background concentration for the same chemical, the background concentration becomes the EIL.

6 Uncertainty

There are inherent limitations in ERAs similar to those facing any science-based endeavour. Given the stochastic nature of ecosystems, we cannot expect to predict the precise outcome for a population, community or functional process, as small changes in initial conditions can result in large differences in outcomes. The best we can do is estimate the probability of some outcome occurring.

Uncertainty also arises from the limitations we have in the data available. The scale of processes, the difficulty in understanding what the system should look like without the contamination, the limitations of our understanding and measurement of toxicity as well as our estimation of exposure, together with the fact that there are usually multiple and complex stressors involved, all contribute to uncertainty in any ERA. An informative discussion on these limitations is presented in Kapustka (2008).

Risk assessors need to be mindful of all of these issues in considering the reliability of their risk estimates. In some cases, the risks will clearly be present or clearly not present. In these situations, a risk characterisation decision can still be reached, even with very limited data. In other situations, even a large database may not provide sufficient information to permit a risk characterisation decision to be made about whether site contamination poses an unacceptable risk. The importance of uncertainty in an ERA is quite site-specific.

There is also some level of error in all the sampling, the measurements made and the modelling undertaken. These are additional aspects of uncertainty that need to be considered in any ERA.

Every ERA report should discuss the uncertainty in the risk estimate and the impact that uncertainty has on the decision.

Detailed discussion on the mathematical analysis of uncertainty may be found in Cox and Baybutt (1981), Hoffman and Gardener (1983) and Gardener et al. (1981). A number of uncertainty analysis computing programs have also been developed that may be useful in this context (for example, PRISM, @ RISK and Crystal Ball).

Depending on the site uncertainty, sensitivity analyses could be conducted to identify which sets of data are contributing the most to the uncertainty in the ERA. This could be used to direct subsequent work and thus reduce the overall uncertainty in the ERA.

7 Reporting

This section provides information about the recommended structure and content of both a Preliminary ERA and Definitive ERA report. Comments on the contents of ERA reports were included in previous sections about Preliminary and Definitive ERAs. The following is intended as guidance only, as the structure and content of reports will be heavily influenced by site-specific issues as well as client and regulatory requirements. The basic intent of this guidance is to provide a logical structure in a report that will facilitate understanding of the outcomes of the risk assessment by the risk managers, decision-makers and other readers of the reports (for example, stakeholders).
The ERA report should have the following main components:

- summary
- table of contents
- introduction
- problem identification
- receptor identification
- exposure assessment
- toxicity assessment
- risk characterisation
- uncertainty
- conclusions and recommendations
- references
- appendices.

Some of the components of a report are self-evident (such as the table of contents, introduction and references) and will not be further discussed.

The level of ERA will also determine the degree of complexity and completeness of the information and data analysis in each of these sections.

7.1 Summary

The summary should include the following information:

- the background to the site
- the rationale and objectives for conducting the ERA
- a description of the type of ERA conducted
- a description of the elements of the risk assessment
- a summary of the key conclusions of the risk assessment and recommendations arising from it.

The summary should be written in non-technical language and contain sufficient information to enable a non-technical reader to understand the approach and results of the risk assessment, independent of the rest of the document.

7.2 Problem identification

The problem identification section should include the following information:

- the objectives of the risk assessment
- DQOs and CSM considerations

- the background to the events leading to the conduct of a risk assessment
- the level of ERA being conducted
- a site description and history
- a summary of site information and data contained in any previous site assessment reports. This could include information about land use, site geology, soil contaminant concentrations and distribution, background concentrations, and regional and local hydrology
- an evaluation of quality assurance/quality control data on any previous field measurements and laboratory analysis contained in site assessment reports
- uncertainty estimates with respect to the site assessment data
- identification of key contaminants of concern (based on site history and any previous site assessment reports)
- conclusions that can be drawn about problem identification.

7.3 Receptor identification

The receptor identification section should include the following information:

- ecological values to be protected
- CSM considerations
- the approach used to identify ecological values that are potentially at risk
- an assessment of the possible spatial and temporal overlap of receptors and contaminants of concern (this would link in with the exposure assessment)
- basic life history and behaviour information about species identified as key receptors
- the sources and estimates of uncertainty
- conclusions that can be drawn about receptor identification.

7.4 Exposure assessment

The exposure assessment section should include the following information:

- the sources of the contaminants (if not already discussed in problem identification)
- the environmental fate and transport of the contaminants
- the magnitude, duration and frequency of exposure
- the applicable pathways with respect to the ecological receptors
- the sources and estimates of uncertainty
- conclusions that can be drawn about exposure assessment.

7.5 Toxicity assessment

The toxicity assessment section should include the following information:

- the toxicity of the contaminants
- the potential ecological effects at the individual organism, population and community levels
- known toxicity modifying factors (both synergistic and antagonistic resulting from exposure to multiple contaminants)

- indicators of ecological responses (e.g. suitable end-points)
- the sources and estimates of uncertainty
- conclusions that can be drawn about toxicity assessment.

7.6 Risk characterisation

The risk characterisation section of the report should use information gathered during the exposure and toxicity assessments to estimate the magnitude, probability and significance of ecological impacts occurring as a result of the concentration of contaminants present. An analysis of uncertainty should accompany this risk estimate.

7.7 Uncertainty

The uncertainty section of the report should include the following information:

- a summary of the analyses of uncertainty that have been undertaken for each component of the ERA and documented in various sections of the ERA report
- a discussion of overall uncertainty based on an assessment of all levels of uncertainty
- a discussion of the implications of the uncertainty for the findings of the report
- methods and indicative costs of reducing uncertainty (e.g. moving to higher levels of data collection, exposure assessment, etc.)
- conclusions that can be drawn about uncertainty.

7.8 Conclusions and recommendations

The conclusion section of the ERA should be brief and use the conclusions that have been drawn for each component of the ERA and documented in various sections of the ERA report. This section should summarise the results of the ERA in the context of the objectives of the study. Recommendations by the risk assessor to the risk manager/decision-maker regarding the characterisation of risk and possible ERA outcomes should be summarised in this section. Conclusions should be integrative in nature, combining all aspects of the assessment.

7.9 Appendices

Supporting documentation and information, such as previous site assessment reports, summary tables of all data used in the ERA, and maps/diagrams showing sampling locations, should be provided in the appendices of the report.

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9 Appendix A

Table A1: Summary of the EILs for fresh and aged contamination in soil with various land uses. Presented ranges are the EILs for a range of soil characteristics.

Contaminant	Age of contam	Added contaminant limits (mg added/kg soil) or EIL (mg/kg) for various land uses		
		Area of ecological significance[3]	Urban residential/ public open space[4]	Commercial & industrial[5]
Zinc[1]	fresh	7–130	25–500	45–800
	aged	15–280	70–1300	100–2000
Arsenic[2]	fresh	20	50	80
	aged	40	100	160
Naphthalene[2]	fresh	10	170	370
DDT[2]	fresh	3	180	630
Chromium (III)[1]	fresh	25–50	75–160	120–270
	aged	60–130	190–400	310–660
Copper[1]	fresh	15–60	30–120	45–200
	aged	20–80	60–230	85–340
Lead[1]	fresh	110	270	440
	aged	470	1100	1800
Nickel[1]	fresh	1–25	10–170	20–350
	aged	5–95	30–560	55–960

Notes:

[1] = the values presented for zinc, chromium (III), copper and lead are added contaminant limits (ACLs) based on added concentrations. The EIL is calculated from summing the ACL and the ambient background concentration (ABC).

[2] = the values presented for arsenic, naphthalene and DDT are generic EILs based on total concentrations. Insufficient information was available to calculate ACLs for these contaminants.

[3] = The standard protection level is 99%

[4] = The standard protection level is 80%

[5] = The standard protection level is 60%

Refer to:

- **Schedules B5b, and B5c for further details on the EIL methodology and derivation of the EILs including their reliability**
- **Schedule B1 for examples on the application of EILs**
- **the EIL calculation spreadsheet for calculating EILs specific to site soils.**

10 Appendix B: Mixtures of chemicals

A number of different types of joint action exist for mixtures of contaminants. Of these there are only predictive models for concentration addition (also called simple similar joint action) and response addition (also referred to as independent joint action). When all the chemicals in the mixture have the same mechanism of action, (that is, they exert their toxicity in the same manner at the same location), and they do not affect each other's biological activity in the organism, then the toxicity should conform to concentration addition (Plackett & Hewlett 1952). If, however, the chemicals have different mechanisms of action and they affect each other's biological activity, then the toxicity of the mixture should conform to response addition (Plackett & Hewlett 1952). Other types of joint action include synergism, antagonism, supra-addition, complex similar and dependent joint action.

The available literature shows that for the vast majority of mixtures, the toxicity conforms to concentration addition with relatively small numbers of antagonistic and synergistic mixtures. For example, Deneer (2000), Faust et al. (1994), Warne and Hawker (1995) and Ross and Warne (1997) found that approximately 10–30% of mixtures (regardless of the type of chemical, but focusing predominantly on organic chemicals) were antagonistic or synergistic, with each type of joint action being equally frequent and the remaining 70–90% conformed to concentration addition, based on aqueous concentration toxicity data. Similar values but with higher percentages of antagonistic and synergistic mixtures, (that is, 43% antagonistic, 27% additive and 29% synergistic), were found in a recent review by Norwood et al. (2003) of the aquatic toxicity of mixtures of metals.

It has also been shown (Backhaus et al. 2000a, 2000b; Chevre et al. 2006; Dyer et al. 2000; Faust et al. 1994; Junghans et al. 2006) that concentration addition overestimated the toxicity of mixtures and yielded slightly higher estimates of the toxicity of mixtures than response addition when chemicals had different mechanisms of action.

A two-step mixed model independently proposed by Junghans (2004), Altenberger et al. (2004), and De Zwart & Posthuma (2005) is, however, theoretically superior to the concentration addition method to estimate the toxicity of mixtures. In this model, the first step is to estimate the combined toxicity of components that have the same mechanism of action using concentration addition and then, if necessary, to estimate the combined toxicity of components or groups of components that have different mechanisms of action using the response addition model. But as the concentration addition method results in higher estimates of toxicity than the response addition method, it is not necessary to use the more complicated two-step mixed model method.

Given the above, it is appropriate to use the concentration addition model to estimate the toxicity of mixtures irrespective of the type of joint action, unless there is specific information in the literature about a mixture that shows that this model is inappropriate.

The hazard quotient (HQ) method described below is a modification of the concentration addition model that takes into account the use of EILs in the ERA framework. The HQ method requires the ratio of existing soil contaminant concentrations and the **EIL** for each individual chemical to be calculated.

$$HQ = X/E$$

where X is the concentration of a contaminant in soil, and E is the $\mathbf{EIL}_{SOIL}$ for that contaminant.

The sum of the HQ for each contaminant is calculated. The total toxicity of the contaminants present at a site, assuming they conform to concentration addition, is calculated by summing the HQs for each contaminant. The resulting value is called the Hazard Index (HI).

$$HI = HQ_A + HQ_B + HQ_C$$

where HQ_A is the HQ for contaminant A (that is, X_A/E_A), HQ_B is the HQ for contaminant B (that is, X_B/E_B), and HQ_C is the HQ for contaminant C (that is, X_C/E_C).

Where HI is equal to or less than 1, ecological values are assumed to be protected.

Where HI is greater than 1, there is potential for adverse impacts to ecological values.

That is, the sum of effects of simultaneous sub-threshold exposures to several contaminants may induce an effect equivalent to greater than the maximum tolerable dose for a single contaminant given in isolation.

11 Glossary

Added contaminant limit (ACL) values are generated in the process of deriving EILs and comprise the non-ambient background concentration in the EIL for the contaminant.

Aged applies to a soil that has contained a contaminant for more than two years.

Ageing is the natural process that occurs over time whereby the bioavailability of contaminants decreases due to binding to minerals, clays and organic carbon.

Ambient background concentration (ABC) of a contaminant is the soil concentration in a specified locality that is the sum of the naturally occurring background and the contaminant levels that have been introduced from diffuse or non-point sources by general anthropogenic activity not attributed to industrial, commercial, or agricultural activities.

Area of ecological significance is an area where the planning provisions or land-use designation is primarily for the intention of conserving and protecting the natural environment. This would include national parks, state parks, wilderness areas and designated conservation areas.

Bioavailability is the ability of a contaminant to interact with the biological system of an organism. Not all of a contaminant that is present in environmental compartments (for example, soil, sediment, water and air) is biologically available—rather, only a fraction of the total (the bioavailable fraction) is available.

Biomagnification is the accumulation and transfer of chemicals via the food web due to ingestion, resulting in an increase of the internal concentration in organisms at the succeeding trophic levels.

Biota of supporting ecological processes is the biota associated with supporting ecological processes that provide habitat, shelter, food and water and permit other organisms to reproduce and ultimately survive as a viable species. Examples include bacteria, fungi and soil invertebrates that sustain the nutrient cycling processes necessary for plant growth.

Contaminant is any chemical existing in the environment above background levels and representing, or potentially representing, an adverse health or environmental risk.

Contaminant of concern means a contaminant that is present at a site at concentrations that may result in adverse impacts to ecological values. Exactly how this is determined varies depending on the current situation and its place in the ecological risk assessment (ERA) framework. In the site contamination assessment phase, a chemical is considered a contaminant of concern when the concentration is greater than the background concentration of the chemical. At the conclusion of a Preliminary ERA contaminants of concern are those chemicals that have soil concentrations greater than the most appropriate ecological investigation levels (EILs). On completing a Definitive ERA, contaminants of concern are those chemicals that exceed the site-specific EILs.

Contamination means the condition of land or water where any chemical substance or waste has been added at above background level or bioavailability of a chemical substance has increased and represents, or potentially represents, an adverse health or environmental impact. This does not apply where materials are added in accordance with relevant government approvals or endorsements such as to improve its suitability for agriculture.

Definitive ecological risk assessment (Definitive ERA) is the second level of ecological risk assessment that can be conducted within the ERA framework of this Measure. This type of ERA is more detailed and provides a site-specific assessment of the risk posed by the contaminants.

Ecological investigation level (EIL) is the concentration of a contaminant above which further appropriate investigation and evaluation of the impact on ecological values will be required. The EILs are calculated using EC_{30} or lowest observed effect concentrations (LOEC) toxicity data. EILs are the sum of the added contaminant limit (ACL) and the ambient background concentration (ABC) and the limit is expressed in terms of total concentration. EILs depend on specific soil physicochemical properties and land use scenarios and generally apply to the top 2 m of soil.

Ecological risk assessment (ERA) is a set of formal, scientific methods for defining and estimating the probabilities and magnitudes of adverse impacts on plants, animals and/or the ecology of a specified area posed by a particular stressor(s) and the frequency of exposure to the stressor(s). Stressors include chemicals, changes in physicochemical properties such as temperature, other human actions and natural catastrophes.

Ecological risk management in the context of this Measure is a decision-making process that involves consideration of political, social, economic, scientific and engineering information together with risk-related information in order to determine the appropriate response to environmental contamination.

Ecological significance is the consideration of ecological significance and should include the impact of the contaminated site on the species, population or community and on-flowing impacts on the structure and function of the ecosystem.

Ecological values means plants, animals, fungi or ecological processes associated with a defined area that are considered to be of significant societal, ecological or economic significance.

Economic significance is the economic importance (for example, the contribution of local biota to tourism) and cost of maintaining biota.

ECx means effective concentration—the concentration which affects X% of a test population after a specified exposure time.

Exposure assessment is the estimation (qualitative or quantitative) of the magnitude, frequency, duration, route and extent (for example, number of organisms) of exposure of organisms present at a site to one or more contaminated media.

Exposure is the contact of a contaminant with any portion of an organism, system or sub-population. The organism may be exposed by inhalation, ingestion or dermal contact.

Generic ecological investigation levels (EILs) are EILs that are derived without considering any physicochemical properties of soil. When a generic EIL is developed for a contaminant there is a single numerical maximum concentration that is applicable to all Australian soils within each specified land-use.

Generic ecological value is an ecological value associated with a state, region, local area or standardised land-use category.

Hazard is the intrinsic capacity of a chemical, biological, physical or social agent to produce a particular type of adverse health or environmental effect. For example, one hazard associated with dichlorodiphenyltrichloroethane (DDT) is that it can cause the thinning of eggshells of some predatory birds.

Hazardous substance is a chemical that has the capacity to produce adverse effects. For the purposes of this framework, hazardous substance does not include radioactive, physical or biological agents.

High ecological value (see area of ecological significance)

Introduced flora and fauna are biota that are not native to Australia but which are desired to inhabit the site. Such biota may include wildlife, domestic animals, flowering plants, conifers and ferns.

Land use is based on the human purposes or economic activities that are conducted on a piece of land. This Measure specifies three land-use categories: (1) areas with high ecological value, (2) urban residential and public open space, and (3) commercial and industrial land.

Mixture ecological investigation levels ($EIL_{mixture}$) are EILs that take into account the joint action (toxicity) of mixtures of contaminants. If the $EIL_{mixture}$ is not exceeded, then no further investigation is required, whereas, if the $EIL_{mixture}$ is exceeded, then further investigation is triggered. If the $EIL_{mixture}$ is not exceeded in a Definitive ERA, it is considered that the mixture will not pose an adverse ecological impact, whereas if the $EIL_{mixture}$ is exceeded, then it is consider**ed that the mixture will** pose an adverse ecological impact.

National Environment Protection Measure (Measure) means a Measure made under section 14(1) of the *National Environment Protection Council Act 1994* (Cwlth) and the equivalent provisions of the corresponding Acts of participating states and territories.

Native flora and fauna are biota that would naturally inhabit the site in the absence of the chemical contamination. Such biota may include flowering plants, ferns and terrestrial, subterranean or arboreal fauna.

Octanol–water partition coefficient (K_{ow}) is the ratio of a chemical's solubility in n-octanol and water at equilibrium. This is widely used as a surrogate for the ability of a contaminant to accumulate in organisms and to biomagnify. These are often expressed in the logarithmic form (i.e. log K_{ow}). Chemicals with a log K_{ow} value ≥ 4 are considered in this report to have the potential to biomagnify. There is a linear relationship between log Kow and log Koc values. Thus, K_{ow} can also be used to indicate the ability of chemical to leach to groundwater. A log K_{ow} value <2 indicates a chemical has the potential to leach to groundwater.

Preliminary ecological risk assessment (Preliminary ERA) is the first level of assessment conducted in the ERA framework of this Measure. A Preliminary ERA is a generic assessment of the risk posed as it involves comparison of measured concentrations to the generic or soil-specific EILs for the relevant land use.

Receptor is the entity (organism, population, community, or set of ecological processes) that may be adversely affected by contact with, or exposure to, a contaminant of concern.

Risk means the probability in a certain timeframe that an adverse outcome will occur in a person, a group of people, plants, animals and/or the ecology of a specified area that is exposed to a particular dose or concentration of a hazardous agent, (that is, it depends on both the level of toxicity of the hazardous agent and the level of exposure).

Site means the parcel of land being assessed for contamination.

Site-specific ecological investigation levels are EILs that have been derived during a Definitive ERA. These EILs have taken into account various factors of the site; they are therefore site-specific and may not apply to any other particular site.

Site-specific ecological value is an ecological value that is specific to the site under investigation.

Societal significance is the significance that societies place on preserving biota and ecological processes. This can vary markedly depending on cultural issues and the type of species that are being considered (for example, cute and cuddly biota often have greater societal significance than insects, microorganisms and other invertebrates) and is not constant over time (for example, the importance of tree hollows as bird and arboreal species habitats has only relatively recently been appreciated by the broad community).

Soil is a complex heterogeneous medium that consists of variable amounts of mineral material, organic matter, pore water and pore air, and is capable of supporting organisms, including plants, bacteria, fungi, protozoans, invertebrates and other animal life. For the purposes of this guideline, soil includes geological materials (gravels, sands, silts, clays and porous rock), and anthropogenically deposited fill material (for example, crushed rock, broken bricks, gasworks ash, foundry sand, 'clean' fill.).

Soil-specific ecological investigation levels are EILs that are specific for a specified set of soil physicochemical properties. These would apply to all soils or sites that have this combination of soil properties and have the same land use.

Soil quality guideline (SQG) is a collective term used to describe any quantitative or qualitative limit that controls the concentration of contaminants in soils. Ecological investigation levels are a type of SQG. This term was used in Schedule B5b and B5c for contaminant limits as a range of values were derived using various toxicity data. Only SQGs derived from LOEC and EC_{30} toxicity data are adopted as EILs.

Toxicity assessment means the overall process of evaluating the type and magnitude of toxicity caused by a hazardous substance.

Toxicity means the quality or degree of being poisonous or harmful to plant, animal or human life.

Transitory or permanent wildlife includes wildlife that lives permanently or spends part of their life cycle on the site in question (for example, the site may be part of a bird's territory).

12 Shortened forms

ABC	ambient background concentration
ACL	added contaminant level
ANZECC	Australian and New Zealand Environment and Conservation Council
ASTM	American Society of Testing and Materials
CSM	conceptual site model
DQOs	data quality objectives
EC	Environment Canada
EC_{30}	30% effect concentration
EIL	ecological investigation level
ERA	ecological risk assessment
ESD	ecologically sustainable development
HI	hazard index
HQ	hazard quotient
ISO	International Standards Organisation
K_{ow}	octanol water partition coefficient
LOEC	Lowest observed effect concentration
OECD	Organisation of Economic Cooperation and Development
NEPM	National Environment Protection Measure
NHMRC	National Health and Medical Research Council
SSD	species sensitivity distribution
US EPA	United States Environmental Protection Agency

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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This compilation has been split into 22 volumes

Volume 1: sections 1-6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of

Schedule B5b

GUIDELINE ON

Methodology to Derive Ecological Investigation Levels in Contaminated Soils

Explanatory note

The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

The original Schedule B5 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document, together with Schedule B5a and Schedule B5c.

The National Environment Protection Council (NEPC) acknowledges the contribution of the Commonwealth Scientific and Industrial Research Organisation (CSIRO), the NSW Environment Protection Authority and the NSW Environmental Trust to the development of this Measure.

Contents
Methodology to derive ecological investigation levels in contaminated soils

1 Introduction

This guideline presents the methodology for deriving terrestrial ecological investigation levels (EILs) for three groups of land uses: (1) areas of ecological significance (2) urban residential/public open space, and (3) commercial/industrial. The methodology was developed to protect soil processes, soil biota (flora and fauna) and terrestrial invertebrates and vertebrates and is presented in this Schedule. Also addressed is the strength and limitations of the EIL derivation methodology. Technical notes on the methods used in the methodology are also provided. In developing the EIL derivation methodology, the approaches used by other countries were investigated and a summary of these is presented in Appendix A.

This methodology should be considered together with *National water quality management strategy – Australian and New Zealand guidelines for fresh and marine water* (ANZECC& ARMCANZ 2000) where there are risks of impact to the aquatic ecosystem.

2 EIL derivation methodology

2.1 Overview of the EIL derivation methodology

The methodology was developed being cognisant of both the methods used in other jurisdictions and of the existing methods used in Australia to derive water and sediment quality guidelines (ANZECC & ARMCANZ 2000; Simpson et al. 2005; Simpson & Batley 2007). The methodology is flexible and can deal with a variety of different land uses, risk pathways and toxicity data. It could be used to derive not just EILs but also other soil quality guidelines (SQGs) that have different purposes and/or different land uses. Examples of other SQGs include negligible risk target values, clean-up guidelines (goals that a site remediation must meet), intervention values (guidelines that, if exceeded, require immediate action in the form of remediation), and agricultural guidelines (guidelines to protect the long-term sustainability of agricultural land). The same basic methodology could also be used to derive guidelines for contaminants in products that are added to soil such as soil amendments, biosolids, fertilisers and re-use of wastes or by-products. In fact, guidelines for cadmium, copper and zinc for Australian biosolids applied to agricultural land have been developed using a very similar method (Warne et al. 2007, Heemsbergen et al. 2009). While the methodology can be used to derive other SQGs, this guideline will henceforth only focus on EILs.

An overview of the EIL derivation methodology is given in Figure 1. It consists of three main steps:

1. choosing the level of protection desired for the site
2. assessing exposure pathways
3. collating appropriate data for the selected exposure pathways and deriving EILs.

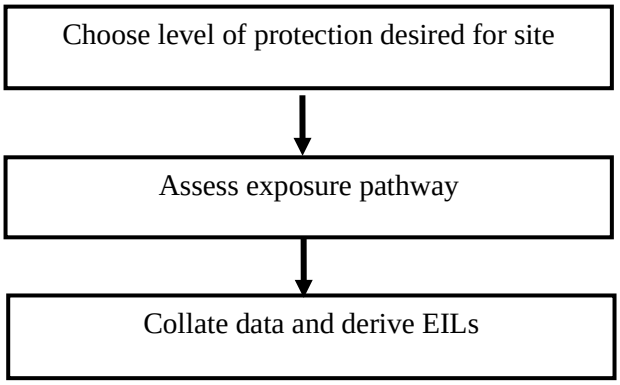


Figure 1. Overview of the methodology for the derivation of EILs.

2.2 Levels of protection

Selecting the level of protection to be provided to a site or soil is one of the most important steps in the EIL derivation methodology.

The level of protection provided will depend on:

1. **The species and ecological functions that should be protected—**every land use has specific functions and species that should be protected in order to ensure the land can continue to be used for that purpose. These functions and species include plants, soil microbial processes, soil and terrestrial invertebrates and vertebrates. For example, it would not be expected that all terrestrial species would be protected in an urban residential setting but it would be in national parks and areas of high ecological value.

2. **The exposure pathways that are relevant for the land use—**for terrestrial ecosystems in general, there are multiple potential exposure pathways. However, not all exposure pathways will be relevant for any particular land use. For example, exposure pathways that involve biomagnification are unlikely to be relevant to small industrial sites, as their surface area is limited.
3. **The extent to which the species and ecological functions will be protected—**using the preferred method for deriving EILs (that is, species sensitivity distribution (SSD) methods), it is possible to protect a hypothetical percentage of species/ecological functions (e.g. 99% or 95%) by an EIL. The extent of protection (that is, the percentage of species protected) can be changed depending on land use. For example, relatively low protection could be provided for commercial/industrial areas, and high protection for national parks and other high ecological value lands.

The land use-based approach has been adopted by several countries (for example, Germany and Canada). The Canadian soil quality guidelines (CCME 2006, Appendix A3) include four land-use types—agricultural, residential/parkland, commercial and industrial. Each land use has a list of relevant ecological receptors of concern to be included in the derivation of the Canadian SQGs. Furthermore, at industrial and commercial sites, a low level of adverse effects would be expected to occur in less than half of the species in the terrestrial community, as the CCME set the species protection level at 50%. Therefore, each land use type has its own SQG (CCME 2006).

The Australian and New Zealand water quality guidelines (WQGs) (ANZECC and ARMCANZ 2000) include a similar approach, which provides different levels of protection (that is, percentage of species) to aquatic ecosystems depending on how pristine the ecosystem is (that is, their current conservation status).
For pristine and thereby high conservation value ecosystems, slightly to moderately disturbed, and highly disturbed ecosystems, the default levels of protection in Australian aquatic ecosystems are 99% (PC_{99}), 95% (PC_{95}) and 90% (PC_{90}) or 80% (PC_{80}) of species, respectively (ANZECC & ARMCANZ 2000).

The EIL derivation methodology was used to derive a series of SQGs for eight contaminants using three different sets of toxicity data and thus providing three different levels of protection (Schedule B5c). For practicable application, the National Environment Protection (Assessment of Site Contamination) Measure (the NEPM) has adopted a combination of lowest observed effect concentration (LOEC) and 30% effect concentration data (EC_{30}) for derivation of the EILs[1]. For further information about this toxicity data refer to the Glossary and relevant Section.

2.2.1 Levels of protection for specific land uses

For all land uses (urban residential, public open space, commercial, industrial, agricultural, national parks/areas with high ecological value), with the exception of agriculture (see paragraph below on agricultural land), the following ecological receptors are relevant:

- biota supporting ecological processes, including microorganisms and soil invertebrates
- native flora and fauna
- introduced flora and fauna
- wildlife, i.e. secondary poisoning in birds and small rodents.

[1] In cases where the LOEC and/or EC_{30} data are not available but other measures of toxicity are then the latter will be divided by conversion factors to obtain the former (refer to Glossary, Section 2.4.2.1 and Table 8 for further details).

Henceforth, the above list of protected organisms will be referred to as 'species and soil microbial processes'.

The level of protection provided varies depending on the land use and whether the contaminant in question biomagnifies. Different levels of protection are aimed at protecting certain percentages of species and soil microbial processes[2]. The percentages of species to be protected will apply to the land uses irrespective of the purpose of the SQG. If a protection level is set at 80%, then theoretically 20% of the species and soil processes are at risk of experiencing adverse effects.

The toxic effects that this 20% of species/soil processes may experience will vary depending on the type of toxicity data that was used to derive the SQG. For example, for SQGs derived using NOEC (no observed effect concentration) or EC_{10} data, the potentially affected 20% of species/soil processes would experience toxic effects that were not significantly different to the controls or up to a 10% effect respectively. For SQGs based on EC_{50} data, the potentially affected 20% of species/processes could experience a 50% effect.

Biomagnification and the corresponding levels of protection should be enacted only when:

- the contaminant meets the criteria for biomagnification
- the surface area of the contaminated land exceeds a certain minimum surface area. The minimum surface area for urban residential/public open space is 250 m^2 and the minimum surface area for commercial, industrial and agricultural land is 1,000 m^2.

A summary of the percentages of species and soil microbial processes to be protected in soil with different land uses is given in Table 1 below.

Table 1. Percentage of species and soil processes to be protected for different land uses

Land use	Standard % protection	Biomagnification[a] % protection
Urban residential	80	85[b]
Public open space	80	85[b]
Commercial	60	65[c]
Industrial	60	65[c]
Agricultural	95[d] and 80[e]	98[c,d] and 85[c,e]
Areas of ecological significance	99	99

[a] if a contaminant meets the criteria for biomagnification, [b] if surface area exceeds 250 m^2, [c] if surface area exceeds 1,000 m^2, [d] agricultural crops, [e] for soil processes and terrestrial fauna.

The level of protection for some of the land uses are the same. Therefore, some of the land uses have been combined. Thus, in essence, there are only four different land uses: 1) national park/area with high ecological value, 2) urban residential/public open space, 3) commercial/industrial, and 4) agricultural. The NEPM focuses on the first three groups.

2.2.1.1 National parks and areas with high ecological value

National parks and areas with high ecological value are near-pristine ecosystems and should remain in that condition. As far as possible, it should be ensured that these ecosystems are not affected by soil contamination. Therefore, the appropriate level of protection is 99% of species. As this is the

[2] Protection is provided in terms of the percentage of species and soil microbial processes because the method used to derive EILs is a species sensitivity distribution method.

maximum percentage of protection possible (due to the statistical method used to calculate SQG), 99% is also the species protection setting for contaminants that biomagnify.

2.2.1.2 Urban residential and public open space

Henceforth, this grouping of land uses will be referred to as 'urban residential'. Urban residential lands are not pristine, rather, they are extensively modified, but they still retain many important functions and species. Stakeholders would expect these to be maintained. For example, it would be reasonable to expect that such land uses should sustain plant growth of both introduced (ornamental) and native species. To ensure viable growth of plant species, not only should plant toxicity data be considered but also soil health (for example, nutrient cycling and microbial functions). Nutrient cycling in soil ecosystems is essential for plant growth and therefore both microorganisms and soil invertebrates should be protected. Microorganisms are responsible for many processes regarding nutrient cycling—decomposition of organic matter, and N and P cycling processes (Marschner & Rengel 2007). Soil invertebrates have a number of important functions, including interacting with microorganisms regarding nutrient cycling, and modifying soil structure. In addition, many birds and small terrestrial animals feed on plants and soil invertebrates in urban areas. Therefore, secondary poisoning for some contaminants should be assessed to ensure adequate protection is provided to organisms high in urban food chains.

As urban residential lands are modified ecosystems, it would not be warranted or realistic to protect 95% of species and functions. Yet a reasonably high degree of protection is required in order to maintain the desired receptors and ecological functions. It has therefore been decided to protect 80% of species and soil microbial processes appropriate to this land use. For contaminants with a potential for biomagnification, the percentage of species protected should be raised by 5% to 85%.

2.2.1.3 Commercial and industrial land

Henceforth, these two land uses will be referred to as commercial/industrial land use. Ecosystems in commercial/industrial lands can be highly artificial. However, soils should still support the basic soil processes and should be able to recover if land use changes. Therefore, 60% of species will be protected for non-biomagnifying contaminants present in commercial/industrial land and 65% for contaminants that show biomagnification potential.

2.2.1.4 Agricultural land

The protection of crop species is vital to maintaining the sustainability of agricultural land and therefore 95% of the crop and grass species will be protected for this land use. Other plant species will not be used in the derivation of agricultural SQGs and therefore it will not be known what level of protection is provided by the SQG to native flora. Soil processes and soil invertebrates are highly important to ensure nutrient cycling to sustain crop species. However, tillage and the use of pesticides/herbicides make it unrealistic to protect 95% of soil processes and soil invertebrates and therefore only 80% of these will be protected. If a contaminant shows biomagnification potential, the percentage of species protected should be raised to 98% for crop species and 85% for soil processes and soil invertebrates. The lower of these two derived SQG values has been adopted as the agricultural SQG, and is included for information purposes only.

2.3 Determining the most important exposure pathways

It is important to determine the relevant exposure pathways for the combination of specific contaminants at a specific land use. For the sake of simplicity, many of the exposure pathways have been grouped into three pathways:

1. Direct toxicity – this is where the exposure to the organism occurs directly from either soil, soil pore water or air in soil pores. This includes pathways 1, 2 and 4 in Box 1 below.

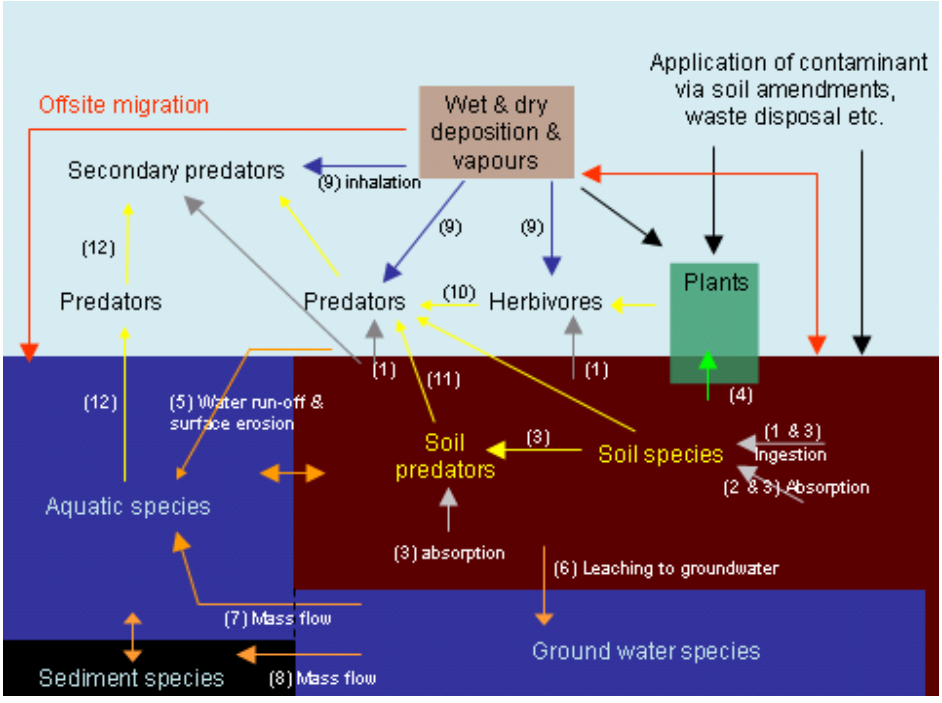
2. Biomagnification – this includes all exposure pathways where the source of the contaminant is food (organisms lower in the food chain). This includes pathways 3, 10, 11 and 12 in Box 1.
3. Metabolites – Metabolites are the breakdown products of the parent contaminant and require their own exposure pathway assessment.

The importance of the various exposure pathways can be determined by categorising the physicochemical properties of the toxicant and those of the receiving soil that control the environmental fate of chemicals. An overview of compartments within soil and the physicochemical properties that determine the fate of contaminants is given in Box 2 below. Several of the physicochemical properties shown are soil-dependent, for example, soil pH, cation exchange capacity, organic matter, clay content and dissolved organic carbon.

However, others are physicochemical properties of the contaminant itself, for example, partitioning between octanol and water (K_{ow}), its soil to water partition coefficient (K_d), Henry's law constant (H). These physicochemical properties can be used to determine the most important exposure pathways for contaminants. Organic and inorganic contaminants have different physicochemical properties that control their environmental fate and therefore different schemes for assessing exposure routes have been developed.

The EIL derivation methodology aims to protect soil and terrestrial species and soil processes. Potential off-site migration and its potential impacts are not included in the methodology. A recommended method for deriving EILs and/or other SQGs that also protects aquatic ecosystems is presented as an Appendix. Another issue that was considered for incorporation into the EIL derivation methodology was the bioavailability of the contaminants before addition to soil; for example, soluble contaminants versus those bound in insoluble forms. While this is a central issue in the management of contamination, it is not currently possible to incorporate this into the derivation of EILs and/or SQGs and the derivation assumes contaminants are 100% bioavailable. Some information on potential methods for assessing bioavailability and how it could be incorporated into a more detailed site-specific risk assessment is provided as an Appendix.

Box 1. Overview of potential exposure pathways in terrestrial ecosystems



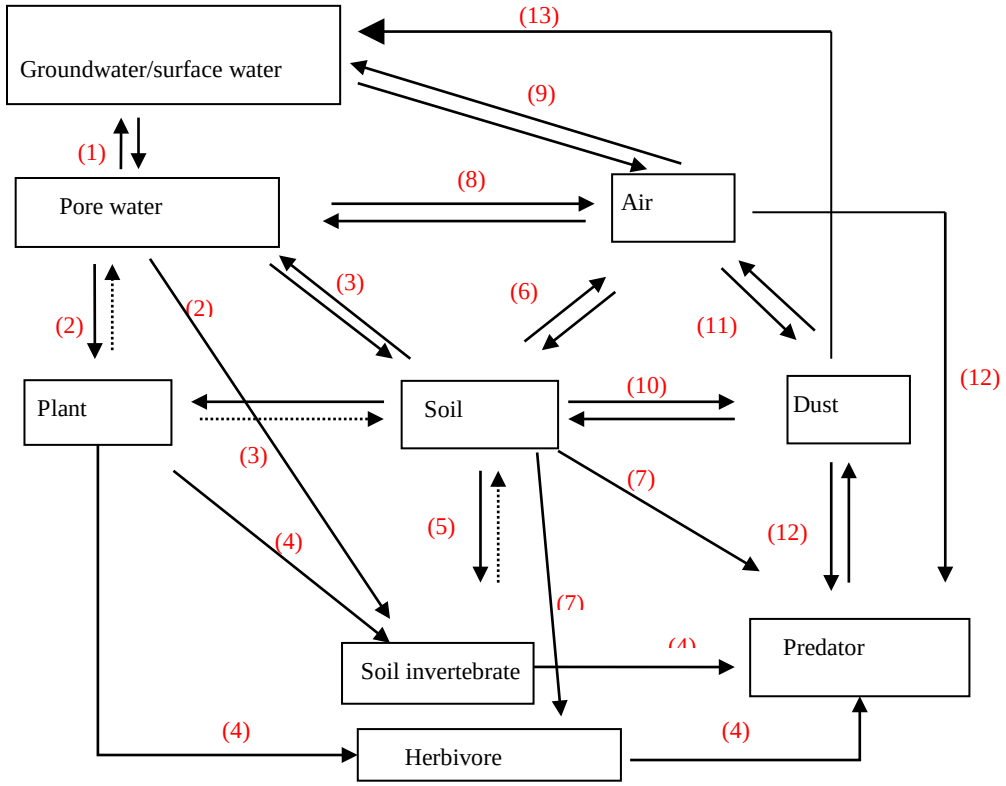
Exposure pathways

1. Soil – organism (via ingestion, organisms include herbivores and soil dwellers)
2. Soil – soil organism (passive absorption)
3. Soil – soil organisms – soil predators
4. Soil – plants
5. Soil – surface water – aquatic organisms
6. Soil – groundwater – stygofauna
7. Soil – groundwater – surface water – aquatic organisms
8. Soil – groundwater – sediment – mieofauna
9. Soil – air – terrestrial species
10. Soil – plant – herbivores – carnivores
11. Soil – soil organisms and/or soil predators – terrestrial predators
12. Soil – groundwater – surface water – aquatic organisms – aquatic predators

The exposure pathways can be grouped together:

- The direct toxicity pathways are 1, 2 and 4 and should be addressed for all contaminants.
- Leaching pathways include pathways 6, 7 and 8 and are relevant for site-specific ecological risk assessment. It will not be considered for general EIL derivation.
- Secondary poisoning includes pathways 3, 10, 11 and 12 and should be addressed for contaminants having biomagnification potential in the food web.
- A site-specific pathway for sloping land is pathway 5 and this should be assessed for contamination situated on slopes where down-slope migration of the contamination is possible. It will not be considered for general EIL derivation.
- Pathway 9 requires harmonisation of air quality guidelines with the soil quality guidelines but will not be used in the current process. Inhalation is more a human health issue and therefore the health investigation levels (HILs) using human toxicology assessment of inhalation is a much more accurate measurement of potential risk.

Box 2. Soil compartments, routes of environmental exposure and the key physicochemical properties that govern the distribution of a contaminant



Properties controlling the environmental fate and exposure routes of chemicals:

(1) soil porosity, water holding capacity (WHC), soil–water partition coefficient (K_d), precipitation

(2) octanol–water partition coefficient (K_{ow}), soil pH, pM^{n+} (free ion), ionic activity, electrical conductivity, and dissolved organic carbon (DOC)

(3) soil pH, cation exchange capacity (CEC), K_d, organic matter (OM), clay, DOC

(4) diet, metabolism, octanol–water partition coefficient (K_{ow})

(5) ingestion rate (diet), metabolism, absorption through skin, soil pH, CEC, K_d, OM, clay, DOC, K_{ow}

(6) sublimation constant (K_s)

(7) amount soil ingested, K_d, metabolism

(8) boiling point, K_{ow}, Henry's gas law constant (K_H)

(9) boiling point, K_{ow}, surface area, turbulence, wind speed

(10) erosion, plant coverage, WHC, % moisture

(11) sublimation constant (dust to air), K_d (air to dust), density of dust

(12) lung type, K_d, K_{ow}, breathing rate x volume

(13) wind speed, vicinity of water body.

2.3.1 Exposure pathway assessment for organic contaminants

The environmental fate of organic contaminants is largely controlled by three physicochemical properties:

1. half-life ($t_{1/2}$)
2. Henry's law constant (H)
3. octanol–water partition coefficient (K_{ow}) which, in general, determines a contaminant's potential to cause secondary poisoning.

2.3.1.1 Half-life

The half-life ($t_{½}$) of a contaminant is a measure of persistence of the contaminant in the environment. It represents the time taken for 50% of the contaminant to be lost from the environment. The loss may occur through biodegradation (microbially mediated degradation) or abiotic pathways (hydrolysis, oxidation, reduction, etc.). The more persistent a contaminant in the environment (that is, larger $t_{½}$), the longer is the potential exposure time of species to the contaminant and the more deleterious the effects that could occur[3].

In order to classify contaminants in terms of their half-lives, the most relevant comparison is their persistence (based on half-life) to the generation time of soil organisms. Soil organisms do vary greatly, with some microbes having generation times of hours, while earthworms have a generation time of approximately one year. A generic generation time of three months for soil organisms (microorganisms were not considered) was selected and the resulting categories of biodegradation rates can be found in Table 2 below.

Half-lives of contaminants depend on the soil physicochemical properties and therefore preference should be given on half-life values based on Australian soils. However, if this information is not available for Australian soils, then appropriate overseas studies can be used.

Table 2. Biodegradation rates, half-lives and the classification to be used in assessing the importance of the various exposure pathways for organic contaminants.

94% of contaminant degraded in (months)	$t_{1/2}$ (days)	$t_{1/2}$ Classification
<3	<22.5	Fast (F)
3–6	22.5–45	Moderately fast (M)
>6	>45	Slow (S)

2.3.1.2 Henry's law constant

Henry's law constant (H) is a measure of the volatility of the contaminant. The higher the volatility (or value of H) the more of the contaminant will volatilise and be found in the soil air and in the atmosphere. H is a temperature-dependent constant.

[3] This occurs because as exposure to a toxicant increases, the external ambient concentration needed to cause a toxic effect decreases.

Together with the $t_{1/2}$ of the contaminant, H is used to assess the transfer and persistence of the contaminant in the soil, as vapour transport for many contaminants may constitute an important pathway of loss and exposure to organisms.

Several researchers have used different cut-off values of H to class contaminants into volatile and non-volatile categories but, in most cases, for aquatic environments. Jury et al. (1983, 1984) categorised the behaviour of trace organic contaminants in soils using H (among other properties) and this is useful to assess the importance of the various exposure pathways for organic contaminants (see Table 3 below). Jury et al. (1983) used the Henry's law constant in dimensionless form as the ratio of concentration in the gas phase to concentration in the liquid phase, both in units of molar concentration, that is, H = (molar concentration in air)/(molar concentration in water)[4]. This is the most relevant form for estimation of the mass distribution of a chemical.

The dimensionless form of H based on concentrations (on a molar concentration basis) is the most commonly used of the dimensionless values (Staudinger & Roberts, 1996). The US EPA has published a calculator where Henry's law constant, H, can be estimated in different unit forms and at different temperatures. This can be accessed at www.epa.gov/athens/learn2model/part-two/onsite/esthenry.htm.

Table 3. Henry's law constant (H dimensionless) values to be used in assessing the importance of the various exposure pathways for organic contaminants

Henry's constant value (cm^3 air/cm^3 solution)	Classification
$>2.5 \times 10^{-3}$	Highly volatile (H)
2.5×10^{-7}–2.5×10^{-5}	Moderately volatile (M)
$<2.5 \times 10^{-7}$	Not volatile (L)

2.3.1.3 Octanol-water partition

The octanol–water partition (K_{ow}) is the ratio of the concentration of a contaminant that is dissolved in n-octanol to that dissolved in water at equilibrium and at a specified temperature. It is used as a surrogate to estimate the potential for contaminants to accumulate in tissue, both plant and animal (Connell 1989, Posthumus & Slooff 2001). The K_{ow} values can often be so large that the values are usually expressed as the logarithm to base 10 (that is, log K_{ow}). Contaminants with high log K_{ow} values are more likely to accumulate in plants and soil invertebrates than contaminants with low K_{ow} values (Connell 1989, Posthumus & Slooff 2001). If further magnification of these contaminants occurs in the food chain, the predators might experience toxicity while its prey does not. This effect is known as secondary poisoning.

Contaminants with log K_{ow} values below 4 are not considered to biomagnify, while highly fat soluble, lipophilic contaminants with log K_{ow} values equal to or greater than 4 are most likely to biomagnify. For most contaminants, it is expected that metabolism, excretion and degradation rates exceed the bioaccumulation rates at concentrations equivalent to the trigger values for protecting aquatic ecosystems (ANZECC & ARMCANZ 2000). Hence, only for contaminants with log K_{ow} values equal to or greater than 4 should secondary poisoning be considered. This approach is also consistent with the starting point to consider biomagnification used in the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000).

For the purpose of this methodology, the log K_{ow} values of contaminants are divided into two classes. These are:

- low, log K_{ow} <4: the contaminant has a low potential to biomagnify

[4] The H can also be expressed on a mass basis (e.g. mg/kg).

- high, log K_{ow} ≥ 4: the contaminant has a high potential to biomagnify.

2.3.1.4 Overview of the main exposure pathways for organic contaminants

Table 4 below presents the various combinations of the three physicochemical properties of organic contaminants described above and the resulting two exposure routes that are considered the most important for deriving EILs and/or SQGs.

Slowly degrading contaminants (that is, $t_{1/2}$ = slow, Table 2) with high log K_{ow} values and low H will have biomagnification as the most important exposure pathway followed by direct toxicity. If, however, these slowly degrading, high log K_{ow} contaminants have a high H, then direct toxicity will be the most important exposure pathway, followed by biomagnification.

For rapidly degrading contaminants (that is, $t_{1/2}$ = fast), the metabolites of the contaminant might have a larger impact on the environment than the parent contaminant. Therefore, it is necessary to assess the toxicity of the parent contaminant and to separately assess the toxicity and exposure pathways of the metabolites, as these can be markedly different from the parent contaminant. It would be preferable for metabolites to have their own EIL and/or SQG values. However, in practice, the number of EILs and/or SQGs for metabolites will be very limited due to a lack of knowledge of their toxicity and environmental fate.

Table 4. The properties (half-life $t_{½}$; logarithm of the octanol–water partition coefficient log K_{OW}; Henry's gas law constant H) used to assess the importance of the various exposure pathways for organic contaminants and the corresponding two most important routes

$t_{½}$ [a]	Log K_{ow} [b]	H [b]	Exposure routes to be considered	
			Primary	Secondary
S	H	L–M	Biomagnification	Direct toxicity
S	H	H	Direct toxicity	Biomagnification
S	L	L–M	Direct toxicity	Metabolites
S	L	H	Direct toxicity	Metabolites
M or F	H	L–M	Direct toxicity	Metabolites
M or F	H	H	Direct toxicity	Metabolites
M or F	L	L–M	Direct toxicity	Metabolites
M or F	L	H	Direct toxicity	Metabolites

[a.] S = slow, M = moderately fast, F = fast. [b.] H = high, M = medium, L = low

2.3.2 Exposure pathway assessment for inorganic contaminants

2.3.2.1 Biomagnification

There is no straightforward physicochemical property of inorganics that will predict their biomagnification potential, unlike organic contaminants. In the past, the bioconcentration, bioaccumulation and biomagnification factors (BCF, BAF and BMF respectively) have been used for this purpose, but this is not appropriate (Luoma & Rainbow 2008). Unless there is clear evidence that an inorganic element does not biomagnify, it should be considered to biomagnify and therefore secondary poisoning should be considered when deriving the EIL and/or SQG for that contaminant. A preliminary list of inorganic elements that do and do not biomagnify is given in Table 5 below.

Table 5. A preliminary list of inorganics known to biomagnify or known to not biomagnify based on information in the literature.

Biomagnification status	Inorganic contaminants
Known to biomagnify	Cd, Hg (especially methyl forms), Se
Known to not biomagnify	As, Cu, Fe, Mg, Pb, Zn

Only three biomagnification classes for inorganics should be used: known biomagnifiers, known non-biomagnifiers, and unknown biomagnifiers (which are then treated as biomagnifiers pending further investigation).

2.3.2.2 Henry's law constant

Henry's law constant (H) is a measure of the volatility of the element, as described previously. Inorganic elements and contaminants in general have very low volatility. Therefore, exposure pathways involving volatility should only be considered for mercury. These have not been included in the method used to determine the important exposure routes for inorganics.

2.3.2.3 Overview of main exposure pathways for inorganic contaminants

Table 6 below presents the two exposure routes for inorganic contaminants that are considered the most important for deriving EILs and/or SQGs, depending on whether the contaminant biomagnifies or not.

For unknown and known biomagnifying inorganics, secondary poisoning should be addressed. For all inorganic contaminants, direct toxicity to relevant species and soil processes should be addressed.

Table 6. The property used to conduct the inorganic contaminant exposure pathway assessment with the corresponding two most important exposure routes

Biomagnifies	Exposure routes to be considered	
	Primary	Secondary
Yes	Biomagnification	Direct toxicity
No	Direct toxicity	–
Unknown	Biomagnification	Direct toxicity

2.4 Derivation of EIL values

A schematic of the methodology to derive EILs for contaminants is given in Figure 2 below. The main steps in the methodology are:

1. collation and screening of the data
2. standardisation of the toxicity data
3. incorporation of an ageing/leaching factor for aged contaminants
4. calculation of the added contaminant limit (ACL) by either the SSD or assessment factor (AF) approach, depending on the toxicity data
5. normalisation of the toxicity data to an Australian reference soil. This is only done if the SSD approach is used to calculate the ACL
6. accounting for secondary poisoning for those contaminants that are considered to biomagnify in the food web
7. calculation of the ambient background concentration (ABC) of the contaminant in the soil (if appropriate)
8. calculation of the EIL or SQG by summing the ACL and ABC values

EIL = ABC + ACL *(equation 1)*

The separation of naturally occurring concentrations of a contaminant and the added contaminant in deriving EILs and/or SQG is based on the 'added risk approach' (Struijs et al. 1997; Crommentuijn et al. 1997). This approach assumes that the availability of the ABC of a contaminant is zero or

sufficiently close that it makes no practical difference. But, more importantly, it assumes that the background 'has resulted in the biodiversity of ecosystems or serves to fulfil the needs for micronutrients for the organisms in the environment' (Traas 2001). Therefore, the approach views only the effect of added contaminants to the environment as adverse. This approach is mostly relevant for ecological risk assessment (ERA) but less relevant for human risk assessment.

Evidence supporting the assumptions of the added risk approach has been provided by Posthuma (1997) and Crommentuijn et al. (2000b) and by work showing that the availability of metal salts decreases over time through aging processes (Posthuma 1997; Song et al. 2006). However, for microbial communities the background might be important regarding the development of tolerance to the metals (Díaz-Raviña & Bååth 1996; Bååth et al. 1998; Rutgers et al. 1998; McLaughlin & Smolders 2001; Rusk et al. 2004; Fait et al. 2006; Broos et al. 2007). Some of these studies found positive relationships between metal background concentration and effect concentrations, which could indicate that microbial communities in soils with relatively high background metals have evolved to be more tolerant to additional metal. Although these studies have shown that background concentration might not be completely inactive, adaptation of microbial communities does not lead to an underestimation of the ACL; rather, it is more likely to cause overprotection for microorganisms.

2.4.1 Collation and screening of data

2.4.1.1 Toxicity data collation

The first step in the methodology of deriving an EIL and/or SQG is to conduct a literature review and/or to search databases, such as the US EPA ECOTOX database (US EPA 2004), Australasian ecotoxicology database[5] (Warne et al. 1998; Warne & Westbury, 1999; Markich et al. 2002; Langdon et al. 2009) or the ECETOC database (ECETOC 1993), for available toxicity data for the contaminant in question. Unlike the situation in the derivation of HILs, it is not appropriate to have a hierarchy of data sources to be used in deriving EILs and/or SQGs. For most metals and well-known organic contaminants, toxicity data in addition to that found in the above databases will be available in the literature. Therefore, one should not rely solely on these databases.

For many organic contaminants there will be no toxicity data available. If there is no toxicity data available, models can be used to predict toxicity. These models include quantitative structure–activity relationships (QSARs) and quantitative activity–activity relationships (QAARs). The Australian and New Zealand WQGs (ANZECC and ARMCANZ 2000) used QSARs to derive trigger values (TVs) for narcotic organic contaminants (for example, ethanol for marine waters) when there was insufficient data. If QSARs or QAARs are not available, the equilibrium partitioning method (Van Gestel 1992; ECB 2003) can be used if toxicity data is available for aquatic species.

[5] The Australasian Ecotoxicology Database is in the process of being placed on the CSIRO website. It should be available in 2011.

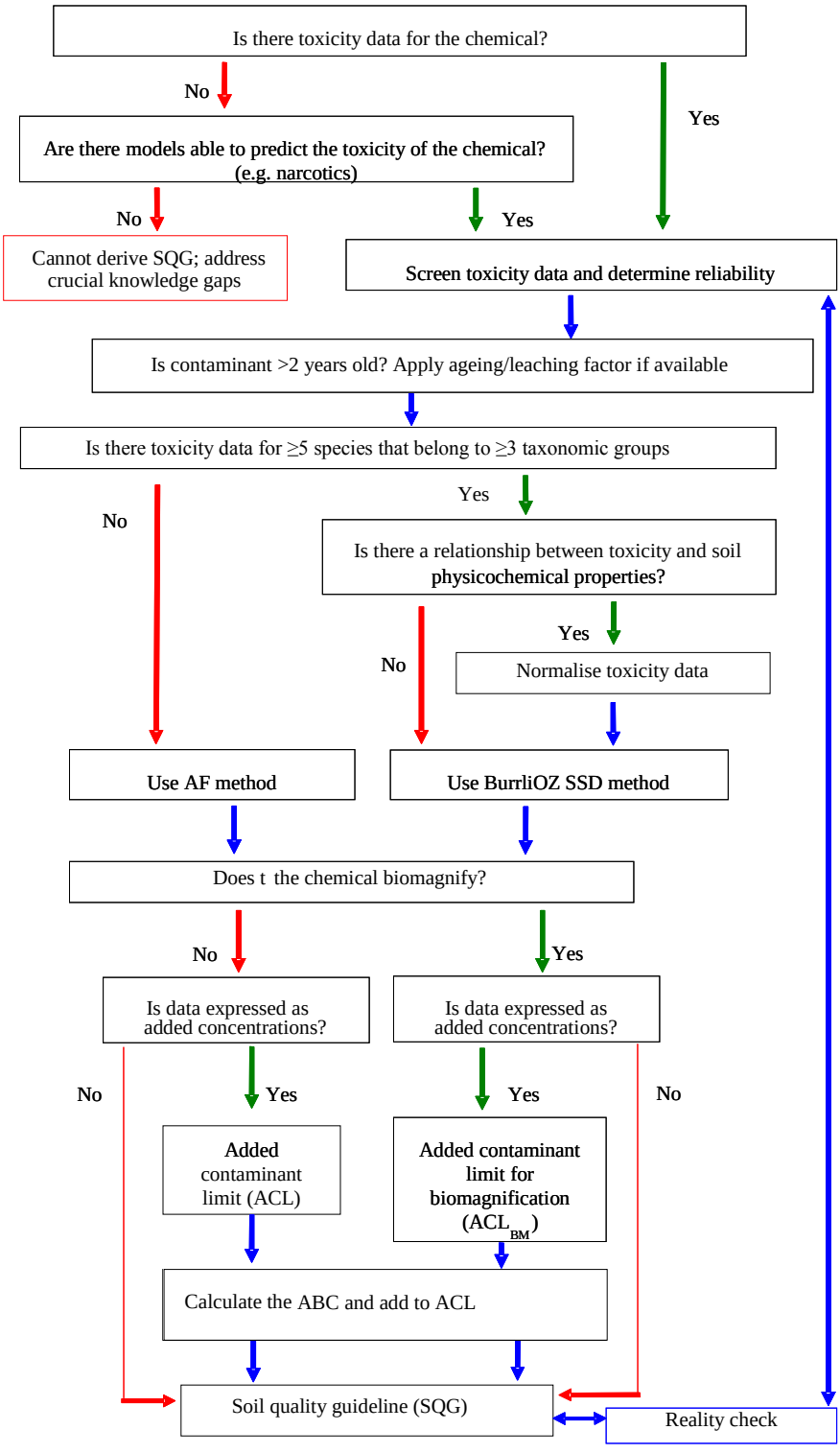


Figure 2. Schematic of the methodology for deriving ecological investigation levels (EILs) for Australian soils.

2.4.1.2 Quantitative structure–activity relationships

QSARs are empirical relationships between the toxicity of contaminants to a particular test organism and one or more physicochemical properties of the contaminant. QSARs are derived for contaminants

with either the same mechanism of action or similar contaminant structures. The most widely used physicochemical property is log K_{ow}. An example of a typical QSAR is presented below:

$$\log EC_{50} = -0.72 \log K_{ow} + 3.37 \quad \textit{(equation 2)}$$

where log EC_{50} (µmol/L) is the concentration at which 50% growth inhibition of lettuce (*Lactuca sativa)* was observed (Hulzebos et al. 1991).

The toxicity of contaminants with the same mechanism of action or chemical structure as those in the QSAR can be predicted based on their physicochemical properties. The prediction is made by substituting the value of the contaminant into the QSAR. If equation 2 was being used, the log K_{ow} of a contaminant would be substituted into the equation.

QSARs have been developed for terrestrial plants (Hulzebos et al. 1991) and invertebrates (Van Gestel et al. 1991); however, they are not as widely available as for aquatic species (Posthumus & Slooff 2001). Only QSARs derived using terrestrial species should be used to derive EILs and other SQGs.

2.4.1.3 Quantitative activity–activity relationships

The simplest forms of QAARs are empirical relationships that model the toxicity of contaminants with the same mechanism of action to one species using toxicity data of another species. These are termed binary relationships. An example (Westbury et al. 2004) is provided below:

$$\log EC_{50} (\textit{C. d.}) = 0.848 \log LC_{50} (\textit{P. r.}) + 0.047 \quad \textit{(equation 3)}$$

where log EC_{50} (*C. d.*) is the log of the concentration that causes a 50% immobilisation of the cladoceran *Ceriodaphnia dubia*, and log LC_{50} (*P. r.*) is the log of the concentration that kills 50% of the fish *Poecilia reticulata*.

More complex QAARs have been developed that relate the toxicity of contaminants simultaneously to multiple species (Raimondo et al. 2007; Morton et al. 2008). Both the simple and more complex QAARs allow toxicity data for one or more species to be used to estimate the toxicity to another species. Thus they can fill some of the data gaps that often occur in deriving EILs or their equivalents.

2.4.1.4 Equilibrium partitioning method

The equilibrium partitioning method (EqP) is used to predict the toxicity of a contaminant in soils based on aquatic toxicity data. The EqP is based on the assumption that the main route of exposure for soil organisms is the soil pore water concentration (Van Gestel 1992; ECB 2003). Therefore the EqP is not suitable for:

- contaminants with log K_{ow} values >4 (as they partition to soil rather than soil pore water)
- contaminants with a specific mode of action (e.g. endocrine disruptors)
- species that are exposed primarily through food
- aquatic species that have no direct terrestrial equivalent (e.g. fish)
- species where the main exposure pathway in terrestrial systems is dissimilar to that in water.

Therefore, the EqP method should only be used to assess the toxicity of the following taxonomic groups, as they meet the above criteria: annelida, bacteria, fungi, hexapoda (larvae only), nematoda, protozoa and tardigrades.

The EqP estimate of a NOEC for a contaminant in soil ($NOEC_{soil}$) is calculated from the NOEC of aquatic species as indicated below:

$$NOEC_{soil} = \frac{K_d}{RHO_{soil}} \cdot NOEC_{water} \cdot 1000$$

(equation 4)

where RHO_{soil} is the bulk density of the saturated soil and K_d is the soil–water partitioning coefficient (L/kg) (ECB 2003).

While there has been work done overseas to assess the validity of the EqP method (Van Beelen et al. 2003), there has been no such work undertaken in Australia. This is not a preferred method as Australian soils are relatively old, have low concentrations of nutrients, low organic carbon contents and different clay mineralogy (Taylor 1983), and are thus quite different from European and North American soils.

2.4.1.5 Screening and selection of toxicity data

The next step in the methodology is to determine the suitability of the available toxicity data. Toxicity data is considered acceptable when the:

- difference between tested concentrations was not greater than five-fold
- exposure duration was greater than or equal to 24 hours
- toxicity end point measured was growth, seedling emergence, lethality, immobilisation, reproduction, population growth or the equivalent
- measured toxic effect was a given percentage effect concentration (e.g. LC_{10}, EC_{50}) or were NOEC, LOEC or MATC (see the Glossary) values.

Biomarker end points, like enzyme production, lysosomal damage and avoidance responses, are considered to be less ecologically relevant and therefore they should not be used for the derivation of EILs unless data is limited and the predictive methods discussed in the previous section are not suitable. Biomarker tests are very sensitive and are therefore considered as early warning tests. However, if such data is used to derive EILs, this should be clearly stated. Biomarker data can be highly relevant for site-specific ecological risk assessment.

Once the unsuitable toxicity data has been removed, the next step is to assess the quality of the remaining data. Such screening methods are used in the methodologies of most countries to derive environmental quality guidelines (EQGs); for example, in Denmark, the Netherlands and the USA. However, in most cases, how the data was screened is not described. A screening method was used for the Australian and New Zealand WQGs (Warne et al. 1998; Warne 2001). This method assessed whether appropriate experimental designs, chemical analyses and statistics were used to obtain the toxicity data.

The method was based on the method used within the US EPA AQUIRE database, which was later renamed the US EPA ECOTOX database (US EPA 1994, 2004) but was improved by Warne et al. (1998).

These methods were subsequently reviewed and further improved by Hobbs et al. (2005). The Hobbs et al. (2005) data quality assessment procedures were modified so they were suitable for terrestrial ecotoxicity data (see Table 7) for use in this guideline.

Table 7. Scheme to assess the quality of terrestrial ecotoxicology data. This has been modified from the aquatic scheme of Hobbs et al. (2005).

	Question	Marks awarded
1	Was the duration of the exposure stated (e.g. 48 or 96 hours)?	10 or 0

	Question	Marks awarded
2	Was the biological end point (e.g. immobilisation or population growth) stated and defined (10 marks)? Award 5 marks if only the biological end point is stated.	10, 5 or 0
3	Was the biological effect stated (e.g. LC or NOEC)?	5 or 0
4	Was the biological effect quantified (e.g. 50% effect, 25% effect)? The effect for NOEC and LOEC data must be quantified.	5 or 0
5	Were appropriate controls (e.g. a no-toxicant control and/or solvent control) used?	5 or 0
6	Was each control and contaminant concentration at least duplicated?	5 or 0
7	Were test acceptability criteria stated (e.g. mortality in controls must not exceed a certain percentage) (5 marks)? or Were test acceptability criteria inferred (e.g. test method used (US EPA, OECD, ASTM, etc.)) (award 2 marks). Note: Invalid data must not be included in the database.	5, 2 or 0
8	Were the characteristics of the test organism (e.g. length, mass, age) stated?	5 or 0
9	Was the type of test media used stated?	5 or 0
10	Were the contaminant concentrations measured?	4 or 0
11	Were parallel reference toxicant toxicity tests conducted?	4 or 0
12	Was there a concentration–response relationship either observable or stated?	4 or 0
13	Was an appropriate statistical method or model used to determine the toxicity?	4 or 0
14	For NOEC/LOEC data, was the significance level 0.05 or less? or For LC/EC/BEC data, was an estimate of variability provided?	4 or 0
15	Were the following parameters measured and stated? (3 marks if measured and stated, 1 if just measured) pH OM or OC content clay content CEC	 3, 1 or 0 3, 1 or 0 3, 1 or 0 3, 1 or 0
16	Was the temperature measured and stated?	3 or 0
17	Was the grade or purity of the test contaminant stated?	3 or 0
18	Were other cations and/or major soil elements measured? or Were known interacting elements on bioavailability measured (e.g. Mo for Cu and Cl for Cd)?	3 or 0
19	For spiked soils with metal salts: were the soils leached after spiking?	3 or 0
20	Were the incubation conditions and duration stated?	3, 1 or 0
	Total score Total possible score for the various types of data and contaminants: 102	
	Quality score (%) (Total score /102 * 100)	
	Quality class (H ≥80%, A 51%–79%, U ≤ 50%)[a]	

[a] H = high quality, A = acceptable quality and U = unacceptable quality.

Each experimentally derived toxicity datum should have its quality assessed by the data quality assessment scheme (Table 7), which asks 20 questions, with marks awarded depending on the answer to the questions. The quality score for each datum is determined by expressing the total score obtained as a percentage of the maximum possible score. The toxicity data is then classified into three classes

depending on the quality score. Data with a quality score ≤50%, between 51% and 79% and ≥80% were classed as unacceptable (U), acceptable (A), and high (H) quality respectively. Only acceptable and high quality data should be used to derive EILs.

Only toxicity data expressed as either added or total soil concentrations should be used to derive EILs. There is considerable evidence both from overseas (Smolders et al. 2003; Smolders et al. 2004; Oorts et al. 2006; Zhao et al. 2006) and within Australia (Broos et al. 2007; Warne et al. 2008b) that chemical extract concentrations; for example, calcium chloride, ammonium nitrate and soil solution extracts, are not necessarily better measures of bioavailability than total concentrations for inorganic contaminants where contamination occurred in soluble forms. Furthermore, there is also considerably more toxicity data expressed as total metal concentration, and there is regulatory acceptance and understanding of this concentration measure.

2.4.2 Standardisation of the toxicity data

By this point in the methodology, the available toxicity data has been collated or models used to derive estimates and the data has been assessed for its appropriateness and quality. The obtained data requires standardisation in terms of four factors:

1. measures of toxicity
2. the toxicity expressed in terms of added concentrations
3. duration of exposure
4. use of toxicity data for endemic or overseas species.

Please note that this is not the normalisation step that accounts for the effect that soil characteristics have on toxicity values.

2.4.2.1 Measures of toxicity

There are many different measures of toxicity. The most frequently used toxicity measures to derive EQGs are NOECs and EC/LC_{50}-type data. However, not all studies report these particular measures of toxicity; for example, the toxicity may be reported as an EC_{25} or an LC_{40}. Therefore, in order to maximise the data available to derive EILs, it may be necessary to estimate the reported toxic effect.

A number of studies (Moore & Caux 1997; US EPA 1991; Hoekstra &Van Ewijk 1993) have shown that NOECs, while not statistically different from the control, typically correspond to a 10–30% effect, with 75% of NOECs corresponding to less than a 20% effect (Moore & Caux 1997). LOEC values would of necessity cause higher percentage effects and have a median of 30% (Moore & Caux 1997). For the purposes of this methodology, toxicity data that caused less than a 20% effect; for example, EC_0 to ≤EC_{19}, are considered equivalent to NOEC data and for brevity are referred to as NOEC and EC_{10} data. Toxicity data that cause a 20–40% effect are considered equivalent to LOEC data and are referred to throughout this guideline as LOEC and EC_{30} data. Toxicity data that cause >40–60% effect are considered equivalent to EC_{50} data and are referred to as EC_{50} data.

Due to the general paucity of terrestrial ecotoxicology data, if toxicity data is not expressed as a single value but instead is given as ranges, then the lowest value of the range should be used in order to provide a conservative estimate of the toxicity. In certain studies, the lowest toxicant concentration had already caused significant toxic effects and therefore toxicity data are given as a < or ≤ value. If possible, the percentage effect that the reported concentration caused should be determined and, using the ranges stated in the previous paragraph, be considered equivalent to NOEC, LOEC or EC_{50} data, and they should be converted accordingly. Toxicity with an effect greater than 60% should not be used to derive EILs. If, in studies, the highest tested concentration did not cause an effect or a statistically significant effect on the test species (that is, an unbounded NOEC), then the toxicity data should be given a > value and treated as an EC_{10}. This is done as it is a conservative approach and will result in more toxicity data available for EIL and/or SQG derivation.

As stated earlier, EILs are to be derived using LOEC and EC_{30} toxicity data. But such data is not always generated in toxicity studies. Therefore, in order to maximise the data available to derive EILs, toxicity data can be converted to LOEC and EC_{30} data. Two different approaches were applied to the different measures of toxicity data in the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000). For organics, only chronic NOEC data was considered acceptable to derive high reliability TVs, while only acute EC/LC_{50} values were suitable for moderate reliability TVs and either NOEC or EC/LC_{50} data was suitable for low reliability TVs (Warne 2001). In contrast, for metals, chronic NOEC, LOEC, EC/LC50 and maximum acceptable toxicant concentrations (MATC) values could be used provided all non-NOEC values were converted to NOEC values (Warne 2001). This was done using a series of default conversion factors (see Table 8 below). The reason for the different approaches was that for the organic contaminants, generally the chronic data was NOEC values, whereas the vast majority of the chronic metal toxicity data was EC/LC_{50} values (Warne 2001).

Table 8. Default conversion factors used to convert different chronic measures of toxicity to chronic NOECs in the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000). Values are from Warne (2001).

Toxicity data[a]	Conversion factor
EC_{50} to NOEC or EC_{10}	5
LOEC or EC_{30} to NOEC or EC_{10}	2.5
MATC* to NOEC or EC_{10}	2

[a] EC_{50}, EC_{30} and EC_{10} values are the concentrations that cause a 50%, 30% or 10% effect, NOEC = the no observed effect concentration, LOEC = lowest observed effect concentration, MATC = the maximum acceptable toxicant concentration and is the geometric mean of the NOEC and LOEC.

The more flexible method that was applied to the metals in the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000) and the conversion factors that were used (see Table 8) were used in the EIL derivation methodology. It should be noted that these conversion factors are based on expert judgement (Warne *pers. comm.*). Therefore, if sufficient terrestrial data is available to derive terrestrial conversion factors then these should be used. For example, data from the Australian National Biosolids Research Program indicates that the phytotoxicity chronic EC_{10} to chronic EC_{50} conversion factor for cations such as Cu and Zn was 3 (unpublished data).

Compared to aquatic toxicity studies, there is a limited number of terrestrial toxicity studies. Therefore, maximum use must be made of the available toxicity data and data should be converted from one measure to another (see above).

However, if more data become available then it should be used in the following descending order of preference:

1. 30% effect data (e.g. EC_{30}, LC_{30})
2. LOEC data
3. 10% or 50% effect data (e.g. EC_{10}, LC_{50})
4. NOEC and MATC.

There are a number of well-acknowledged limitations to NOEC and LOEC data (Newman 2008; Fox 2008; Warne & Van Dam 2008). Some scientists (Chapman et al. 1996) have argued that they should not be used to derive EQGs. However, they continue to be used for that purpose because no regulatory authority has recommended an alternative measure of toxicity be used and because a large amount of this type of data is available. For these reasons, the Australian and New Zealand WQGs (ANZECC &

ARMCANZ 2000) used NOEC data but suggested that the use of NOEC data 'be phased out' as EC_{10}-type data become available. Warne and Van Dam (2008) have gone one step further by calling for a ban on the generation and use of NOEC and LOEC data in Australia. Since the Australian and New Zealand WQGs were published, more researchers are reporting EC/LC_{10} to EC/LC_{20}-type toxicity data. The use of point estimate toxicity data is therefore preferred.

2.4.2.2 Conversion from total to added concentrations

The EIL derivation methodology makes a clear distinction between natural background concentration, which is the natural level of contaminants in the soil, and ABC, naturally occurring background and the contaminant levels that have been introduced from diffuse or non-point sources by general anthropogenic activity not attributed to industrial, commercial, or agricultural activities. Therefore, it is preferable that all toxicity data is expressed as an added concentration. If the toxicity data is not expressed in terms of added contaminant then they should be converted to that form, if possible. This can be achieved by subtracting either the ABC, if it is known, or the average concentration in the control soil (that is, the test soil with no addition of the test contaminant) from the total concentrations and then re-calculating the toxicity. If background concentrations are not given then, for some inorganics, the method of Hamon et al. (2004) can be used to estimate ABC in Australian soils or the Dutch background correction equations (Lexmond et al. 1986) can be used to estimate the background concentration. Alternatively, one can set a default background level or assume that the background concentration was zero.

2.4.2.3 Duration of exposure

The Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000) make a clear distinction between chronic and acute toxicity data and convert TVs derived using acute EC/LC-type data to chronic TVs by using, in order of decreasing preference, acute to chronic ratios (ACRs) or a default AF of 10. This approach is very common and widely used in water quality guidelines (ANZECC & ARMCANZ 2000; CCME 1991; US EPA 1991) but is not used in soil guidelines. This is due mostly to the fact that the exposure duration of most terrestrial ecotoxicity tests is three to four weeks. Therefore, conversion factors should only be used for short-term exposure tests. If ACR values are available then they should be used to convert acute terrestrial toxicity data. Only if ACR values are not available should a default AF of 10 be used, which is consistent with the approach adopted by the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000).

2.4.2.4 The use of toxicity data for endemic or overseas species

In deriving any EQGs, the question always arises as to whether toxicity data for overseas species should be used. By using toxicity data for overseas species, the assumption is made that they have the same sensitivity as endemic species. The validity of this assumption has been questioned and examined in a number of studies using aquatic species (Dyer et al. 1997; Markich & Camilleri 1997; Brix et al. 2001; Hobbs et al. 2004; Hose & Van den Brink 2004; Maltby et al. 2005; Chapman et al. 2006; Kwok et al. 2007). However, the evidence is conflicting, with some studies (Maltby et al. 2005; Hose & Van den Brink 2004) finding no differences while others have found differences (Dyer et al. 1997; Markich & Camilleri 1997; Brix et al. 2001; Hobbs et al. 2004; Chapman et al. 2006; Kwok et al. 2007). Kwok et al. (2007) combined results from SSD analysis with ERA principles to determine that, in order to protect 95% of tropical aquatic species, toxicity data for temperate aquatic species should be divided by a factor of 10. Using a similar methodology, Hobbs (2006) found that if Australasian species were to be protected from 95% of chemicals, then toxicity data for northern hemisphere freshwater and marine/estuarine species would have to be divided by 6.2 and 2.2 respectively. The inconsistency in the published results led Chapman et al. (2006) to conclude that 'toxicity data from one geographic region will not be universally protective of other regions'.

The other factor that needs to be considered in resolving this issue is that from a statistical point of view EILs and/or SQGs become increasingly reliable as the number of species for which there is toxicity data increases. Therefore, as a pragmatic compromise, toxicity data for both endemic and overseas species should be used to derive EILs. This is consistent with the Australian and New

Zealand WQGs (ANZECC & ARMCANZ 2000). However, if there are four or more toxicity data measurements in Australia for a species; that is, they meet the minimum data requirements to derive EILs and SQGs, then this should be used in preference to toxicity data for the same species tested overseas.

2.4.3 Incorporation of an ageing and leaching factor

Typically, soil toxicity tests use soils that have been freshly spiked with the contaminant in question. There are very limited amounts of toxicity data available for soils where the contaminant was added some time prior to testing, let alone field-aged soils contaminated by a variety of sources of contaminants with varying bioavailability. The predominance of laboratory-spiked toxicity data has implications for the derivation of EILs due to ageing and leaching.

Ageing or natural immobilisation (attenuation) is the process by which many contaminants (both inorganic and organic), when added to soil, will bind over time to various soil components (Barrow 1986; Hamon et al. 2007; Smolders & Degryse 2007) and this can reduce the concentration of the contaminant that is biologically available (McLaughlin et al. 2000a). Leaching is a process that removes readily soluble soil components such as salinity from soils. Most laboratory-spiked toxicity tests do not leach the soils after the spiking and this has the effect of increasing the ionic strength, decreasing soil pH, increasing aqueous concentrations of dissolved cations (such as Ca, Mg, K, Cd, Cu, Ni, Pb, etc.) and anions (Cl, SO_4, NO_3, etc.), and ultimately increasing the toxicity (Stevens et al. 2003). A study by Oorts et al. (2006) examined the magnitude of the ageing and leaching effects on the toxicity of Cu and concluded that leaching accounts for the majority of the observed difference in toxicity between freshly spiked and aged soils. A study by Smolders et al. (2009), the findings of which have been incorporated into the Flemish SQGs (VLAREBO 2008), derived ageing/leaching factors (ALFs) for Zn^{2+} (3), Cu^{2+} (2), Ni^{2+} (1–3), Co^{2+} (1.1–3.5), Pb^{2+} (4.2), Cd^{2+} (1) based on toxicity measures in a variety of European field and freshly spiked soils.

This is the only study that has generated such ALFs across a wide range of soils and ecotoxicity end points. These ALFs were developed based on a maximum of 18 months ageing and leaching (Smolders et al. 2009). These ALFs should be used in deriving EILs when the contaminants have been present in the soil for at least 2 years. This would be achieved by multiplying the non-aged and non-leached toxicity data by the appropriate ageing/leaching factor, thus decreasing their 'effective' toxicity. Thus, EILs for both fresh (contaminants have been in the soil for less than 2 years) and aged (the contaminants have been in the soil for greater than 2 years) contamination can be derived. Currently, there are very few ALFs available, particularly for Australian soils. There are no ALFs for organic chemicals. When ALFs are not available, it is not possible to derive EILs for aged contamination. In such cases, there are two potential approaches. Firstly, conduct research to derive ALFs for the contaminant of concern or, secondly, conduct direct toxicity assessments (DTA) using soil from the site under investigation. If sufficient toxicity tests are conducted, then site-specific EILs could be derived in much the same manner as deriving site-specific WQGs (ANZECC & ARMCANZ 2000).

2.4.4 Comparison of available toxicity data to the minimum data requirements

There are two potential methods that can be used to derive ACLs: the AF method — a worst-case scenario approach, and the SSD method — a risk-based approach. Both approaches require a minimum amount of toxicity data to derive EILs. The preferred methodology to calculate EILs is the SSD approach because this is a risk-based approach. However, which method is used to derive EILs depends on the number of species and taxonomic groups for which there are toxicity data (see Table 9 below).

Unlike the toxicity data for terrestrial species, toxicity data for soil processes is not based on single species but rather a community of microbial species that perform that soil process. Thus, strictly speaking, it is not suitable for use in SSD methods. However, these processes are important measures

of soil ecosystem health and should be protected. The preferred method for deriving EILs is therefore to use the normal single species toxicity data but also soil process toxicity data.

SSD methods require a minimum set of toxicity data for the aquatic environment, which is usually specified in terms of a minimum number of species and taxonomic groups for which data is required. However, such an approach is not suitable for soil processes where the desirable data types are the number of soil processes and the number of nutrient groups.

A nutrient group is considered to be all toxicity end points measured that relate to a particular nutrient (see Table 11 below). For example, toxicity data for substrate-induced nitrification, potential nitrification rate and denitrification would all belong to the nitrogen nutrient group.

As the number of species and taxonomic groups or soil processes and nutrient groups for which toxicity data is available decreases, the confidence that the resulting EIL will provide the desired level of protection also decreases. In an attempt to compensate for this, the percentage of species and/or soil processes to be protected by the EILs increases as the number of species or soil processes and taxonomic groups or nutrient groups for which toxicity data is available decreases (see Table 9 below).

Table 9. Number of species or functional processes and number of taxonomic groups or nutrient groups needed for the SSD and AF approaches and the corresponding level of protection provided for residential land. The same principle of increasing the level of protection as the amount of toxicity data decreases also applies to other soil quality guidelines and for other land uses (i.e. the default level of protection would increase by 5% if there was data for 5 to 8 species or functional processes)

Number of species or functional processes	Number of taxonomic or nutrient groups	Methodology to derive EIL	Percentage of species to be protected
≥9	≥3	SSD Burr III	80% [a]
5–8	≥3	SSD Burr III	85% [a]
3–8	<3	AF	Not relevant[b]

[a] add 5% to the percentage of the species or soil processes to be protected if the contaminant is a biomagnifier.

[b] The AF does not determine EILs based on protecting a certain percentage of species.

The decision by regulatory agencies about the minimum data requirements is often arbitrary (Pennington 2003) and is based on pragmatic considerations. The US EPA requires at least eight species (US EPA 1999), the Dutch suggests ten species for EQGs (van Vlaardingen & Verbruggen 2007) although some studies have used five species (Van de Plassche et al. 1993; ANZECC & ARMCANZ 2000) and four species (Crommentuijn 2000a), and between five and eight species (OECD 1992, 1994). Since 2000, a number of publications have shown the importance of having larger data sets. For example, Newman et al. (2000) used non-parametric methods to estimate for 30 toxicants that approximately 15 to 55 (with a median of 30) species were needed per toxicant in order produce reliable EQGs. In another example, Wheeler et al. (2002) estimated that a minimum of 10 to 15 species per toxicant are needed. Subsequently, the European Union (EU) has recommended in the technical guidance document on aquatic risk assessment (ECB 2003) that the minimum toxicity data requirement is ten species that belong to eight taxonomic groups. Thus, while it is preferable to use toxicity data sets containing more species and taxonomic groups (or more soil processes and nutrient groups), this must be weighed against the fact that for soil and terrestrial ecosystems there is a general lack of toxicity data. If it were decided to use the same minimum data requirements as the EU, then EILs could be derived for only a limited amount of contaminants using the preferred SSD method.

Other contaminants would have to be derived using the second choice AF method, likely to generate highly conservative criteria. It is imperative to acknowledge the situation for terrestrial systems and to set reasonable minimum data requirements for the SSD method, in order that the majority of the EILs are derived by the preferred SSD method.

Studies by the Danish EPA (Pedersen et al. 1994) and the OECD (1995) indicated that WQGs derived using data sets containing less than five values were very dependent on the spread of the values, whereas for data sets containing five or more values, this effect was markedly reduced. Therefore, the recommended minimum number of species and/or soil processes required to use the SSD approach is five. The minimum number of taxonomic or nutrient groups for toxicity data required in order to use the SSD method was reduced to three. Between five and eight species and/or soil processes, the SSD approach still has a large variation and uncertainty and therefore the protection level should be increased by 5% of species and/or soil processes in order to be more certain that the desired level of protection is achieved. If toxicity data for more than eight species and/or soil processes is available, the SSD approach is deemed to be sufficiently robust to set the protection limit for the appropriate land use (Table 9 above).

In order to determine which method (either the SSD method or the AF method) can be used to derive the EIL, the screened toxicity data should firstly be grouped together on the basis of species or soil processes. Then, using the information presented in Tables 10 and 11 below, the number of taxonomic groups and/or nutrient groups for which toxicity data is available can be determined.

If there is sufficient terrestrial toxicity data for a contaminant, toxicity data derived by models like QSARs or QAARs and the equilibrium partitioning approach should not be used. However, if there is insufficient terrestrial toxicity data available to meet the SSD requirements, the modelled data should be used in combination with measured toxicity data. The minimum data requirements to use the SSD and AF methods are the same when using a data set containing both measured and modelled toxicity data as when using only measured toxicity data. However, only low reliability EILs can be generated using modelled toxicity data (Section 2.4.11).

Table 10. The taxonomic groups for terrestrial species

Taxonomic group	Examples of species in this group
Mollusca	Snails, slugs
Annelida	Enchytraeids, earthworms
Nematoda	Nematodes
Hexapoda	Insects, springtails
Myriapoda	Centipedes, millipedes
Chelicerata	Mites, spiders
Crustaceans	Woodlice
Algae	Algae
Plantae	Plants
Fungi	Fungi
Bacteria	Bacteria
Protozoa	Amoebas, ciliates, flagellates
Tardigrada	Water bears
Chordata	Reptiles, mammals, birds

Table 11. The nutrient groups for soil (i.e. microbial and fungal) processes

Nutrient group	Soil process	Examples of end points
C cycle	Aerobic decomposition	Basal respiration, substrate-induced respiration
N cycle	N mineralisation/ammonification	Urease activity, NH_4 production
	Nitrification	NO_3 production, substrate-induced respiration
	Denitrification	Nitrate reductase
	Nitrogen fixation	Nitrogenase activity
P cycle	P mineralisation	Phosphatase, Py-phosphatase
S cycle	S mineralisation	Aryl-sulfatase

2.4.5 Calculation of the added contaminant limit using a species sensitivity distribution approach

The SSD approach is a statistical method to calculate a soil concentration that theoretically protects a specified percentage of species and/or soil processes. The SSD method used to derive the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000) was the Burr Type III method (Shao 2000), which was incorporated into the BurrliOZ program (Campbell et al. 2000) that is available from: www.cmis.csiro.au/Envir/burrlioz/
Download1.htm.

If there are screened toxicity data values for a contaminant to at least five species or soil processes for three taxonomic or nutrient groups, then there is sufficient data to calculate an ACL using the Burr Type III SSD method.

All SSD methods use a single numerical value to describe each species or soil process for which toxicity data is available. The means by which a single value was obtained for each species or soil process (Van de Plassche et al. 1993) are set out below:

- if there were only one toxicity datum, that was taken to represent the species or process
- if there were several toxicity values for the same end point, the geometric mean of the values was calculated and was taken to represent the species or process

- if there were several toxicity values for different end points (e.g. mortality or reproduction), the end point with the lowest geometric mean was taken to represent the species or process.

SSD methods require the toxicity data to have a uni-modal distribution. If the data set is not uni-modal (for example, insecticides are more toxic to insects than mammals), then the toxicity data belonging to the most sensitive distribution should be used for ACL derivation, as recommended by Warne (1998, 2001) when deriving WQGs.

2.4.6 Normalisation of toxicity data to an Australian reference soil

The use of normalisation relationships is an attempt to minimise the effect of soil characteristics on the toxicity data so the resulting toxicity data will more closely reflect the inherent sensitivity of the test species to the contaminant. If toxicity data more closely reflects species sensitivity, then a more accurate calculation of the soil concentration that should protect a certain percentage of species and soil processes can be made. Derivation of soil-specific EILs and the use of normalisation relationships to normalise toxicity data can only be done if there is sufficient data to use the SSD method. Toxicity data should not be normalised if the available toxicity data is only sufficient to meet the minimum data requirements of the AF approach.

If the toxicity data for a contaminant has been demonstrated to be affected by soil characteristics, (that is, by statistically significant ($p \leq 0.05$) normalisation relationships between toxicity data and soil characteristics), then the toxicity data must be normalised to the Australian reference soil (see Table 12 below).

Table 12. Values of soil characteristics for the Australian reference soil to be used to normalise toxicity data

Soil property	Value
pH:	6
Clay:	10%
CEC:	10 cmol/kg
Org. Carbon:	1% or equivalent OM

Normalisation relationships are currently limited to a few combinations of contaminants, species and countries from which the soils are obtained (Smolders et al. 2004; Li et al. 2003; McLaughlin et al. 2006; Song et al. 2006; Broos et al. 2007; Warne et al. 2008a; 2008b). This is predominantly due to the concept of developing normalisation equations for terrestrial ecotoxicity data being relatively recent and the size of, and cost of conducting, such work.

The lack of normalisation equations for a wide variety of species can be overcome by applying the relationships across species within the following groupings of the taxonomic groups:

- plants, algae
- annelids, nematode, mollusca, protozoa
- hexapoda, myriapoda, chelicerata, tardigrada
- microbial and fungal functional end points.

These groupings are based on the basic body design of the organisms and the likely exposure route of organisms to the contaminant; that is, being exposed by the direct environment or through food. The following four derivation steps are listed in order of descending order of preference:

1. If normalisation relationships for all four taxonomic groupings are available and each grouping meets the minimum data requirements to use the SSD approach, derive a set of soil-specific ACL values for each grouping and then the lowest ACL for the soil in question is adopted.
2. If normalisation relationships for all four taxonomic groupings are available but at least one grouping does not meet the minimum data requirements to use the SSD approach, apply the normalisation relationships and combine all the data in one SSD calculation. Then use the normalisation relationships to derive a set of ACLs for each taxonomic grouping and the lowest ACL for the soil in question is adopted.
3. If normalisation relationships are available for some groupings then apply them to the appropriate data and combine all the data (including the non-normalised toxicity data) in one SSD calculation. Then use the normalisation relationships to derive a set of ACLs for each grouping of organisms that have a normalisation relationship and the lowest ACL for the soil in question is adopted.
4. If normalisation relationships are not available, then pool all data and derive one generic ACL.

The above steps are used to standardise the derivation of realistic EILs that are protective but at the same time ensure that the EILs do not become too conservative.

If the toxicity data shows a significant relationship with specific soil characteristics; for example, soil pH, organic carbon or clay content, cation exchange capacity (CEC), soil-specific ACL values can be calculated using those relationships. The toxicity data is first normalised to the reference Australian soil using the methods described above, and the ACL value derived using the SSD approach is valid for the Australian reference soil. Using the normalisation relationships, ACL values can then be

calculated for different soil types. For example, if toxicity data showed a relationship with pH, different ACL values can be calculated for a range of soil pH conditions.

The lack of normalisation equations for soils from Australia can be overcome by using normalisation relationships developed with soils from other countries, particularly Europe and America. However, these normalisation relationships should only be used when they are derived from soils similar to Australian soils and/or their validity for Australian soils has been assessed and found suitable. The importance of this was shown by a study of Broos et al. (2007), which assessed the normalisation relationships of Smolders et al. (2004) and Oorts et al. (2006) for microbial nitrification in soils. They re-analysed the overseas data after removing microbial toxicity data for soils with organic compound concentrations greater than those found in Australian soils. This resulted in a change of soil characteristics, explaining the variance in the toxicity data.

A second option to overcome the lack of normalisation relationships in the literature is to examine the currently available toxicity data and use regression analyses on the collated data to determine if a significant relationship exists between toxicity and soil characteristics.

Normalisation relationships from field studies are preferred over those from laboratory studies. All the normalisation relationships for toxicity, apart from those developed by Broos et al. (2007) and Warne et al. (2008b), model laboratory-based data (Rooney et al. 2006; Smolders et al. 2003; Smolders et al. 2004; Oorts et al. 2006; EU 2006b; Song et al 2006, Warne et al 2008a). Warne et al. (2008b) found that field-based normalisation relationships gave much more accurate estimates of field phytotoxicity than laboratory-based normalisation equations. Therefore, field-based normalisation relationships should be used in preference to laboratory-based normalisation relationships. It is, however, realised that the current lack of field-based normalisation relationships will unavoidably necessitate the use of laboratory-based relationships, despite their limitations.

If multiple normalisation relationships are available within a taxonomic group of organisms, then the most geographically appropriate normalisation relationship should be applied to the toxicity data. For example, a European normalisation relationship would be applied to European data and an Australian normalisation relationship would be applied to Australian data. If there are multiple geographically appropriate normalisation relationships for a group of organisms, then the relationship with the lowest slope should be used, as this will give the most conservative normalised toxicity data (EC 2008).

2.4.7 Calculation of the added contaminant level using an assessment factor approach

If the minimum data requirements for the SSD approach cannot be met, the AF approach should be used to derive EILs. The AF is a 'worst-case scenario' type of approach. In this approach the lowest toxicity value for a contaminant; that is, the most sensitive data point, is divided by an AF in order to derive an ACL.

$$ACL = \frac{lowest\ NOEC\ or\ EC10}{Assessment\ \ factor} \qquad \textit{(equation 5)}$$

Equation 5 applies to the derivation of EILs; if other SQGs were to be derived, then different toxicity data would be substituted in the equation. The magnitudes of the AFs depend on the available toxicity data and are given in Table 13 below. If there is toxicity data for less than three species, the AF is 500, due to the lack of information and thereby the high uncertainty in estimating the risk posed by the contaminant in the soil. If there is toxicity data for more than three species the AF decreases, depending on how many taxonomic or nutrient groups are represented (see Tables 10 and 11 above for taxonomic and nutrient groups respectively). If field data or model ecosystems with multiple species tested are available, an assessment has to be made as to how well the study represents the field situation and how protective the toxicity data is. An AF of 10 should be used if the EIL is calculated using mesocosm or microcosm data.

Table 13. Assessment factors to be used to derive ACL using the AF approach (adapted from ANZECC & ARMCANZ 2000).

Toxicity data available for derivation of ACL		
Number of species	**Number of taxonomic or nutrient groups**	**Assessment factor**
<3 species	N/A	500
≥3 species	1	100
	2	50
<5 species	3	10
Field data/data of model ecosystems		10

N/A = not applicable

2.4.8 Accounting for secondary poisoning and biomagnification

Secondary poisoning can occur if contaminants accumulate from the ambient environment (for example, soil) into the tissue of organisms (bioaccumulation) that are then consumed by other organisms and the concentration in tissue increases in the journey up the food chain (for example, soil, earthworms, birds and predatory birds). In such a situation, the species at most risk are the species higher in the food web (the predators). Examples of contaminants that biomagnify and have shown adverse effects on predators include DDT, cadmium and PCBs (Morrissey et al. 2005; Jongbloed et al. 1996; Luoma & Rainbow 2008). Biomagnification and secondary poisoning should only be addressed for contaminants that show biomagnification potential.

Secondary poisoning should be addressed for residential EILs. Residential areas cover a large area and can harbour many birds and small land species that can potentially be at risk from contaminants that biomagnify. For site-specific risk assessment, secondary poisoning EILs may not be relevant for contaminated sites of limited area.

The vast majority of ecotoxicological data is derived from direct exposure from the ambient environment and not from food. Thus, if a contaminant biomagnifies, then normal toxicity data and EILs derived using such data may underestimate the impact the contaminant has on the environment and communities. Therefore, a more protective measure is needed for biomagnifying contaminants.

If an SSD approach were used to derive the EIL for contaminants that biomagnify, the level of protection (that is, percentage of species and/or soil processes to be protected) should be increased by 5%, i.e. to 85% (or to 90% if <8 taxonomic species or functional processes are used). This approach is consistent with that used in the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000) to deal with secondary poisoning.

If the EIL were derived using the AF approach, then a BMF will have to be applied in order for the EIL to account for biomagnification.

The ACL for biomagnification will be calculated by:

$$ACL_{Biomagnification} = \frac{ACL}{BMF} \qquad (equation\ 6)$$

If there is sufficient BMF data available for an organic contaminant, then the 80th percentile of these values should be used in equation 6 above. For those organic contaminants that have no BMF values, BMF values for organic contaminants with similar chemical structures should be collated and then a specific percentile value could be adopted. The percentile of BMF values to be used is set at 80%.

For inorganic contaminants, grouping of BMF values is not recommended and biomagnification should be dealt with on an individual chemical basis.

For organic contaminants, the BMF values depend on the K_{ow} of the contaminant and increase to 10 for organic contaminants having a log K_{ow} of 5–8. For inorganic contaminants, the K_{ow} values of the contaminant should not be used but the literature should be searched for BAF or BMF for terrestrial species, or fish if no terrestrial data is available. If BMF values are not available for an inorganic contaminant or a group of organic chemicals, a conservative biomagnification factor should be used. The biomagnification factors for organic contaminants, from the European technical guidance for risk assessment (ECB 2003), which are shown in Table 14 below, should be used.

Table 14. Default BMF values for organic substances that correspond to the logarithm of the octanol–water coefficients and the BCFs adapted from ECB (2003).

log K_{ow} of contaminant	BCF (fish)	BMF
<4.0	<2,000	1
4.0–5	2,000–5,000	2
5–8	>5,000	10
>8–9	2,000–5,000	3
>9	<2,000	1

2.4.9 Calculation of the ambient background concentrations

To calculate a site-specific EIL, ABCs for soils should be determined, as the ACL is based on added toxicity values. If possible, the ABCs should be directly measured at a clean reference site with a comparable soil type to the site being examined. However, such sites are not always available or easy to identify.

2.4.9.1 Inorganic contaminants

For metal contaminants, if reliable ABCs cannot be measured, then either the estimation method of Hamon et al. (2004) or collations of ABC values such as Olszowy et al. (1995) could be used. The equations for calculating ABC values are presented in Table 15 below. Estimates of ABCs for several metals based on example soil iron or manganese concentrations (determined by aqua regia digestion) are presented in Table 16 below. To use the Hamon et al. (2004) method, it is necessary to ascertain that the iron and manganese concentrations of the soil at the site in question are not elevated by co-contamination—these elements are normally determined in chemical analysis of soils to determine total metal concentrations and therefore minimal extra cost is involved. These Hamon et al. (2004) relationships are based on soils from sites with no known history of contamination apart from farming.

Therefore, this approach would be suitable for predicting the ABC in otherwise uncontaminated areas including new suburbs; that is, suburbs less than 20 years old (Olszowy et al. 1995). In fact, for the inorganic contaminants where comparison is possible, the ABC values predicted by the Hamon et al. (2004) method are very similar to the 25th percentile of the ABC values for new suburbs from Olszowy et al. (1995).

Olszowy et al. (1995) conducted a stratified random sampling study to determine the ABCs in residential areas of the capitals of New South Wales, Queensland, Victoria and South Australia. A total of 320 soil samples collected at 0–150 mm depth were collected and analysed. If the Hamon et al. (2004) method cannot calculate an ABC, then the Olszowy et al. (1995) values for new suburbs would be appropriate to use for new suburbs or areas with no known history of contamination. In old-established urban areas (i.e. suburbs more than 20 years old), it would be appropriate to use the 25th percentile of the ABC values from Olszowy et al. (1995).

Table 15. Equations from Hamon et al. (2004) and the corresponding coefficient of determination (r^2) used to estimate ABCs for arsenic (As), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb) and and zinc (Zn)

Element	Normalising element	Gradient	y intercept	r^2
As	Fe	0.547	0.507	0.50
Co	Mn	0.894	-1.409	0.71
Cr	Fe	0.750	1.242	0.58
Cu	Fe	0.612	0.808	0.61
Ni	Fe	0.702	0.834	0.64
Pb	Fe	1.039	0.118	0.66
Zn	Fe	0.589	1.024	0.61

Table 16. Predicted ambient background soil concentrations (mg/kg) for arsenic (As), cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb) and zinc (Zn) at different soil iron concentrations, based on the equations from Hamon et al. (2004)

Soil Fe%	As (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Ni (mg/kg)	Pb (mg/kg)	Zn (mg/kg)
0.1	<1	<3	<2	<1	<0.1	<3
1	<3	<17	<6	<7	<1	<11
10	<12	<98	<26	<34	<14	<41
20	<18	<165	<40	<56	<29	<62

2.4.9.2 Organic contaminants

Most organic contaminants of interest to contaminated sites are xenobiotics, hence they have no natural background concentration. Notable exceptions to this include lipids and fats, hormones (for example, oestrogen, testosterone), fatty acids, alcohols, hydrocarbons, polycyclic aromatic hydrocarbons (PAHs) and dioxins. Therefore, ABCs will have to be generated by direct measurement or a default ABC of zero could be assumed (Crommentuijn et al. 2000b). There are no equivalent models to that of Hamon et al. (2004) available for organic contaminants.

For dioxins, regional ABC values are available (Muller et al. 2004) and could be used or, alternatively, site-specific assessments could be conducted. For other pyrogenic organic contamination (for example, PAHs), a site-specific assessment should be conducted to determine if the measured concentrations are background concentrations for that region. If a site-specific assessment is conducted, then the upper 80th percentile of the ABCs should be used as the background as per the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000). However, even if they are considered ABCs, this does not imply that there is no risk to terrestrial biota.

2.4.10 Calculation of the EIL

If biomagnification is not considered, then the EIL for a contaminant is calculated as follows:

EIL = ABC + ACL *(equation 7)*

where ABC is the ambient background concentration (mg/kg) and ACL is the added contaminant limit (mg/kg).

If biomagnification is considered and is significant for that contaminant, then the EIL is calculated as follows:

$$EIL = ABC + ACL_{BM} \quad \textit{(equation 8)}$$

where ACL_{BM} is the contaminant added limit that accounts for biomagnification.

2.4.11 The reliability of the EIL

Classifying the EIL based on the amount and type of toxicity data is important to provide users with an indication of the reliability of the EIL values and also for prioritising future re-assessments of EILs. Methods for determining the reliability of TVs were developed and used in the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000; Warne 2001) and this formed the basis of the soil EIL reliability assessment system. The number of data points, the type of toxicity data, the number of species/soil processes for which there is data, and whether or not there are normalisation relationships are all used to assess the reliability. The three classes of EIL reliability are high, moderate and low. The requirements for an EIL to receive these classifications are provided below.

High reliability:

- The toxicity database contains sufficient toxicity data for the SSD approach and at least one normalisation relationship (that is, relationships that describe the effects of soil characteristics on toxicity) is available.

Moderate reliability:

- The toxicity database meets the minimum data requirements for the SSD approach but normalisation relationships are not available.

Low reliability

- The toxicity database meets the minimum data requirements for the SSD approach but contains modelled toxicity data (that is, from QSARs, QAARs or the equilibrium partitioning method) or ecologically less relevant end points (e.g. biomarker end points).

or

- The toxicity database meets the minimum data requirements for the AF approach.

In the Australian and NZ WQGs (ANZECC & ARMCANZ 2000), low reliability TVs were only used for interim guidance. A similar approach should be adopted regarding low reliability EILs—that such values should be considered to be a knowledge or data gap that requires further work to resolve.

For organic contaminants with low reliability EILs, the EILs are only as good as the QSARs and QAARs they were derived from. Therefore, further research is only necessary if the QSARs and QAARs are of relatively poor quality.

2.4.12 Evaluation of the appropriateness of the derived EILs

Once the EILs have been derived, their appropriateness should be evaluated. A similar process was also conducted as the last step in the derivation of the Australian and New Zealand WQGs (Warne 2001). Their appropriateness is determined by comparing each EIL with the toxicity data used to derive them, any available field-, mesocosm- or microcosm-based toxicity data, plant or crop nutritional requirements (for essential elements), and background concentrations. The aim of the comparison is to determine which species, if any, are likely to experience toxic effects if exposed to the EIL. If the species that potentially may be affected are considered rare or endangered, are keystone species, or are commercially important, then it may be appropriate to decrease the EIL (that is, increase the level of protection being provided). This evaluation or 'ground-truthing' process is, by necessity, done on a case-by-case basis.

2.4.13 Strengths and limitations of EIL derivation methodology

A discussion of the strengths and limitations of the methodology is presented below.

2.4.13.1 Strengths

The EIL derivation methodology:

- is risk-based and enables protection of a selected percentage of species
- incorporates assessment of all major exposure scenarios for terrestrial ecosystems, including secondary poisoning
- can handle different types of toxicity data, thereby maximising the number of EILs that can be derived for contaminants
- can be used to derive SQGs for a variety of different land uses and purposes
- considers bioavailability and can therefore derive soil-specific EILs if the necessary data is available for the contaminant to ensure a uniform protection level for different types of soils
- considers ageing and leaching for aged soil contamination
- accounts for the ambient background concentration issue
- is consistent and incorporates the most recent advances in risk assessment, terrestrial toxicity and soil chemistry
- is consistent with the Australian and NZ water quality guidelines.

2.4.13.2 Limitations

The EIL derivation methodology:

- does not incorporate the different sources and types of contamination, and the bioavailability of different sources of contamination
- is relatively complex and will require researchers with expertise to derive reliable EILs
- uses a secondary poisoning method that is not optimal and may require improving in the future. The methodology does not use complex secondary poisoning models due to a serious lack of data necessary for these models, especially a lack of Australian data. If, in the future, the data is available, it is recommended that these types of models for EIL derivation be considered for contaminants showing biomagnification potential.

3 Technical notes on methods used in the EIL derivation methodology

In this section, the various methods used in the EIL derivation methodology are more thoroughly explained and their strengths and limitations discussed. Recommendations on which methods should be used are also provided. The methods addressed in this section are:

- to account for the effect that soil characteristics have on toxicity and bioavailability
- for calculating ACLs
- for measuring and incorporating ABCs
- to account for bioaccumulation and secondary poisoning effects.

3.1 Methods to account for the effect of soil characteristics on toxicity and bioavailability

Soil characteristics are known to affect bioavailability and therefore the toxicity of contaminants to organisms (Lexmond 1980; McBride 1989; Alloway 1995; Basta et al. 2005). An example of the strong effects that soil characteristics have on toxicity is provided in Table 17. This shows laboratory-based toxicity data (EC_{10}) for Cu and Zn to wheat grown in 14 different Australian soils (Warne et al. 2008a). The lowest and highest EC_{10} values vary 20–30 fold for both Cu and Zn. As the conditions were standardised and only one test species was used, the cause for the differences in toxicity can only be soil type and soil properties.

Table 17. Total added concentrations (mg metal/kg soil) of Cu and Zn that cause a 10% reduction in growth for wheat seedlings (EC_{10}) grown in 14 Australian soils (Warne et al. 2008a)

Site	Cu EC_{10}	Zn EC_{10}
Avon	945	755
Brennans	205	275
Bundaberg	260	235
Cecil Plains	3,300	5,855
Dalby	885	655
Dookie	490	965
Dutson Downs	-	875
Esk	465	565
Flat Paddock	115	250
Kingaroy	810	505
Night Paddock	110	530
Spalding	930	620
Tintinara	430	430
Wilsons	465	335

There are two methods that attempt to address the issue of the effects of soil characteristics. These are to express toxicity data in terms of a contaminant estimate of the bioavailable fraction of a contaminant and to express toxicity data in terms of total concentrations and develop relationships

(termed normalisation relationships) between toxicity and soil characteristics that account for bioavailability (see McLaughlin et al. 2000a for a discussion of these two philosophies).

3.1.1 Chemical estimates of bioavailability

A number of soil extraction methods have been developed with the aim of providing a better estimate of the bioavailable fraction than total concentrations. These include calcium chloride ($CaCl_2$) extracts, ammonium nitrate (NH_4NO_3) extracts, soil solution and other extracts and diffusion-based methods (for a review, see McLaughlin et al. 2000b). The extraction methods assume that they only extract that portion of the total amount of a chemical that is biologically available. This is a chemical approach to estimating the bioavailable fraction.

Available information suggests that only Germany (BBodSchV 1999) and Switzerland (Gupta et al. 1996) use a measure of chemical concentration other than the total contaminant concentration in soil. The German guidelines (BBodSchV 1999) have some soil TVs based on concentrations in NH_4NO_3 extracts for some inorganic contaminants (that is, TVs for cadmium) in the soil-to-plant pathway. This was only done if NH_4NO_3 extracts were better predictors (that is, showed better correlations) for internal plant concentrations from soil than the total soil concentration. The ammonium nitrate extract is considered by the German guidelines to be the bioavailable concentration of inorganics in soil.

The perfect chemical measure of bioavailability should give very similar toxicity values (for example, LC_{50}) in a range of different soils for a given chemical tested on a given species. For soils, the perfect measure of bioavailability should overcome the effects that different soil characteristics have on toxicity and truly reflect the available fraction of the contaminant that causes the toxicity to the organism. Therefore, the ability of techniques to determine the bioavailable fraction can be assessed by comparing the variability of the toxicity values for one species across different soils—the measure with the smallest variability in toxicity values being the best measure of the bioavailable fraction (McLaughlin et al. 2000b). This approach was adopted by Broos et al. (2007) and Warne et al. (2008b) using microbial and plant toxicity data for Cu and Zn in 14 different Australian soils (field-based) using one source of contamination (soluble metal salts). In both cases, the variation in toxicity values based on total concentrations was smaller than or as small as those based on soil solution and $CaCl_2$ extracts. Unpublished work from the Australian National Biosolids Research Program (NBRP) showed that the concentrations in ammonium nitrate and calcium chloride extracts were very highly related with coefficients of determination (r^2) greater than 0.9. Therefore, although it is untested, it is highly likely that the data from the NBRP would reveal that variation in toxicity values across soils based on total concentrations would be lower than those based on ammonium nitrate.

A number of authors from Europe (Smolders et al. 2003; Smolders et al. 2004; Oorts et al. 2006; Zhao et al. 2006) have also found that extractable or soil solution measurements were not useful predictors of plant and microbial toxicity in soils and thus used total metal concentrations to develop normalisation relationships. In contrast, a number of other studies have reported various extractable measures to be better than total concentrations (Posthuma & Notenboom 1996; Vijver et al. 2001; Nolan et al. 2005; Menzies et al. 2007).

McLaughlin et al. (2010) in a review of how to derive soil standards for trace elements concluded ‘it is difficult to conclude that one particular extractant or type of extractant is superior to others in predicting the trace element bioaccumulation and toxicity across a range of soils or organism endpoints’. There is also considerably more toxicity data expressed as total metal concentration. A further issue to be considered in development of EILs using extractable concentrations of contaminants would be the significant analytical challenge for many laboratories to consistently extract and accurately determine the low concentrations of contaminants found in partial extracts of soil.

One disadvantage of using total contaminant concentrations instead of a partial extract of soil designed to measure bioavailability is that different sources of contamination, having differing bioavailability, are not differentiated. However, for a screening level risk assessment such as the use of EILs, use of total concentrations is protective.

For the purposes of developing EILs (which are used across soils with a wide range of properties), there is some evidence from both overseas and Australia that, at least for metals, extractable concentrations in soil may not necessarily be better measures of bioavailability than total concentrations.

3.1.2 Normalisation relationships

The use of normalisation relationships is an attempt to minimise the effect of soil characteristics on the toxicity data so the resulting toxicity data will more closely reflect the inherent sensitivity of the test species. If toxicity data more closely reflects species sensitivity, then a more accurate estimate of the soil concentration that should protect a certain percentage of species and soil processes can be derived. Normalisation relationships are also used to extrapolate ACL values determined for the Australian reference soil out to soils with a range of physicochemical properties (that is, different soils). To normalise toxicity data, empirical relationships are needed between soil characteristics and toxicity data. An example of a relationship between toxicity and a soil property is given in Figure 3, which shows how toxicity values increase with increasing soil pH.

Normalisation relationships are relatively simple empirical relationships between the toxicity or plant uptake data for a single contaminant to one species and the physicochemical properties of the soils where the tests were conducted. These empirical relationships are usually obtained using data from laboratory studies in which a single species is exposed to a single contaminant in different soils. Normalisation relationships have generally been developed using linear regression analysis techniques including forward and backward step-wise regression (Smolders et al. 2004; Rooney et al. 2006; Broos et al. 2007; Warne et al. 2008a) or partial least squares (PLS) regression (Lock & Janssen 2001). It is important that only soil physicochemical properties that are not significantly correlated to each other are used to develop normalisation equations. Although there are no generally accepted rules, researchers have generally only reported or recommended the use of normalisation equations that have coefficients of determination (r^2) or adjusted coefficients of determination (adj r^2) greater than 0.5 (that is, they explain more than 50% of the variation in toxicity values). This is quite reasonable as, if a relationship does not explain at least 50% of the variation, then using it to normalise other toxicity data could introduce considerable error.

A number of studies have successfully developed normalisation relationships for plants, microbial processes and soil invertebrates. The main soil characteristics affecting the toxicity of inorganic contaminants appear to be pH, clay content, cation exchange capacity and organic matter content (Lock & Janssen 2001; Smolders et al. 2003; Smolders et al. 2004; Rooney et al. 2006; Song et al. 2006; Broos et al. 2007; Warne et al. 2008a, 2008b).

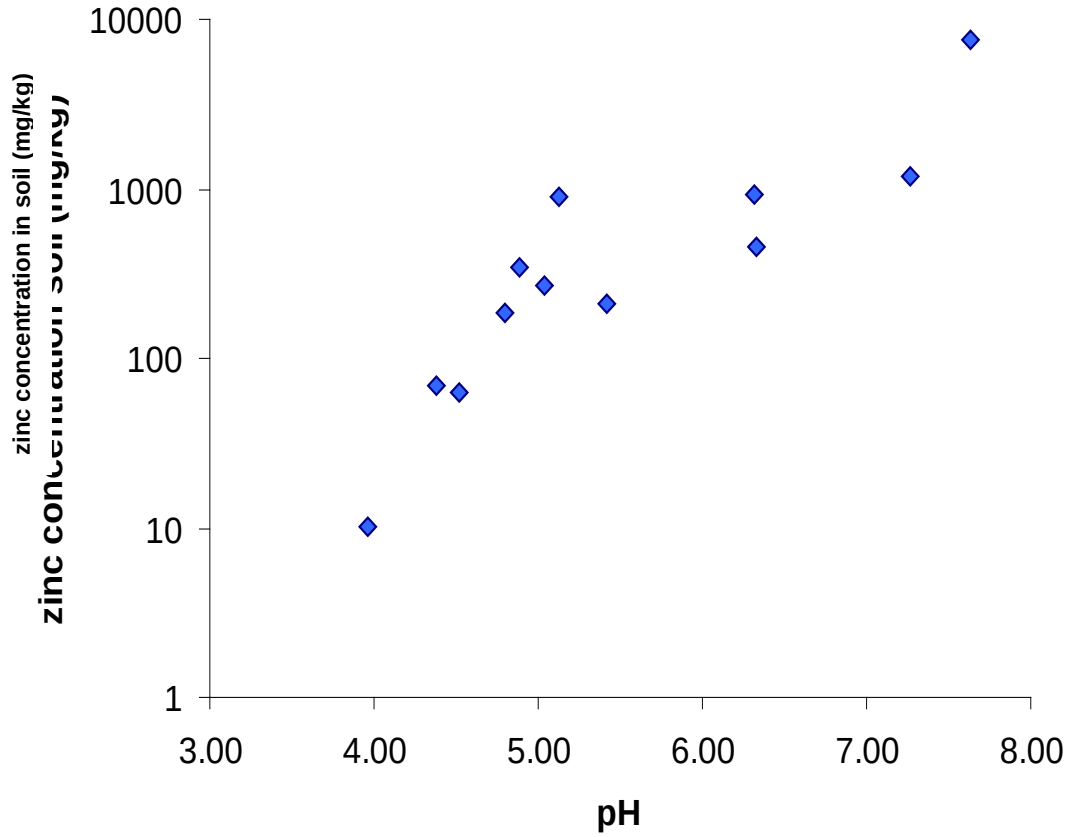


Figure 3. An example of the effect that soil pH can have on toxicity values (shaded diamonds). Toxicity data shown are SIN EC 10 from the NBRP program.

Normalisation equations can, in principle, be developed for any combination of contaminant, species, and toxicity end point. However, they should only be developed using ecologically relevant species, measures and toxicity end points for the ecosystem that is being protected. In addition, it is preferable from an implementation point of view, that relatively easy and relatively cheap-to-measure, accurate, repeatable soil characteristics are used to derive normalisation relationships. Otherwise, the costs and difficulty of determining unusual soil characteristics will inhibit application of the relationships.

In Australia, empirical relationships have been obtained between soil characteristics and toxicity data for a limited set of contaminants and end points to date. Examples of relationships between toxicity and soil characteristics from the NBRP program are:
Microbial (substrate induced nitrification – SIN) *see also Figure 3*
SIN log EC_{10} Zn = 0.55*pH – 0.55 $R^2 = 0.74$ *(equation 9)*
Plant (toxicity)
log EC_{10} Zn = 0.271 * pH + 0.702 * log CEC *adj.* $R^2 = 0.66$ *(equation 10)*

where pH is the soil pH (0.01 M $CaCl_2$), CEC is the cation exchange capacity, EC_{10} is the concentration that causes a 10% effect.

Normalisation relationships are currently limited to a few combinations of contaminants, species and countries from which the soils are obtained. The lack of normalisation equations for a wide variety of species can be overcome by applying the relationships to species other than those for which they were derived (EU 2006b). However, this practice should only be conducted if it could be expected that the contaminant would exert its toxicity in the same manner as to the other species and the application of the normalisation relationship leads to a decrease in the range of toxicity values for the other species (EU 2006b).

The lack of normalisation equations for Australian soils can be overcome by using relationships developed with soils from other countries, particularly Europe and America. However, these normalisation relationships should only be used when they are derived from soils similar to Australian soils and/or if their validity for Australian soils has been assessed and found suitable[6]. The importance of this was shown by a study of Broos et al. (2007). This study assessed the normalisation relationships of Smolders et al. (2004) and Oorts et al. (2006) and re-analysed the data after removing microbial toxicity data for soils with OC concentrations greater than those found in Australian soils. This resulted in a change of soil characteristics, mainly explaining the variance in the toxicity data. For the initial data set, OC was the most important factor explaining the toxicity of Zn and Cu to nitrifying microorganisms but without the high OC soils, pH became the main explanatory soil property.

Normalisation relationships usually take the form of:
*Toxicity data = a * soil property ± b* *(equation 11)*

where *a* is the gradient of the regression and *b* is the y-intercept. The y-intercept is a measure of the inherent sensitivity of the test species used to derive the normalisation relationship—and each species will have a unique y-intercept. Thus, when applying normalisation relationships to other species, the toxicity data should only be transformed using the gradient (that is, *a* in equation 11) of the normalisation relationship (EU 2006).

A second option to overcome the lack of normalisation relationships in the literature is to examine the currently available toxicity data, and use regression analyses on the collated data to determine if a significant relationship exists between toxicity thresholds and soil characteristics.

Normalisation relationships from field studies are preferred over those from laboratory studies. All the normalisation relationships for toxicity apart from those developed by Broos et al. (2007) and Warne et al. (2008b) model laboratory-based data (Rooney et al. 2006; Smolders et al. 2003; Smolders et al. 2004; Oorts et al. 2006; EU 2006; Song et al. 2006; Warne et al. 2008a). Warne et al. (2008b) found that field-based normalisation relationships gave much more accurate estimates of field phytotoxicity than laboratory-based normalisation equations. Therefore, field-based normalisation relationships should be used to model field-based phytotoxicity data in preference to laboratory-based normalisation relationships. It is, however, realised that the current lack of the field-based normalisation relationships will unavoidably necessitate the use of laboratory-based relationships, despite their limitations.

3.1.3 Normalisation of toxicity data to a reference soil

If there are normalisation relationships for a toxicant, then the toxicity data should be normalised to a reference soil with a specified set of soil characteristics before the data is used in the SSD to derive the ACL value. Therefore, a reference soil for Australia should be used to normalise all the toxicity data (see Table 12). The specific setting of the Australian reference soil does not affect the EILs; however, all data should be normalised to the same chosen setting. Furthermore, it does not matter if all data is normalised to different settings and then an ACL value is calculated using the SSD method, or if one Australian setting is used, an ACL value is calculated and then the normalisation equation is used to calculate ACLs for different soil settings. This is because of the statistical methodology behind the SSD and normalisation approach.

Figure 4 shows how normalisation of toxicity data leads to a significant decrease in variation in toxicity values for a species (from the blue to the purple points in the figure). Therefore, the normalised toxicity data more accurately reflects the inherent sensitivity of each species.

[6] This is done by comparing values predicted by the non-Australian normalisation relationships to Australian toxicity data.

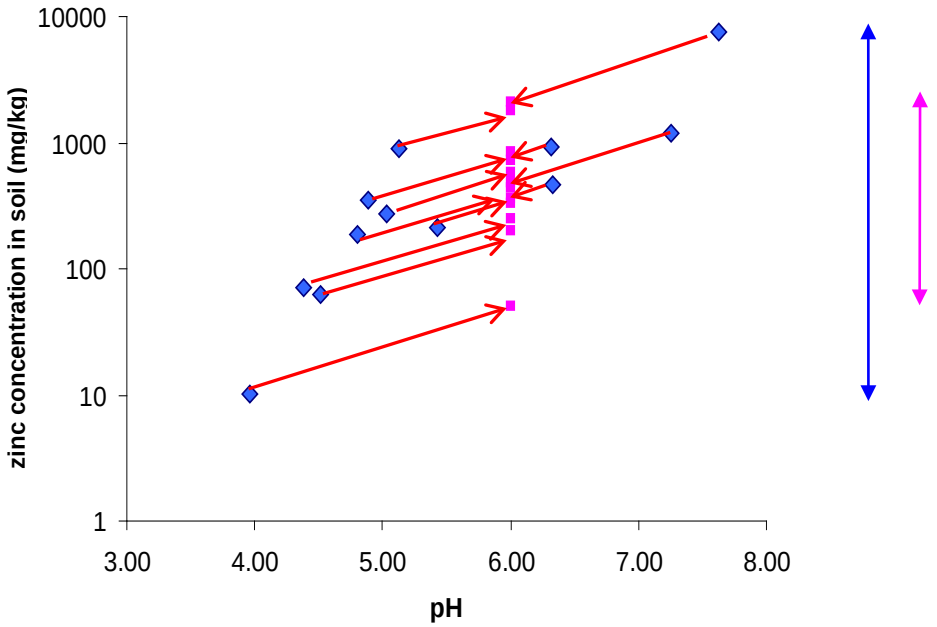


Figure 4. Example of the effect of normalising using microbial toxicity data from the National Biosolids Research Program. The red arrows show how each toxicity value was normalised. The blue and pink arrows show the variation in toxicity values for the non-normalised and normalised data respectively. In this case the toxicity data was normalised to a pH of 6.

3.2 Methods to calculate soil quality guidelines

In general, there are three main methods to derive SQGs. These are in order of increasing complexity: the geometric mean method, AF methods and SSD methods. They are discussed below.

3.2.1 Species sensitivity distribution methods

The SSD methods are statistical methods to calculate a soil concentration that protects a specified number of species and/or soil processes. Briefly, all SSD methods use toxicity data obtained from tests on individual species and fit a statistical distribution to the data to derive a concentration that should protect any selected percentage of species in the ecosystem being considered.

There are essentially four different SSD methods that have been used to derive EQGs:

- the Stephan et al. (1985) method, which fits a log-triangular distribution to the data
- the Aldenberg and Slob (1993) method, which is an enhancement of the Kooijman (1987) and Van Straalen and Denneman (1989) methods, which fits a log-logistic distribution to the data
- the Wagner and Løkke (1991) and Aldenberg and Jaworska (2000) methods, which fit a log-normal distribution to the data
- the Burr type III (Shao 2000; Campbell et al. 2000) method, which fits the best of the Burr type III family of distributions to the data.

The Stephan et al. (1985) method was the first SSD method developed. It is used by the USA to derive its WQGs (US EPA 1986) and was adopted by South Africa to derive freshwater guidelines (Roux et al. 1996).

Limitations of this method are that by using a log-triangular distribution it assumes there is a threshold toxicity value, below which no detrimental effects will occur, and the scientific literature and risk

assessment theory does not support such an concept (Okkerman et al. 1991; OECD 1992; Emans et al. 1993; Pedersen et al. 1994; NZ Ministry of the Environment 1996) and it uses an arbitrary AF of two without any justification (Hart et al. 1995; NZ Ministry of the Environment 1996). As early as 1995, the US EPA recognised that the method required updating (Delos 1995). For the above reasons, this method was not considered for the derivation of the Australian and New Zealand WQGs (Warne 1998). At least partially due to the limitations of the Stephan et al. (1985) method, South Africa has adopted the more advanced Burr type III SSD method (Shao 2000) for its marine water quality guidelines (Warne et al. 2004a, 2004b).

In the late 1990s, the Aldenberg and Slob (1993) method was viewed as the preferred and most scientifically defensible SSD method. It was recommended over the Wagner and Løkke method by the OECD and subsequently adopted (OECD 1995). The Dutch used the Aldenberg and Slob method to derive their WQGs and SQGs. This reflected the research that the Dutch had undertaken to assess the scientific validity of this method (Emans et al. 1993; Okkerman et al. 1991, 1993). One drawback of the Aldenberg and Slob method compared to the Wagner and Løkke method was its use of the log-logistic distribution. There is no theoretical basis for the sensitivity of species to conform to a logistic distribution (Forbes & Forbes 1993). In fact, Aldenberg and Slob (1993) stated that the log-logistic distribution was chosen because it has 'practical mathematical features that make the calculations of statistical confidence intervals relatively easy'. Aldenberg and Jaworska (2000) overcame the mathematical difficulties associated with using the normal distribution to develop a log-normal equivalent method to the Aldenberg and Slob method. The Aldenberg and Jaworska method has since been adopted by the Dutch to derive their WQGs and SQGs (Crommentuijn 2000a, 2000b). All of the above methods attempt to fit a single statistical distribution to the toxicity data.

The draft Australian and New Zealand WQGs (ANZECC & ARMCANZ 1999) adopted the Aldenberg and Slob SSD method. However, during the derivation of the TVs it was found that in more than 33% of cases where the Aldenberg and Slob method could be used, based on meeting the minimum data requirements of the method, the data did not have a log-logistic distribution. Therefore, strictly speaking, it was invalid to use the Aldenberg and Slob SSD method. This meant that for many contaminants an AF method had to be used. As there is no theoretical reason why species sensitivity must conform to a logistic distribution, there is no reason why other distributions cannot be considered. This issue was first realised by Shao (2000) and he therefore recommended that a family of distributions, the Burr type III (BT III) be used to fit to the toxicity data, rather than a single distribution as with the other SSD methods. Other authors (Maltby et al. 2003; Kwok et al. 2007) have since also adopted a more flexible approach to the statistical distributions being fitted to the data, whereby the distribution that best fits the data is used to derive the EQG or to determine the ecological risk.

The variety of shapes that BT III distributions can have is large (Shao 2000), including the log-logistic distribution and approximations of the log-normal and log-triangular distributions. Thus, attempting to fit a BT III distribution to any given toxicity data set has a greater probability of success than attempting to fit only the log-logistic distribution.

This method is guaranteed to fit a statistical distribution to the toxicity data at least as well as the Aldenberg and Slob method because the log-logistic distribution is a BT III distribution (Shao 2000). Greater detail about the BT III method is provided in Shao (2000).

3.2.2 How do SSD methods work?

The main difference between the various SSD methods is the statistical distribution that they fit to the data. For that reason, the following explanation of how SSDs work is generic.

In SSD methods, each species is given equal weighting and a single value is used to represent the sensitivity of each species. However, there is usually multiple toxicity data for each species which

requires some manipulation. The rules governing this manipulation were presented earlier in this Schedule. The data is then entered into SSD software such as ETx (Aldenberg & Jaworska 2000) or BurrliOZ (Campbell et al. 2000). The SSD calculates the cumulative frequency of the species sensitivity data by ranking the data from lowest to highest and then using the formula:

cumulative frequency = rank * [100/(n + 1)] (*equation 12*)

The cumulative frequency for each species is then plotted against the concentration that represents the sensitivity of each species. A typical SSD plot is shown in Figure 5 below. In the case of the Stephan et al. (1985), Wagner and Løkke (1991), Aldenberg and Slob (1993), and Aldenberg and Jaworska (2000) methods that fit one specific distribution to the toxicity data, statistical tests (for example, the Kolmorogorov Smirnov test or the Anderson-Darling test) are used to determine if the toxicity data fits the selected distribution. The more flexible SSD methods, for example, BT III and the approach adopted by Maltby et al. (2003) and Kwok et al. (2007), use statistical methods (for example, maximum likelihood methods, Anderson-Darling test) to determine which particular statistical distribution best fits the toxicity data. In doing this, the SSD methods estimate the parameters that mathematically describe the selected distribution. Because the equation that describes the selected distribution is known, it is very simple to calculate the concentration that should theoretically protect any chosen percentage of species or permit any chosen percentage of species to experience toxic effects. To do this, the cumulative frequency that corresponds to the percentage of species to be protected is entered into the equation for the distribution that best fitted the toxicity data. Thus, the 5th percentile of the selected distribution becomes the concentration that, if not exceeded, will protect 95% of species and the 10th percentile will protect 90% of species, and so on. The resulting concentrations are generally referred to in Europe as hazardous concentration (HC) values, while in Australia and NZ, Hong Kong and South Africa they are termed protective concentration (PC) values. The number following HC or PC indicates the percentage of species that should be harmed or should be protected respectively. More detailed information on each SSD method can be obtained from the original documents cited above and in the thorough review of SSD methods by Posthuma et al. (2002).

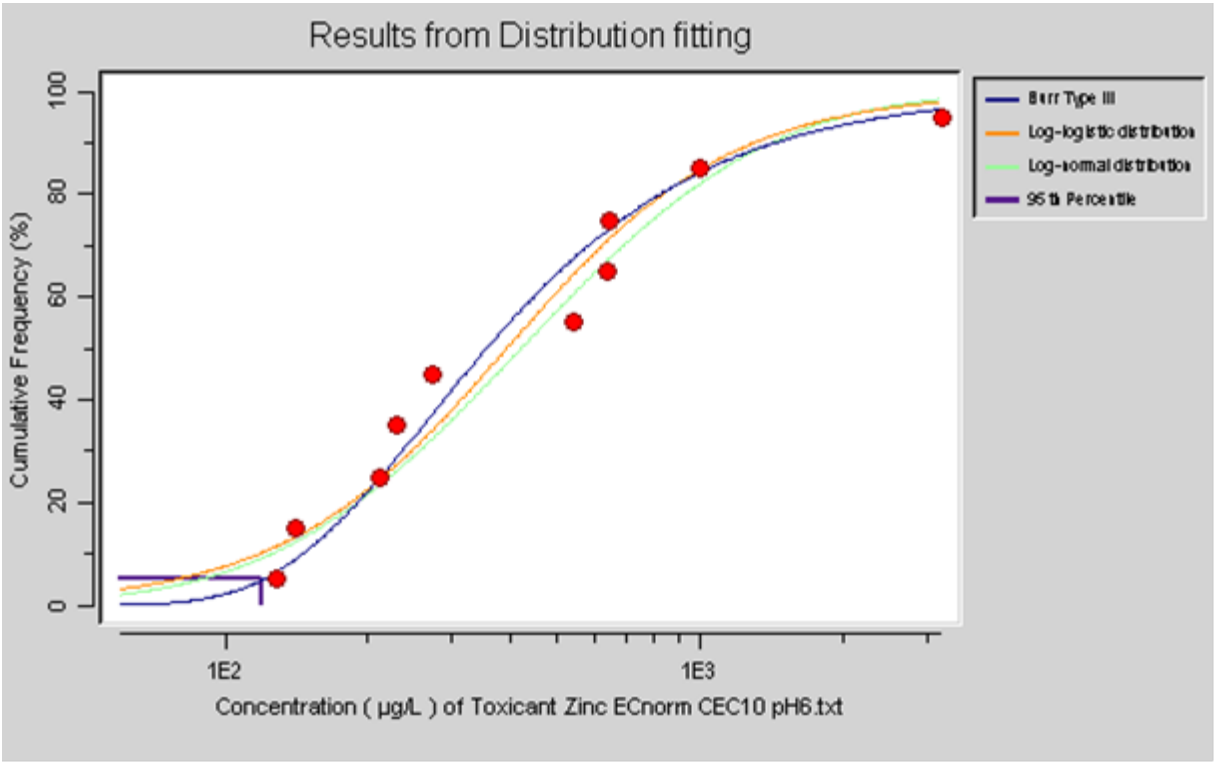


Figure 5. A typical SSD plot. The example provided is output from the BurrliOZ program using EC_{10} values for plants (field data NBRP)

The toxicity data used to derive a PC value is only a sample of the total species in the ecosystem being protected. As with any sampling program, different distributions could be obtained depending on the species that form the sample. Therefore, different samples could lead to different PC values for the same contaminant being calculated. Aldenberg and Slob (1993) overcame this problem by developing two confidence limits: 95% and 50% for the HC or PC values respectively. These confidence limits indicate the degree of certainty that the calculated HC value will protect the selected percentage of species. Thus, a HC_5 95 value means that there is a 95% certainty that the concentration will protect at least 95% of species in an ecosystem. The Dutch used the HC_5 95 values as their long-term aspirational goal for water quality. In the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000) and in jurisdictions that have adopted their methodologies (that is, Hong Kong and South Africa in its marine water quality guidelines) confidence intervals are not used. This was developed because the 95% confidence limits were not deemed to be statistically robust (Fox 1999). Additionally, if the sample size is large, the 50^{th} percentile will approximate the median of estimates of the PC value. Thus, the 50^{th} percentile should equal the HC_5 50.

3.2.2.1 Criticisms

All SSD methods make a series of assumptions. In the early 1990s, the SSD methods received considerable criticism from Calabrese and Baldwin (1993), Forbes and Forbes (1993), Schudoma (1994), and Smith and Cairns (1993), and some doubted whether the SSD methods were in fact better than the AF methods.

The key criticisms were:

- whether ecosystems are sufficiently protected by protecting a given percentage of the species comprising that particular ecosystem
- whether the distribution of species sensitivities in ecosystems is closely approximated by the distributions used in the various SSD methods
- whether the SSD methods yield environmental quality guidelines that are conservative by nature.

A number of the other assumptions made by SSD methods were also attacked by these authors; however, these were assumptions made by all methods of deriving EQGs. There is considerable experimental support for the SSD methods (Emans et al. 1993; Okkerman et al. 1991, 1993). In addition, organisations such as the OECD compared both the SSD methods and AF methods and concluded by recommending the SSD methods (OECD 1995). An overview of the criticisms and support for the SSDs is provided in Warne (1996) and a more condensed version in Warne (1998). Several authors including Forbes and Calow (2002a) have now changed their position considerably and support SSDs while acknowledging their limitations. SSD methods are now well established and widely used in deriving EQGs and conducting ERAs. For example, SSD methods are the preferred method of deriving the EU soil and water quality guidelines (ECB 2003; EU 2006b).

A potential weakness of SSD methods, and indeed of all modelling methods, is that as the quantity of data used decreases the effect of the sample used increases dramatically. Initial studies by the Danish EPA (Pedersen et al. 1994) and the OECD (1995) indicated that WQGs derived using data sets containing less than five values were very dependent on the spread of the values, whereas for data sets containing five or more values this effect was markedly reduced. Subsequent more rigorous work by Newman et al. (2000), Forbes and Calow (2002b) and Wheeler et al. (2002) indicated that toxicity data for between 10 and 30 species was necessary for the resulting limit values to be stable irrespective of the sample. To calculate an HC_5/PC_{95} value using empirical methods, at least 20 species are needed, and 100 species are needed for an HC_1/PC_{99} value (Forbes & Calow 2002b). Using non-parametric methods, Newman et al. (2000) estimated for 30 toxicants that between 15 and 55 (median of 30) species per toxicant were needed, while Wheeler et al. (2002) estimated a minimum of 10 to 15 species per toxicant were needed. The decision by the regulating agency about the appropriate number

of species is often arbitrary (Pennington 2003): US EPA requires at least eight species (US EPA 1999), the Dutch suggest ten (van Vlaardingen & Verbruggen 2007), the OECD between five and eight (OECD 1992, 1994) and Australia and New Zealand—five species (Warne 2001). It is worth remembering that the above estimates are based on available SSDs that tend to include data from only a small fraction of taxonomic and other groups present in nature. If data were available for a larger range of organisms, the number of species for which data is required may increase. If this occurred, then the findings of Newman et al. (2000), Wheeler et al. (2002), and others would have underestimated the number of species required for estimating the SSDs. Reflecting these findings, the EU has required that future WQGs need toxicity data for at least ten species that belong to at least eight taxonomic groups and an additional assessment factor of 1–5 to the PC_{95} should be considered (ECB 2003).

3.2.2.2 Strengths and limitations

SSD methods have a number of strengths:

- they use toxicity data for all species that is available, thus conforming to risk-assessment principles
- they have a sound statistical basis providing the assumptions of the method are met
- they are flexible methods, can use any measure of toxicity, and can calculate HC or PC values to protect any chosen percentage of species except 0% and 100%
- the methods are transparent and allow the level of protection to be chosen. The approach also enables a more informed debate to occur over the level of protection to be offered
- they can be used in the reverse manner to determine what level of protection (i.e. percentage of species) is offered when a certain concentration of a contaminant occurs in the environment. This should be useful in ERAs and site-specific investigations
- several aspects of the methodology have been validated.

The limitations of the methods include:

- the data requirements may limit the number of guideline values that can be derived
- it is more complex to understand how the guideline values are derived than with the AF or geometric mean methods
- several of the assumptions made by SSD methods may be compromised. For instance, SSD assumes that the species are representative of the totality of the ecosystem and all species are equally as important to ecosystem functioning (that is, no consideration is given to keystone species).

3.2.3 Assessment factor methods

In AF methods, all available toxicity data for a contaminant is collated. Then the lowest toxicity value is divided by a constant that is variously called an assessment factor, uncertainty, application or safety factor. Typically the AFs are 10, 100 or 1000. The magnitude of the AF used to derive an EQG is inversely related to the perceived environmental relevance of the toxicity data; that is, the more environmentally realistic the toxicity data, the smaller the AF and vice-versa. This approach for deriving EQGs was first proposed by Hart et al. (1995) and was adopted from methods used in human health to derive average daily intakes (Cotruvo 1988; Calabrese & Baldwin 1993). The AF method is used to derive both soil and water quality guidelines in numerous countries.

Depending on the toxicity data available, up to three extrapolations can be made by AF methods, with each extrapolation typically given an AF of 10. The extrapolations are laboratory-to-field, acute-to-

chronic, and interspecies, and are designed to compensate for inadequacies in the available toxicity data. The magnitude of the various AFs and the type and magnitude of the extrapolations that are inherently assumed by the AFs used in the modified US EPA (OECD 1992) and CCME (1991) methods are presented in Table 18 below.

The field-to-laboratory extrapolation accounts for the supposition that laboratory studies tend to underestimate the toxicity in the field. Proposed reasons for this include: laboratory tests being conducted on animals that are robust and easily bred/maintained in the laboratory rather than 'sensitive' species, life stages that are not tested in the laboratory may be more sensitive to toxicants (Hart 1996), and all the limitations associated with single species toxicity tests that are discussed in Warne (1998).

It is also possible for laboratory-based experiments to overestimate the toxicity in field situations. This can arise if laboratory experiments use freshly spiked soils with minimal ageing period, which overestimates the bioavailability compared to field bioavailability.

The acute-to-chronic extrapolation is extensively used to derive WQGs because the vast majority of toxicity data is acute whereas chronic data is preferred for environmental protection. The CCME method (CCME 1991), like the original US EPA method (US EPA 1986), uses an ACR derived from another species for the same contaminant in preference to a generic ACR. When a contaminant-specific ACR is not available, then CCME (1991) and the US EPA (1986) use a generic ACR. CCME (1991) uses an ACR of 2 or 10 depending on the environmental persistence of the contaminant, while the modified (OECD 1995) and unmodified US EPA (1986) methods use one generic ACR of 10.

However, an acute-to-chronic extrapolation is not used in soil guideline value derivation. An acute-to-chronic extrapolation should only be used for short-term contact exposure studies. Such tests are a very short-term acute toxicity test performed on direct dermal contact using earthworms, which might not represent exposure in soils accurately. The test will very likely give toxicity values that are an underestimation of chronic exposure toxicity data.

Most AF methods used in most jurisdictions have minimum data requirements. When these are not met then an interspecies extrapolation is used. This is used because there is increased uncertainty in deriving guideline values from such a small sample size.

Table 18. The assessment factors, types and magnitudes of the extrapolations used in the modified US EPA and CCME methods

Available toxicity data	**Type of extrapolation**	**Modified US EPA method[a]**	**CCME method[b]**
Chronic NOEC (for the US EPA) or LOEC (for CCME)	Field-to-laboratory	10	10
Acute LC_{50} or EC_{50}	Field-to-laboratory and acute-to-chronic	100 (10 x 10)	ACR or 20 or 100[c]
Acute LC_{50} or EC_{50} for one or two species	Field-to-laboratory and acute-to-chronic and interspecies	1000 (10 x 10 x 10)	ACR or 20 or 100[d]

[a] It is assumed toxicity data is available for at least an algae, a crustacean and a fish (OECD 1992).

[b] Assumes that toxicity data is available for at least three species of fish, of which two must be chronic; two invertebrates, one of which should be planktonic; and a freshwater vascular plant or algae (CCME 1991).

[c] An AF of 50 is used for non-persistent contaminants while 100 is used for persistent contaminants when no ACR is available.

[d] Where data is not sufficient to meet the requirements set in b, then interim WQGs are derived (CCME, 1991).

3.2.3.1 Criticisms

Criticisms of the AF approach revolve around the scientific validity of AFs, the magnitude of the AFs, and whether or not the method is consistent with a risk framework and the principle of ecologically sustainable development. Many scientists and organisations have acknowledged the arbitrary nature of AFs, that they have no theoretical scientific basis and are purely empirical (Hart 1974; Nicholson 1984; Kooijmand 1987; Okkerman et al. 1991; OECD 1992; Schudoma 1994; Rand et al. 1995; OECD 1995; Warne 1998). Goldberg (1975) asserted that using AFs was tantamount to admitting that information essential for risk assessments was lacking.

Nicholson (1984) considered that:
'There is little scientific basis for application factors except that they are the result of careful judgement … there is little evidence, in most cases, that the arbitrary value chosen is indeed the best choice, i.e. whether a particular value for an application factor will provide 'adequate' protection and whether a less (or more) stringent value would be more appropriate.'

The fact that there is no universally accepted magnitude for AFs (as seen in Table 18) confirms their arbitrary nature. The AF method ignores all other data except the lowest and is therefore an example of the 'worst-case scenario' type of approach. Such a procedure is at odds with a risk-based approach, which requires an array of data in order to derive estimates of the probability of certain toxicological events occurring. Risk-based concepts and procedures are central to many of the more recently adopted scientific, social and political paradigms within Australia including the current Australian and New Zealand *Guidelines for fresh and marine water quality* (ANZECC & ARMCANZ 2000).

There has been considerable discussion in the scientific literature about the appropriate size of AFs. There are numerous examples of where AFs should be less than 10 and equally numerous examples of where they should be considerably larger (refer to Warne (1998) and Chapman et al. (1998) for detail). Chapman et al. (1998) concluded that the discussion about the size of the AFs is 'to some extent futile … because no one set of factors has universal applicability'. Ultimately, AFs are a measure to address a lack of knowledge and as soon as that knowledge is available, AFs should no longer be used.

3.2.3.2 Strengths and weaknesses

The strengths of AF methods are that:

- they are simple to use
- they are easily understood
- EILs can be derived with as little as one toxicity value
- the more unreliable the data the larger the AF becomes – thus taking into account the increased uncertainty
- the magnitude of the AFs can easily be modified to reflect new toxicological findings but this is invariably not done.

The weaknesses of AF methods are that:

- the AFs have no theoretical basis; they are purely empirical
- there is debate over the scientific validity of acute-to-chronic ratios
- the method is at odds with risk assessment principles
- the method is not transparent, as it does not state the degree of protection provided by an AF of a certain magnitude and thus does not permit informed decisions and debate over the level of protection to occur.

Reflecting the above limitations, many countries only use AF methods to derive SQGs and/or WQGs when SSD methods cannot be used. For example, the Australian and New Zealand (ANZECC & ARMCANZ 2000), OECD (1995), the Netherlands (Crommentuijn 2000), Canadian (CCME 2006), Danish (Bro-Rasmussen et al. 1994) and South African (Roux et al. 1996) guidelines all now use a statistical extrapolation method in preference to an AF method, which is only used when there is insufficient data.

3.2.4 Geometric mean methodology of the US EPA

The US EPA has developed ecological soil screening levels (Eco-SSLs) for sites where terrestrial organisms may be exposed directly or indirectly to contaminated soil, using the geometric mean method. The geometric mean[7] method uses all the toxicity values at the highest relative bioavailability score for which sufficient data existed (that is, ≥3 data points). Thus, the Eco-SSL is really the geometric mean of the sensitivities of all organisms for which there is toxicity data in the most bioavailable situation. By using the geometric mean approach, there is no consistent level of protection being provided (that is, different percentages of species will be protected). This is not a particularly conservative approach for the soil ecosystems where the contaminant is most bioavailable. However, the percentage of species that could experience toxic effects will be less and the degree of conservatism greater in the soils where the contaminant is less bioavailable.

Geometric means are also used in the manipulation of toxicity data prior to use within SSD methods. However, the manner in which the geometric means are implemented is quite different to that of the US EPA Eco-SSLs. The geometric mean approach is a combination of the worst-case scenario and risk-based approaches. It is a worst-case scenario as it derives Eco-SSLs for the soil in which the contaminant is most bioavailable. It is consistent with risk-based approaches as it does not attempt to protect all species.

3.2.4.1 Strengths and limitations

The strengths of the geometric mean method are that:

- it is simple to use
- it is easily understood
- limit values can be derived with as little as three toxicity values
- it is at least partially consistent with risk-based concepts.

The limitations of the method are that:

- the resulting limit does not reflect the uncertainty in the toxicity data used in deriving the limit, e.g. a limit based on three acute laboratory-based toxicity data is treated the same as 25 field-based chronic toxicity data—whereas the latter data set is considerably more environmentally relevant than the former

[7] The geometric mean is analogous to the normal arithmetic mean except that the values are logged before summing and being divided by the number of data points. The value is then anti-logged to provide the geometric mean. The formula for this is

Geometric mean = anti-log [(logA + log B +…..logN)/n] *(equation 13)*

In determining the geometric mean the data can be logged to any base (e.g. $\log_{10}$, $\log_2$ or the natural log) as long as the same base is used throughout equation 13.

The reason for using the geometric mean rather than the arithmetic mean is that the geometric mean is not affected as much by extremely low or high values. For example, the geometric and arithmetic means of a data set consisting of 10, 25, 40 are 21.5 and 25 respectively. If a value of 400 was added to the same data set then the geometric and arithmetic means would be 45 and 119 respectively.

- the resulting limit is not transparent as it does not state the degree of protection and thus does not permit informed decisions and debate over the level of protection to occur.

3.2.5 Methods for calculating EILs

In deciding which of the above methods would be best to derive EILs, it is important to recognise the role of EILs. They are a concentration above which further investigation should be conducted. Therefore, if the contaminant concentration does not exceed the EIL, then it is assumed that the situation does not warrant further investigation and is, in fact, safe. Therefore, EILs need to be reasonably conservative. Other considerations are scientific validity, ease of use and interpretation and consistency with existing Australian environmental management systems.

3.2.6 Secondary poisoning and biomagnification

Secondary poisoning can occur if a contaminant biomagnifies, that is, it accumulates in organisms' tissue and the concentration increases with each trophic level in a food web (for example, soil—earthworms—birds—predatory birds). The species most at risk are those in the higher trophic levels in a food web, i.e. the predators. Examples of contaminants that biomagnify and have deleterious effects on predators include DDT, Cd and PCBs (Morrissey et al. 2005; Jongbloed et al. 1996).

The vast majority of environmental toxicity data is on direct exposure to contaminants from the ambient environment (that is, soil, water or air) and not from food. Therefore, if contaminants are biomagnified, then normal toxicity data and EILs based on such data may underestimate the impact the contaminant has on the environment and communities. To overcome this problem, contaminants that biomagnify need to be identified and biomagnification needs to be considered in deriving the EIL for those contaminants.

3.2.7 Methods for accounting for secondary poisoning

Secondary poisoning is taken into account in the soil quality guidelines of several countries, including Canada (CCME 2006), USA (US EPA 1996) and the Netherlands (Van de Plassche 1994). However, not all countries consider secondary poisoning in their SQGs, for example, Germany (BBodSchV 1999).

There are three methods for deriving EILs that account for biomagnification:

1. biomagnification algorithms
2. default biomagnification factors
3. increasing the percentage of species to be protected.

These methods are critically assessed below.

3.2.8 Using biomagnification algorithms

There are three slightly different biomagnification algorithms. The main difference between them is whether ingestion of soil is considered (the US EPA and Canadian methods) or not (the Dutch method).

The US EPA methodology (US EPA 1996), which accounts for soil ingestion, calculates the secondary poisoning SQG (SQGsp) by:

$$SQGsp = \frac{Toxicity\ reference\ value}{FIR \cdot (P_s + BAF_{ij})} \qquad \textit{(equation 14)}$$

where SQGsp is the soil quality guideline that accounts for secondary poisoning and is expressed in mg/kg, the toxicity reference value is expressed in mg contaminant/kg prey tissue, FIR is food intake

rate (kg food [dry weight]/ kg body weight [wet weight] /day), Ps is the proportion of the diet that is soil (%) and BAFij is the bioaccumulation factor for contaminant 'i' by species 'j' (unitless).

The Canadian methodology (CCME 2006) is based on daily intake models similar to derivation of maximum human daily uptake models. The Canadian methodology takes into account direct soil ingestion and bioaccumulation through the food chain.

SQGs are thereby calculated using the following equation:

$$SQG_{2C} = \frac{0.75 \cdot DTED_{2C} \cdot BW_{2C}}{(SIR_{2C} \cdot BF) + (FIR_{2C} \cdot BAF_{2})}$$

(equation 15)

where SQG_{2C} refers to the soil quality guideline for soil and food ingestion for the secondary consumer (mg/kg dry weight soil), $DTED_{2C}$ is the daily threshold effects dose for the secondary consumer (mg/kg body weight-day), BW_{2C} is the body weight of the species used in the $DTED_{2C}$ (kg), SIR_{2C} is the soil ingestion rate for the species used in the $DTED_{2C}$ (kg dry weight soil/day), BF is the bioavailability factor (unitless), FIR_{2C} is the food ingestion rate for the species used in the $DTED_{2C}$ (kg dw food/day) and BAF_2 is the bioaccumulation factor (unitless) (CCME 2006).

The Dutch methodology developed by Van der Plassche (1994) or Romijn et al. (1991) does not account for soil ingestion and calculates the SQG by:

$$SQGsp = \frac{NOEC\ predator}{BCFprey}$$

(equation 16)

where SQGsp is the soil quality guideline that accounts for secondary poisoning expressed in mg/kg, NOEC predator is the NOEC for a predator expressed as mg contaminant/kg prey tissue, BCF_{prey} is the bioconcentration factor of the contaminant for a prey species expressed as a ratio of concentration in the prey and in the soil. If the BCF_{prey} is unknown, the BCF was predicted based on the log Kow of the contaminant using QSARs.

The above methods were not adopted in the Australian and NZ WQGs because of 'the lack of relevant data' and as there is 'no formal and specific guidance on how to take information on bioaccumulation into account when deriving water quality guidelines' (ANZECC & ARMCANZ 2000). Food web approaches were not advocated because they are 'very complex and require extensive data sets, which are not available for the majority of contaminants' (ANZECC & ARMCANZ 2000). These data sets include toxicity data for top predators, biomagnification and bioaccumulation data and dietary information for the species. For terrestrial ecosystems, Australian data needed for a food web modelling approach is even scarcer. The paucity of Australian data was the main reason why a proposed food web methodology for deriving EILs was not incorporated into this guideline.

However, biomagnification algorithms are currently the best available methodology to set EILs that protect top predators if the necessary data sets are available.

3.2.9 Using a default biomagnification factor

The biomagnification default factor method refers to dividing the normal SGQ by a biomagnification factor to protect the higher predators. Predators are assumed to have the same sensitivity to the contaminant as other species, but as biomagnification occurs in the food web, the SQG is divided by a default biomagnification factor to protect the predators. This default biomagnification factor could be derived by collating biomagnification values for similar contaminants and then a specific percentile value on a log-normal basis could be adopted as the default BMF. If biomagnification values are not known, a conservative default biomagnification factor could be set (for example, 10). This is a simple

and easily understood method but it could under-protect for some combinations of species and contaminants and over-protect for others. This methodology can also result in very conservative limit values.

3.2.10 Increasing the percentage of species to be protected

Increasing the percentage of species to be protected is an indirect way of addressing biomagnification and was used in the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000). For example, the level of protection was raised from 95% to 99% for slightly to moderately modified ecosystems. It is a simple method but not necessarily scientifically rigorous. As it does not directly address biomagnification, it cannot be guaranteed that the resulting limit values will provide sufficient protection. Furthermore, this methodology might give very conservative limit values which in some cases could be lower than background concentrations. This occurred when PC_{99} values were derived for some metals (Warne *pers. comm.*).

3.3 Determining ambient background concentrations

3.3.1 Inorganics

Metals and metalloids are naturally present in soils. Natural (background) concentrations of metals in soils depend on the parent rock from which the soil originated and are highly variable. Some authors (Reimann & Garrett 2005) argue that natural background concentrations no longer exist anywhere in the world due to man-made activities and global transport of contaminants. Therefore, the term ambient background concentration (ABC) as suggested by Zhao et al. (2007) is used rather than background concentration.

Metal concentrations in soils are easily and quickly measured; therefore, the preference is to directly measure the ABC in known unpolluted reference soils. However, finding a similar unpolluted reference soil to the contaminated soil is not always possible for a wide variety of reasons. The complexity and problems associated with measuring the ABC are discussed in a series of papers in *Human and Ecological Risk Assessment*, vol. 9 (2003) and by Reimann and Garrett (2005). Reliable ABC values for a soil with similar physicochemical and structural properties to the soil being investigated cannot always be obtained or the measured values are compromised in one or more ways. If reliable background concentrations cannot be obtained, then a modelling method should be used.

3.3.2 Background concentration models

A model able to predict the background concentrations of metals in Australian soils was developed by Hamon et al. (2004). In this study, a large number of remote sites in Australia and South-East Asia were surveyed for metal concentrations in soil. Principal component analysis revealed strong associations of many metals (for example, As, Co, Cr, Cu, Ni, Pb) with structural elements of soil minerals (Fe and Mn). Linear regressions were developed that permit the prediction of background soil metal concentrations using only Fe or Mn concentrations (Figure 6).

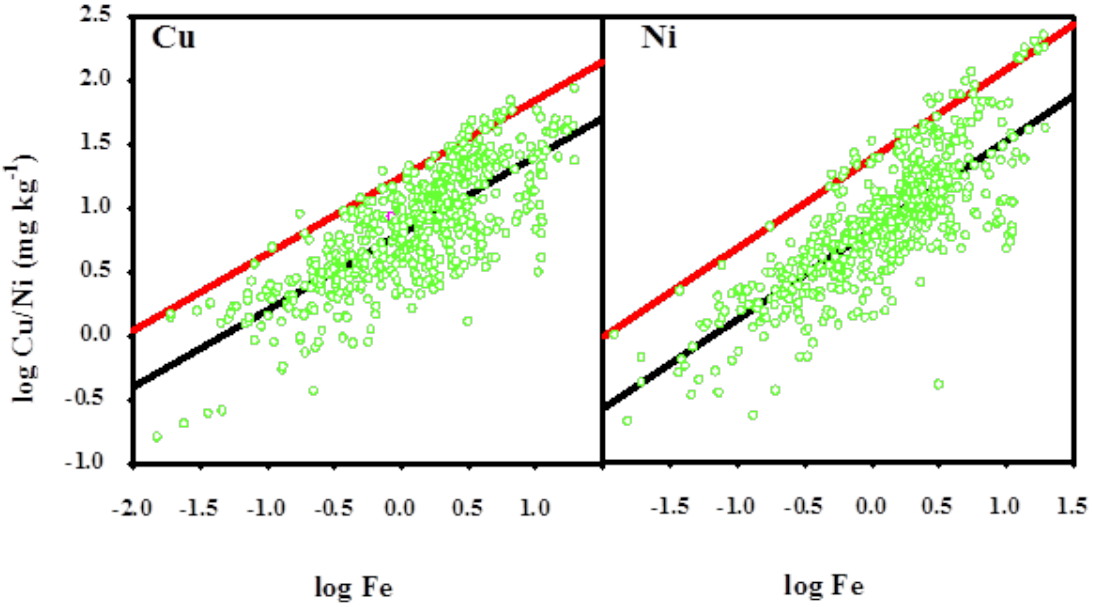


Figure 6. Example relationships between the logarithm of iron concentration of soil and background Cu and Ni concentrations (modified from Hamon et al. 2004). The red and black lines are the 95th and 50th percentile of the relationships respectively.

The equations developed by Hamon et al. (2004 [Table 15, Section 2.4.9.1]) can be used to estimate the background concentration. Hamon et al. (2004) calculated the 'background concentrations' using the equation that encompassed the upper 95th percentile of the data. However, Zhao et al (2007) argued that this approach is not conservative as the poorer the relationships, the larger the 95th percentile will be and hence the larger the estimates of ABC will be. They argue that this may lead to under-protection of soils (by deriving larger ABCs which are added to limit values base on added metal concentrations). Given the above and the purpose of EILs, the 50th percentile of the data (that is, the regression equation) should be used to estimate ABC values.

The relationships developed by Hamon et al. (2004) take the form

ABC = a* log Fe or Mn + b *(equation 17)*

To calculate the ABC, measure the Fe and Mn concentration in the soil (expressed in %) using aqua regia digestion (Hamon et al. 2004), and substitute the appropriate metal concentration into the appropriate equation. It is, however, necessary to ascertain that the Fe and Mn content of the soil at the site in question is not elevated by contamination. These elements are normally determined in chemical analysis of soils to determine total metal concentrations and therefore minimal extra cost is involved.

3.3.3 Organics

Most organic contaminants of interest to contaminated sites are xenobiotics, hence they have no natural background concentration. Notable exceptions to this include lipids and fats, hormones (for example, oestrogen, testosterone), fatty acids, alcohols, hydrocarbons, polycyclic aromatic hydrocarbons and dioxins. Therefore, ABCs will have to be generated by direct measurement or a default ABC of zero (Crommentuijn et al. 2000b) could be assumed. There are no equivalent models to that of Hamon et al. (2004) available for organic contaminants.

For pyrogenic and naturally occurring organic contamination, a site-specific assessment should be conducted to determine if the measured concentrations are background concentrations for that region. If a site-specific assessment is conducted, then the upper 80th percentile of the ABCs should be used as the background as per the Australian and New Zealand WQGs (ANZECC & ARMCANZ 2000).

However, even if they are considered ABCs, this does not imply that there is no risk to terrestrial biota.

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5 Appendices

5.1 Appendix A: Review and comparison of frameworks for deriving soil quality guidelines in other countries

5.1.1 A1: USA

The US EPA has developed a series of Eco-SSLs (www.epa.gov/ecotox/ecossl/) to protect terrestrial organisms from soil contamination.

Eco-SSLs apply to sites where terrestrial organisms may be exposed directly or indirectly to contaminated soil. Eco-SSLs were developed to support risk management decisions for Superfund sites (orphaned contaminated sites identified as having significant contamination potentially present for many years or even decades). This was undertaken to avoid repetitious risk assessment and literature reviews of toxicity data for the same contaminants at each contaminated site, and to allow risk assessors to focus their efforts on the main contaminants of concern.

Seven types of receptors were initially considered in the development of the Eco-SSLs (mammals, birds, reptiles, amphibians, soil invertebrates, plants, and soil microbes and their processes) but final SSLs were produced without consideration of amphibians and reptiles due to insufficient data being available, in the view of the US EPA, to derive screening levels. Soil microorganisms and microbial processes were also not included in the derivation of Eco-SSLs but the rationale for this was over the variability of the data and their ecological significance.

For plants and invertebrates, the methodology used to develop Eco-SSLs was to review the relevant toxicity literature for each contaminant, screen the data for quality, and only use toxicity data representing high bioavailability conditions in upland aerobic soils (that is, avoiding consideration of flooded soil conditions). Because of the different behaviour of many contaminants in soils, high bioavailability was defined for three broad groups of contaminants—cationic metals, anionic metals and non-ionising organic contaminants. For example, high bioavailability for cationic metals was defined as low soil pH and organic matter content. Where literature data did not exist for a contaminant, this was developed by experimentation.

The Eco-SSL for a contaminant was calculated as the geometric mean of all the toxicity values at the highest relative bioavailability score for which sufficient data existed (that is, $\geq$3 data points). If less than three data values were available at the highest relative bioavailability level, data from the next highest bioavailability score was included in that Eco-SSL data set. This process proceeded until a combined data set of three or more data values was identified for calculating the Eco-SSL. If there were less than three acceptable studies, an Eco-SSL was not calculated.

For wildlife Eco-SSLs, three avian and three mammalian species were chosen to represent some of the most highly exposed species at contaminated sites (meadow vole, short-tailed shrew, long-tailed weasel, mourning dove, American woodcock and red-tailed hawk). Wildlife Eco-SSLs were developed by back-calculating from a hazard quotient (HQ) of 1.0, calculated by dividing the estimated exposure dose by the toxicity reference value (TRV). When the HQ was 1.0, the exposure dose equalled the Eco-SSL.

A generic food-chain model was used to estimate the relationship between the concentration of the contaminant in soil and the critical dose (TRV). TRVs were developed using a literature screening process similar to that of the plant and invertebrate Eco-SSLs.

Twenty-four Eco-SSLs have been produced for aluminium, antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, iron, lead, manganese, nickel, selenium, silver, vanadium, zinc, dieldrin, hexahydro -1,3,5-trinitro-1,3,5-triazine (RDX), trinitrotoluene (TNT), dichloro-diphenyl-

trichloroethane (DDT) and its metabolites (DDE and DDD), pentachlorophenol, PAHs, and polychlorinated biphenyls (PCBs).

5.1.2 A2: The Netherlands

As part of the Dutch Soil Protection Act (VROM 2000), the Netherlands Ministry of Housing, Spatial Planning and the Environment (VROM) has developed a series of soil-screening values for contaminated sites, remediation and long-term soil concentration goals, based on protection of soil health.

Soil quality is assessed and managed using three soil screening values—the target and intervention value and a value between these two termed the intermediate value. These values are independent of land use. Soils with contaminant concentrations below target value are considered to be at no risk and no restrictions on their use have been set. Soils with contaminant concentrations below the intermediate values can have certain restrictions set on soil and site management. Soils with contaminant concentrations exceeding intermediate but below the intervention value require further investigation of the site to assess the hazard posed by the contaminants. Soils with contaminant concentrations exceeding the intervention value require remediation as a matter of urgency.

Remediation levels for contaminants in soils have a separate set of values, the so-called reference values. These values are land use-specific, but site-specific reference values can be derived. Land uses are grouped into four clusters: 1) residential and intensively used parkland, 2) extensively used parkland, 3) buildings and paved areas, and 4) agriculture and nature reserves.

The intervention and target values are preferably derived using an SSD method with a log-normal distribution. Toxicity data used in the SSD approach are NOECs and LOECs but if these are not available, higher adverse effect data is used and converted to NOECs using a safety factor of 10. Toxicity data is normalised to a reference soil of 10% organic matter and 25% clay. The equations used to normalise the toxicity data (that is, normalisation equations) are based on the studies by Lexmond et al. (1986) and Van Straalen and Denneman (1989), where background levels of contaminants showed a positive relationship with organic matter and/or clay. Intervention values are designed to protect 50% of the species. In other words, the permitted concentration is hazardous to 50% of species and hence referred to as the HC_{50}. Target values are equal to the HC_5 (that is, the concentration that should permit only 5% of species to be affected) divided by 100. This factor 100 is applied to take into account combination toxicity (Crommentuijn 2000a).

If limited toxicity data is available, equilibrium partitioning (EqP) methods are used to derive soil screening values by extrapolation of aquatic toxicity data. If no data is available, the Dutch guidelines use QSARs to estimate toxicity data from contaminants that have the same mechanism of action.

Intervention and target values have been set for 75 contaminants and a further 20 contaminants have target values and/or indicative levels of serious contaminant levels (VROM 2000).

5.1.3 A3: Canada

The Canadian SQGs were developed by CCME to assess in-place contaminants in soil (CCME 1999, 2006) and can be found at: www.ccme.ca/publications/list_publications.html#link2.

SSQs and the level of protection for terrestrial species and soil processes depend on land use (that is, agriculture, residential/parkland, commercial and industrial sites). Using potential exposure scenarios, ecological receptors that sustain the primary activities for each land use are identified. These include soil invertebrates, soil nutrient cycling processes, plants, wildlife for all four land uses, soil and food ingestion by herbivores and consumers (including biomagnification) for residential and agricultural, and crops and livestock for agricultural land use.

SSQs were derived using laboratory and field-based toxicity data. This data measures the effects that undermine a species' ability to survive and reproduce under normal living conditions for soils that represented typical Canadian soils. The preferred measures of toxicity are 25% effect concentrations (IC_{25} or EC_{25}). A second option is to use LOECs divided by an uncertainty factor (safety factor) if there is insufficient 25% effect data (SSD method). A third option is to use median effect data (LC_{50} or EC_{50}) divided by an uncertainty factor (for agricultural and residential/parkland only, not for commercial and industrial sites). Depending on the quantity of toxicity data available, the weight-of-evidence (SSD) approach, LOEC method or median effects method was used to obtain SQGs. SSD was the preferred methodology if sufficient data was available. The output from the SSD might be divided by an uncertainty factor, depending on the type and amount of toxicity data used in the SSD. For the agricultural and residential/parkland land uses, the SQGs derived using an SSD (IC_{25} and/or EC_{25} data) are set to protect 75% of species and soil processes while, for commercial and industry land uses, 50% of the species are protected. A full description of the methodology can be found online at www.ccme.ca/assets/pdf/sg_protocol_1332_e.pdf.

If sufficient toxicity data is available, the SQGs distinguish between two generic soil types: coarse-textured soils (soils containing predominantly sand and gravel) and fine-textured soils (soils containing predominantly silt and clay). This separation has been made as contaminant fate, transport and bioavailability are dependent to varying degrees on soil texture, moisture content and other factors. Separation of the two soil types can thereby minimise the uncertainty in guideline derivation introduced by soil variability.

Thirty-two SQGs have been produced using the 1999 or 2006 derivation protocol, and 34 interim remediation criteria in soils remain (established in 1991) that have not yet been replaced by the SQG protocol. A complete list of SSQs and interim remediation criteria can be viewed at www.documents.ccme.ca.

The SQGs include: arsenic (inorganic), barium, benzene, benzo(*a*)pyrene, cadmium, chromium (total and Cr VI), copper, cyanide (free), DDT (total), di-isopropanolamine, ethylbenzene, ethylene glycol, lead, mercury (inorganic), naphthalene, nickel, nonylphenol (and its ethyloxylates), pentachlorophenol, phenol, PCBs, polychlorinated dibenzo-*p*-dioxins/dibenzofurans (PCDD/Fs), propylene glycol, selenium sulfolane, tetrachloroethylene, thallium, toluene, trichloroethylene, uranium, vanadium, xylenes, and zinc.

The interim remediation criteria include: conductivity, pH, sodium adsorption ratio, antimony, beryllium, boron (hot water soluble), cobalt, fluoride (total), molybdenum, silver, sulfur (elemental), tin, chlorobenzene, 1,2-dichlorobenzene, 1,3-dichlorobenzene, 1,4-dichlorobenzene, styrene, chlorophenols, nonchlorinated phenolic compounds, benzo(*a*)anthtracene, benzo(*b*)fluoranthene, benzo(*k*)fluoranthene, dibenz(*a*,*h*)anthracene, indeno(1,2,3-c,d)pyrene, phenanthrene, pyrene, chlorinated aliphatics, chlorobenzenes, hexachlorobenzene, hexachlorocyclohexane, nonchlorinated aliphatics, phthalic acid esters, quinoline, and thiophene.

5.1.4 A4: EU and UK

European Union Regulation 1488/94 and Directive 98/8 require that an environmental risk assessment be carried out on notified new substances, on priority existing substances and active substances and substances of concern in a biocidal product. Neither the regulation nor directive provides soil guideline values, but a technical guidance document (TGD) on ERA (ECB 2003) and soil guideline derivation was published as part of EU Directive 93/67 and is available online at ecb.jrc.it/Documents/TECHNICAL_GUIDANCE_DOCUMENT/EDITION_2/tgdpart2_2ed.pdf.

Several member states, including the UK, have adopted the methodology for deriving their national SQGs given in the technical guidance document (ECB 2003). Eventually, all EU member states will develop SQGs and use the method recommended in the TGD (ECB 2003).

In the UK, soil guideline values (SGVs) represent 'intervention values' which, if exceeded, indicate potentially unacceptable risks to site users and therefore trigger further investigation. SGVs aim to be precautionary to ensure that all the potential sites of concern are captured at the screening stage.

The SGVs are derived by calculating a predicted no-effect concentration (PNEC) preferably using NOEC data or estimates of NOECs (larger effect toxicity data, for example, EC_{50}, divided by a safety factor). The TGD (ECB 2003) recommends that, if possible, toxicity data should be normalised for the effect soil characteristics have on the toxicity of a contaminant.

The PNEC can be derived by three methodologies:

1. the EqP methodology if no or very limited terrestrial toxicity data is available
2. the AF approach if a limited data set is available
3. a statistical extrapolation using an SSD method if sufficient data (more than 10 species from 8 taxonomic groups) is available.

For the SSD, the TGD does not recommend a particular statistical distribution to be used in the SSD method. The output of the SSD is the HC_5. Whether the HC_5 value is protective is then assessed by the amount and type of toxicity data used in the SSD divided by an AF of between 1 and 5, depending on the uncertainties around the HC_5.

Currently, the EU is performing environmental risk assessments on all the existing chemicals and these reports can be found online at www.ecb.jrc.it/.

An overview document is available for methodologies used for deriving soil screening values for individual European countries (Carlon 2007) and is available online at www.ies.jrc.cec.eu.int/fileadmin/Documentation/Reports/RWER/EUR_2006-2007/EUR22805-EN.pdf.

5.1.5 A5: Germany

The German Federal Soil Protection and Contaminated Sites Ordinance (BBodSchV 1999) provides a series of precautionary, trigger and action values to protect terrestrial ecosystems from adverse effects from soil contamination. These values are used to prevent future soil contamination and for remediation of contaminated sites.

Precautionary values indicate a potential future soil impairment that should be averted. For inorganic chemicals, precautionary values are derived for three soil types: sandy, silt–loam and clay soils. For organic chemicals, precautionary values are derived for two soil types: soils with a humus content >8% and with a humus content ≤8 %. The ordinance does not give guidance on how to derive precautionary values.

Once the precautionary values have been exceeded, the ordinance (BBodSchV 1999) provides additional annual loading limits of the contaminants to prevent the soil concentration reaching the trigger or action values and causing adverse effects.

Trigger values trigger the investigation of the contaminated site to ascertain if the contaminant poses a hazard. Action values represent a direct hazard situation which should be prevented and therefore soils exceeding action values should be remediated. Action and trigger values are land use-dependent and specific exposure pathways are assigned to each land use. Trigger and action values are developed for three exposure pathways: soil to human, soil to plant and soil to groundwater. Trigger values for inorganic contaminants and the soil to plant pathway are, if possible, based on an estimate of the bioavailable concentration (that is, measured in 1 M NH_4NO_3 soil extraction). The soil to plant values

are based on regression analyses between soil and plant concentrations of the contaminant. A maximum internal plant concentration is set, either based on human health issues or plant toxicity, and the corresponding soil concentration, based on the linear regression, is the trigger or action value.

5.1.6 A6: New Zealand

The New Zealand Ministry for the Environment has developed environmental guideline values (EGVs) for contaminated land assessment, which are available online at www.mfe.govt.nz/publications/hazardous/contaminated-land-mgmt-guidelines-no2/contaminated-land-mgmt-guidelines-no2.pdf. The contaminated land management guidelines are not regulations but a guideline to obtain the most appropriate EGVs for a contaminated site.

New Zealand EGVs contain values with some derived within New Zealand and others by international regulators (for example, Canada, the Netherlands, USA, Australia). Therefore, a suite of methods was used to derive these values. A distinction was made between risk-based and threshold-based EGVs which is based on quality and quantity of the data available and the method used to derive the values.

Risk-based values are derived from a given exposure scenario; for example, protection of human health or the protection of a nominal proportion of species in an ecosystem and thus are calculated using a SSD method.

Threshold values may be derived from toxicological data where insufficient data is available to calculate risk-based values. The EGVs may also be classified as threshold values where insufficient information on their derivation is provided.

A hierarchy was established to determine the order in which EGVs should be used in a contaminated site assessment. The hierarchy in descending order of use is:

1. New Zealand-derived risk-based EGVs
2. risk-based EGVs from other national regulators
3. New Zealand-derived threshold EGVs
4. threshold EGVs from other national regulators.

Although EGVs are provided, the New Zealand framework stresses that the original reference document for an EGV must be referred to in order to assess if the EGV is relevant for the contaminated soil being investigated. Therefore, the EGVs and the framework are guidelines to obtain the most relevant EGV for a contaminated site.

5.2 Appendix B: method for deriving EILs that protect aquatic ecosystems

5.2.1 Determining the leaching potential of inorganic contaminants

The key physicochemical property of inorganic contaminants that controls their potential movement to ground and/or surface waters is the soil–water partition coefficient (K_d). This is the ratio of the concentration of a contaminant bound to the soil to that dissolved in soil pore water at equilibrium and therefore is related to the aqueous solubility of that contaminant. The lower the K_d, the more of a contaminant that will be present in the soil pore water. This may increase the potential for plants and soil invertebrates to be exposed via the pore water and increase the potential for leaching to groundwater and for groundwater organisms to be exposed. Although K_d is soil- and contaminant-dependent, a conservative cut-off point for inorganics at a log K_d of 3 is used in the methodology. The log K_d thresholds are presented in Table B1.

Table B1. Classification system used for the mobility of inorganic contaminants in soil, based on the logarithm of the soil–water partition coefficient (log K_d).

Log K_d value	Leachability
<3	High potential to leach (H)
≥3	Low potential to leach (L)

For inorganics with a log K_d <3, leaching of the contaminant should be addressed if there is a water source in the immediate vicinity.

5.2.2 Determining the leaching potential of organic contaminants

There are two partition coefficients related to the leaching potential of organic contaminants. The first is the octanol–water partition coefficient (K_{ow}), that is, the ratio of the concentration of a contaminant that is dissolved in *n*-octanol to that dissolved in water at equilibrium and at a specified temperature. It is used as a surrogate to estimate the potential for contaminants to accumulate in tissue—both plant and animal (Connell 1989, Posthumus & Slooff 2001). The second is the organic carbon–water partition coefficient (K_{oc}). Both K_{ow} and K_{oc} are chemical-specific values and collations of values for contaminants are widely available e.g. at http://www.epa.gov/region9/superfund/prg/.
Contaminants with a high log K_{oc} preferentially partition to soil organic matter rather than water and thus have a low potential to leach. Conversely, contaminants with a low log K_{ow} tend to have a high potential to leach. Log K_{ow} and log K_{oc} have a linear relationship (Briggs 1981, Connell 1989).

$$\log K_{oc} = 0.9 \times \log K_{ow} + 0.62 \qquad \textit{(equation B1)}$$

Therefore log K_{ow} (which is much more readily available than log K_{oc}) can act as a surrogate of the potential for contaminants to leach from soil to groundwater. On this basis, Wilson et al. (1996) used log K_{oc} and log K_{ow} to classify the mobility of organic contaminants in soil (Table B2).

Table B2. The classification system used for the mobility of organic contaminants in soil based on the logarithm of the organic carbon–water partition coefficient (log K_{oc}) and logarithm of the octanol–water partition coefficient (log K_{ow}). Modified from Wilson et al. 1996.

Corresponding log K_{ow} values[1]	log K_{oc}	Classification of mobility
<2	<2.4	Mobile (M)
2.0–2.7	2.4–3.05	Medium mobility (MM)
2.7–3.7	3.05–3.95	Low mobility (LM)
>3.7	>3.95	Immobile (IMM)

[1.] log K_{ow} values corresponding to the log K_{oc} values were derived using equation B1.

Many organic contaminants can degrade either biologically or chemically. Thus, it is recommended that EILs derived for organic contaminants with a slow degradation rate (that is, large half-life, refer to Table 2) and a log K_{oc} (or log K_{ow}) <4 should consider the protection of aquatic ecosystems where appropriate.

5.2.3 Calculation of EILs that protect aquatic ecosystems

The US EPA methodology (US EPA 1996) may be used to calculate EILs that account for the potential of contaminants to leach and affect aquatic ecosystems. Although the method has its limitations due to several simplifications, it is a robust method where the required information is available for Australian soils.

5.2.3.1 Inorganic contaminants

The potential leaching of inorganic contaminants to the groundwater depends on the soil to water partitioning of the contaminant, K_d, which is contaminant- and soil-dependent. Furthermore,

volatilisation can reduce the soil concentration of the inorganic contaminant and this amount will reduce the potential of the contaminant to leach to the groundwater. For essentially all inorganic contaminants, volatilisation is limited; however, for Hg, a substantial amount can be volatilised.

Because groundwater catchments will most likely contain both contaminated and uncontaminated soils, pore water concentrations of the contaminant in question will not always equal the groundwater concentration. Therefore, a dilution and attenuation factor (DAF) is used to take this into account. The fraction of contaminated land to the total area of the local groundwater/aquifer catchment can be used to calculate the DAF, as indicated by equation B1 below.

DAF = 100 ÷ percentage of contaminated soil in local catchment *(equation B2)*

Therefore, for inorganic contaminants the EIL is calculated as follows (US EPA 1996):

$$EIL = C_w \cdot \left(K_d + \frac{\theta_w + \theta_a \cdot H'}{\rho_b}\right) \cdot DAF \qquad \textit{(equation B3)}$$

where EIL is the ecological investigation level in soil (mg/kg), C_w is the target soil leachate concentration (mg/L) (that is, the appropriate WQG), K_d is the soil to water partition coefficient (L/kg), θ_w is the water-filled soil porosity L_{water}/L_{soil}), θ_a is the air-filled soil porosity (L_{air}/L_{soil}), ρ_b is the dry soil bulk density (kg/L), H is the Henry's law constant (unitless), and DAF is the dilution and attenuation factor.

5.2.3.2 Organic contaminants

Organic contaminants can bind to the organic carbon in soil. The extent of this depends on the properties of the contaminant and the amount and type of organic carbon in the soil. For organic contaminants the equation for soil to groundwater migration becomes (US EPA 1996):

$$EIL = C_w \cdot \left\{ \left(K_{oc} \cdot f_{oc}\right) + \frac{\theta_w + \theta_a \cdot H'}{\rho_b} \right\} \cdot DAF \qquad \textit{(equation B4)}$$

where EIL is the ecological investigation level in soil (mg/kg), C_w is the target soil leachate concentration (mg/L) (that is, the appropriate WQG), K_{oc} is the organic carbon to water partition coefficient (L/kg), f_{oc} is the organic carbon content of soil (kg/kg), θ_w is the water-filled soil porosity (L_{water}/L_{soil}), θ_a is the air-filled soil porosity (L_{air}/L_{soil}), ρ_b is the dry soil bulk density (kg/L), H is the Henry's law constant (unitless), and DAF is the dilution and attenuation factor that is calculated as per equation B2.

The target soil leachate concentration (C_w) should be set as the relevant WQG for that contaminant in groundwater systems, which currently is the surface freshwater TV (ANZECC & ARMCANZ 2000).

5.3 Appendix C: Methods for determining the bioavailability of contaminants and how this could be incorporated into the ERA framework

The methodology for deriving EILs outlined in this Schedule accounts for the effects of soil reactions that modify the bioavailability of soluble contaminants. However, it does not take into account the form or bioavailability of the contaminant. The EIL derivation framework also makes the assumption that ecotoxicity data in the literature is derived using highly bioavailable forms of contaminants (for example, soluble metal salts or soluble organic molecules), and indeed this is generally the case for most ecotoxicity studies. Thus, the framework is reasonably conservative in its assumptions and protective, and is appropriate for a screening level risk assessment.

Soil contamination can occur from a variety of sources, and not all these sources have 100% bioavailability when they are initially added to soil; for example, vitreous slags, tyre debris, massive metal, encapsulated materials, etc.

When total concentrations of contaminants are determined in a soil containing these materials, these contaminants will be solubilised, assumed to be bioavailable, and therefore some sites may exceed the EILs, yet the actual risk is negligible. Further chemical investigation of the bioavailability of the contaminants should be undertaken prior to direct toxicity assessment.

For a detailed review of methods to assess metal bioavailability in soils, see McLaughlin et al. (2000b). For detailed reviews of methods to assess bioavailability of organic contaminants in soils see Stokes et al. (2005) and Dean and Scott (2004).

Information on leachability tests applicable to contaminated sites can be found in Schedule B3.

6 Glossary

Adaptation is (1) change in an organism in response to changing conditions of the environment (specifically chemical), which occurs without any irreversible disruption of the given biological system and without exceeding the normal (homeostatic) capacities of its response, and (2) a process by which an organism stabilises its physiological condition after an environmental change.

Added contaminant limit (ACL) is the added concentration of a contaminant above which further appropriate investigation and evaluation of the impact on ecological values will be required. ACL values are generated in the process of deriving ecological investigation levels (EILs).

Adsorption is the adhesion of molecules to surfaces of solids.

Ambient background concentration (ABC) of a contaminant is the soil concentration in a specified locality that is the sum of the naturally occurring background and the contaminant levels that have been introduced from diffuse or non-point sources by general anthropogenic activity not attributed to industrial, commercial, or agricultural activities.

An **area of ecological significance** is one where the planning provisions or land use designation is for the primary intention of conserving and protecting the natural environment. This would include national parks, state parks, and wilderness areas and designated conservation areas.

Bioaccumulation is the net result of the uptake, distribution and elimination of a substance due to all routes of exposure, that is, exposure to air, water, soil/sediment and food.

Bioaccumulation factor is a partition coefficient for the distribution of a chemical between an organism exposed through all possible routes and an environmental compartment or food.

Bioavailability is the ability of a contaminant to interact with the biological system of an organism. Not all of a contaminant that is present in environmental compartments (for example, soil, sediment, water and air) is biologically available—rather, only a fraction of the total (the bioavailable fraction) is available.

Bioconcentration factor (BCF) is a quantitative measure of a chemical's tendency to be taken up from the ambient environment (for example, water for aquatic organisms and soil or soil pore water for soil organisms). The BCF is the ratio of the concentration of the chemical in tissue (or a specific organ) and the concentration in the ambient environment.

Bioconcentration is the net result of the uptake, distribution and elimination of a substance due to exposure in the ambient environment (for example, water for aquatic organisms and soil or soil pore water for soil organisms).

Biological half life is the time needed to reduce the concentration of a test chemical in the environmental compartment or organisms to half the initial concentration, by transport processes, (for example, diffusive elimination), transformation processes (for example, biodegradation or metabolism) or growth.

Biomagnification factor is the quantitative measure of a chemical's tendency to be taken up through the food web.

Biomagnification is the accumulation and transfer of chemicals via the food web due to ingestion, resulting in an increase of the internal concentration in organisms at the succeeding trophic levels.

Chronic is the extended or long-term exposure to a stressor, conventionally taken to include at least a tenth of the life-span of a species.

Concentration–response curve is a curve describing the relationship between response in the test population and exposure concentration.

Contaminant is any chemical existing in the environment above background levels and representing, or potentially representing, an adverse health or environmental risk.

Contamination means the condition of land or water where any chemical substance or waste has been added at above background level and represents, or potentially represents, an adverse health or environmental impact.

Control is treatment in a trial that duplicates all the conditions of the exposure treatments but contains no test material.

Default conversion factors are numerical values that are used to convert a measure of toxicity to another measure of toxicity (for example, EC_{50} to a NOEC) when no experimentally determined values are available.

Ecological investigation level (EIL) is the concentration of a contaminant above which further appropriate investigation and evaluation of the impact on ecological values will be required. The EILs are calculated using EC_{30} or lowest observed effect concentrations (LOEC) toxicity data. EILs are the sum of the added contaminant limit (ACL) and the ambient background concentration (ABC) and the limit is expressed in terms of total concentration. All EILs, whether generic, soil-specific or site-specific, only apply to soil to a depth of two metres below the current soil surface.

ECx means **effective concentration**; the concentration which affects X% of a test population after a specified exposure time.

End point assessment is a quantitative or quantifiable expression of the environmental value considered to be at risk in a risk analysis.

Environmental fate means the destiny of a chemical or biological pollutant after its release into the natural environment.

Environmental quality guideline is a generic term that applies to any guidelines that control the concentration of contaminants in various environmental compartments (for example, water, sediment, soil).

Freundlich adsorption isotherm is an empirical equation that describes the adsorption of a contaminant to soil. The equation for this is $x/m = K_f C_e^{\ l/n}$, where x/m is the concentration of the contaminant in soil (mg/kg), C_e is the contaminant concentration in the aqueous phase at equilibrium (mg/L), K_f is the equilibrium constant (the Freundlich adsorption constant) and l/n is the contaminant-specific exponent.

Generic soil quality guidelines describe a single concentration-based value that applies to all Australian soils that have a particular land use. These are derived when normalisation relationships are not available. Compare these with soil-specific soil quality guidelines.

Indicator means a biotic characteristic of the environment, for example, a plant end point that provides evidence of the occurrence or magnitude of exposure or effects.

K_d (see **water to soil partition coefficient**).

K_{oc} (see **organic carbon–water partition coefficient**)

K_{ow} (see **octanol–water partition coefficient**)

Leaching involves the dissolving of contaminants in soil and subsequent downward transport to groundwater or surface water bodies.

Leachate is water that has percolated through a column of soil.

LOEC is the lowest observed effect concentration (level); the lowest concentration of a material used in a test that has a statistically significant effect on the exposed population of test organisms compared to the control.

Logistic curve is a function fitting the general equation $y = k/ (1+e^{a+bt})$ where t represents time, y the body weight or population size, a and b are model-specific parameters. This mathematical function with parameters can be adjusted so that the function closely describes a set of empirical data. Statistical models are curve-fitted to data where the mathematical function used is selected for its numerical properties.

NOEC means no observed effect concentration; the highest concentration of a test substance to which organisms are exposed that does not cause any observed and statistically significant adverse effects on the organisms compared to the controls.

Normalisation relationships are empirical, generally linear, relationships that can predict the toxicity of a contaminant to an organism using soil physicochemical properties. These are used in the EIL derivation methodology to generate soil-specific soil quality guidelines.

Octanol–water partitioning (K_{ow}) means the ratio of a chemical's solubility in n-octanol and water at equilibrium. This is widely used as a surrogate for the ability of a contaminant to

accumulate in organisms and to biomagnify. These are often expressed in the logarithmic form (that is, log K_{ow}). Chemicals with a log K_{ow} value ≥4 are considered in this guideline to have the potential to biomagnify. There is a linear relationship between log K_{ow} and log K_{oc} values. Thus, K_{ow} can also be used to indicate the ability of chemical to leach to groundwater. A log K_{ow} value <2 indicates a chemical has the potential to leach to groundwater.

Organic carbon–water partition coefficient (K_{oc}) means the ratio of a chemical's solubility in organic carbon and water at equilibrium. This is widely used as a surrogate for the ability of a contaminant to accumulate in soils and conversely to leach to groundwater or to be removed by surface run-off. These are often expressed in the logarithmic form (that is, log K_{oc}). Chemicals with a log K_{oc} <2.4 were considered, in this guideline, to be mobile and therefore have the ability in some soils to leach to groundwater.

Precautionary principle is the general principle by which all that can reasonably be expected is done to prevent unnecessary risks.

Reference site is a relatively uncontaminated site used for comparison with contaminated sites in environmental monitoring studies or used for the assessment of ambient background concentrations of contaminants.

Risk assessment is a process intended to calculate or estimate the risk to a given target organism, system or sub-population, including the identification of attendant uncertainties, following exposure to a particular agent, taking into account the inherent characterisations of the agent of concern as well as the characterisation of the specific target.

Risk means the probability in a certain timeframe that an adverse outcome will occur in a person, a group of people, plants, animals and/or the ecology of a specified area that is exposed to a particular dose or concentration of a chemical substance; that is, it depends on both the level of toxicity of the chemical substance and the level of exposure to it.

Secondary poisoning is the product of biomagnification and toxicity.

Soil quality guideline (SQG) is a collective term used to describe any quantitative or qualitative limit that controls the concentration of contaminants in soils. Ecological investigation levels (EILs) are a type of SQG.

Soil-specific soil quality guidelines is a suite of concentration-based values, where each value applies to a soil with different physicochemical properties. These values take into account properties of soils that modify the bioavailability and toxicity of contaminants. These can only be derived if normalisation relationships are available. Compare these to generic SQGs.

Speciation is the exact chemical form of contaminant in which an element occurs in a sample.

Species sensitivity distribution (SSD) is a suite of methods that are the main method used to derive quality guidelines for contaminants in different compartments of the environment (for example, soil, water, sediment). Basically, these plot toxicity data (one value per species) as a cumulative frequency distribution against the concentration at which the toxic effect occurs. A statistical distribution is then fitted to the plot from which it can be estimated what concentration is required to protect any chosen percentage of species. In Australia, the SSD method used to derive guidelines uses the Burr type III family of distributions and is called the BurrliOZ method.

Statistically significant effects are effects (responses) in the exposed population which are different from those in the controls at a statistical probability level of p <0.05.

Steady state is the non-equilibrium state of a system in which matter flows in and out at equal rates so that all of the components remain at constant concentrations (dynamic equilibrium).

Water to soil partition coefficient (K_d) is the ratio of the concentration of a contaminant in soil pore water to that in the solid phase of soil at equilibrium. The units are L/kg. This contaminant property is affected by physicochemical properties of the contaminant and the soil. This property is usually expressed as a logarithm (that is, log K_d). In this guideline, chemicals with log K_d <3 are considered to have the potential to leach.

7 Shortened forms

ABC	ambient background concentration
ACL	added contaminant limit
ACR	acute-to-chronic ratio
AF	assessment factor
ALF	ageing/leaching factor
ANZECC	Australia and New Zealand Environment and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
BAF	bioaccumulation factor
BCF	bioconcentration factor
BMF	biomagnification factor
CCME	Canadian Council of Ministers of the Environment
CEC	cation exchange capacity
DAF	dilution and attenuation factor
DOC	dissolved organic carbon
DTA	direct toxicity assessment
Eco-SSL	ecological soil screening level
ECB	European Chemicals Bureau
EC_{30}	30% effect concentration
EGV	environmental guideline value
EIL	ecological investigation level
ERA	ecological risk assessment
EqP	equilibrium partitioning method
EQG	environmental quality guideline
EU	European Union
HC	hazardous concentration
HIL	health investigation level
LOEC	lowest observed effect concentration

MATC	maximum acceptable toxicant concentration
NBRP	National Biosolids Research Program
NEPC	National Environment Protection Council
NEPM	National Environment Protection Measure
NHMRC	National Health and Medical Research Council
NOEC	no observed effect concentration
OECD	Organisation for Economic Cooperation and Development
OM	organic matter
PC	protective concentration
PLS	partial least squares
PNEC	predicted no-effect concentration
QAAR	quantitative activity–activity relationship
QSAR	quantitative structure–activity relationship
QSPR	quantitative structure–property relationship
SGV	soil guideline value
SIN	substrate-induced nitrification
SQG	soil quality guideline
SQV	soil quality value
SSD	species sensitivity distribution
US EPA	United States Environmental Protection Agency
TGD	technical guidance document
TRV	toxicity reference value
TV	trigger value
USA	United States of America
VROM	Ministry of Housing, Spatial Planning, and the Environment (Netherlands)
WHC	water holding capacity
WQG	water quality guideline

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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Volume 1: sections 1-6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of

Schedule B5c

GUIDELINE ON

Ecological Investigation Levels for Arsenic, Chromium (III), Copper, DDT, Lead, Naphthalene, Nickel & Zinc

Explanatory note

The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

The original Schedule B5 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document, together with Schedule B5a and Schedule B5b.

The National Environment Protection Council (NEPC) acknowledges the contribution of the Commonwealth Scientific and Industrial Research Organisation (CSIRO), the NSW Environment Protection Authority and the NSW Environmental Trust to the development of this Measure.

Contents
Guideline on ecological investigation levels

1 Introduction

1.1 Objectives

The objective of this guideline is to derive EILs for arsenic (As), copper (Cu), chromium III (Cr (III)), dichlorodiphenyltrichloroethane (DDT), naphthalene, nickel (Ni), lead (Pb) and zinc (Zn) using the methodology detailed in Schedule B5b to:

- illustrate the flexibility of the methodology—being able to derive soil contaminant limits that provide different levels of protection, and use different toxicity data
- illustrate the magnitude and appropriateness of the soil contaminant limits
- compare the EILs with those of overseas jurisdictions.

1.2 Terminology

The term 'soil quality guideline' (SQG) is used in this guideline to describe any concentration-based limit for contaminants in soils.

A combination of lowest observed effect concentration (LOEC) and 30% effect concentration data (EC_{30}) has been adopted in the NEPM for the derivation of EILs. Equivalent data for EC_{10} and EC_{50} is included for information purposes only.

2 Overview of the method for deriving soil quality guidelines

Soil quality guidelines can have various purposes. The National Environment Protection (Assessment of Site Contamination) Measure (NEPM) contains a specific type of SQG, the ecological investigation level (EIL), to guide the assessment of contaminated sites in Australia. The EILs were derived in such a manner that when they are exceeded it indicates that terrestrial ecosystems may experience harmful effects due to the presence of contaminants. The EILs are thus used to indicate when further investigation is necessary.

However, SQGs with other purposes can and have been developed. For example, the Dutch have three sets of SQGs, each with a different purpose. These are target levels (their purpose is to indicate the long-term goals for the concentration of contaminants), maximum permissible levels (their purpose is to define the maximum level of contamination that is considered acceptable), and intervention levels (their purpose is to define the maximum permitted concentration before some immediate action is required).

As a result of consultation conducted in developing the Australian methodology in November 2008, three different sets of ecotoxicity data were used to derive SQGs. The three sets of SQGs are termed $SQG_{(NOEC\ \&\ EC10)}$, $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ reflecting the type of ecotoxicity data that was used in their generation. A summary of the three types of SQGs, the data used and likely ecotoxicological effects that would be expected to occur if these are met is presented in Table 1. A combination of lowest observed effect concentration (LOEC) and 30% effect concentration data (EC_{30}) has been adopted in the NEPM for the derivation of EILs.

Table 1. The relationship between the three types of soil quality guidelines (SQGs), the data that is used to derive the SQGs and the type of toxic effects that would be experienced if the SQGs are met.

Type of SQG	Toxicity data used to calculate the SQGs	Expected toxic effects if the SQG is not exceeded
$SQG_{(NOEC\ \&\ EC10)}$	NOEC and EC_{10}	slight toxic effects
$SQG_{(LOEC\ \&\ EC30)}$	LOEC and EC_{30}	moderate toxic effects
$SQG_{(EC50)}$	EC_{50}	significant toxic effects

An overview of the SQG derivation methodology (detailed in Schedule B5b) is presented in Figure 1. One of the key aims in developing the methodology was to account for the availability and toxicity of the contaminant in the soil being studied. To do this, key soil and site-specific factors that are known to modify the toxicity of contaminants had to be accounted for. One factor that was incorporated into the methodology was the background concentration. In order to do this, the data used to derive the SQGs was expressed in terms of the amount of contaminant that had to be added to the soil to cause toxicity. When this toxicity data was used in accordance with the methodology, the resulting value was termed the added contaminant level (ACL). An ambient background concentration (ABC) specific to the soil being investigated was then added to the ACL to calculate the SQG.

ACL values are generated as part of the methodology of deriving SQGs. Thus, it is necessary to differentiate the ACLs generated in deriving $SQG_{(NOEC\ \&\ EC10)}$ from those generated in deriving $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values. The ACL generated in deriving an $SQG_{(NOEC\ \&\ EC10)}$ is termed

the NOEC and EC_{10}-based ACL ($ACL_{(NOEC \& EC10)}$). Similarly, ACLs generated in deriving $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values are referred to as the LOEC and EC_{30}-based ACL ($ACL_{(LOEC \& EC30)}$) and the EC_{50}-based ACL ($ACL_{(EC50)}$).

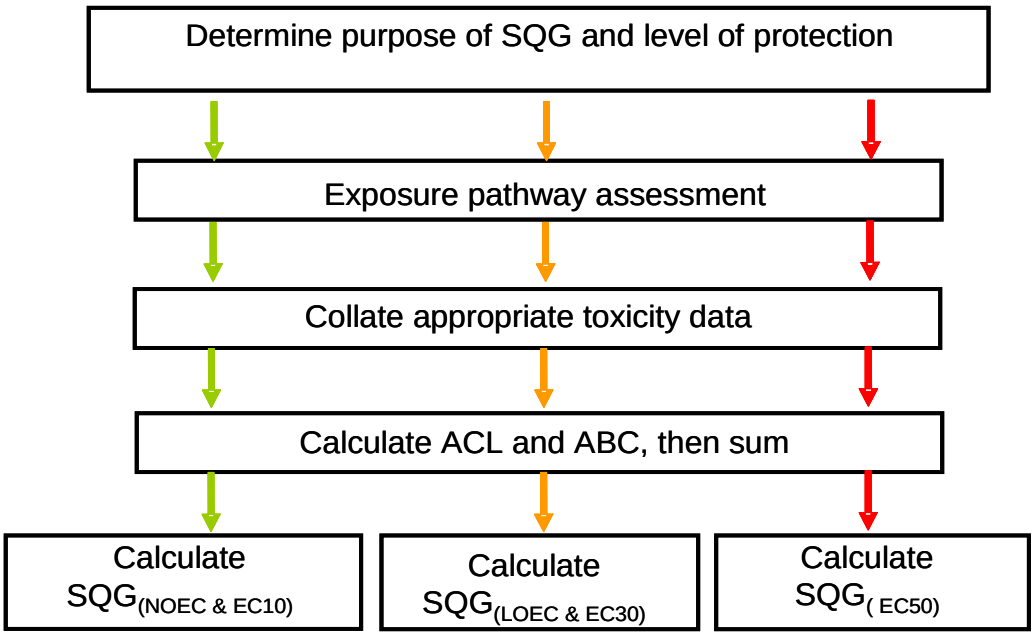


Figure 1. Overview of the methodology for deriving soil quality guidelines based on NOEC and EC_{10} data ($SQG_{(NOEC \& EC10)}$) indicated by the green (far left) arrows, based on LOEC and EC_{30} data ($SQG_{(LOEC \& EC30)}$) indicated by the orange (middle) arrows and based on EC_{50} data ($SQG_{(EC50)}$) indicated by the red (far right) arrows. As part of this process, ACLs and ABCs are calculated. The differences between the three SQGs are presented in Table 1.

The key steps in the methodology are:

1. determining the purpose of the SQG and the appropriate level of protection
2. determining the most important exposure pathways
3. collating and screening the toxicity data
4. determining whether the contamination is fresh or aged and whether there are ageing/leaching factors available to account for this
5. normalising the toxicity data
6. calculating the ACL
7. accounting for biomagnification
8. measuring or calculating the ABC
9. calculating $SQG_{(NOEC \& EC10)}$, $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values for fresh contamination in soils with different land uses
10. calculating $SQG_{(NOEC \& EC10)}$, $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values for aged contamination in soils with different land uses.

These key steps and the decision pathway involved in deriving $ACL_{(NOEC \& EC10)}$ and $SQG_{(NOEC \& EC10)}$ values are provided in Figure 2 below. Exactly the same procedure would be used to derive $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values, except that different toxicity data would be used (Table 1). Details of the methodology for calculating SQGs are provided in Schedule B5b.

Land has a variety of potential uses, and the level of protection that is appropriate for each land use varies. For example, it is appropriate for a higher level of protection to be applied to areas of ecological significance compared to industrial land. The recommended levels of protection for various land uses are provided in Schedule B5b and are used in this guideline. For contaminants that do not biomagnify, the recommended level of protection of species for areas of ecological significance, urban residential/public open space and commercial/industrial land are 99%, 80% and 60% respectively. For contaminants that biomagnify, the recommended levels of protection of species for areas of ecological significance, urban residential/public open space and commercial/industrial land are 99%, 85% and 65% respectively. SQGs were generated for areas of ecological significance, urban residential land/public open space, and commercial/industrial land uses.

The contamination at many contaminated sites is not fresh, rather it has been there for some years. The biological availability (bioavailability) and toxicity of many contaminants decreases over time (that is, it ages) due to binding to soil particles, chemical and biological degradation and a range of other processes. Furthermore, in many laboratory-based ecotoxicity experiments that spike soils with soluble metal salts, ecotoxicity is overestimated due to a lack of leaching of soluble salts which affect metal sorption. These factors have been addressed in recent risk assessments for metals in soils using 'ageing/leaching' factors, and can be accounted for by multiplying the toxicity data by an ageing/leaching factor and thus deriving SQGs for aged contamination. Site-specific assessments of a contaminant's bioavailability can also be made, but these are usually conducted as part of a more detailed site-specific (Tier 2) ecological risk assessment. When ageing/leaching factors were available for the test chemicals examined in this study, SQGs were derived for aged contamination.

When contaminants are introduced to soil, some will bind strongly to the soil while others are mobile and will move off-site. Leaching to groundwater is a key off-site migration pathway and can result in aquatic ecosystems being exposed to contaminants. Therefore, the potential of contaminants to leach is an important characteristic that affects the environmental fate and effect they cause. The leaching potential is not controlled solely by the physicochemical properties of contaminants, but also by the properties of the soil containing the contaminant and climatic conditions. It is not possible or appropriate to account for the potential to leach in deriving practical SQGs at a generic level, rather this should be done as part of a more detailed site-specific ecological risk assessment.

Given the available data, the most complete set of SQGs was derived for each of the eight contaminants. A summary of what SQGs could be derived is presented below.

- For chromium (III), copper, nickel and zinc, it was possible to derive a set of soil-specific SQGs using each of the three types of toxicity data for each of the three land uses for both fresh and aged contamination.
- For arsenic and lead, it was possible to derive generic (not soil-specific) SQGs using each of the three types of toxicity data for each of the three land uses and for both fresh and aged contamination.
- For DDT and naphthalene, it was possible to derive generic (not soil-specific) SQGs using each of the three types of toxicity data for each of the three land uses but only for fresh contamination.

In addition, SQGs that account for the potential of contaminants to leach (and therefore should protect aquatic ecosystems) were derived for arsenic and zinc. This was only done for these contaminants to illustrate how this is done and what effect it has on the resulting SQGs compared to the SQGs that do not account for leaching.

2.1 Precision of estimates and rounding of added contaminant limits

In order to increase the readability and ease of use of this report the ACL, ABC and SQG values presented in the various tables have all been rounded off using the following scheme:

- all values <1 were rounded off to the nearest 0.1
- all values between 1 and 10 were rounded off to the nearest whole number

- all values between 10 and 100 were rounded off to the nearest multiple of 5
- all values between 100 and 1000 were rounded off to the nearest multiple of 10
- all values greater than 1000 were rounded off to the nearest 100 units.

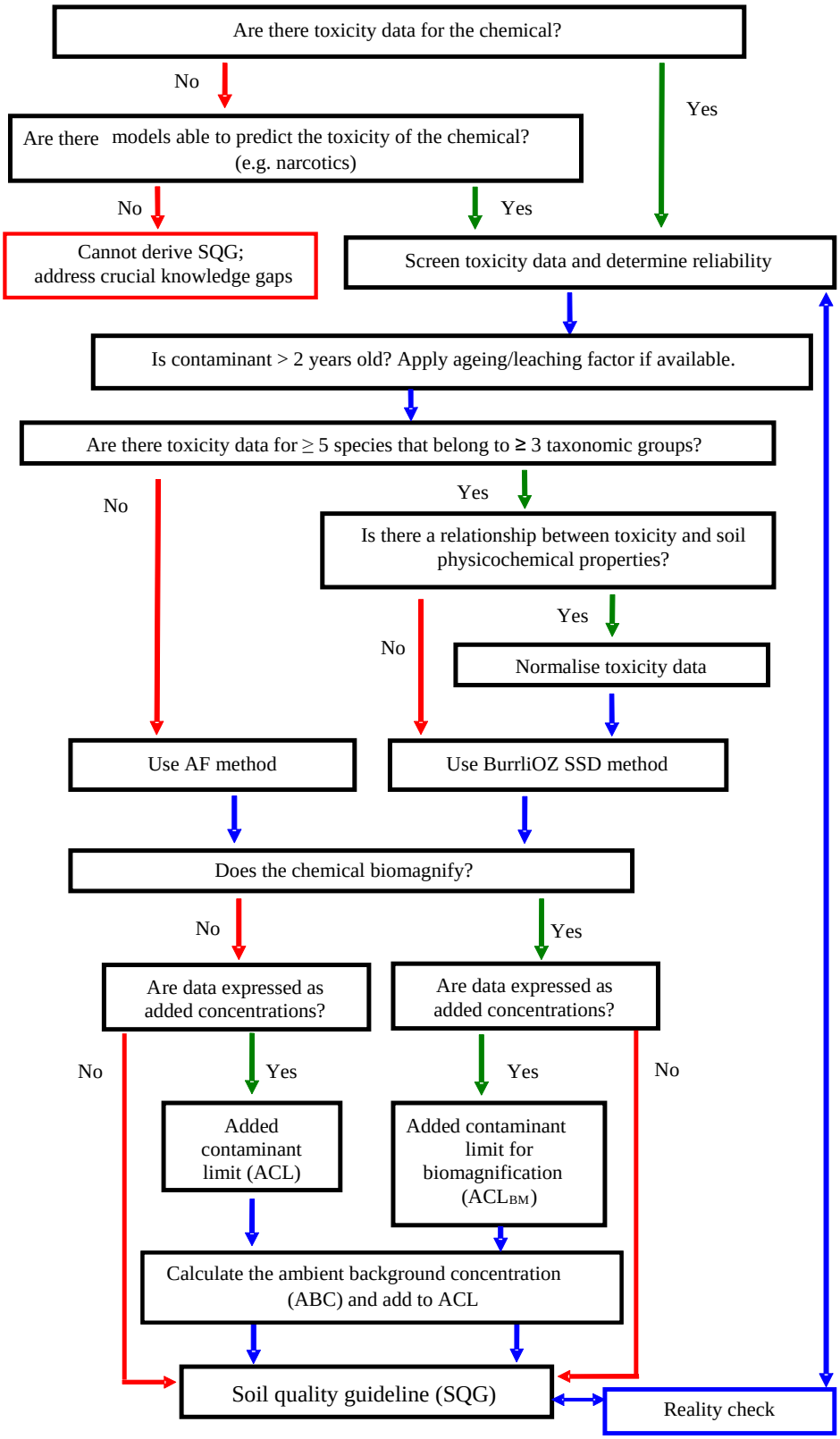


Figure 2. Schematic of the methodology for deriving soil quality guidelines (SQGs) (modified from Heemsbergen et al. 2008). Green arrows show the path when the preceding question was answered with a 'yes' while the red arrows indicate the path when the answer was 'no'. Blue arrows indicate the path when there is no choice.

3 Zinc

3.1 Zinc compounds considered

The SQGs for Zn were derived using data for the following:

- zinc metal (CAS No. 7440-66-6)
- zinc oxide (CAS No. 1314-13-2)
- zinc distearate (CAS Nos 557-05-1/91051-01-3)
- zinc chloride (CAS No. 7646-85-7)
- zinc sulphate (CAS No. 7733-02-0).

3.2 Exposure pathway assessment

The two key considerations in determining the most important exposure pathways for inorganic contaminants are whether they biomagnify (see Glossary) and whether they have the potential to leach to groundwater.

A surrogate measure of the potential for a contaminant to leach is its water–soil partition coefficient (K_d). If the logarithm of the K_d (log K_d) of an inorganic contaminant is less than 3 then it is considered to have the potential to leach to groundwater (Schedule B5b). The Australian National Biosolids Research Program (NBRP) measured the log K_d of Zn in 17 agricultural soils throughout Australia. These measurements showed that in most soils the log K_d of Zn was below 3 L/kg (unpublished data). The log K_d value for Zn reported by Crommentuijn et al. (2000) was 2.2 L/kg. Therefore, there is the potential for Zn in some soils to leach to groundwater and affect aquatic ecosystems. However, the methodology for EIL derivation (Schedule B5b) does not advocate the routine derivation of EILs that account for leaching potential. Rather, it advocates that this is done on a site-specific basis as appropriate. However, the calculations of Zn SQGs that account for leaching have been included here as an illustration of the process and the effect that this has on the resulting soil quality guidelines.

Zinc is an essential element and, as such, concentrations of Zn in tissue are highly regulated and it does not biomagnify (Louma & Rainbow 2008; Schedule B5b). Therefore, the biomagnification route of exposure does not need to be considered for Zn and the SQGs will only account for direct toxicity.

3.3 Toxicity data

Zinc is a well-studied inorganic contaminant and therefore a large dataset of toxicity values was available. Most studies presented their toxicity data in terms of added concentration (that is, the concentration of the contaminant added to the soil that causes a specified toxic effect) and so could be used without further modification. Some toxicity data was expressed in terms of total contaminant concentration but the background concentrations were reported. In such cases, the toxicity data was converted to an added concentration basis by subtracting the background from the total concentration. If toxicity data was expressed in terms of total contaminant concentration but the background concentration was not reported then the Dutch background correction equation (Lexmond et al. 1986) was used to estimate the background concentration.

$$\text{background Zn} = 1.5 * [2 * \text{organic matter (\%)} + \text{clay content (\%)}] \qquad (\textit{equation 1})$$

The background concentration was then subtracted from the total concentration data to derive the added concentration toxicity value.

The toxicity database used to calculate the $SQG_{(NOEC\ \&\ EC10)}$ values for Zn included EC_{10} and NOEC toxicity data for nine soil processes (Table 2), 14 invertebrate species and 1 invertebrate community measurement (Table 3) and 22 plant species (Table 4). The raw data used to generate Tables 2–4 is provided in Appendix A. There was sufficient data (that is, toxicity data) for at least five species or soil processes that belong to at least three taxonomic or nutrient groups (Schedule B5b) available to derive $SQG_{(NOEC\ \&\ EC10)}$ values using a species sensitivity distribution (SSD) methodology. Given that

Zn does not biomagnify, the level of protection recommended for non-biomagnifying contaminants was used to generate the SQG for each land use.

Table 2. The geometric mean values of the zinc toxicity data (expressed in terms of added Zn) for individual soil processes.

Soil process	Geometric means (mg/kg added Zn)		
	EC_{10} or NOEC	EC_{30} or LOEC	EC_{50}
Acetate decomposition	187	280	560
Amidase	121	182	364
Ammonification	98	148	295
Arylsulphatase	289	434	868
Glucose decomposition	274	1169	2904
Nitrate reductase	56	84	168
Nitrification	455	706	930
Phosphatase	674	1011	2022
Respiration	104	157	313

Table 3. The geometric mean values of zinc (Zn) toxicity data (as added Zn) for soil invertebrate species and an invertebrate community.

Species/endpoint		Geometric means (mg/kg added Zn)		
Common name	Scientific name	EC_{10} or NOEC	EC_{30} or LOEC	EC_{50}
Earthworm	*Aporrectodea caliginosa*	223	274	391
Earthworm	*Aporrectodea rosea*	390	407	436
Earthworm	*Eisenia fetida*	201	296	575
Earthworm	*Lumbriculus rubellus*	220	285	443
Earthworm	*Lumbriculus terrestris*	1062	1257	1675
Nematode	*Acrobeloides sp.*	221	332	663
Nematode	*Caenorhabditis elegans*	122	183	366
Nematode	*C. elegans* (dauer larvae)	689	1034	2068
Nematode	Community nematodes	306	459	919
Nematode	*Eucephalobus sp.*	135	202	403
Nematode	*Plectus sp.*	23	35	70
Nematode	*Rhabditidae sp.*	199	299	597
Potworm	*Enchytraeus albidus*	121	181	363
Potworm	*Enchytraeus crypticus*	276	414	828
Springtail	*Folsomia candida*	188	283	565

Table 4. The geometric mean values of the zinc (Zn) toxicity data (expressed in terms of added Zn) for individual plant species.

Plant species		Geometric means (mg/kg added Zn)		
Common name	**Scientific name**	**EC_{10} or NOEC**	**EC_{30} or LOEC**	**EC_{50}**
Alfalfa	*Medicago sativa*	198	297	595
Barley	*Hordeum vulgare*	83	233	495
Beet	*Beta vulgaris*	198	297	595
Black or white lentil	*Vigna mungo*	95	142	284
Canola	*Brassica napus*	230	328	409
Common vetch	*Vicia sativa*	42	63	127
Cotton	*Gossypium sp.*	272	288	293
Fenugreek	*Trigonella foenum graecum*	106	159	318
Lettuce	*Latuca sativa*	264	396	793
Maize	*Zea mays*	202	304	581
Millet	*Panicum milaceum*	540	1580	2026
Oats	*Avena sativa*	222	333	667
Onion	*Allium cepa*	66	99	198
Pea	*Pisum sativum*	264	396	793
Peanuts	*Arachis hypogaea*	140	224	280
Red clover	*Trifolium pratense*	39	59	117
Sorghum	*Sorghum sp.*	123	254	444
Spinach	*Spinacia oleracea*	132	198	396
Sugar cane	*Sacharum*	3220	4830	9661
Tomato	*Lycopersicon esculentum*	264	396	793
Triticale	*Tritosecale sp.*	998	1364	1658
Wheat	*Triticum aestivum*	640	928	1172

3.4 Normalisation relationships

A normalisation relationship is an empirical model that predicts the toxicity of a single contaminant to a single species using soil physicochemical properties (for example, soil pH and organic carbon content). Seven normalisation relationships were reported in the literature for Zn toxicity (Table 5). Three were developed for Australian soils (Broos et al. 2007; Warne et al. 2008a; Warne et al. 2008b) and four have been derived for European soils (Lock & Janssen 2001; Smolders et al. 2003). Three of the relationships were for plants, two for microbial functions and two for soil invertebrates. Of these, relationships 1–4, 6 and 7 were used to derive Zn SQGs. Relationship number 5 for wheat was not used, as an equivalent field-based relationship for Australian soils was available and field-based normalisation relationships provide better estimates of toxicity in the field (Warne et al. 2008a) and thus are preferred to laboratory-based relationships (Schedule B5b).

Normalisation relationships are used to account for the effect of soil characteristics on toxicity data, so the resulting toxicity data more closely reflect the inherent sensitivity of the test species. All the Zn toxicity data in Tables 2–4 was normalised to their equivalent toxicity in the recommended Australian

reference soil (Schedule B5b) (Table 6). Depending on the conditions under which the toxicity tests were conducted, the normalised toxicity data could be higher or lower in the reference soil compared to the original toxicity data in the test soil.

Table 5. Normalisation relationships for the toxicity of zinc to soil invertebrates, soil processes and plants.

Eqn	Species/soil process	Y parameter	X parameter(s)	Reference
1	*E. fetida* (earthworm)	log EC_{50}	0.79 * log CEC	Lock and Janssen 2001
2	*F. candida* (collembola)	log EC_{50}	1.14 * log CEC	Lock and Janssen 2001
3	PNR	log EC_{50}	0.15 * pH	Smolders et al. 2003
4	SIN	log EC_{50}	0.34 * pH + 0.93	Broos et al. 2007
5	*T. aestivum* (wheat)	log EC_{10}	0.14 * pH + 0.89 * log OC + 1.67	Warne et al. 2008a
6		log EC_{10}	0.271 * pH +0.702 * CEC + 0.477	Warne et al. 2008b
7		log EC_{50}	0.12 * pH +0.89 * log CEC + 1.1	Smolders et al. 2003

CEC = cation exchange capacity ($cmol_c/kg$); OC = organic carbon content (%); PNR = potential nitrification rate; SIN = substrate induced respiration.

Table 6. Values of soil characteristics for the recommended Australian reference soil to be used to normalise toxicity data

Soil property	Value
pH	6
Clay (%)	10
CEC ($cmol_c/kg$)	10
OC (%)	1

3.5 Sensitivity of organisms to zinc

The toxicity data (geometric means) used by the SSD method to calculate the ACL is shown in Table 2 for soil processes, Table 3 for soil invertebrates and Table 4 for plants. Figure 3 shows the SSD (that is, a cumulative distribution of the geometric means of the species) for all species for which there was Zn toxicity data. Toxicity data for plants, soil processes and soil invertebrates was evenly spread in the SSD, which indicates that these groups of organisms all have a similar sensitivity to Zn. Therefore, all the toxicity data was used to derive the ACLs, thus increasing the quantity of data used in the SSD method and increasing the reliability of the ACL values.

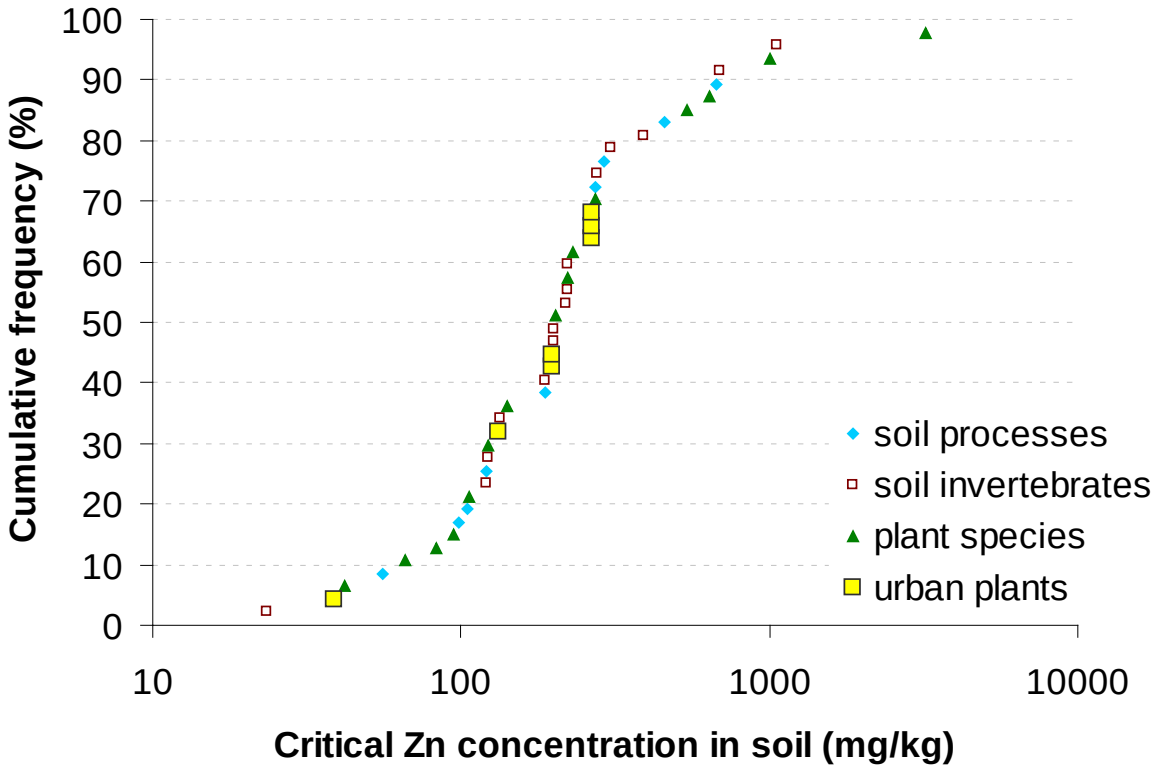


Figure 3. The species sensitivity distribution (plotted as a cumulative frequency against added zinc (Zn) concentration) for soil processes, soil invertebrates and plant species to Zn.

3.6 Calculation of soil quality guidelines for fresh zinc contamination

Soil quality guidelines were derived for fresh zinc contamination using three different sets of toxicity data: NOEC and EC_{10}; LOEC and EC^{30}; and EC^{50}. The methods by which they were calculated and the resulting ACL and SQG values are presented in the following sections.

3.6.1 Calculation of soil quality guidelines for fresh zinc contamination based on no observed effect concentration and 10% effect concentration toxicity data

3.6.1.1 Calculation of soil-specific added contaminant limits

The NOEC and EC_{10} toxicity data were normalised using the equations presented in Table 5 to the Australian reference soil (Table 6) and then the lowest geometric mean for each species/soil microbial process was entered into the BurrliOZ species sensitivity distribution (Campbell et al. 2000) method. The SSD generated a single numerical value (that is, the $ACL_{(NOEC\ \&\ EC10)}$ for each desired level of protection. These $ACL_{(NOEC\ \&\ EC10)}$ values only apply to the Australian reference soil.

The $ACL_{(NOEC\ \&\ EC10)}$ value for the Australian reference soil with an urban residential land/public open space use was approximately 100 mg/kg. These $ACL_{(NOEC\ \&\ EC10)}$ values for the reference soil were then used to calculate $ACL_{(NOEC\ \&\ EC10)}$ values for a range of soils (that is, soil-specific $ACL_{(NOEC\ \&\ EC10)}$) for each group of organisms using the same normalisation relationships as before but in the reverse manner. The following explains how the soil-specific $ACL_{(NOEC\ \&\ EC10)}$ values for soils with an urban residential /public open space land use were calculated as an example of how this was done for each of the land uses.

Soil-specific $ACL_{(NOEC\ \&\ EC10)}$ values for soil processes varied with soil pH and ranged from 20 to 330 mg/kg added Zn for soils with pHs between 4 and 7.5 (Table 7). The soil-specific $ACL_{(NOEC\ \&\ EC10)}$

values for invertebrates (Table 8) varied with cation exchange capacity (CEC), with values ranging from 60 to 420 mg/kg for soils with CEC values ranging from 5 to 60 $cmol_c$/kg. Soil-specific $ACL_{(NOEC\ \&\ EC10)}$ values for plants (Table 9) were pH- and CEC- specific and ranged from 20 to 910 mg/kg for soils with pHs between 4 and 7.5 and CEC values between 5 and 60 $cmol_c$/kg.

Table 7. Soil-specific ACL values for zinc (Zn) based on no observed effect concentration and 10% effect concentration toxicity data that should theoretically protect 80% of soil processes in soils with pH values ranging from 4.0 to 7.5.

Soil pH	Zn ACL (mg/kg) for soil processes
4.0	20
4.5	30
5.0	45
5.5	70
6.0	100
6.5	150
7.0	220
7.5	330

Table 8. Soil-specific ACL values for zinc (Zn) based on no observed effect concentration and 10% effect concentration toxicity data that should theoretically protect 80% of invertebrate species in soils with CEC ranging from 5 to 60 $cmol_c$/kg.

Cation exchange capacity ($cmol_c$/kg)	Zn ACL (mg/kg) for invertebrates
5	60
10	100
20	180
30	240
40	300
60	420

Table 9. Soil-specific ACL values for zinc (Zn) based on no observed effect concentration and 10% effect concentration toxicity data that should theoretically protect 80% of plant species in soils with pH values ranging from 4.0 to 7.5 and CEC values ranging from 5 to 60 $cmol_c$/kg.

pH	CEC ($cmol_c$/kg)					
	5	10	20	30	40	60
4.0	20	30	50	65	75	100
4.5	25	40	65	85	110	140
5.0	35	55	90	120	140	190
5.5	45	75	120	160	200	260
6.0	65	100	170	220	270	360
6.5	85	140	230	300	370	490
7.0	120	190	310	410	500	670

7.5	160	260	420	560	690	910

These soil-specific $ACL_{(NOEC \& EC10)}$ values for each organism group (presented in Tables 7 to 9) were then merged into a single set of soil-specific $ACL_{(NOEC \& EC10)}$ values—so that the lowest $ACL_{(NOEC \& EC10)}$ value for each combination of soil pH and CEC was adopted (Table 10). The $ACL_{(NOEC \& EC10)}$ values presented in Table 10 should protect at least 80% of soil processes, soil invertebrate and plant species and these ranged from 20 to 330 mg/kg in soils with pH values between 4 and 7.5 and CEC values between 5 and 60 $cmol_c/kg$. The $ACL_{(NOEC \& EC10)}$ values presented in Tables 7–9 are the ACLs for individual groups of organisms and should not be used as $ACL_{(NOEC \& EC10)}$ values.

Table 10. Soil-specific added contaminant limits based on no observed effect concentration and 10% effect concentration toxicity data ($ACL_{(NOEC \& EC10)}$, mg/kg) for zinc (Zn) that theoretically protect at least 80% of soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.0 to 7.5 and CEC values ranging from 5 to 60 cmolc/kg. These values may be used as $ACLs_{(NOEC \& EC10)}$ for Zn in freshly contaminated soils with an urban residential /public open space land use.

pH	CEC ($cmol_c/kg$)					
	5	10	20	30	40	60
4.0	20	20	20	20	20	20
4.5	25	30	30	30	30	30
5.0	35	45	45	45	45	45
5.5	45	70	70	70	70	70
6.0	60	100	100	100	100	100
6.5	60	100	150	150	150	150
7.0	60	100	180	220	220	220
7.5	60	100	180	240	300	330

The same methods as described above were used to generate the $ACL_{(NOEC \& EC10)}$ values for areas of ecological significance and commercial/industrial land uses. The $ACL_{(NOEC \& EC10)}$ values for these land uses are presented in Tables 11 and 12.

Table 11. Soil-specific added contaminant limits based on no observed effect concentration and 10% effect concentration toxicity data ($ACL_{(NOEC \& EC10)}$, mg/kg) for zinc (Zn) that theoretically protect at least 99% of soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.0 to 7.5 and CEC values ranging from 5 to 60 cmolc/kg. These **values may be used as $ACLs_{(NOEC \& EC10)}$ for Zn in freshly contaminated soils for areas of ecological significance.**

pH	CEC ($cmol_c/kg$)					
	5	10	20	30	40	60
4.0	4	5	5	5	5	5
4.5	6	8	8	8	8	8
5.0	8	10	10	10	10	10
5.5	10	15	15	15	15	15
6.0	15	25	25	25	25	25
6.5	15	25	35	35	35	35
7.0	15	25	45	55	55	55
7.5	15	25	45	60	75	80

Table 12. Soil-specific added contaminant limits based on no observed effect concentration and 10% effect concentration toxicity data ($ACL_{(NOEC\ \&\ EC10)}$, mg/kg) for zinc (Zn) that theoretically protect at least 60% of soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.0 to 7.5 and cation exchange capacity (CEC) values ranging from 5 to 60 $cmol_c$/kg. These values may be used as $ACLs_{(NOEC\ \&\ EC10)}$ for Zn in freshly contaminated soils with a commercial/industrial land use.

pH	CEC ($cmol_c$/kg)					
	5	**10**	**20**	**30**	**40**	**60**
4.0	30	35	35	35	35	35
4.5	40	50	50	50	50	50
5.0	55	75	75	75	75	75
5.5	75	110	110	110	110	110
6.0	95	160	160	160	160	160
6.5	95	160	240	240	240	240
7.0	95	160	280	350	350	350
7.5	95	160	280	390	480	520

3.6.1.2 Calculation of ambient background concentration values

To convert ACLs to SQGs, the ambient background concentration (ABC) needs to be added to the ACL. Three methods of determining the ABC were recommended in the methodology for deriving SQGs (Schedule B5b). The preferred method is to measure the ABC at an appropriate reference site. However, where this is not possible the methods of Olszowy et al. (1995) and Hamon et al. (2004) were recommended, depending on the situation.

For sites with no history of contamination the method of Hamon et al. (2004) was recommended to estimate the ABC. In this method, the ABC for Zn varies with the soil iron concentration (Table 13). Predicted ABC values for Zn range from 3 to 60 mg/kg in soils with iron concentrations between 0.1 and 20%.

Table 13. Zinc (Zn) ABC calculated using the Hamon et al. (2004) method.

Soil iron content (%)	Zn ABC (mg/kg)
0.1	3
1	10
10	40
20	60

For aged contaminated sites (i.e. the contamination has been in place for at least two years, see Schedule B5b) the methodology recommends using the 25th percentiles of the ABC data for the 'old suburbs' of Olszowy et al. (1995) (see Table 14). The ABC values for Zn in 'new suburbs' were similar to the values predicted by the Hamon et al. (2004) method. Therefore it is recommended that the Hamon et al. (2004) method be used to generate ABC values for new suburbs (that is, <2 years old) as soil-specific values will be generated, while for old suburbs with aged contamination (that is,

>2 years) it was recommended that the 25th percentile of the ABC data from old suburbs (Olszowy et al. 1995) be used.

Table 14. Zinc (Zn) ABC based on the 25th percentiles of Zn concentrations in 'old suburbs' (i.e. >2 years old) from various states of Australia (Olszowy et al. 1995).

Suburb type	25th percentile of Zn ABC values (mg/kg)			
	NSW	QLD	SA	VIC
New suburb, low traffic	25	15	25	15
New suburb, high traffic	45	30	30	20
Old suburb, low traffic	75	80	55	40
Old suburb, high traffic	120	160	90	55

3.6.1.3 Examples of soil quality guidelines for fresh zinc contamination based on no observed effect concentration and 10% effect concentration data

To calculate an $SQG_{(NOEC\ \&\ EC10)}$, the ABC value is added to the $ACL_{(NOEC\ \&\ EC10)}$. ABC values vary with soil type. Therefore, it is not possible to present a single set of $SQG_{(NOEC\ \&\ EC10)}$ values. Thus, two examples of $SQG_{(NOEC\ \&\ EC10)}$ values for urban contaminated soils are provided below. These examples would be at the low and high end of the range of SQGs values (but not the extreme values) generated for Australian soils.

Example 1	
Site descriptors – urban residential/public open space land use in a new suburb.	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with a 1% iron content.	
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:	
$ACL_{(NOEC\ \&\ EC10)}$:	45 mg/kg
ABC:	10 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$:	55 mg/kg

Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.	
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:	
$ACL_{(NOEC\ \&\ EC10)}$:	480 mg/kg[1]
ABC:	40 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$:	520 mg/kg

3.6.2 Calculation of soil quality guidelines based on protecting aquatic ecosystems from leaching of fresh zinc contamination

As indicated in the exposure pathway assessment, the log K_d values for Zn measured in a range of Australian soils were below 3 and therefore there is the potential in some soils for Zn to leach to groundwater and effect aquatic ecosystems. Although the calculation of SQGs based on protecting aquatic ecosystems from the effects of leached contaminants is not included in the EIL derivation

1 The soil-specific Zn ACLs for commercial/industrial land use are provided in Appendix B, Table 1.

methodology (Schedule B5b), the calculations are presented here to illustrate the recommended approach and what effect this has on the resulting SQGs. The following SQGs were based on the $ACL_{(NOEC\ \&\ EC10)}$ values for urban residential/public open space land use.

The soil-specific SQGs for Zn that accounted for leaching potential were calculated using the US EPA method (US EPA 1996).

$$SQG = C_w \,.\, (K_d + (\theta_w + \theta_a \,.\, H) / \rho_b) \,.\, DAF \qquad \textit{(equation 2)}$$

where SQG is the appropriate soil quality guideline in soil (mg/kg), C_w is the target soil leachate concentration (mg/L) (that is, the Australian and New Zealand freshwater quality guideline for Zn, (ANZECC and ARMCANZ 2000)), K_d is the soil–water partition coefficient (L/kg), θ_w is the water-filled soil porosity L_{water}/L_{soil}), θ_a is the air-filled soil porosity (L_{air}/L_{soil}), ρ_b is the dry soil bulk density (kg/L), H is the Henry's law constant (unitless), and DAF is the dilution and attenuation factor[2]. The values of DAF used in the calculations were 1 and 20. There is a linear relationship between the DAF and the SQGs, thus the SQGs calculated using a DAF of 20 are 20 times larger than those calculated using a DAF of 1.

The value for θ_w was set to 0.1 L_{water}/L_{soil}, θ_a was set to 0.1 L_{air}/L_{soil} and ρ_b was set to 1.3 kg/L. The calculated SQG values when DAF was 1 and 20 are presented in Tables 15 and 16 respectively.

Table 15. Soil-specific zinc (Zn) soil quality guidelines ($SQG_{(NOEC\ \&\ EC10)}$, mg total Zn/kg) based on protecting groundwater ecosystems from groundwater leaching when the dilution and attenuation factor (DAF) was 1.

pH	CEC ($cmol_c/kg$)					
	5	**10**	**20**	**30**	**40**	**60**
4	0.1	0.1	0.3	0.6	0.9	2
5	0.1	0.3	0.9	2	2	4
6	0.3	0.8	2	4	6	10
7	0.8	2	6	10	15	30
8	2	5	15	25	40	75

2 Soil pore water is the predominant source of groundwater. As the soil pore water leaches it passes through material that can bind the contaminants (attenuation), thus reducing their concentration. Also, in the majority of cases groundwater catchments will contain both contaminated and uncontaminated soils; pore water from the contaminated soil will be diluted by that from the uncontaminated (dilution). Therefore a a dilution and attenuation factor (DAF) is used to convert soil pore water concentrations to groundwater concentrations. The fraction of contaminated land to the total area of the groundwater/aquifer catchment can be used to calculate the DAF as indicated below:

DAF = 100 ÷ percentage of contaminated soil in catchment

Table 16. Soil-specific zinc (Zn) soil quality guidelines ($SQG_{(NOEC \& EC10)}$, mg total Zn/kg) based on protecting groundwater ecosystems from groundwater leaching when the dilution and attenuation factor (DAF) was 20.

pH	CEC ($cmol_c$/kg)					
	5	10	20	30	40	60
4	1	2	7	10	20	35
5	2	6	15	30	50	85
6	6	15	45	80	120	220
7	15	40	115	210	310	570
8	40	110	300	530	810	1500

3.6.3 Calculation of soil quality guidelines for fresh zinc contamination based on lowest observed effect concentration and 30% effect concentration toxicity data, and based on 50% effect concentration toxicity data

In addition to calculating $SQG_{(NOEC \& EC10)}$ values, two other sets of SQGs corresponding to two other levels of protection were generated. T hese were the $SQG_{(LOEC \& EC30),}$ which indicate concentrations above which moderate toxic effects would occur and the $SQG_{(EC50),}$ which indicate concentrations above which marked toxic effects would occur.

3.6.3.1 Calculation of soil-specific added contaminant limits

The Zn $SQG_{(LOEC \text{ and } EC30)}$ and $SQG_{(EC50)}$ and associated ACL values were calculated using the methodology, except the input data for the SSD was changed to the appropriate type (Table 1). This data is presented in Tables 2–4 and the raw data can be found in Appendix A. These measures of toxicity were not available in all instances, so, to maximise the data available to calculate $SQG_{(LOEC \text{ and } EC30)}$ and $SQG_{(EC50)}$ values, the available toxicity data was converted to these measures using conversion factors. The NBRP (cited in Heemsbergen et al. 2008) derived a set of conversion factors for Cu and Zn (Table 17). These experimentally-based conversion factors were used rather than the generic conversion factors presented in Heemsbergen et al. (2008), which is consistent with the approach recommended in the methodology for deriving SQGs. Table 18 shows the $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ values for the Australian reference soil (that is, a pH of 6 and a CEC of 10 $cmol_c$/kg) with areas of ecological significance, urban residential/public open space and commercial/industrial land uses. The set of soil-specific Zn $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ values for each land use are presented in Tables 19 and 20.

Table 17. Conversion factors used to convert various measures of toxicity for cations such as copper and zinc. The conversion factors were obtained from unpublished data from the Australian National Biosolids Research Program and were cited by Heemsbergen et al. (2008).

Data being converted	Conversion factor
NOEC or EC_{10} to EC_{50}	x 3
NOEC or EC_{10} to LOEC or EC^{30}	x 1.5
LOEC or EC_{30} to EC_{50}	x 2

Table 18. Zinc (Zn) added contaminant levels based on lowest observed effect concentration and 30% effect concentration data ($ACL_{(LOEC\ \&\ EC30)}$), and based on 50% effect concentration data ($ACL_{(EC50)}$) for the Australian reference soil with various land uses.

Land use	$ACL_{(LOEC\ \&\ EC30)}$ values (mg/kg added Zn)	$ACL_{(EC50)}$ values (mg/kg added Zn)
Areas of ecological significance	40	80
Urban residential/public open space	160	290
Commercial/industrial	250	450

Table 19. Soil-specific added contaminant limits based on lowest observed effect concentration and 30% effect concentration toxicity data ($ACL_{(LOEC\ \&\ EC30)}$, mg/kg) for fresh zinc (Zn) that should theoretically provide the appropriate level of protection (that is, 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.0 to 7.5 and CEC values ranging from 5 to 60 $cmol_c/kg$. These are the recommended $ACL_{(LOEC\ \&\ EC30)}$ values in freshly contaminated soils with each land use.

Areas of ecological significance						
	CEC ($cmol_c/kg$)					
pH	**5**	**10**	**20**	**30**	**40**	**60**
4.0	7	8	8	8	8	8
4.5	10	10	10	10	10	10
5.0	15	20	20	20	20	20
5.5	20	25	25	25	25	25
6.0	25	40	40	40	40	40
6.5	25	40	60	60	60	60
7.0	25	40	70	90	90	90
7.5	25	40	70	95	120	130
Urban residential/public open space land use						
	CEC ($cmol_c/kg$)					
pH	**5**	**10**	**20**	**30**	**40**	**60**
4.0	25	30	30	30	30	30
4.5	35	50	50	50	50	50
5.0	50	70	70	70	70	70
5.5	70	100	100	100	100	100
6.0	90	150	150	150	150	150
6.5	90	150	230	230	230	230
7.0	90	150	270	340	340	340
7.5	90	150	270	370	460	500

Commercial/industrial land use						
	CEC ($cmol_c/kg$)					
pH	5	10	20	30	40	60
4.0	45	50	50	50	50	50
4.5	60	75	75	75	75	75
5.0	80	110	110	110	110	110
5.5	110	170	170	170	170	170
6.0	140	250	250	250	250	250
6.5	140	250	360	360	360	360
7.0	140	250	420	540	540	540
7.5	140	250	420	590	730	800

Table 20. Soil-specific added contaminant limits based on 50% effect concentration toxicity data ($ACL_{(EC50)}$, mg/kg) for fresh zinc (Zn) that should theoretically provide the appropriate level of protection (that is, 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.0 to 7.5 and cation exchange capacity (CEC) values ranging from 5 to 60 $cmol_c/kg$. These are the recommended $ACL_{(EC50)}$ for Zn in freshly contaminated soils with each land use.

Areas of ecological significance						
	CEC ($cmol_c/kg$)					
pH	5	10	20	30	40	60
4.0	15	15	15	15	15	15
4.5	20	25	25	25	25	25
5.0	25	35	35	35	35	35
5.5	35	55	55	55	55	55
6.0	45	80	80	80	80	80
6.5	45	80	110	110	110	110
7.0	45	80	130	170	170	170
7.5	45	80	130	190	230	250
Urban residential/public open space land use						
	CEC ($cmol_c/kg$)					
pH	5	10	20	30	40	60
4.0	50	60	60	60	60	60
4.5	70	90	90	90	90	90
5.0	95	130	130	130	130	130
5.5	130	200	200	200	200	200
6.0	170	290	290	290	290	290
6.5	170	290	430	430	430	430
7.0	170	290	500	640	640	640
7.5	170	290	500	690	870	940

Commercial/industrial land use						
	CEC ($cmol_c/kg$)					
pH	5	10	20	30	40	60
4.0	80	95	95	95	95	95
4.5	100	150	150	150	150	150
5.0	150	200	200	200	200	200
5.5	200	300	300	300	300	300
6.0	250	450	450	450	450	450
6.5	259	450	650	650	650	650
7.0	259	450	750	1000	1000	1000
7.5	259	450	750	1100	1300	1400

3.6.3.2 Calculation of ambient background concentration values

The ABC values for freshly contaminated soils were calculated using the method set out in this Schedule and presented in Table 13.

3.6.3.3 Examples of soil quality guidelines for fresh zinc contamination based on lowest observed effect concentration and 30% effect concentration data, and based on 50% effect data

In order to calculate the $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values the soil-specific ABC has to be added to the $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ values, respectively. Therefore, the $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values will always be at least as large as those presented in Tables 19 and 20. Examples of the $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values are provided below.

$SQG_{(LOEC\ \&\ EC30)}$ – Example 1

Site descriptors – urban residential/public open space land use in a new suburb.

Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with a 1% iron content.

The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:

$ACL_{(LOEC\ \&\ EC30)}$	70	mg/kg
ABC	10	mg/kg
$SQG_{(LOEC\ \&\ EC30)}$	80	mg/kg

$SQG_{(LOEC\ \&\ EC30)}$ – Example 2

Site descriptors – commercial/industrial land use in a new suburb.

Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.

The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:

$ACL_{(LOEC\ \&\ EC30)}$	730	mg/kg
ABC	40	mg/kg
$SQG_{(LOEC\ \&\ EC30)}$	770	mg/kg

$SQG_{(EC50)}$ — Example 3		
Site descriptors – urban residential/public open space land use in a new suburb.		
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with a 1% iron content.		
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:		
$ACL_{(EC50)}$	130	mg/kg
ABC	10	mg/kg
$SQG_{(EC50)}$	140	mg/kg

$SQG_{(EC50)}$ — Example 4		
Site descriptors – commercial/industrial land use in a new suburb.		
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.		
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:		
$ACL_{(EC50)}$	1300	mg/kg
ABC	40	mg/kg
$SQG_{(EC50)}$	1340	mg/kg

3.7 Calculation of soil quality guidelines for aged zinc contamination

3.7.1 Calculation of an ageing and leaching factor for zinc

In addition to calculating SQGs in recently contaminated soils (that is, contamination is <2 years old), an equivalent set of levels was derived for soils where the contamination is aged (that is, it has been present for ≥2 years). The Zn $SQG_{(NOEC\ \&\ EC10)}$, $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ for aged sites were calculated using the methods set out in Schedule B5b and this Schedule, the only difference being that laboratory toxicity data based on freshly spiked soils or soils that had not been leached were multiplied by an ageing/leaching factor. A factor (3 for Zn) was developed by Smolders et al. (2009) that accounted for ageing and leaching of various metals. This ageing and leaching factor (ALF) has been incorporated into the methodology to derive the Flemish soil quality guidelines (VLAREBO 2008). Therefore, the raw toxicity data (Appendix A) for Zn that was generated using freshly spiked and non-leached soils was multiplied by this conversion factor and the geometric means for each species and soil process recalculated (Tables 21–23). It should be noted that the values in Tables 21–23 are not simply the data from Tables 2–4 multiplied by 3, as the correction factor was not applied to all the data (for example, data from the field-based NBRP was not adjusted).

3.7.2 Calculation of soil quality guidelines for aged zinc contamination based on no observed effect concentration and 10% effect concentration toxicity data

3.7.2.1 Calculation of added contaminant limits for aged zinc contamination based on no observed effect concentration and 10% effect concentration toxicity data

The lowest geometric mean of the age-corrected toxicity data for each species/soil microbial process that was used to derive the aged $ACL_{(NOEC\ \&\ EC10)}$ values is presented in Table 21 for soil processes, Table 22 for soil invertebrate species and Table 23 for plant species. The conversion of the fresh toxicity data to account for ageing/leaching and the resulting toxicity values are presented in Appendix A.

Table 21. The geometric mean values of the aged and age-corrected zinc (Zn) toxicity data (expressed in terms of added Zn) for soil processes.

Soil process	Geometric means (mg/kg added Zn)		
	EC_{10} or NOEC	EC_{30} or LOEC	EC_{50}
Acetate decomposition	561	841	1681
Amidase	363	545	1091
Ammonification	295	443	885
Arylsulphatase	868	1303	2605
Glucose decomposition	274	1169	2904
Nitrate reductase	168	252	504
Nitrification	455	706	930
Phosphatase	2022	3033	6066
Respiration	313	470	940

Table 22. The geometric mean values of the aged and age-corrected zinc (Zn) toxicity data (expressed in terms of added Zn) for soil invertebrate species.

Invertebrate species		Geometric means (mg/kg added Zn)		
Common name	Scientific name	EC_{10} or NOEC	EC_{30} or LOEC	EC_{50}
Earthworm	*A. caliginosa*	669	823	1172
Earthworm	*A. rosea*	1172	1221	1308
Earthworm	*E. fetida*	602	888	1726
Earthworm	*L. rubellus*	659	855	1328
Earthworm	*L. terrestris*	3187	3771	5026
Nematode	*Acrobeloides sp.*	663	995	1989
Nematode	*C. elegans*	366	550	1099
Nematode	*C. elegans* (dauer larval stage)	2068	3103	6205
Nematode	Community nematodes	919	1378	2756
Nematode	*Eucephalobus sp.*	404	605	1210
Nematode	*Plectus sp.*	70	105	210
Nematode	*Rhabditidae sp.*	597	896	1791
Potworm	*E. albidus*	363	544	1088
Potworm	*E. crypticus*	828	1241	2483
Springtail	*F. candida*	566	848	1696

Table 23. The geometric mean values of the aged and age-corrected zinc (Zn) toxicity data (expressed in terms of added Zn) for plant species.

Species	Scientific name	Geometric means (mg/kg added Zn)		
		EC_{10} or NOEC	EC_{30} or LOEC	EC_{50}
Alfalfa	*M. sativa*	595	892	1784
Barley	*H. vulgare*	110	306	652
Beet	*B.vulgaris*	595	892	1784
Black or white lentil	*V. mungo*	284	426	852
Canola	*B. napus*	230	328	409
Common vetch	*V. sativa*	127	190	380
Cotton	*Gossypium sp.*	272	288	293
Fenugreek	*T. foenum graecum*	318	477	953
Lettuce	*L. sativa*	793	1189	2379
Maize	*Z. mays*	460	694	1324
Millet	*P. milaceum*	540	1580	2026
Oats	*A. sativa*	667	1000	2000
Onion	*A. cepa*	198	297	594
Pea	*P. sativum*	793	1189	2379
Peanuts	*A. hypogaea*	140	224	280
Red clover	*T. pratense*	117	176	351
Sorghum	*Sorghum sp.*	256	528	924
Spinach	*S. oleracea*	396	595	1189
Sugar cane	*Sacharum*	3220	4830	9661
Tomato	*L. esculentum*	793	1189	2379
Triticale	*Tritosecale sp.*	998	1364	1658
Wheat	*T. aestivum*	640	928	1172

For each urban residential/public open space land use, soil-specific $ACL_{(NOEC\ \&\ EC10)}$ values were derived separately for soil processes, soil invertebrate species and plant species (data not shown). Within each land use type, the soil-specific $ACL_{(NOEC\ \&\ EC10)}$ values for each organism group were then merged so that the lowest $ACL_{(NOEC\ \&\ EC10)}$ value for each combination of soil pH and CEC was adopted (Table 24). These should theoretically protect 99%, 80% and 60% of all soil processes, soil invertebrate species and plant species that are exposed to aged Zn contamination in soils that are in an area of ecological significance, or have an urban residential/public open space, commercial/industrial land use, respectively.

Table 24. Soil-specific added contaminant limits based on no observed effect concentration and 10% effect concentration toxicity data ($ACL_{(NOEC\ \&\ EC10)}$, mg/kg) for aged zinc (Zn) contamination that should theoretically provide the appropriate level of protection (i.e. 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.0 to 7.5 and CEC values ranging from 5 to 60 $cmol_c$/kg. These are the recommended $ACL_{(NOEC\ \&\ EC10)}$ values for Zn in aged contaminated soils with each land use.

Areas of ecological significance						
	CEC ($cmol_c$/kg)					
pH	5	10	20	30	40	60
4.0	10	10	10	10	10	10
4.5	15	20	20	20	20	20
5.0	20	25	25	25	25	25
5.5	25	40	40	40	40	40
6.0	35	55	55	55	55	55
6.5	35	55	85	85	85	85
7.0	35	55	100	125	125	125
7.5	35	55	100	130	170	180
Urban residential/public open space land use						
	CEC ($cmol_c$/kg)					
pH	5	10	20	30	40	60
4.0	45	55	55	55	55	55
4.5	60	80	80	80	80	80
5.0	85	110	110	110	110	110
5.5	110	170	170	170	170	170
6.0	150	250	250	250	250	250
6.5	150	250	370	370	370	370
7.0	150	250	440	550	550	550
7.5	150	250	440	600	750	800
Commercial/industrial land use						
	CEC ($cmol_c$/kg)					
pH	5	10	20	30	40	60
4.0	70	85	85	85	85	85
4.5	100	120	120	120	120	120
5.0	125	180	180	180	180	180
5.5	180	270	270	270	270	270
6.0	230	400	400	400	400	400
6.5	230	400	590	590	590	590
7.0	230	400	690	870	870	870
7.5	230	400	690	940	1200	1300

3.7.2.2 Calculation of ambient background concentration values

The ABC values for aged Zn contamination used to calculate aged $SQG_{(LOEC\ and\ EC30)}$ and $SQG_{(EC50)}$ values were obtained from Olszowy et al. (1995) and are presented in Table 14.

3.7.2.3 Examples of soil quality guidelines for Australian soils with aged zinc contamination based on no observed effect concentration and 10% effect concentration data

SQGs are the sum of the ABC and ACL values, both of which are soil-specific. It is, therefore, not possible to present a single set of aged SQGs. Thus, some examples of aged SQGs for aged urban contaminated soils are provided below. The presented examples represent SQGs that would be at the low and high end of the range of SQGs that would be generated for Australian soils, but are not extreme values.

Example 1		
Site descriptors – urban residential/public open space land use in an old NSW suburb with low traffic volume.		
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron and aged Zn contamination.		
The resulting $ACL_{(NOEC \& EC10)}$, ABC and $SQG_{(NOEC \& EC10)}$ values are:		
$ACL_{(NOEC \& EC10)}$	110	mg/kg
ABC	75	mg/kg
$SQG_{(NOEC \& EC10)}$	185	mg/kg, which would be rounded off to 180 mg/kg.

Example 2		
Site descriptors – commercial/industrial land use in an old Queensland suburb with a high traffic volume.		
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron and aged Zn contamination.		
The resulting $ACL_{(NOEC \& EC10)}$, ABC and $SQG_{(NOEC \& EC10)}$ values are:		
$ACL_{(NOEC \& EC10)}$	1200	mg/kg
ABC	160	mg/kg
$SQG_{(NOEC \& EC10)}$	1360	mg/kg, which would be rounded off to 1400 mg/kg.

3.7.3 Calculation of soil quality guidelines for aged zinc contamination based on lowest observed effect concentration and 30% effect concentration toxicity data and based on 50% effect concentration toxicity data

3.7.3.1 Calculation of added contaminant limits for aged zinc contamination based on lowest observed effect concentration and 30% effect concentration and based on 50% effect concentration toxicity data

The Zn $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values for aged sites were calculated using the method described in this Schedule with the exception that aged or age-corrected Zn toxicity data was used (Tables 21–23). Table 25 presents the $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ values for the Australian reference soil (Table 6) for areas of ecological significance, urban residential/public open space, and commercial/industrial land uses.

The soil-specific $ACL_{(LOEC \text{ and } EC30)}$ and $ACL_{(EC50)}$ values for aged Zn contamination and the various land uses are presented in Tables 26 and 27 respectively. As with the $ACL_{(NOEC \& EC10)}$ values for aged Zn contamination, the $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ values must have the soil-specific ABC added. Therefore, the $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values will be larger than the corresponding ACL values presented in Tables 26 and 27, respectively. Examples of the $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values are provided below.

Table 25. Zinc (Zn) ACLs for the Australian reference soil (pH = 6, CEC = 10 cmolc/kg) based on lowest observed effect concentration and 30% effect concentration toxicity data, and based on 50% effect concentration toxicity data.

Land use	$ACL_{(LOEC \& EC30)}$ values (mg/kg added Zn)	$ACL_{(EC50)}$ values (mg/kg added Zn)
Areas of ecological significance	90	140
Urban residential/public open space	400	700
Commercial/industrial	630	1100

Table 26. Soil-specific added contaminant limits based on lowest observed effect concentration and 30% effect concentration toxicity data ($ACL_{(LOEC \& EC30)}$, mg/kg) for aged zinc (Zn) contamination that should theoretically provide the appropriate level of protection (i.e. 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.0 to 7.5 and CEC values ranging from 5 to 60 $cmol_c$/kg. These are the recommended $ACL_{(LOEC \& EC30)}$ values for Zn in aged contaminated soils with each land use.

Areas of ecological significance						
	CEC ($cmol_c$/kg)					
pH	**5**	**10**	**20**	**30**	**40**	**60**
4.0	15	20	20	20	20	20
4.5	20	25	25	25	25	25
5.0	30	40	40	40	40	40
5.5	40	60	60	60	60	60
6.0	50	90	90	90	90	90
6.5	50	90	130	130	130	130
7.0	50	90	150	190	190	190
7.5	50	90	150	210	260	280
Urban residential/public open space land use						
	CEC ($cmol_c$/kg)					
pH	**5**	**10**	**20**	**30**	**40**	**60**
4.0	70	85	85	85	85	85
4.5	100	120	120	120	120	120
5.0	130	180	180	180	180	180
5.5	180	270	270	270	270	270
6.0	230	400	400	400	400	400
6.5	230	400	590	590	590	590
7.0	230	400	700	880	880	880
7.5	230	400	700	960	1200	1300

Commercial/industrial land use						
	CEC ($cmol_c/kg$)					
pH	5	10	20	30	40	60
4.0	110	130	130	130	130	130
4.5	150	190	190	190	190	190
5.0	210	290	290	290	290	290
5.5	280	420	420	420	420	420
6.0	360	620	620	620	620	620
6.5	360	620	920	920	920	920
7.0	360	620	1100	1400	1400	1400
7.5	360	620	1100	1500	1900	2000

Table 27. Soil-specific added contaminant limits based on 50% effect concentration toxicity data ($ACL_{(EC50)}$, mg/kg) for aged zinc (Zn) contamination that should theoretically provide the appropriate level of protection (i.e. 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.0 to 7.5 and cation exchange capacity (CEC) values ranging from 5 to 60 $cmol_c/kg$. These are the recommended $ACL_{(EC50)}$ values for Zn in aged contaminated soils with each land use.

Areas of ecological significance						
	CEC ($cmol_c/kg$)					
pH	5	10	20	30	40	60
4.0	25	30	30	30	30	30
4.5	35	45	45	45	45	45
5.0	45	65	65	65	65	65
5.5	65	95	95	95	95	95
6.0	85	140	140	140	140	140
6.5	85	140	210	210	210	210
7.0	85	140	250	310	310	310
7.5	85	140	250	340	430	460
Urban residential/public open space land use						
	CEC ($cmol_c/kg$)					
pH	5	10	20	30	40	60
4.0	130	150	150	150	150	150
4.5	170	220	220	220	220	220
5.0	230	330	330	330	330	330
5.5	320	480	480	480	480	480
6.0	410	710	710	710	710	710
6.5	410	710	1100	1100	1100	1100
7.0	410	710	1200	1600	1600	1600
7.5	410	710	1200	1700	2100	2300

Commercial/industrial land use						
	CEC ($cmol_c/kg$)					
pH	5	10	20	30	40	60
4.0	200	230	230	230	230	230
4.5	270	350	350	350	350	350
5.0	370	510	510	510	510	510
5.5	510	760	760	760	760	760
6.0	650	1100	1100	1100	1100	1100
6.5	650	1100	1700	1700	1700	1700
7.0	650	1100	1900	2500	2500	2500
7.5	650	1100	1900	2700	3400	3600

3.7.3.2 Calculation of ambient background concentrations

The ABC values used for aged Zn contamination are presented in Table 14.

3.7.3.3 Examples of soil quality guidelines for Australian soils with aged zinc contamination based on lowest observed effect concentration and 30% effect concentration data, and based on 50% effect concentration toxicity data

Both the ACL and ABC values for aged zinc contamination are soil-specific therefore a single set of SQGs could not be presented. Thus, examples from the low and high portions of the range of SQG(LOEC & EC30) and SQG(EC50) are presented below.

$SQG_{(LOEC\ \&\ EC30)}$ — Example 1

Site descriptors – urban residential/public open space land use in an old NSW suburb with low traffic volume.

Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.

The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:

$ACL_{(LOEC\ \&\ EC30)}$	180	mg/kg
ABC	75	mg/kg
$SQG_{(LOEC\ \&\ EC30)}$	255	mg/kg

This $SQG_{(LOEC\ \&\ EC30)}$ would then be rounded off using the rules in section 2.1 to a value of 250 mg/kg.

$SQG_{(LOEC\ \&\ EC30)}$ — Example 2

Site descriptors – commercial/industrial land use in an old Victorian suburb with high traffic volume.

Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.

The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:

$ACL_{(LOEC\ \&\ EC30)}$	1900	mg/kg
ABC	55	mg/kg
$SQG_{(LOEC\ \&\ EC30)}$	1955	mg/kg

This $SQG_{(LOEC\ \&\ EC30)}$ would then be rounded off using the rules in section 2.1 to a value of 2000 mg/kg.

$SQG_{(EC50)}$ — Example 3

Site descriptors – urban residential/public open space land use in an old NSW suburb with low traffic

volume.

Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.

The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:

$ACL_{(EC50)}$	330	mg/kg
ABC	75	mg/kg
$SQG_{(EC50)}$	405	mg/kg

This $SQG_{(EC50)}$ would then be rounded off using the rules in section 2.1 to a value of 400 mg/kg.

$SQG_{(EC50)}$ – Example 4

Site descriptors – commercial/industrial land use in an old Victorian suburb with high traffic volume.

Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.

The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:

$ACL_{(EC50)}$	3400	mg/kg
ABC	55	mg/kg
$SQG_{(EC50)}$	3455	mg/kg

This $SQG_{(EC50)}$ would then be rounded off using the rules in section 2.1 to a value of 3500 mg/kg.

3.8 Reliability of the zinc soil quality guidelines

Based on the criteria established in the methodology for SQG derivation (Schedule B5b), the Zn SQGs were considered to be of high reliability. This occurred as the toxicity data set easily met the minimum data requirements to use the SSD method and normalisation relationships were available to account for soil characteristics.

3.9 Comparison with other guidelines

A compilation of SQGs for Zn from a number of jurisdictions is presented in Table 28. These SQGs have a variety of purposes and levels of protection and therefore comparison of the SQGs between each other and with the Zn SQGs is problematic. The guidelines for Zn range from 20 mg/kg (added Zn) for the Netherlands to 200 mg/kg (total Zn) for Canada. The superseded interim urban EIL (NEPC 1999) was 200 mg/kg total Zn and therefore at the top of the range of the international Zn guidelines.

The Zn $ACL_{(NOEC\ \&\ EC10)}$ values in freshly contaminated urban residential/public open space soils ranged from 20–330 mg/kg (added Zn) (Table 10). The corresponding values for urban residential/public open space soils with aged Zn contamination ranged from 45–810 mg/kg (Table 24). The lowest ACLs (for sandy acidic soils) were very similar to the lowest of the international SQGs, but considerably lower than the superseded interim urban EIL. However, the largest ACLs (for neutral to alkaline, high CEC soils) were considerably larger than any of the international SQGs apart from the Dutch intervention level, which has a different purpose from the ACLs. Thus, in soils where the Zn has a low bioavailability, higher concentrations of Zn are permitted under the methodology than under the superseded interim urban EIL.

The intervention value in the Netherlands is 720 mg/kg total Zn. The range of $ACL_{(EC50)}$ values (which most closely relate to the Dutch intervention value) in freshly contaminated urban residential/public open space soils was 50–940 mg/kg (Table 20). While the range for aged Zn contamination was 125–2,300 mg/kg (Table 27), the Dutch value corresponds to the 60^{th} and 25^{th} percentile of the range of $ACL_{(EC50)}$ values for fresh and aged Zn contamination respectively. Therefore, depending on soil physicochemical properties, the $ACL_{(EC50)}$ values would permit considerably less (in high bioavailability soils) to considerably more (in low bioavailability soils) Zn than in the Netherlands.

Table 28. Soil quality guidelines for zinc (Zn) from international jurisdictions.

Name of zinc limit	Numerical value of the limit (mg/kg)
Dutch intervention level[1]	720 (added Zn)
Dutch maximum permissible addition[1]	20 (added Zn)
Canadian SQG (residential)[2]	200 (total Zn)
Eco-SSL plants[3]	160 (total Zn)
Eco-SSL soil invertebrates[3]	120 (total Zn)
Eco-SSL avian[3]	46 (total Zn)
Eco-SSL mammalian[3]	79 (total Zn)
EU soil guidelines using negligible risk[4]	67–150 (total Zn)

1 = VROM, 2000

2 = CCME, 1999a and 2006 and http://www.ccme.ca/publications/list_publications.html#link2

3 = http://www.epa.gov/ecotox/ecossl/

4 = Carlon, 2007

4 Arsenic

4.1 Arsenic compounds considered

The metalloid As occurs in a number of oxidation states: -3 (-III), 0, +3 (III) and +5 (V). Arsenic (III) is the dominant form under reducing conditions and As (V) is the dominant form in oxidised soils. The SQG derivation methodology (Schedule B5b) is only suitable for the aerobic portion of soils. SQGs for As were therefore calculated using only well oxidised soil studies. Therefore, arsenic will predominantly be present as As (V) but, as all the toxicity studies expressed toxicity in terms of total arsenic, the SQGs generated in this study are for total arsenic. For waterlogged soils, a separate As SQG should be derived, due to the difference between As (III) and As (V) in both toxicity and bioavailability in these soils. The chemical abstract service number (a unique identification number for each chemical) for As is 7440-38-2.

4.2 Exposure pathway assessment

The two key considerations in determining the most important exposure pathways for inorganic contaminants such as As are whether they biomagnify and whether they have the potential to leach to groundwater. A surrogate measure of the potential for a contaminant to leach is its water–soil partition coefficient (K_d). If the logarithm of the K_d ($\log K_d$) of an inorganic contaminant is less than 3 then it is considered to have the potential to leach to groundwater (Schedule B5b). The $\log K_d$ reported by Crommentuijn et al. (2000) was 2.28 L/kg, so As has the potential in some soils to leach to groundwater. This is consistent with information regarding human health problems experienced in Bangladesh from the presence of As in groundwater. The methodology for EIL derivation (Schedule B5b) does not advocate the routine derivation of EILs that account for leaching potential. Rather, it advocates that this is done on a site-specific basis as appropriate. However, the calculations are presented here to illustrate the recommended approach and the effect that this would have on the resulting SQGs.

Arsenic is not known to biomagnify in oxidised soils (Heemsbergen et al. 2009) and therefore only direct toxicity routes of exposure were considered in deriving the SQGs.

4.3 Toxicity data

The raw toxicity data for As is presented in Appendix B. The toxicity data (geometric means for each species) used to calculate the SQGs is presented in Table 29. There was toxicity data for three soil invertebrate species, five terrestrial animal species and 13 species of plants. These meet the minimum data requirements recommended by Heemsbergen et al. (2008) to use the BurrliOZ SSD method (Campbell et al. 2000).

Table 29. Geometric mean values of arsenic (As) toxicity data (expressed in terms of total As) for soil invertebrate species, terrestrial bird and mammal species and plant species.

Test species		Geometric mean (mg/kg)		
Common name	**Scientific name**	**EC_{10} or NOEC**	**EC_{30} or LOEC**	**EC_{50}**
Bean	*Phaseolus vulgaris*	22.6	84	168
Blueberry	*Vaccinium sp.*	22.2	55	111
Common rat	*Rattus norvegicus*	10.0	25	50
Corn	*Z. mays*	25.1	67	123
Cotton	*Gossypium sp.*	20.8	52	104
Deer mouse	*Peromyscus maniculatus*	320	1600	1600
Earthworm	*Eisenia fetida*	20.0	100	100

Earthworm	*L. rubellus*	76.1	381	381
Earthworm	*L. terrestris*	100	250	500
Fulvous whistling duck	*Dendrocygna bicolour*	229	1145	1145
Grass		13.4	81	161
Northern bobwhite	*Colinus virginianus*	54.0	270	270
Oat	*A. sativa*	22.7	44	70
Pea	*Pisum sativum*	20.8	52	104
Pine		292	731	1462
Potato	*Solanum tuberosum*	36.3	108	181
Radish	*Raphanus sativa*	67.7	169	339
Sheep	*Ovis aries*	25.0	63	125
Soyabean	*Glycine max*	9.7	24	35
Tomato	*L. esculentum*	62.5	166	263
Wheat	*T. aestivum*	43.4	153	307

In order to maximise the use of the available toxicity data, conversion factors (adopted from the *Australian and New Zealand guidelines for fresh and marine water quality* (ANZECC & ARMCANZ 2000) by Heemsbergen et al. (2008)) were used to permit the inter-conversion of NOEC, LOEC, EC_{50}, EC_{30} and EC_{10} data. Conversion factors for cations (for example, Cu and Zn) were developed by the NBRP and recommended by Heemsbergen et al. (2008) in preference to the default conversion factors adopted from the WQGs. However, as As is predominantly found in anionic form in soils, the default conversion factors were used (Table 30).

Table 30. The default conversion factors used to convert different measures of toxicity to chronic no observed effect concentrations (NOECs) or 10% effect concentrations (EC_{10}). Sourced from Heemsbergen et al. (2008), who adopted the values from the Australian and New Zealand guidelines for fresh and marine water quality (ANZECC & ARMCANZ 2000).

Toxicity data[a]	Conversion factor
EC_{50} to NOEC or EC_{10}	5
LOEC or EC_{30} to NOEC or EC_{10}	2.5
MATC* to NOEC or EC_{10}	2

[a] EC_{50} is the concentration that causes a 50% effect, EC_{30} is the concentration that causes a 30% effect, EC_{10} is the concentration that causes a 10% effect, NOEC = no observed effect concentration, LOEC = lowest observed effect concentration, *MATC = the maximum acceptable toxicant concentration and is the geometric mean of the NOEC and LOEC.

4.4 Normalisation relationships

It is well known that soil physicochemical properties affect the toxicity and bioavailabiity of As. However, this knowledge is qualitative. For example, Sheppard (1992) reviewed the existing literature and concluded that the toxicity of As was five times more toxic in sands and loams than in clay soils. There is only one set of published normalisation relationships for As toxicity (Song et al. 2006). This relates the toxicity of As (i.e. barley root elongation) expressed in terms of total added As, ammonium sulphate [$(NH_4)_2SO_4$]-extractable As or ammonium phosphate ($NH_4H_2PO_4$)-extractable As to soil properties such as oxalate-extractable Mn and oxalate-extractable Fe concentrations. The

normalisation relationships for EC_{10} and EC_{50} toxicity data expressed in terms of total added As (from Song et al. 2006) are:

EC_{10} = 0.1 (oxalate-extractable Mn) + 1.03 (% clay) – 9.25 *(equation 3)*

(r^2 adj = 0.89, p = <0.001, n = 16)

EC_{50} = 0.21 (oxalate-extractable Mn) + 0.016 (oxalate-extractable Fe)
+ 4.29 (% clay) – 48.2 *(equation 4)*

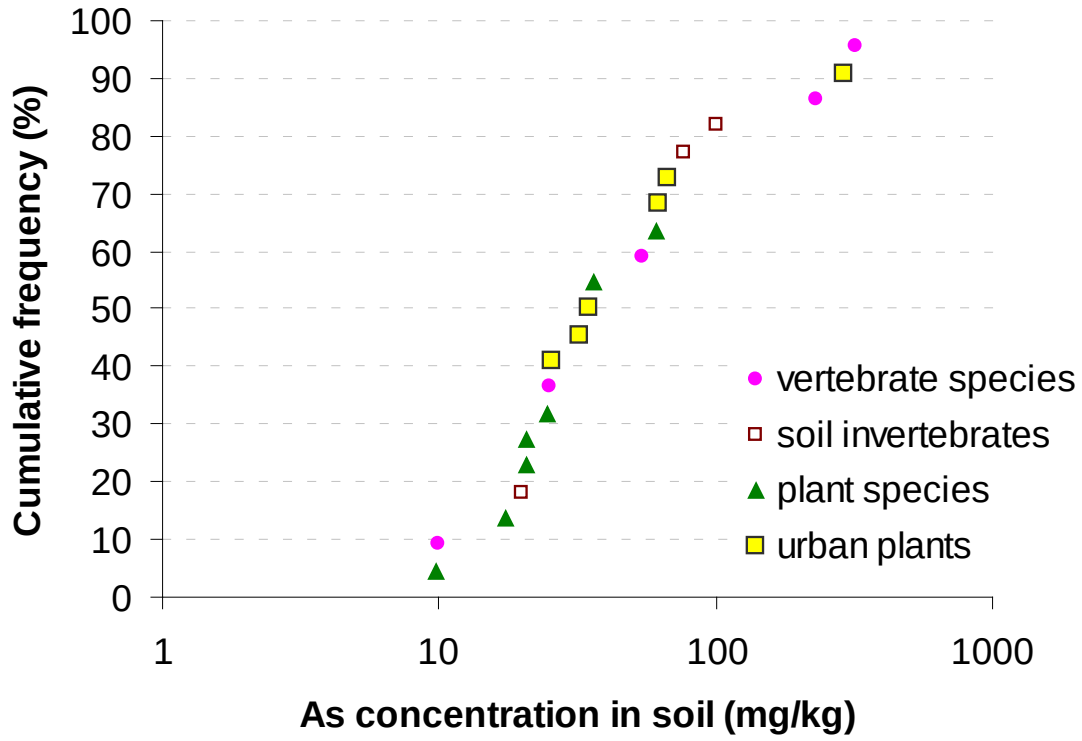
(r^2 adj = 0.91, p = <0.001, n = 16)

However, with the exception of the Song et al. (2006) data, none of the available As toxicity studies had expressed the toxicity in the units of the normalisation relationships nor had the studies measured the soil properties used in the normalisation relationships. Therefore, the normalisation relationships could not be used.

4.5 Sensitivity of organisms to arsenic

Figure 4 shows the SSD (that is, the cumulative distribution of the geometric means of species sensitivities to As) for all species for which As toxicity data was available. The distribution of the major groups of organisms along the SSD is uniform—thus all of the organism groups have a smilar sensitivity to As.

Figure 4. The species sensitivity distribution (plotted as a cumulative frequency against total arsenic (As) concentration) of As for soil invertebrate species, terrestrial vertebrate species and plant species.



4.6 Calculation of soil quality guidelines for fresh arsenic contamination

The As toxicity data could not be normalised to the Australian reference soil because none of the publications had reported the properties required by the one normalisation relationship available for As. Thus, soil-specific ACLs could not be derived. Rather, a single generic ACL for each land use was

derived. These generic ACLs would apply to all Australian soils of the appropriate land use. For example, the single ACL for urban residential /public open space land use would apply to all Australian urban residential/public open space soils.

4.6.1 Calculation of soil quality guidelines for fresh arsenic contamination based on no observed effect concentration and 10% effect concentration toxicity data

All the available As toxicity data (apart from that of Song et al. 2006) were reported as total concentrations without making a distinction between added and background concentrations. The Hamon et al. (2004) method can predict the ABC for As in Australian soils. For European soils or toxicity studies, the Dutch background standardisation equation for As can be used (Lexmond et al. 1986):

As= 0.4*(clay content + organic matter content) *(equation 5)*

However, the As toxicity studies did not report the Fe and Mn contents (required by the Hamon et al., 2004 method) or the organic matter or clay content (required by the Lexmond et al. 1986 method) of the soils in which the toxicity was determined. Therefore, it was not possible to estimate the ABC nor express toxicity in terms of added concentrations. As a result, no ACL values could be calculated.

The situation for As was that:

- there were sufficient toxicity data to use the BurrliOZ software
- the data could not be normalised to the Australian reference soil
- the toxicity data could not be expressed in terms of added concentrations
- a background concentration for As could not be calculated.

Therefore, only a single numerical value was generated by the BurrliOZ SSD method for each of the three land uses (that is, areas of ecological significance, urban residential/public open space, and commercial/industrial).

The output was the $SQG_{(NOEC \& EC10)}$ for that particular land use and no soil-specific $SQG_{(NOEC \& EC10)}$ values could be calculated. The As $SQG_{(NOEC \& EC10)}$ values for the three land uses are presented in Table 31.

Table 31. Generic soil quality guidelines based on no observed effect concentration and 10% effect concentration toxicity data ($SQG_{(NOEC \& EC10)}$) for fresh arsenic (As) contamination in soil with different land uses.

Land use	$SQG_{(NOEC \& EC10)}$ (mg/kg total As)
Areas of ecological significance	8
Urban residential/public open space	20
Commercial/industrial	30

It should be noted, because As has generic $SQG_{(NOEC \& EC10)}$ values, that they should be applied to all Australian soils that have the particular land use.

4.6.1.1 Calculation of ambient background concentration values

Despite the fact that ACLs could not be derived for As, the issue of background concentrations of As in Australian soils will be discussed as the situation could change in the future if additional data becomes available. If, in the future, toxicity data can be expressed in terms of added concentrations, it is recommended that the method of Hamon et al. (2004) be used to derive ABC values. Examples of the ABC values generated by the Hamon et al. (2004) method are presented in Table 32. The soil-

specific estimate of ABC could be added to a generic ACL (if toxicity data could be expressed as added As, but no normalisation relationships were suitable) or it could be added to a soil-specific ACL (if it were possible to express the toxicity data in terms of added As and if normalisation relationships could be applied to the data).

Table 32. Ambient background concentrations of arsenic (As) estimated using the method of Hamon et al. (2004) as a function of the iron content of the soil.

Soil iron (%)	As (mg/kg)
0.1	1
1	3
10	12
20	18

4.6.2 Calculation of soil quality guidelines for fresh arsenic contamination based on protecting aquatic ecosystems from leaching

The log K_d value for As (Crommentuijn et al. 2000) was below 3 and therefore in accordance with the SQG derivation methodology (Schedule B5b) $SQG_{(NOEC\ \&\ EC10)}$ values were derived to protect aquatic ecosystems from the effects of leached As from freshly contaminated soils.

The As $SQG_{(NOEC\ \&\ EC10)}$ values based on protecting groundwater ecosystems were calculated using the US EPA method (US EPA 1996). The generic $SQG_{(NOEC\ \&\ EC10)}$ values were calculated using DAF values of one and 20 and these are presented in Table 33. There is a linear relationship between the DAF and the SQGs, thus the SQGs calculated using a DAF of 20 are 20 times larger than those calculated using a DAF of 1.

Table 33. Generic arsenic (As) soil quality guidelines (SQGs, mg total As/kg) based on no observed effect concentration and 10% effect concentration toxicity data to protect groundwater ecosystems from leaching.

Dilution factor	1	20
SQG (mg/kg)	4.6	91

4.6.3 Calculation of soil quality guidelines for fresh arsenic contamination based on lowest observed effect concentration and 30% effect concentration toxicity data, and based on 50% effect concentration toxicity data

The $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values were calculated using the same method as for the As $SQG_{(NOEC\ \&\ EC10)}$ values ,except that different toxicity data was used. The data used is presented in Table 29. To maximise the data available to generate the $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values, the available toxicity data was converted to the appropriate measure of toxicity using the default conversion factors presented in Table 30.

As with the $SQG_{(NOEC\ \&\ EC10)}$ values for As, soil-specific $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values could not be generated, but rather a single generic $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ value was generated for each of the three land uses (Table 34). Also, all toxicity data was expressed as total As rather than added As. As these are generic $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values ,they should be applied to all Australian soils with a particular land use.

Table 34: Generic soil quality guidelines based on lowest observed effect concentration and 30% effect concentration toxicity data ($SQG_{(LOEC\ \&\ EC30)}$), and based on 50% effect concentration toxicity data ($SQG_{(EC50)}$) for soil with different land uses.

Land use	$SQG_{(LOEC \& EC30)}$ (mg/kg total As)	$SQG_{(EC50)}$ (mg/kg total As)
Areas of ecological significance	20	30
Urban residential/public open space	50	90
Commercial/industrial	80	140

4.7 Calculation of soil quality guidelines for aged arsenic contamination

4.7.1 Calculation of an ageing and leaching factor for arsenic

Song et al. (2006) conducted some experiments to determine the effect of ageing As over three months in four soils. They found that in all soils the toxicity and extractability decreased and the extent of the decrease ranged from 2- to 12-fold (Song et al. 2006). Yang et al. (2002) and Fendorf et al. (2004) also found that As aged in soils with the majority occurring within the first few months. Yang et al. (2002) also found that As ageing did not always occur—it occurred in only 47% (i.e. 17 out of 36) of the soils they examined. Song et al. (2006) found that the extent of ageing was significantly correlated with oxalate-extractable iron and Olsen-P concentrations in the four test soils. However, they also noted that data on more soils was needed in order for the relationships to be considered more robust. Song et al. (2006) concluded that ageing of As 'should be taken into account during risk assessment'. Therefore, in order to account for ageing in a conservative manner (that is, one that is protective of the environment), the lowest ALF factor (2) determined by Song et al. (2006) was used to derive the aged SQGs. This ALF was applied to all the toxicity data.

4.7.2 Calculation of soil quality guidelines for aged arsenic contamination

As the available toxicity data can only be expressed as total As concentrations, ACL values could not be derived, so SQGs were derived. The ALF of 2 was applied to all the toxicity data; therefore the aged $SQG_{(NOEC \& EC10)}$, $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values are exactly twice the corresponding fresh SQGs for arsenic. The resulting aged $SQG_{(NOEC \& EC10)}$, $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values are presented in Table 35.

Table 35. Generic soil quality guidelines based on no observed effect concentration and 10% effect concentration toxicity data ($SQG_{(NOEC \& EC10)}$), lowest observed effect concentration and 30% effect concentration toxicity data ($SQG_{(LOEC \& EC30)}$), and based on 50% effect concentration toxicity data ($SQG_{(EC50)}$) for soil with different land uses.

Land use	$SQG_{(NOEC \& EC10)}$ (mg/kg total As)	$SQG_{(LOEC \& EC30)}$ (mg/kg total As)	$SQG_{(EC50)}$ (mg/kg total As)
Areas of ecological significance	15	40	60
Urban residential/public open space	40	100	180
Commercial/industrial	60	160	290

4.7.3 Calculation of ambient background concentration values

Background levels of As are not elevated by historic pollution in urban residential/public open space soils, as can be seen by data from Olszowy et al. (1995) (Table 36). Therefore, in the future, if toxicity data can be expressed in terms of added concentrations, it is recommended that the method of Hamon et al. (2004) be used to estimate background values, as they are soil-specific. Examples of the ABC values generated by the Hamon et al. (2004) method are presented in Table 32.

Table 36. Background concentrations of arsenic (As) from Olszowy et al. (1995) in suburbs of different age and with different intensities of traffic in various states of Australia.

Suburb type	25th percentile As (mg/kg)			
	NSW	QLD	SA	VIC
New suburb, low traffic	5	3	5	NA
New suburb, high traffic	5	3	5	NA
Old suburb, low traffic	5	4	5	5
Old suburb, high traffic	5	3	5	5

NA = not available

4.8 Reliability of the soil quality guidelines

The As toxicity dataset met the minimum data requirements to use the SSD method but there were no normalisation relationships available to account for soil characteristics. Based on the criteria for assessing the reliability of SQGs (Schedule B5b), this means that the As SQGs were considered to be of moderate reliability.

4.9 Comparison with other guidelines

A compilation of SQGs for As from a number of jurisdictions is presented in Table 37. These guidelines have a variety of purposes and levels of protection and therefore comparison of the values is problematic. The SQGs for As range from 4.5 mg/kg (added As) for the Dutch to 110 mg/kg (total As) for another European country. The superseded interim urban EIL (NEPC 1999) was 20 mg/kg total As and lies in the lower portion of the range of As SQGs. The As $SQG_{(NOEC\ \&\ EC10)}$ for freshly contaminated urban residential/public open space soils was 20 mg/kg (total As) and thus identical to the superseded interim urban EIL. The $SQG_{(NOEC\ \&\ EC10)}$ for aged contamination at 40 mg/kg is twice the superseded interim urban EIL for As.

The $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values for As in freshly contaminated urban residential/public open space soils are 50 and 80 mg/kg respectively. The $SQG_{(LOEC\ \&\ EC30)}$ is in the middle of the range of SQGs for other jurisdictions, while the $SQG_{(EC50)}$ is in the upper portion of the range of SQGs. The aged As $SQG_{(LOEC\ \&\ EC30)}$ for urban residential/public open space soils lies in the upper part of the range of international SQGs while the aged As $SQG_{(EC50)}$ value for urban residential/public open space soils is markedly larger than any other international SQG.

Table 37. Soil quality guidelines for arsenic (As) from international jurisdictions.

Name of arsenic soil quality guideline	Numerical value of the guidelines (mg/kg)
Dutch target value[1]	29 (total As)
Dutch maximum permissible addition[1]	4.5 (added As)
Canadian SQG[2]	12 (total As)
Eco-SSL plants[3]	18 (total As)
Eco-SSL soil invertebrates[3]	NA
Eco-SSL avian[3]	43 (total As)
Eco-SSL mammalian[3]	46 (total As)
EU screening values potential risk in residential areas[4]	5–110 (total As)

1 = VROM 2000
2 = CCME, 1999b, and 2006 and http://www.ccme.ca/publications/list_publications.html#link2
3 = http://www.epa.gov/ecotox/ecossl/
4 = Carlon 2007
NA = not available

5 Naphthalene

5.1 Compounds considered

Unlike Zn and As, which can occur in several forms in soil, naphthalene is a unique compound and only information relating to it was used in the derivation of the SQG values. Naphthalene ($C_{10}H_8$) is the smallest of the family of compounds collectively termed polycyclic aromatic hydrocarbons (PAHs). The chemical abstract service number for naphthalene is 91-20-3 (HSDB 2004).

5.2 Exposure pathway assessment

Selected physicochemical properties of naphthalene are:

Molecular weight:	128.17 (O'Neil 2001)
Log K_{ow}	3.29 (US EPA 1982),
	3.01–3.45 (Verschueren 1983),
	3.30 (Hansch et al. 1995)
Log K_{oc}	2.97 (US EPA 1982; GDCH 1992; Kenaga 1980)
Vapour pressure	0.087 mm Hg (US EPA 1982)
	0.085 mm Hg at 25°C (Ambrose et al. 1975)
Aqueous solubility	31 mg/L at 25°C (Pearlman et al. 1984)
Henry's law constant	4.6×10^{-4} atm-m^3/mol (US EPA 1982; Yaws et al. 1991)
	4.4×10^{-4} atm-m^3/mol (Shiu & Mackay 1997)
Half-life (in soil)	2–18 days (ATSDR 2005)

The minimum log K_{ow} value at which biomagnification should be considered in the derivation of SQGs is 4 (Schedule B5b). As the reported log K_{ow} values for naphthalene were below 4 and it has a relatively short half-life (see above), it is not considered a biomagnifying compound and the normal protection levels were used. Therefore only the direct toxicity exposure route was considered in the derivation of SQGs for naphthalene. The log K_{oc} value for naphthalene is moderate (~3) and therefore there is only a moderate potential for naphthalene to be leached to groundwater or surface water. Soil quality guidelines to protect aquatic ecosystems were therefore not generated.

5.3 Toxicity data

Toxicity data for naphthalene was available for two plant species, eight species of soil invertebrates and four species of terrestrial vertebrates (Table 38). In total, there was data for 14 species that belonged to five taxonomic groups and thus this met the minimum data requirements recommended by the methodology to use the BurrliOZ SSD method (Campbell et al. 2000). Table 38 shows the geometric means of individual species used to derive the naphthalene SQGs. The raw toxicity data used to generate the species geometric means are presented in Appendix E.

In order to maximise the use of the available toxicity data, default conversion factors were used to permit the inter-conversion of NOEC, LOEC, EC_{50}, EC_{30} and EC_{10} data (Table 30).

Table 38. Geometric means of the toxicity of naphthalene (expressed in terms of total naphthalene) to soil invertebrates, terrestrial vertebrates and plants.

Test species		Geometric mean (mg/kg)		
Common name	**Scientific name**	**NOEC or EC10**	**LOEC or EC30**	**EC50**
Earthworm	*Eisenia fetida*	54	135	270
European rabbit	*Oryctolagus cuniculus*	2000	5000	10 000
House mouse	*Mus musculus*	407	1018	2036
Lettuce	*L. sativa*	21	54	107
Mite	*Acari spp*	232	580	1160
Mite	*Mesostigmata spp.*	195	487	973
Mite	*Oribatida sp.*	219	547	1094
Northern bobwhite	*C. virginianus*	1000	2500	5000
Common rat	*R. norvegicus*	1000	2500	5000
Radish	*R. sativa*	61	153	305
Spider	*Grammonata inornata*	177	443	886
Springtail	*Collembola spp.*	214	535	1070
Springtail	*F. fimetaria*	20	50	100
Springtail	*Poduromorpha spp.*	203	508	1016

5.4 Normalisation relationships

It is well known that the organic carbon (OC) or organic matter content of soils affects the toxicity and bioavailabiity of organic contaminants such as naphthalene. European guidelines use normalisation relationships for organic contaminants (ECB 2003), but these have not yet been verified for Australian soils. In fact, when data for soils with OC contents greater than typical Australian soils was removed, OC was no longer a useful descriptor of toxicity (Broos et al. 2007). While the above example is for an inorganic contaminant, it shows the potential for European normalisation relationships to be inappropriate for Australia. As Australian soils are in general low in organic carbon, it was not recommended to use European normalisation relationships (Schedule B5b). There were no normalisation relationships available for naphthalene. Therefore, the toxicity data could not be normalised to the Australian reference soil, nor could soil-specific SQGs be derived.

5.5 Sensitivity of organisms to naphthalene

The SSD for the naphthalene toxicity data is presented in Figure 5. As there was only toxicity data for 14 different species, insufficient data was available to make a robust assessment of the relative sensitivity of the groups of organisms. Nonetheless, it appears that plant and soil invertebrate species were more sensitive to naphthalene than vertebrate species, as the vertebrate toxicity data was all higher than those for other species. An argument could be mounted to exclude the terrestrial vertebrates from the calculation of the SQGs; however, this was not adopted, for three reasons. Firstly, the data was sparse and therefore the differences in the relative sensitivity of the groups of organisms may not be real. Secondly, the terrestrial vertebrates represent a major group of organisms that most people would wish to be able to maintain in urban residential/public open space settings. Thirdly, removal of these species only had a minor effect on the resulting $SQG_{(NOEC\ \&\ EC10)}$ (i.e. the PC_{80} for all species was 68 mg/kg while the corresponding value when the vertebrates were removed was 60 mg/kg).

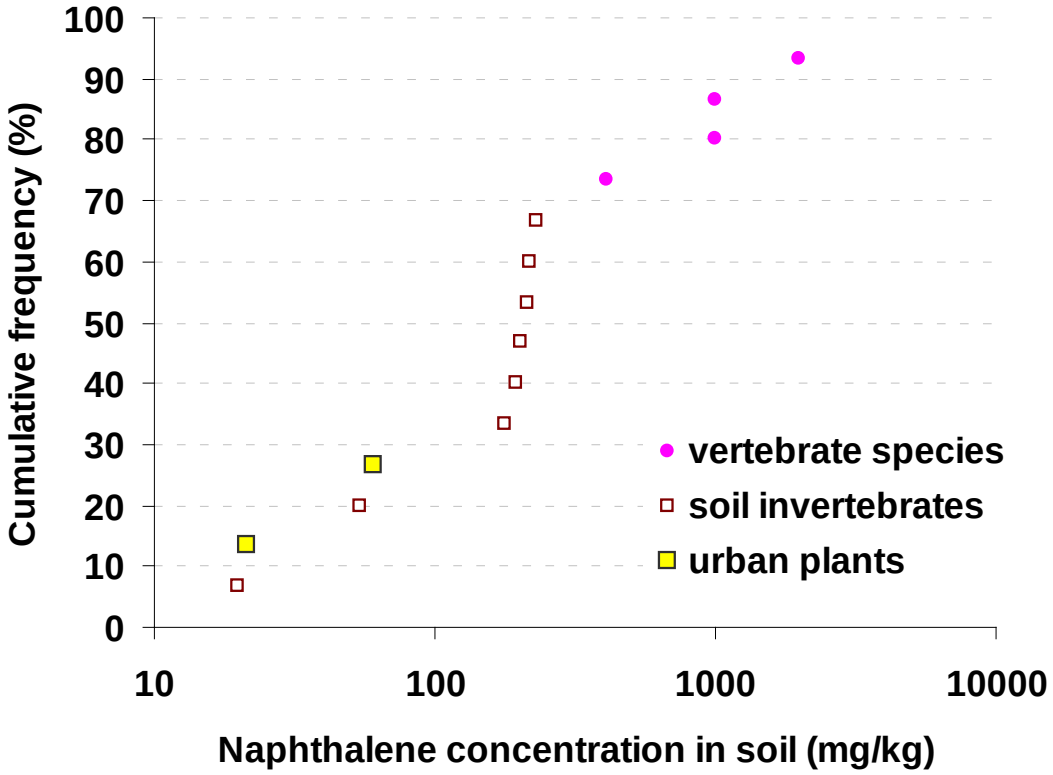


Figure 5. The species sensitivity distribution (plotted as a cumulative frequency of the toxicity data against naphthalene soil concentration) of soil invertebrates, plants and terrestrial vertebrates to naphthalene.

5.6 Calculation of soil quality guidelines for fresh naphthalene contamination

Given that (a) there was sufficient toxicity data to use the BurrliOZ software, (b) the data could not be normalised to the Australian reference soil, and (c) the toxicity data could not be expressed in terms of added concentrations, it meant that there was a single output from the BurrliOZ SSD for each of the three land uses (that is, areas of ecological significance, urban residential/public open space, and commercial/industrial). Therefore, the output from the SSD was a single generic (not soil-specific) SQG for each land use.

5.6.1 Calculation of soil quality guidelines for fresh naphthalene contamination based on no observed effect concentration and 10% effect concentration toxicity data

The generic SQGs for naphthalene in soils with each of the three land uses are presented in Table 39.

Table 39. Generic soil quality guidelines for naphthalene in freshly contaminated soils with different land uses based on no observed effect concentration and 10% effect concentration toxicity data.

Land use	$SQG_{(NOEC \& EC10)}$ (mg/kg total naphthalene)
Areas of ecological significance	5
Urban residential/public open space	70
Commercial/industrial	150

5.6.1.1 Calculation of ambient background concentration values

There is no equation available to estimate the background concentration of naphthalene. Naphthalene is produced by some organisms (for example, magnolias and termites) but at very low concentrations, which are negligible in terms of ABC values. Naphthalene can also be synthesised as a result of fires and in fire-prone areas and it might be appropriate to determine naphthalene ABC values.

In most soils, naturally occurring naphthalene concentrations will be negligible. For the purpose of this guideline the ABC for naphthalene was assumed to be 0 mg/kg. Therefore, the reported toxicity values which were expressed as total naphthalene were identical to the data when expressed as added naphthalene concentrations (that is, total concentration – 0 = added concentration) and therefore the ACLs derived using the SSD methodology equalled the SQGs.

It should be noted that if a soil-specific ABC for naphthalene is determined then that could be added to the above values to obtain a soil-specific SQG. Otherwise, these generic SQGs are applicable to all Australian soils with these particular land uses.

5.6.2 Calculation of soil quality guidelines for fresh naphthalene contamination based on lowest observed effect concentration and 30% effect concentration data, and based on 50% effect concentration toxicity data

These SQGs were calculated using the same method as that for the $SQG_{(NOEC\ \&\ EC10)}$ values for naphthalene, except that different toxicity data was used (Table 38). To maximise the data available to generate $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values, the available toxicity data was converted to the appropriate measure of toxicity using the default conversion factors recommended in Schedule B5b and presented in Table 30.

As with the $SQG_{(NOEC\ \&\ EC10)}$ values for naphthalene, soil-specific $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ values could not be generated, so rather a single generic $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ was generated for each of the three land uses (Table 40). It should be noted that if a soil-specific ABC for naphthalene is determined then that could be added to the generic SQG values (Table 40) to obtain a soil-specific SQG. Otherwise these generic $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values should apply to all Australian soils with these particular land uses.

Table 40. Generic soil quality guidelines for naphthalene in freshly contaminated soil with different land uses based on lowest observed effect concentration and 30% effect concentration toxicity data and based on 50% effect concentration toxicity data.

Land use	$SQG_{(LOEC\ \&\ EC30)}$ (mg/kg total naphthalene)	$SQG_{(EC50)}$ (mg/kg total naphthalene)
Areas of ecological significance	10	25
Urban residential/public open space	170	340
Commercial/industrial	370	730

5.7 Calculation of soil quality guidelines for aged naphthalene contamination

There is currently no ageing or leaching factor available for naphthalene in the literature and therefore SQGs for aged contamination could not be derived.

5.8 Metabolites of naphthalene

The most well known metabolites of naphthalene are 1-naphthol (CAS no. 90-15-3) or 2-naphthol (CAS no. 135-19-3). These compounds are both known to affect plant growth and are suspected to

have endocrine disrupting properties (Pesticide Action Network at <www.pesticideinfo.org>). There is no toxicity data in soils or SQGs reported for these compounds.

5.9 Reliability of the soil quality guidelines

The naphthalene toxicity dataset met the minimum data requirements to use the SSD method but there were no normalisation relationships available to account for soil characteristics. Based on the criteria for assessing the reliability of SQGs (Schedule B5b), the naphthalene SQGs were considered to be of moderate reliability.

5.10 Comparison with other guidelines

A compilation of SQGs for naphthalene in a number of jurisdictions is presented in Table 41. These SQGs have a variety of purposes and levels of protection and therefore comparison of the values is problematic. The SQGs for naphthalene range from 0.6 mg/kg for Canada to 125 mg/kg for the USA, both expressed as total naphthalene. The original NEPM (NEPC 1999) did not include an EIL for naphthalene. The $SQG_{(NOEC\ \&\ EC10)}$ for areas of ecological significance freshly contaminated with naphthalene is 5 mg/kg and thus is identical to the lower range of values set within the EU, but approximately an order of magnitude higher than the Canadian SQG and $1/25^{th}$ of the USA SQG. The $SQG_{(NOEC\ \&\ EC10)}$ for urban residential/public open space is 70 mg/kg and thus slightly higher than the highest EU SQGs but still approximately half the US EPA screening level for residential land. The $SQG_{(LOEC\ \&\ EC30)}$ for urban residential land use at 170 is 40% larger than the US EPA screening level, while the corresponding $SQG_{(EC50)}$ value is 2.8 times the US EPA screening level.

Table 41. Soil quality guidelines for naphthalene in a number of jurisdictions.

Name of the naphthalene soil quality guideline	Value of the guidelines (mg/kg)
Canadian SQG (residential)[1]	0.6
EU (residential)[2]	5–60
US EPA Screening level (residential)[3]	125

1 = CCME 1999c , 2006 and <http://www.ccme.ca/publications/list_publications.html#link2>

2 = Carlon 2007

3 = http://www.epa.gov/ecotox/ecossl/.

6 DDT

6.1 Compounds considered

DDT is the abbreviation used for dichloro-diphenyl-trichloroethane ($C_{14}H_9Cl_5$). Technical grade DDT (the form used in pesticide formulations) consists of 14 compounds (ATSDR 2002). The active ingredient and the main constituent of DDT is p,p'-DDT (approx 87% of DDT). Other compounds present include o,p'-DDT (15% of DDT), dichloro-diphenyl-dichloroethylene (DDE) and dichloro-diphenyl-dichloroethane (DDD), which are also metabolites and breakdown products of DDT. When DDT is referred to, usually people are referring to p,p'-DDT and this was the form that was used for the derivation of the EIL. The CAS registration number for p,p'-DDT is 50-29-3.

6.2 Pathway risk assessment

Selected physicochemical properties of DDT include:

Molecular weight	354.49 (Howard & Meylan 1997)
Log K_{ow}	6.91 (Howard & Meylan 1997; Hansch et al. 1995)
Log K_{oc}	5.18 (Swann et al. 1981)
Vapour pressure	1.60×10^{-7} at 20°C (Bidleman & Foreman 1987)
Aqueous solubility	0.025 mg/L at 25°C (Howard & Meylan 1997),
	5.5×10^{-3} mg/L at 25°C (Yalkowsky & Dannenfelser 1992)
Henry's law constant	8.3×10^{-6} atm-m^3/mol (Howard & Meylan 1997)
Half-life (in aerobic soil)	range from 2 years (Lichenstein & Schulz 1959) to greater than 15 years (Keller 1970; Stewart & Chisholm 1971)
Half-life (in anaerobic soil)	16–100 days (Castro & Yoshida 1971)
Half-life of DDT	190 years (OMEE 1993)
Bioconcentration factor	2.5–16 (CCME 1999d)
Bioaccumulation factor	0.9–29 (CCME 1999d)

DDT is a well known biomagnifying contaminant and, as the log K_{ow} is higher than 4, both the direct toxicity and biomagnification routes of exposure needed to be accounted for in deriving the SQGs. Therefore, the level of protection (that is, percentage of species to be protected) was increased for urban residential/public open space soils from 80% to 85% as recommended in Schedule B5b. The log K_{oc} value for DDT is >5 and therefore there is a very low potential for DDT to be leached to groundwater or surface water.

6.3 Toxicity data

The raw toxicity data available for DDT is presented in Appendix F. The geometric means of toxicity data for each species and soil process are presented in Table 42. There was toxicity data for a total of 15 species or soil processes that belong to 5 different taxonomic groups or nutrient groups. Thus, there was sufficient toxicity data to use the SSD method to derive SQGs for DDT.

6.4 Normalisation relationships

As with naphthalene, it is well known that the organic carbon or organic matter content of soils affects the toxicity and bioavailabiity of organic contaminants such as DDT. However, there were no normalisation relationships available for DDT. Therefore, the toxicity data could not be normalised to the Australian reference soil (Table 6), nor could soil-specific SQGs be derived.

6.5 Sensitivity of organisms to DDT

Figure 6 shows the SSD (that is, the cumulative distribution of the geometric means of toxicity values) for the species used to derive the DDT SQGs. There is a general paucity of terrestrial toxicity data for

DDT. This is particularly the case for plants and soil invertebrates where each group only has data for two species. It is therefore difficult to assess the relative sensitivity of these groups of organisms. Soil processes had sensitivities to DDT ranging from very sensitive to very tolerant, although most were in the more tolerant part of the distribution. Both plants were tolerant of DDT. Both soil invertebrates had moderate sensitivity while the vertebrate species were generally sensitive. The greater sensitivity of the vertebrates is consistent with the findings on the relative sensitivity of aquatic species.

Table 42. The geometric mean values of the DDT toxicity data for soil invertebrate species, terrestrial vertebrate species, plant species and soil processes.

Test species		Geometric means (mg/kg)		
Common name	**Scientific name**	**NOEC or EC10**	**LOEC or EC30**	**EC50**
Earthworm	*Eisenia fetida*	363	1131	2499
Field mustard	*Brassica rapa*	1000	2500	5000
Helmeted guineafowl	*Numida meleagris*	30	75	150
House sparrow	*Passer domesticus*	600	1500	3000
Japanese quail	*Coturnix japonica*	80	200	400
Mallard duck	*Anas platyrhynchos*	24	59	119
Northern bobwhite	*C. virginianus*	68	170	341
Oats	*A. sativa*	1000	2500	5000
Ring-necked pheasant	*Phasianus colchicus*	104	261	522
Soil process	Ammonification	1250	3125	6250
Soil process	Nitrification	56	141	281
Soil process	Respiration	1000	2500	5000
Soil process	SIN	1000	2500	5000
Soil process	SIR	1000	2500	5000
Springtail	*F. candida*	464	1344	2836

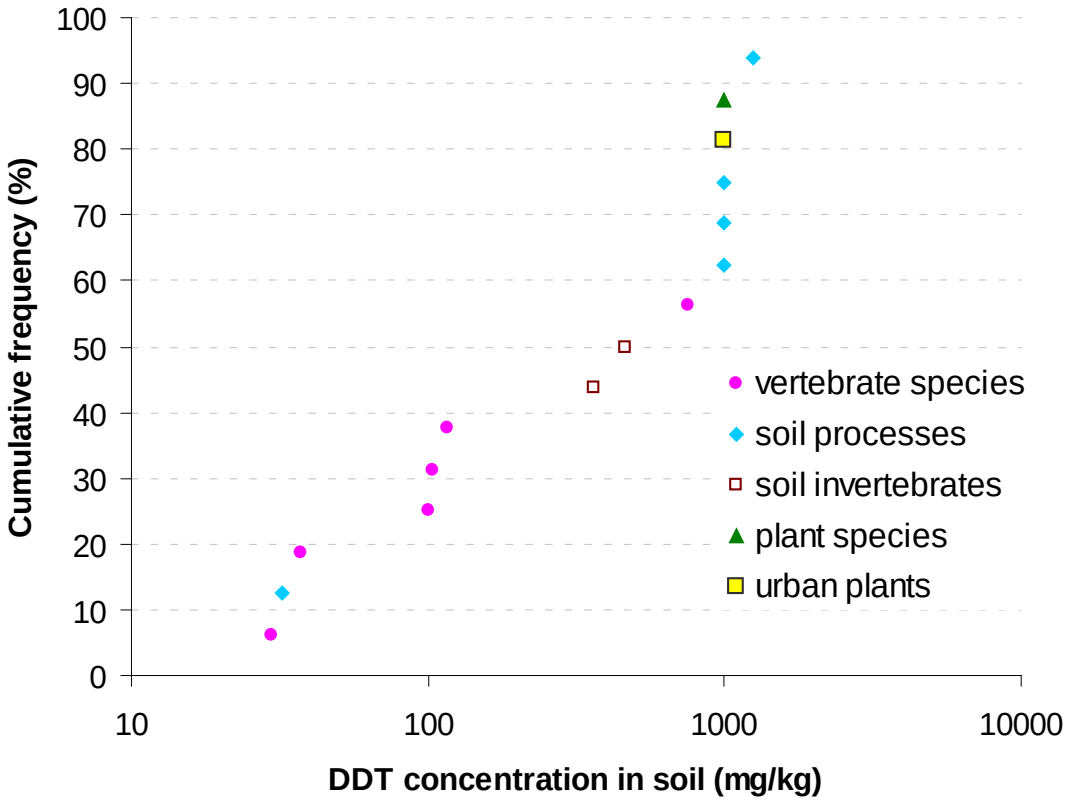


Figure 6. The species sensitivity distribution (plotted as a cumulative frequency of the toxicity data against DDT soil concentration) of soil invertebrate species, soil processes, plant species and terrestrial vertebrate species to DDT.

6.6 Calculation of soil quality guidelines for fresh DDT contamination

All the available DDT toxicity data was reported as total concentrations without making a distinction between added and background concentrations. There was no equation available able to estimate the background concentration of DDT. DDT only occurs due to its synthesis by humans. There is therefore no natural background concentration of DDT. However, due to its persistence and its ability to volatilise, DDT can be subject to long-distance transport. In fact, a global distillation hypothesis was developed and has widely been accepted as the explanation of the presence of DDT and its metabolites and other persistent organic pollutants in polar ecosystems, which have no nearby industrial point sources or non-point sources. Because of this global transport of DDT, it could be argued that there is an ABC. As the DDT toxicity studies did not provide any estimate of the ABC for DDT either at the sites or in the soils that were used, this could not be accounted for in deriving the limits for DDT. Therefore, a default ABC for DDT of 0 mg/kg was adopted.

6.6.1 Calculation of generic soil quality guidelines for fresh DDT contamination based on no observed effect concentration and 10% effect concentration toxicity data

The situation for DDT was that:

- it biomagnifies and this needs to be accounted for in deriving the SQG
- there was sufficient toxicity data to use the BurrliOZ software
- the data could not be normalised to the Australian reference soil as there were no normalisation relationships available for DDT
- the toxicity data could not be expressed in terms of added concentrations
- an ABC of 0 was used.

Therefore, a single value was generated by BurrliOZ (Campbell et al. 2000) for each of the three land uses. The output was the $SQG_{(NOEC\ \&\ EC10)}$ for each particular land use and no soil-specific SQGs could be calculated. As DDT biomagnifies, the SQGs must take this into account. The methodology for deriving SQGs (Schedule B5b) for biomagnifying contaminants is to increase the level of protection (% of species to be protected) by 5% for soils for urban residential/public open space and commercial/industrial land uses to 85% and 65% of species respectively. For areas of ecological significance land uses no increase in the level of protection is recommended (Schedule B5b) as the default level (that is, for non-biomagnifying contaminants) is already 99% protective of species. The methodology was adopted and the resulting $SQG_{(NOEC\ \&\ EC10)}$ values are presented in Table 43.

Table 43. Soil quality guidelines based on no observed effect concentration and 10% effect concentration toxicity data ($SQG_{(NOEC\ \&\ EC10)}$) for DDT in freshly contaminated soils with different land uses.

Land use	$SQG_{(NOEC\ \&\ EC10)}$ (mg total DDT/kg soil)
Areas of ecological significance	1[a]
Urban residential/public open space	70[b]
Commercial/industrial	250[c]

[a] to protect 99% of species, [b] to protect 85% of species, [c] to protect 65% of species.

It should be noted that if a site-specific ABC for DDT is determined (and there is sufficient justification for this ABC to be used instead of the default value of 0 mg/kg) then it may be added to the above generic $SQG_{(NOEC\ \&\ EC10)}$ values to obtain a site-specific $SQG_{(NOEC\ \&\ EC10)}$. As the values in Table 43 are generic $SQG_{(NOEC\ \&\ EC10)}$ values they should be applied to all Australian soils that have the particular land use.

6.6.2 Calculation of soil quality guidelines for fresh DDT contamination based on lowest observed effect concentration data and 30% effect concentration data, and based on 50% effect concentration toxicity data

The $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values were calculated using the same method as that for the corresponding values for Zn, As and naphthalene. The data used to calculate these SQGs is presented in Table 42. To maximise the data available to generate the $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values, the available toxicity data was converted to the appropriate measure of toxicity using the default conversion factors recommended in Schedule B5b and presented in Table 30.

As with the $SQG_{(NOEC\ \&\ EC10)}$ values for DDT, soil-specific $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values could not be generated, so rather a single generic $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ was generated for each of the three land uses (Table 44). As these are generic SQGs, they should be applied to all Australian soils with the particular land use.

Table 44. Soil quality guidelines for DDT in freshly contaminated soil with different land uses based on lowest observed effect concentration and 30% effect concentration toxicity data, and based on 50% effect concentration toxicity data.

Land use	$SQG_{(LOEC \& EC30)}$ (mg/kg total DDT)	$SQG_{(EC50)}$ (mg/kg total DDT)
Areas of ecological significance	3	6
Urban residential/public open space	180	360
Commercial/industrial	640	1300

6.7 Calculation of soil quality guidelines for aged contamination

There is currently no ageing or leaching factor available for DDT and therefore SQGs for aged contamination could not be derived.

6.8 Reliability of soil quality guidelines

The DDT SQGs were considered to be of moderate reliability as the toxicity data set met the minimum data requirements to use an SSD method but there were no normalisation relationships available to account for soil characteristics (Schedule B5b).

6.9 Important metabolites of DDT

The most common metabolites of DDT are shown in Table 45. DDE is a well-known metabolite of DDT and is relatively well studied. However, there is considerably less information available on the environmental fate, metabolism, degradation and toxicity of these metabolites than on DDT. The HILs and some soil quality guidelines use a sum of DDT, DDE and DDD concentration as an SQG , for example, the Dutch and Flemish SQGs. An SQG could be derived for the sum of DDT, DDE and DDD by assuming the compounds have concentration-additive toxicity.

Table 45. Major metabolites of DDT (Sourced from WHO 1989).

Abbreviation of metabolite	Chemical name of metabolite
DDE	1,1'-(2,2-dichloroethenylidene)-bis[4-chlorobenzene]
TDE(DD)	1,1'-(2,2-dichloroethylidene)-bis[4-chlorobenzene]
DDMU	1,1'-(2-chloroethenyldene)-bis[4-chlorobenzene]
DDMS	1,1'-(2-chloroethylidene)-bis[4-chlorobenzene]
DDNU	1,1'-bis(4-chlorophenyl)ethlyene
DDOH	2,2-bis(4-chlorophenyl)ethanol
DDA	2,2-bis(4-chlorophenyl)-acetic acid
Methoxychlor	1,1'-(2,2,2-trichloroethylidene)-bis[4-methoxybenzene]
Perthane	1,1'-(2,2-dichloroethylidene)-bis[4-ethylbenzene]
DFDT	1,1'-(2,2,2-trichloroethylidene)-bis[4-fluorobenzene]

6.10 Comparison with other guidelines

Soil quality guidelines for DDT in a number of jurisdictions are presented in Table 46. These SQGs have a variety of purposes and levels of protection and therefore a comparison of the values is problematic. The SQGs for DDT range from 0.01 to 4 mg/kg total DDT, both from the Netherlands. The original NEPM (NEPC 1999) did not include an EIL for DDT. However, there are four HIL values of 260, 700, 400 and 4,000 mg/kg for land use settings A, B, C and D[3] for the sum of DDT,

[3] A = the standard residential setting with garden/accessible soils and home-grown produce contributing <10% of vegetable and fruit intake. B = residential with minimal opportunities for soil access: includes dwellings with fully and permanently paved yard space such as high rise apartments and flats. C = parks, recreational open

DDD, and DDE (Schedule B1). The SQGs for urban residential/public open space soil contaminated with fresh DDT based on NOEC & EC_{10}, LOEC & EC_{30}, and EC_{50} data were 70, 170 and 350 mg/kg. These values are considerably higher than the SQGs from other jurisdictions and this reflects the different methods that are used to account for biomagnification. The $SQG_{(NOEC\ and\ EC10)}$ and $SQG_{(LOEC\ \&\ EC30)}$ are approximately 27% and 67% respectively, of the HIL for the standard residential setting (setting A) which assumes direct exposure and the consumption of some food grown on the contaminated soil. The SQGs should still offer a considerable degree of protection.

Table 46. Soil quality guidelines for DDT in a number of jurisdictions.

Name of the DDT soil quality guideline	Value of the guideline (mg/kg as total)
Dutch target values[1]	0.01
Dutch intervention value[1]	4
Canadian SQG (residential)[2]	0.7
Eco-SSL plants[3]	NA
Eco-SSL soil invertebrates[3]	NA
Eco-SSL avian[3]	0.093
Eco-SSL mammalian[3]	0.021
EU potentially unacceptable (residential)[4]	1–4

1 = VROM 2000

2 = CCME 1999d, 2006 and http://www.ccme.ca/publications/list_publications.html#link2

3 = http://www.epa.gov/ecotox/ecossl/

4 = Carlon 2007

NA = not available

space and playing fields: includes secondary schools. D = Commercial/industrial: includes premises such as shops and offices as well as factories and industrial sites.

7 Copper

7.1 Copper compounds considered

The following compounds were considered in deriving the SQGs for Cu:

- copper metal (CAS No. 7440-50-8)
- copper (II) sulphate pentahydrate (CAS No. 7758-98-7)
- copper (I) oxide (CAS Nos 1317-3-1)
- copper (II) oxide (CAS No. 1317–38–0)
- dicopper chloride trihydroxide (CAS No. 1332-65-6).

7.2 Exposure pathway assessment

The two key considerations in determining the most important exposure pathways for inorganic contaminants are whether they biomagnify and whether they have the potential to leach to groundwater.

A surrogate measure of the potential for a contaminant to leach is its water−soil partition coefficient (K_d). If the logarithm of the K_d (log K_d) of an inorganic contaminant is less than 3, then it is considered to have the potential to leach to groundwater (Schedule B5b). The Australian National Biosolids Research Program measured the log K_d of Cu in 17 agricultural soils throughout Australia. These measurements showed that, in most soils, the log K_d of Cu was below 3 L/kg (unpublished data). The log K_d value for Cu reported by Crommentuijn et al. (2000) was 2.99 L/kg. Therefore, there is the potential for Cu in some soils to leach to groundwater and affect aquatic ecosystems. However, the methodology for SQG derivation (Schedule B5b) does not advocate the routine derivation of SQGs that account for leaching potential. Rather, it advocates that this be done on a site-specific basis as appropriate (Schedule B5b).

Copper is an essential element for the vast majority of living organisms and, as such, concentrations of Cu in tissue are highly regulated and it does not biomagnify (Louma & Rainbow 2008; Heemsbergen et al. 2008; EC 2008a). Therefore, the biomagnification route of exposure does not need to be considered for Cu and the SQGs will only account for direct toxicity.

7.3 Toxicity data

The ecotoxicology of Cu has been extensively studied both within Australia and internationally. Most studies presented their toxicity data as an added concentration (that is, the concentration of the contaminant added to the soil that causes a specified toxic effect) or in a form that permitted the added concentration to be calculated (that is, by subtracting the background from the total concentration).

The toxicity database used to calculate the SQGs for Cu consisted of over 400 toxicity measures for 11 soil processes (Table 47), 10 invertebrate species (Table 48) and 18 plant species (Table 49). The raw data used to generate Tables 47–49 is provided in Appendix E. There was sufficient data—that is, toxicity data for at least five species or soil processes that belong to at least three taxonomic or nutrient groups (Schedule B5b)—available to derive SQGs using a species sensitivity distribution (SSD) methodology.

Given that Cu does not biomagnify, the level of protection recommended in the SQG derivation methodology for urban residential/public open space land is 80% (that is, 80% of species would be protected) (Schedule B5b).

Table 47. The lowest geometric mean values of the normalised copper (Cu) toxicity data (expressed in terms of added Cu) for soil microbial processes.

Soil process	Geometric means (mg/kg added Cu)		
	EC_{10} or NOEC	EC_{30} or LOEC	EC_{50}
Ammonification	721	1081	2164
Denitrification	59.6	149	179
Glutamic acid decomposition	64.7	329	659
Maize residue mineralisation	199	299	597
Microbial biomass carbon	35.6	80.9	107
Microbial biomass nitrogen	141	90.9	174
N mineralisation	81	84	160
Potential nitrification rate	137	205	282
Respiration	151	916	3165
Substrate induced nitrification	276	421	700
Substrate induced respiration	86	224	589

Table 48. The lowest geometric mean values of the normalised copper (Cu) toxicity data (expressed in terms of added Cu) for soil invertebrate species.

Species		Geometric means (mg/kg added Cu)		
Common name	Scientific name	EC_{10} or NOEC	EC_{30} or LOEC	EC_{50}
Earthworm	*Eisenia andrei*	44.3	66.5	133
Earthworm	*Eisenia fetida*	61.4	129	169
Earthworm	*Lumbriculus rubellus*	42.4	117	656
Mite	*Hypoapsis aculeifer*	195	293	586
Mite	*Platynothrus peltifer*	70.7	106	212
Nematode	*Plectus acuminatus*	27.6	86.4	259
Potworm	*Cognettia sphagnetorum*	36.2	61.7	94.6
Springtail	*Folsomia fimetaria*	265	398	630
Springtail	*Folsomia candida*	205	343	499
Springtail	*Isotoma viridis*	135	202	405

Table 49. The lowest geometric mean values of the normalised copper (Cu) toxicity data (expressed in terms of added Cu) for individual plant species.

Plant species		Geometric means (mg/kg added Cu)		
Common name	**Scientific name**	**EC_{10} or NOEC**	**EC_{30} or LOEC**	**EC_{50}**
Annual meadow grass	*Poa annua*	99.4	90.2	140
Barley	*Hordeum vulgare*	47.5	74.6	187
Canola	*Brassica napus*	825	1157	1125
Cotton	*Gossypium sp.*			
Groundsel	*Senico vulgaris*	27.8	56.4	87.7
Maize	*Zea mays*			
Millet	*Panicum milaceum*			
Oats	*Avena sativa*	147	221	442
Peanuts	*Arachis hypogaea*			
Perennial ryegrass	*Lolium perenne*	69.5	374	690
Smooth hawkesbeard	*Hypochoeris radicata*	98.2	164	186
Sorghum	*Sorghum sp.*			
Sugar cane	*Sacharum sp.*			
Tomato	*Lycopersicon esculentum*	126	196	325
Triticale	*Tritosecale sp.*			
Wheat	*Triticum aestivum*			
Wild buckwheat	*Polygonum convolvulus*	124	196	169
Daisy family	*Andryala integrifolia*	75.5	105	127

7.4 Normalisation relationships

A normalisation relationship is an empirical model that predicts the toxicity of a single contaminant to a single species using soil physicochemical properties (for example, soil pH and organic carbon content). Normalisation relationships are used to account for the effect of soil characteristics on toxicity data. Thus, when toxicity data is normalised the effect of soil properties on the toxicity should be removed, so the resulting toxicity data should more closely reflect the inherent sensitivity of the test species.

Eighteen normalisation relationships were reported in the literature for Cu toxicity and an additional two were derived as part of this study (Table 50), giving a total of 20 normalisation relationships. Six were developed for Australian soils (Broos et al. 2007; Warne et al. 2008a; Warne et al. 2008b) and fourteen have been derived for European soils (Oorts et al. 2006a; Rooney et al. 2006; Criel et al. 2008; EC 2008a). Eight of the relationships were for plants, six for soil invertebrates, and six for microbial functions (Table 50).

The choice of normalisation relationships to be used to normalise the toxicity data was based on (1) regional relevance, (2) whether they are based on field- or laboratory-based toxicity data; preference is given to field-based relationships as they provide better estimates of toxicity in the field (Warne et al. 2008b), (3) providing a conservative SQG—normalisation relationships with lower gradients will provide lower normalised toxicity values and thus lower SQGs (EC 2008a), (4) the quality of the relationship as indicated by the coefficient of determination (r^2), and (5) the number of species to which the relationships apply.

Thus, whenever there were appropriate Australian normalisation relationships, these were applied to Australian toxicity data and the same rule applied to European normalisation relationships.

Of the Australian relationships, number 1 was not used as an equivalent field-based relationship for Australian soils was available (relationship 3) and relationship 2 was not used as ultimately it is the amount of harvestable food that is most important when considering crops. The best relationship developed by Broos et al. (2007) for substrate induced nitrification, (SIN) (relationship 4) was based on EC_{50} and pH. However, to be consistent with all the other normalisation relationships developed, the data was re-analysed using the logarithm of the EC50 data, which resulted in relationship 5, used in this Schedule. Relationship 7 was not used as relationships not explaining at least 60% of the variation are not considered appropriate for normalisation (Warne et al. 2008b). Relationship 3 was used to normalise all the Australian plant toxicity data and relationship 5 was used to normalise all the Australian microbial process toxicity data.

Of the European relationships, 8 rather than 7 was used for barley as it contained fewer parameters and had a marginally higher r^2 value. Relationship 11 was used for tomato rather than relationships 9 and 10, as Fe oxide content of soils was not reported in the vast majority of the toxicity data and as relationship 11 had a lower gradient than relationship 10. For *E. Fetida*, relationship 13 was used as it had a lower gradient than relationship 12. Similarly, relationship 16 for *F. candida* was used rather than relationships 14 or 15 as it had a lower gradient.

All the toxicity data for European plant species, apart from barley, was normalised using relationship 11 for tomato as it was the plant relationship with the lowest gradient. All the European invertebrate toxicity data was normalised using relationship 13 for *E. fetida* as it was the invertebrate relationship with the lowest gradient and relationship 18 for SIR was used to normalise all European microbial process toxicity data (except that for maize residue mineralisation and potential nitrification rate) as it was the microbial process relationship with the lowest positive gradient.

All the Cu toxicity data in Tables 47–49 was normalised to its equivalent toxicity in the recommended Australian reference soil (Schedule B5b) (Table 6). Depending on the conditions under which the toxicity tests were conducted, the normalised toxicity data could be higher or lower in the reference soil compared to the original toxicity data in the test soil.

Table 50. Normalisation relationships for the toxicity of copper (Cu) to plants, soil invertebrates and soil processes. The relationships used to normalise the toxicity data are in bold.

Eqn no.	Species/soil process	Y parameter	X parameter(s)	Reference
Australian relationships				
1	*Triticum aestivum* (wheat)	log EC_{10}[a] (laboratory-based data)	0.98 log CEC[b] – 2.97 EC + 2.01 (r^2 adj = 0.79)	Warne et al. 2008a
2	*T. aestivum* (wheat)	log EC_{50} (field-based 8wk growth)	0.54 pH[c] – 0.16 (r^2 adj = 0.85)	Warne et al. 2008b
3	*T. aestivum* (wheat)	log EC_{10} (field-based grain yield)	**0.31 pH[c] + 1.05 log OC + 0.56** (r^2 adj = 0.80)	Warne et al. 2008b
4	SIN	EC_{50}	434 pH[c] – 1615 (r^2 adj = 0.73)	Broos et al. 2007
5	SIN	log EC_{50}	**0.35 pH[c] + 0.84** (r^2 adj = 0.72)	This study
6	SIR	EC_{50}[d]	22 clay + 641 (r^2 adj = 0.38)	Broos et al. 2007
Northern hemisphere relationships				
7	*Hordeum vulgare* (barley)	log EC_{10}[a]	0.403 log CEC[e] + 0.42 OC + 0.809 (r^2 adj = 0.63)	Rooney et al. 2006
8	*H. vulgare* (barley)	log EC_{50}	**1.06 log CEC[e] + 1.42** (r^2 = 0.66)	EC 2008a
9	*Lycopersicon esculentum* (tomato)	log EC_{10}[a]	0.855 log CEC[e] + 0.388 log Fe oxide – 0.047 (r^2 adj = 0.72)	Rooney et al. 2006
10	*L. esculentum* (tomato)	log EC^{10a}	0.99 log CEC[e, f]	EC 2008a
11	*L. esculentum* (tomato)	log EC^{50}	**0.96 log CEC[e] + 1.47** (r^2 = 0.75)	EC 2008a
12	*Eisenia fetida* (earthworm)	log EC_{10}	0.606 log CEC[e] + 1.56 (r^2 = 0.65)	Criel et al. 2008
13	*E. fetida* (earthworm)	log EC_{50}	**0.58 log CEC[e] + 1.85** (r^2 = 0.75)	EC 2008a
14	*Folsomia candida* (collembola)	log EC_{10}	1.16 log CEC[e] + 1.1 (r^2 = 0.54)	Criel et al. 2008

Eqn no.	Species/soil process	Y parameter	X parameter(s)	Reference
15	*F. candida* (collembola)	log EC_{50}	0.96 log CEC^e + 1.63 (r^2 = 0.63)	EC 2008a
16	*F. candida* (springtail)	Log EC_{10}	**0.8475 log CEC^e + 1.499** (r^2 = 0.56)	This study
17	*F. fimetria* (springtail)	Log EC_{10}	0.7508 log CEC^e + 2.0868 (r^2 = 0.63)	This study
18	SIR	log EC_{50}	0.66 log OC + 1.96 (r^2 = 0.57)	Oorts et al. 2006a
19	MRM	log EC_{20}	-0.26 pH^c + 4.05 (r^2 = 0.52)	Oorts et al. 2006a
20	PNR	log EC_{50}	1.06 log CEC^e + 1.41 (r^2 = 0.66)	Oorts et al. 2006a

a = normalisation relationships were also developed for the same combination of species and endpoint but for different measures of toxicity e.g. log EC_{50} and NOEC and using other soil physicochemical properties.

b = these CEC measurements were made using the ammonium acetate method (Rayment & Higginson 1992).

c = pH measured in 0.01 M calcium chloride (Rayment & Higginson 1992).

d = no statistically significant normalisation relationships could be derived for EC_{10} and EC_{10} SIR data (NBRP unpublished data).

e = these CEC measurements were made using the silver thiourea method (Chhabra et al. 1975).

f = the full normalisation relationship was not provided in EC (2008a) but as only the slope of the relationship is used in the normalising, the constant is not necessary. CEC = cation exchange capacity ($cmol_c/kg$); OC = organic carbon content (%); MRM = maize residue mineralisation; PNR = potential nitrification rate; SIN = substrate induced nitrification, SIR = substrate induced respiration.

7.5 Sensitivity of organisms to copper

The distribution of the sensitivity of species and microbial processes to Cu is presented in Figure 7. Toxicity data for plants, soil processes and soil invertebrates was generally evenly spread in the species sensitivity distribution (SSD); however, the invertebrates did not have the same range of highly tolerant species as the other two organism groups. Nonetheless, the overall distribution of sensitivity to Cu was similar. Therefore, all the toxicity data was used to derive the ACLs and SQGs.

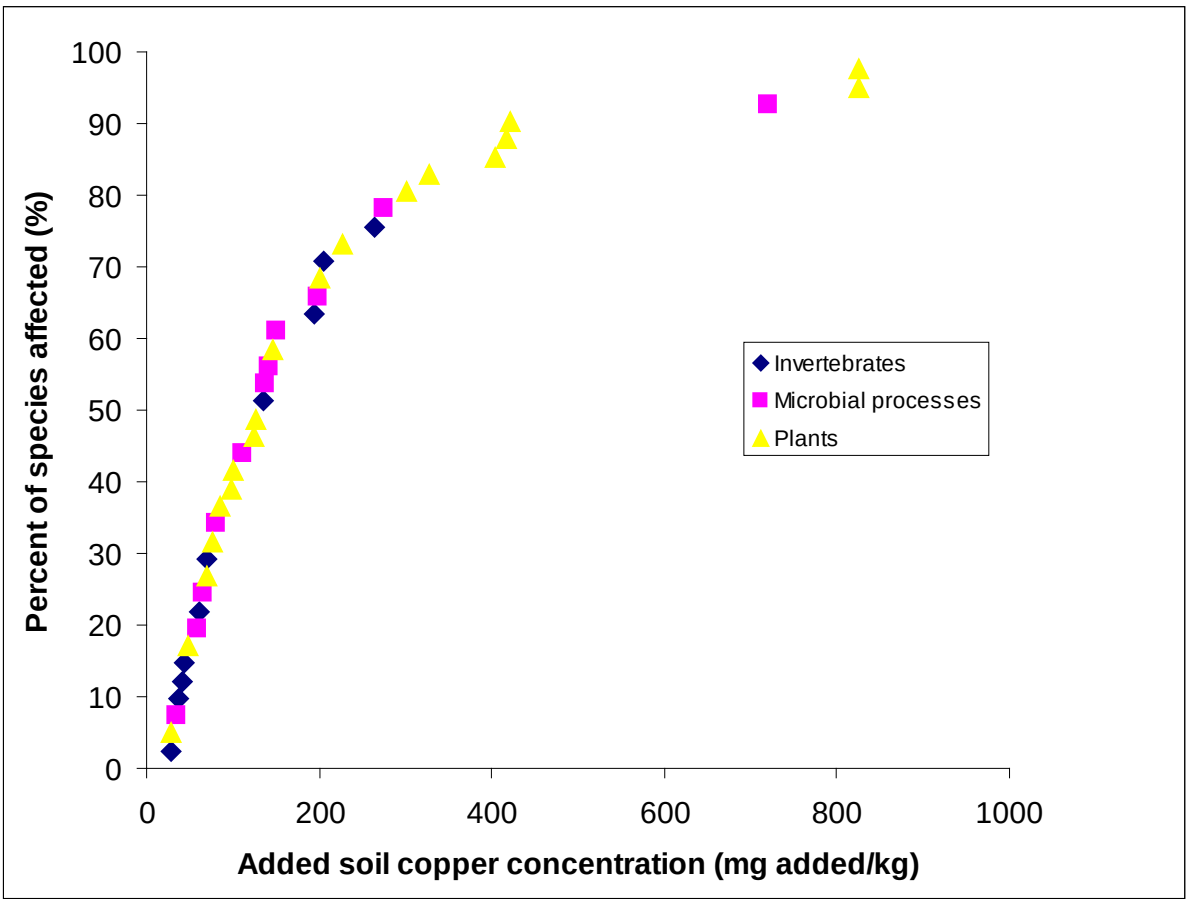


Figure 7. The species sensitivity distribution (plotted as a cumulative frequency against added copper (Cu) concentration) of soil processes, soil invertebrates and plant species to Cu.

7.6 Calculation of soil quality guidelines for fresh copper contamination

As described earlier, SQGs were derived using three sets of toxicity data—NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data.

7.6.1 Calculation of soil quality guidelines for fresh copper contamination based on no observed effect concentration and 10% effect concentration toxicity data

7.6.1.1 Calculation of soil-specific added contaminant limits

The NOEC and EC_{10} toxicity data was normalised as outlined in Heemsbergen et al. (2008). Geometric means for each toxic end point (for example, mortality, reproduction, seedling emergence) for each species were calculated and the lowest geometric mean selected to represent the sensitivity of each species/microbial process. These lowest geometric means were entered into the BurrliOZ software (Campbell et al. 2000) and $ACL_{(NOEC\ \&\ EC10)}$ values calculated that should theoretically protect 99, 80 and 60% of species/microbial processes. The resulting $ACL_{(NOEC\ and\ EC10)}$ values are only applicable to the Australian reference soil (Table 6). In order to generate soil-specific ACLs the normalisation relationships were applied to the $ACL_{(NOEC\ \&\ EC10)}$ values in the reverse manner.

A complicating factor for Cu is that there are different soil physicochemical properties (that is, CEC, pH, OC and a combination of pH and log OC) that control the toxicity of Cu depending on the species or microbial process (Table 50). However, these can be rationalised down to two factors that control the ACL, namely CEC (measured using the silver thiourea method, Chhabra et al. 1975) and pH (measured in 0.01M $CaCl_2$, Rayment & Higginson 1992) (see Appendix F for a detailed explanation of this rationalisation). Thus, there are two sets of ACL values for each land use type (that is, a set that

vary with CEC and a second set that vary with pH). To determine the ACL that applies to a site, it is simply a matter of measuring the CEC and pH of the soil, looking up the tables for the appropriate ACL and then adopting the lower of the two ACL values. In the majority of cases the pH-based ACL values will limit how much Cu can be added to a soil when the soil pH is less than or equal to 6, while the CEC-based ACL values will limit the amount of Cu that can be added to a soil when the soil pH is greater than 6.

The ACL values for areas of ecological significance, urban residential/public open space and commercial/industrial land uses are presented in Tables 51 to 53, respectively.

Table 51. Soil-specific added contaminant limits (ACLs, mg/kg) based on no observed effect concentration (NOEC) and 10% effect concentration (EC_{10}) toxicity data for fresh copper (Cu) contamination that theoretically protect at least 99% of soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.5 to 8 and a cation exchange capacity (CEC) ranging from 5 to 60 $cmol_c$/kg and for an area of ecological significance land use. The lower of the CEC- or the pH-derived ACLs that apply to a soil is the $ACL_{(NOEC\ \&\ EC10)}$ to be used.

Type of ACL	CEC ($cmol_c$/kg)					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	10	20	25	25	25	25
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	7	15	20	30	65	90

Table 52. Soil-specific added contaminant limits (ACLs, mg/kg) based on no observed effect concentration (NOEC) and 10% effect concentration (EC_{10}) toxicity data for fresh copper (Cu) contamination that theoretically protect at least 80% of soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.5 to 8 and a cation exchange capacity (CEC) ranging from 5 to 60 $cmol_c$/kg and an urban residential/public open space land use. The lower of the CEC- or the pH-derived ACLs that apply to a soil is the $ACL_{(NOEC\ \&\ EC10)}$ to be used.

Type of ACL	CEC ($cmol_c$/kg)					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	30	60	65	65	70	70
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	20	40	60	85	170	250

Table 53. Soil-specific added contaminant limits (ACLs, mg/kg) based on no observed effect concentration (NOEC) and 10% effect concentration (EC_{10}) toxicity data for fresh copper (Cu) contamination that theoretically protect at least 60% of soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.5 to 8 and a cation exchange capacity (CEC) ranging from 5 to 60 $cmol_c$/kg and a commercial/industrial land use. The lower of the CEC- or the pH-derived ACLs that apply to a soil is the $ACL_{(NOEC\ \&\ EC10)}$ to be used.

Type of ACL	CEC ($cmol_c$/kg)					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	45	90	100	100	110	110
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	30	60	90	130	270	380

7.6.1.2 Calculation of ambient background concentration values

To convert $ACL_{(NOEC\ \&\ EC10)}$ values to $SQG_{(NOEC\ \&\ EC10)}$ values, the ambient background concentration (ABC) needs to be added to the $ACL_{(NOEC\ \&\ EC10)}$. Three methods of determining the ABC were recommended in the methodology for deriving SQGs (Heemsbergen et al. 2008).
The preferred method is to measure the ABC at an appropriate reference site. However, where this is not possible, the methods of Olszowy et al. (1995) and Hamon et al. (2004) were recommended to predict the ABC where there has been and has not been, respectively, a history of contamination. In the Hamon et al. (2004) method, the ABC for a variety of metal contaminants, including Cu, vary with either the soil iron or manganese content. The equation to predict the ABC for Cu in soils with no history of Cu contamination (Hamon et al. 2004) is:

$$\log \text{Cu conc (mg/kg)} = 0.612 \log \text{Fe content (\%)} + 0.808 \qquad \textit{(equation 7)}$$

Examples of the ABC values predicted by this equation are presented in Table 54.

Table 54. Ambient background concentrations (ABCs) for copper (Cu) predicted using the Hamon et al. (2004) method.

Fe content (%)	Predicted Cu ABC (mg/kg)
0.1	2
0.5	4
1	6
2	10
5	15
10	25
15	35
20	40

Predicted ABC values for Cu range from approximately 2 to 40 mg/kg in soils with iron contents between 0.1 and 20%.

7.6.1.3 Examples of soil quality guidelines for fresh copper contamination based on no observed effect concentration and 10% effect concentration data

To calculate an $SQG_{(NOEC\ \&\ EC10)}$, the ABC value is added to the $ACL_{(NOEC\ \&\ EC10)}$. Ambient background concentration values vary with soil type. Therefore it is not possible to present a single set of SQGs. Thus, two examples of $SQG_{(NOEC\ \&\ EC10)}$ values for urban settings are presented below. These examples would be at the low and high end of the range of $SQG_{(NOEC\ \&\ EC10)}$ values (but not the extreme values) generated for Cu in Australian soils.

Example 1	
Site descriptors – urban residential/public open space land use in a new suburb (that is, fresh Cu contamination).	
Soil descriptors – a sandy acidic soil (pH 5.5, CEC 10) with 1% iron content.	
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:	
$ACL_{(NOEC\ \&\ EC10)}$ CEC-based:	60 mg/kg
$ACL_{(NOEC\ \&\ EC10)}$ pH-based:	40 mg/kg
$ACL_{(NOEC\ \&\ EC10)}$:	40 mg/kg (the lower of the two ACLs that apply to this soil)
ABC:	6 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$:	46 mg/kg, (which would be rounded off to 45 mg/kg).

Example 2	
Site descriptors – commercial/industrial land use in a new suburb (that is, fresh Cu contamination).	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.	
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:	
$ACL_{(NOEC\ \&\ EC10)}$ CEC-based:	110 mg/kg
$ACL_{(NOEC\ \&\ EC10)}$ pH-based:	270 mg/kg
$ACL_{(NOEC\ \&\ EC10)}$:	110 mg/kg (the lower of the two ACLs that apply to this soil)
ABC:	25 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$:	135 mg/kg, which would be rounded off to 130 mg/kg.

7.6.2 Calculation of soil quality guidelines for fresh copper contamination based on lowest observed effect concentration and 30% effect concentration toxicity data, and on 50% effect concentration data

7.6.2.1 Calculation of soil-specific added contaminant limits

In addition to calculating $SQG_{(NOEC\ \&\ EC10)}$ values, Heemsbergen et al. (2008) suggested that two other sets of SQGs could be generated using either a combination of LOEC and EC_{30} data or EC_{50} data. These SQGs are termed the $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ respectively. These additional SQGs were calculated using the method described in Heemsbergen et al. (2008) except the input data for the SSD was changed to the appropriate type (Table 1). The lowest geometric means of the normalised toxicity data used to generate these SQGs are presented in Tables 47–49 and the raw data can be found in Appendix E. Lowest observed effect concentration, 30% effect concentration and 50% effect concentration toxicity data was not available in all instances; therefore, to maximise the data available to calculate $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values, the available NOEC and EC_{10} toxicity data was converted to these measures using conversion factors as necessary. The NBRP developed experimentally derived conversion factors (cited in Heemsbergen et al. 2008) for Cu and Zn (Table 17). These conversion factors were used rather than the generic conversion factors often used to convert toxicity data. This approach is consistent with the recommendation of Heemsbergen et al.

(2008). Tables 55 and 56 show the soil-specific $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ values respectively, for soils with areas of ecological significance, urban residential/public open space and commercial/industrial land uses.

Table 55. Soil-specific ACLs (mg/kg) based on lowest observed effect concentration (LOEC) and 30% effect concentration (EC_{30}) data for fresh copper (Cu) contamination that should theoretically provide the appropriate level of protection (that is, 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.5 to 8 and a CEC ranging from 5 to 60 $cmol_c$/kg for various land uses. The lower of the CEC- or the pH-derived ACLs for a particular land use that apply to a soil is the $ACL_{(LOEC \& EC30)}$ to be used.

Areas of ecological significance land use						
Type of ACL	**CEC ($cmol_c$/kg)[a]**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	25	50	50	55	55	60
	pH[b]					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	15	30	50	70	140	200
Urban residential/public open space land use						
Type of ACL	**CEC($cmol_c$/kg)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	50	100	110	110	120	120
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	30	70	100	140	290	420
Commercial/industrial land use						
Type of ACL	**CEC ($cmol_c$/kg)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	70	150	160	170	170	180
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	45	100	150	210	440	630

a = CEC was measured using the silver thiourea method (Chhabra et al. 1972).

b = pH was measured using the $CaCl_2$ method (Rayment & Higginson 1992).

Table 56. Soil-specific ACLs (mg/kg) based on 50% effect concentration (EC_{50}) data for fresh copper (Cu) contamination that should theoretically provide the appropriate level of protection (that is, 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.5 to 8 and a cation exchange capacity (CEC) ranging from 5 to 60 $cmol_c$/kg for various land uses. The lower of the CEC- or the pH-derived ACLs for a particular land use that apply to a soil is the $ACL_{(EC50)}$ to be used.

Areas of ecological significance land use						
Type of ACL	**CEC ($cmol_c$/kg)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	35	75	85	85	90	95
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	25	50	75	110	230	320
Urban residential/public open space land use						
Type of ACL	**CEC**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	85	170	190	200	200	210
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	50	120	170	250	510	730
Commercial/industrial land use						
Type of ACL	**CEC ($cmol_c$/kg)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	125	260	280	290	310	320
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	80	180	260	380	770	1100

7.6.2.2 Calculation of ambient background concentration values

The ABC values were calculated using the method described earlier and the values presented in Table 54.

7.6.2.3 Examples of soil quality guidelines for fresh copper contamination in Australian soils based on lowest observed effect concentration and 30% effect concentration toxicity data, and on 50% effect concentration data.

As the ACL and ABC values are both soil-specific it is not possible to generate a single set of SQGs. Example SQGs that represent values that at the upper and lower end of the range of values that would be encountered in urban situations are presented. Two examples are presented for SQGs based on LOEC and EC_{30} data and two examples based on EC_{50} data.

$SQG_{(LOEC \& EC30)}$ — Example 1	
Site descriptors – urban residential/public open space land use in a new suburb.	
Soil descriptors – a sandy acidic soil (pH 5.5, CEC 10) with 1% iron content.	
The resulting $ACL_{(LOEC \& EC30)}$, ABC and $SQG_{(LOEC \& EC30)}$ values are:	
$ACL_{(LOEC \& EC30)}$ CEC-based:	100 mg/kg
$ACL_{(LOEC \& EC30)}$ pH-based:	70 mg/kg
$ACL_{(NOEC \& EC10)}$:	70 mg/kg (the lower of the two ACLs that apply to this soil)
ABC:	6 mg/kg
$SQG_{(LOEC \& EC30)}$:	76 mg/kg, which would be rounded off to 75 mg/kg.

$SQG_{(LOEC \& EC30)}$ — Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.	
The resulting $ACL_{(LOEC \& EC30)}$, ABC and $SQG_{(LOEC \& EC30)}$ values are:	
$ACL_{(LOEC \& EC30)}$ CEC-based:	170 mg/kg
$ACL_{(LOEC \& EC30)}$ pH-based:	440 mg/kg
$ACL_{(NOEC \& EC10)}$:	170 mg/kg (the lower of the two ACLs that apply to this soil)
ABC:	25 mg/kg
$SQG_{(LOEC \& EC30)}$:	195 mg/kg, which would be rounded off to 190 mg/kg.

$SQG_{(EC50)}$ — Example 1	
Site descriptors – urban residential/public open space land use in a new suburb.	
Soil descriptors – a sandy acidic soil (pH 5.5, CEC 10) with 1% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$ CEC-based:	170 mg/kg
$ACL_{(EC50)}$ pH-based:	120 mg/kg
$ACL_{(EC50)}$:	120 mg/kg (the lower of the two ACLs that apply to this soil)
ABC:	6 mg/kg
$SQG_{(EC50)}$:	126 mg/kg ,which would be rounded off to 130 mg/kg.

$SQG_{(EC50)}$ - Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$ CEC-based:	310 mg/kg
$ACL_{(EC50)}$ pH-based:	770 mg/kg
$ACL_{(EC50)}$:	310 mg/kg (the lower of the two ACLs that apply to this soil)
ABC:	25 mg/kg
$SQG_{(EC50)}$:	335 mg/kg ,which would be rounded off to 330 mg/kg.

7.7 Calculation of soil quality guidelines for aged copper contamination

7.7.1 Calculation of an ageing and leaching factor for copper

In addition to calculating SQGs in recently contaminated soils (that is, contamination is <2 years old), Heemsbergen et al. (2008) suggested that an identical set of SQGs could be derived for soils where the contamination is aged (that is, it has been present for ≥2 years). The Cu $SQG_{(NOEC\ \&\ EC10)}$, $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values for aged sites were calculated using the methods set out in earlier sections, the only difference being that laboratory toxicity data based on freshly spiked soils or soils that had not been leached were multiplied by an ALF (Schedule B5b). An ALF of 2 was developed by Smolders et al. (2009) while a value of 2.2 was developed and used in the EC ecological risk assessment for Cu (EC 2008a). Given the uniformity of these ALF values and to err on the conservative side (that is to offer greater protection to the environment), an ALF of 2 was adopted in this study.

7.7.2 Calculation of soil quality guidelines for aged copper contamination based on no observed effect concentration and 10% effect concentration toxicity data

7.7.2.1 Calculation of soil-specific added contaminant limits

The raw toxicity data (Appendix E) for Cu that was generated using freshly spiked and non-leached soils was multiplied by the ALF of 2. That data that was field-based and aged and/or leached laboratory-based data was not multiplied by the ALF. In all other ways the aged $ACL_{(NOEC\ \&\ EC10)}$ and $SQG_{(NOEC\ \&\ EC10)}$ values were calculated using the same methods as described in earlier sections. The resulting soil-specific $ACL_{(NOEC\ \&\ EC10)}$ values for aged Cu contamination are presented in Table 57.

Table 57. Soil-specific ACLs (mg/kg) based on no observed effect concentration (NOEC) and 10% effect concentration (EC_{10}) data for aged copper (Cu) contamination that should theoretically provide the appropriate level of protection (i.e., 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.5 to 8 and a CEC ranging from 5 to 60 $cmol_c$/kg for various land uses. The lower of the CEC- or the pH-derived ACLs for a particular land use that apply to a soil is the aged $ACL_{(NOEC\ \&\ EC10)}$ to be used.

Areas of ecological significance land use						
Type of ACL	**CEC ($cmol_c$/kg)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	15	25	30	30	30	35
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	8	20	25	40	80	110
Urban residential/public open space land use						
Type of ACL	**CEC**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	50	110	110	120	120	130
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	30	70	110	150	310	440

Commercial/industrial land use						
Type of ACL	CEC ($cmol_c/kg$)					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	80	160	180	180	190	200
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	50	110	160	230	480	680

7.7.2.2 Calculation of ambient background concentration values

For aged contaminated sites (that is, the contamination has been in place for at least 2 years) the methodology (Schedule B5b) recommends using the 25th percentiles of the ABC data for the 'old suburbs' from Olszowy et al. (1995) (see Table 58).

Table 58. Copper (Cu) ambient background concentrations (ABC) based on the 25th percentiles of Cu concentrations in 'old suburbs' (that is, >2 years old) from various states of Australia (Olszowy et al. 1995).

Suburb type	25th percentile of Cu ABC values (mg/kg)			
	NSW	**QLD**	**SA**	**VIC**
Old suburb, low traffic	20	10	15	10
Old suburb, high traffic	30	15	25	10

7.7.2.3 Examples of soil quality guidelines for aged copper contamination in Australian soils based on no observed effect concentration and 10% effect concentration data.

SQGs are the sum of the ABC and ACL values, both of which are soil-specific. It is, therefore, not possible to present a single set of SQGs. Thus, some examples of $SQG_{(NOEC\ \&\ EC10)}$ values for aged urban soils are provided below. These examples represent $SQG_{(NOEC\ \&\ EC10)}$ values that would be at the low and high end of the range of $SQG_{(NOEC\ \&\ EC10)}$ values that would be generated for Cu in Australian soils, but are not extreme values.

Example 1

Site descriptors – urban residential land /public open space use in an old Victorian suburb with low traffic volume.

Soil descriptors – a sandy acidic soil (pH 5.5, CEC 10) with 1% iron and aged Cu contamination and a low traffic volume.

The resulting aged $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:

aged $ACL_{(NOEC\ \&\ EC10)}$ CEC-based:	110 mg/kg
aged $ACL_{(NOEC\ \&\ EC10)}$ pH-based:	70 mg/kg
aged $ACL_{(NOEC\ \&\ EC10\ \ \ )}$:	70 mg/kg (the lower of the two ACLs that apply to this soil)
aged ABC:	10 mg/kg
aged $SQG_{(NOEC\ \&\ EC10)}$:	80 mg/kg

Example 2
Site descriptors – commercial/industrial land use in an old South Australian suburb with a high traffic volume.
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron and aged Cu contamination.
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:
aged $ACL_{(NOEC\ \&\ EC10)}$ CEC-based: 190 mg/kg
aged $ACL_{(NOEC\ \&\ EC10)}$ pH-based: 480 mg/kg
aged $ACL_{(NOEC\ \&\ EC10)}$: 190 mg/kg (the lower of the two ACLs that apply to this soil)
aged ABC: 25 mg/kg
aged $SQG_{(NOEC\ \&\ EC10)}$: 215 mg/kg, which would be rounded off to 210 mg/kg.

7.7.3 Calculation of soil quality guidelines for aged copper contamination based on LOEC and 30% effect concentration toxicity data, and on 50% effect concentration data.

7.7.3.1 Calculation of soil-specific added contaminant limits

The $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ values for aged Cu contamination were calculated in the same manner as the aged $ACL_{(NOEC\ \&\ EC10)}$ values, except that LOEC and EC_{30} or EC_{50} toxicity data was used respectively. The aged $ACL_{(LOEC\ \&\ EC30)}$ and aged $ACL_{(EC50)}$ values are presented in Tables 59 and 60 respectively.

Table 59. Soil-specific added contaminant limits (ACLs, mg/kg) based on LOEC and 30% effect concentration (EC_{30}) data for aged copper (Cu) contamination that should theoretically provide the appropriate level of protection (i.e. 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.5 to 8 and a CEC ranging from 5 to 60 $cmol_c$/kg for various land uses. The lower of the CEC- or the pH-derived ACLs for a particular land use that apply to a soil is the aged $ACL_{(LOEC\ \&\ EC30)}$ to be used.

Areas of ecological significance land use						
Type of ACL	**CEC ($cmol_c$/kg)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	30	65	70	70	75	80
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	20	45	65	90	190	270
Residential urban /public open space land use						
Type of ACL	**CEC ($cmol_c$/kg)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	95	190	210	220	220	230
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	60	130	190	280	560	800
Commercial/industrial land use						
Type of ACL	**CEC ($cmol_c$/kg)**					

	5	10	20	30	40	60
CEC-based ACLs	140	280	300	320	330	340
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH-based ACLs	85	190	280	400	830	1200

Table 60. Soil-specific ACLs (mg/kg) based on 50% effect concentration (EC_{50}) data for aged copper (Cu) contamination that should theoretically provide the appropriate level of protection (i.e. 99, 80 or 60% of species) to soil processes, soil invertebrate species and plant species in soils with a pH ranging from 4.5 to 8 and a CEC ranging from 5 to 60 $cmol_c/kg$ for various land uses. The lower of the CEC- or the pH-derived ACLs for a particular land use that apply to a soil is the aged $ACL_{(EC50)}$ to be used.

Areas of ecological significance land use						
Type of ACL	**CEC ($cmol_c/kg$)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	80	170	180	190	190	200
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH -based ACLs	50	110	170	240	490	700
Urban residential /public open space land use						
Type of ACL	**CEC ($cmol_c/kg$)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	150	300	350	350	350	400
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH -based ACLs	95	200	300	450	900	1300
Commercial/industrial land use						
Type of ACL	**CEC ($cmol_c/kg$)**					
	5	**10**	**20**	**30**	**40**	**60**
CEC-based ACLs	210	440	470	490	510	530
	pH					
	4.5	**5.5**	**6**	**6.5**	**7.5**	**8.0**
pH -based ACLs	130	290	440	630	1300	1800

7.7.3.2 Calculation of ambient background concentration values

The ABC values for aged Cu contamination were calculated using the data from Olszowy et al. (1995), and are presented in Table 58.

7.7.3.3 Examples of soil quality guidelines for aged copper contamination in Australian soils based on lowest observed effect concentration and 30% effect concentration data

Four examples of SQGs that would apply to aged Cu contamination that represent the range (but not the extremes) of SQGs that would apply to urban residential/public open space and commercial/industrial land uses are presented below.

$SQG_{(LOEC\ \&\ EC30)}$ – Example 1	
Site descriptors – urban residential land/public open space use in an old Victorian suburb with a low traffic volume.	
Soil descriptors – a sandy acidic soil (pH 5.5, CEC 10) with 1% iron content.	
The resulting aged $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
aged $ACL_{(LOEC\ \&\ EC30)}$ CEC-based:	190 mg/kg
aged $ACL_{(LOEC\ \&\ EC30)}$ pH-based:	130 mg/kg
aged $ACL_{(LOEC\ \&\ EC30)}$:	130 mg/kg (the lower of the two ACLs that apply to this soil)
aged ABC:	10 mg/kg
aged $SQG_{(LOEC\ \&\ EC30)}$:	140 mg/kg

$SQG_{(LOEC\ \&\ EC30)}$ – Example 2	
Site descriptors – commercial/industrial land use in an old South Australian suburb with a high traffic volume.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
aged ACL(LOEC & EC30) CEC-based:	330 mg/kg
aged ACL(LOEC & EC30) pH-based:	830 mg/kg
aged ACL(LOEC & EC30):	330 mg/kg (the lower of the two ACLs that apply to this soil)
aged ABC:	25 mg/kg
aged SQG(LOEC & EC30):	355 mg/kg, which would be rounded off to 350 mg/kg.

$SQG_{(EC50)}$ – Example 1	
Site descriptors – urban residential land/public open space use in an old Victorian suburb with a low traffic volume.	
Soil descriptors – a sandy acidic soil (pH 5.5, CEC 10) with 1% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$ CEC based:	300 mg/kg
$ACL_{(EC50)}$ pH based:	200 mg/kg
$ACL_{(EC50)}$:	200 mg/kg (the lower of the two ACLs that apply to this soil)
ABC:	10 mg/kg
$SQG_{(EC50)}$:	210 mg/kg

$SQG_{(EC50)}$ – Example 2	
Site descriptors – commercial/industrial land use in an old South Australian suburb with a high traffic volume.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with a 10% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$ CEC based:	510 mg/kg
$ACL_{(EC50)}$ pH based:	1300 mg/kg
$ACL_{(EC50)}$:	510 mg/kg (the lower of the two ACLs that apply to this soil)
ABC:	25 mg/kg
$SQG_{(EC50)}$:	535 mg/kg, which would be rounded off to 530 mg/kg.

7.8 Reliability of the soil quality guidelines

Based on the criteria established in the methodology for SQG derivation (Schedule B5b), all the Cu SQGs were considered to be of high reliability. This resulted as the toxicity data set easily met the minimum data requirements to use the SSD method and there were normalisation relationships available to account for soil characteristics.

7.9 Comparison with other guidelines

A compilation of SQGs for Cu from a number of jurisdictions is presented in Table 61. These SQGs have a variety of purposes and levels of protection and therefore comparison of the SQGs amongst each other and with the Cu SQGs is problematic. As well, the vast majority of the international SQGs are not soil-specific nor do they account for ageing and leaching. One would therefore expect that the ACLs could be higher than other internationals SQGs. The international guidelines for Cu range from 14 to 1,000 mg/kg (added or total Cu) both being from member countries of the European Union (Carlon 2007). The superseded interim urban EIL (NEPC 1999) for Cu was 100 mg/kg total Cu and therefore in the middle of the range of the international Cu guidelines.

Overall, the superseded interim urban EIL lies in the lower to middle part of the range of ACLs for fresh Cu contamination, while the superseded interim urban EIL lies at the lower third of the range of ACLs for aged contamination.

All of the soil-specific ACL values for urban residential land/public open space land use (irrespective of the toxicity data on which they were based) fell within the range of the international residential SQGs, the one exception being the ACLs based on EC_{50} for soils where the Cu has low bioavailability (that is, high pH and high CEC), which were greater than 1,000 mg/kg added Cu.

However, this was a CEC-based ACL and, as stated earlier, when the soil pH is greater than 6, the pH-based ACLs will limit the amount of Cu that can be present in soil. When this was taken into account, all the soil-specific ACL values for residential land use fell within the range of international SQGs.

Similarly, all the ACLs for commercial/industrial land use, with the exception of the aged ACLs based on EC_{50}, fell within the range of international SQGs for Cu. The one exception was the $ACL(EC_{50})$ value that would permit concentrations nearly twice (that is, 1,800 mg/kg added) that of the collated international limits (1,000 mg/kg). However, in soils with a pH above 6, the pH-based ACL will limit the amount of Cu that is permitted in soil and thus all the ACLs for commercial/industrial land use fell within the range of international SQGs.

The Cu $ACL_{(NOEC\ \&\ EC10)}$ values in freshly contaminated urban residential/public open space soils (which should theoretically protect 80% of species) ranged from 20 to 250 mg/kg (added Cu) (Table 53). The most suitable comparison with these values is with the limits recommended by the EC Cu ecological risk assessment which used NOEC and EC_{10} data and should theoretically protect 95% of

species. These values range from 20 to 173 mg/kg added Cu. The limits derived by these two processes are very similar.

Table 61. Soil quality guidelines for copper (Cu) from international jurisdictions.

Name of Cu limit	Numerical value of the limit (mg/kg)
Dutch target value[1]	36 (added Cu)
Dutch intervention level[1]	190 (added Cu)
Canadian SQG (residential)[2]	63 (total Cu)
Canadian SQG (commercial and industrial)[2]	91 (total Cu)
Eco-SSL plants[3]	70 (total Cu)
Eco-SSL soil invertebrates[3]	80 (total Cu)
Eco-SSL avian[3]	28 (total Cu)
Eco-SSL mammalian[3]	49 (total Cu)
EU minimal risk values (residential)[4]	14–70 (added and total Cu)
EU warning risk values (residential)[4]	100–500 (added and total Cu)
EU potential risk values (residential)[4]	100–1000 (added and total Cu)
EU Cu ecological risk assessment[5]	26–176 (added Cu)

1 = VROM 2000
2 = CCME 1999e, & 2006 and http://ceqg-rcqe.ccme.ca/
3 = http://www.epa.gov/ecotox/ecossl/
4 = Carlon 2007
5 = EC 2008a.

8 Lead

8.1 Lead compounds considered

The following compounds were considered in deriving the SQGs for lead (Pb):

- lead metal (CAS No. 7439-92-1)
- lead oxide (CAS Nos 1317-36-8)
- lead tetroxide (CAS No. 1314-41-6)
- dibasic lead phthalate (CAS No: 69011-06-9)
- basic lead sulphate (CAS No: 12036-76-9)
- tribasic lead sulphate (CAS No: 12202-17-4)
- tetrabasic lead sulphate (CAS No: 12065-90-6)
- neutral lead stearate (CAS No: 1072-35-1)
- dibasic lead stearate (CAS No: 12578-12-0)
- dibasic lead phosphite (CAS No: 12141-20-7)
- polybasic lead fumarate (CAS No: 90268-59-0)
- basic lead carbonate (CAS No: 1319-46-6)
- basic lead sulphite (CAS No: 62229-08-7).

8.2 Exposure pathway assessment

If the logarithm of the K_d (log K_d) of an inorganic contaminant is less than 3 then it is considered to have the potential to leach to groundwater (Schedule B5b). The log K_d reported by Commentuijn et al. (2000) for Pb was 3.28 L/kg so there is little potential for Pb to leach to groundwater. If this exposure pathway were considered important at a site, then the methodology for SQG derivation advocates that this be addressed on a site-specific basis as appropriate (Schedule B5b).

The bioconcentration, bioaccumulation and biomagnification of Pb in aquatic ecosystems have received considerable attention. There has also been considerable attention paid to bioconcentration in terrestrial ecosystems but the biomagnification work has been more limited and often restricted to only examining transfer from food to consumer and not subsequent steps up food chains. One hundred and one terrestrial bioaccumulation factor (BAF) values for Pb have been published (LDA 2008) and these range from 0.00 to 6.86 with a median value of 0.1 kgdw/kgww (where dw = dry weight and ww = wet weight). The EU ecological risk assessment for Pb (LDA 2008) followed the EU technical guidance document (EC 1996), which applies assessment factors to the lowest NOEC for oral exposure of birds and mammals to account for the potential of Pb to biomagnify. However, using this method led to the derivation of limits that were below the concentrations found in control foods (that is, food that would occur in soils with background concentrations of Pb). These limits therefore imply that food (animal or plant) grown in soils with background concentrations poses a risk, which is not consistent with real-world experience. They therefore used an SSD method to determine the predicted no-effect concentration (PNEC) for oral exposure of birds and mammals and obtained a soil limit of 491 mg/kg. This value was higher than the limit based on direct exposure of soil organisms of 333 mg/kg.

Thus, it is apparent that Pb does not pose a biomagnification risk to terrestrial ecosystems. This finding is consistent with the findings for aquatic ecosystems that Pb does not biomagnify (Eisler 1988; Suedel et al. 1994; Demayo et al. 1982; Vighi 1981; Lu et al. 1975; Henney et al. 1991) and is the conclusion reached by the EU Pb ecological risk assessment (LDA 2008). Therefore, only direct toxic effects to soil organisms were considered in the derivation of the SQGs.

8.3 Toxicity data

All the available Pb toxicity data was reported with both the total concentration and ambient background concentration, therefore the data could be converted to added concentrations. A total of ninety-six toxicity measures were available for Pb. These were for eight plant species, five species of soil invertebrates and six microbial processes (Table 62). Thus, this met the minimum data requirements recommended by Heemsbergen et al. (2008) to use the BurrliOZ SSD method (Campbell et al. 2000). Table 62 shows the geometric means of toxicity values of each species or soil microbial process that were used to derive the SQGs for Pb. The raw toxicity data used to generate the species geometric means is presented in Appendix G. In the vaxt majority of cases the geometric means of the toxicity data increase from NOEC or EC_{10} to LOEC or EC_{30} to EC_{50} values. However, for *F. candida*, *Raphanus sativa*, *A. sativa*, *P. tedea* and *L. Sativa*, the EC_{50} values were lower than the LOEC and EC_{30} data. This reflects the fact that the Pb toxicity data was not normalised for soil properties and the toxicity tests were conducted in soils with a variety of physicochemical properties.

In order to maximise the use of the available toxicity data, conversion factors recommended in Schedule B5b to permit the inter-conversion of NOEC, LOEC, EC_{50}, EC_{30} and EC_{10} data were used (Table 17).

Table 62. Geometric means of the toxicity of lead (Pb) (expressed in terms of added Pb) to soil invertebrates, plants and soil microbial processes.

Test species		Geometric mean (mg/kg)		
Common name	**Scientific name**	**NOEC or EC_{10}**	**LOEC or EC_{30}**	**EC_{50}**
Invertebrates				
Earthworm	*Dendrobaena rubida*	129	194	387
Earthworm	*Eisenia andrei*	-	1500	3410
Earthworm	*E. fetida*	761	2026	3829
Earthworm	*L. rubellus*	1000	1500	3000
Springtail	*F. candida*	1797	3749	1866
Microbial processes				
Soil process	ATP	-	-	3018
Soil process	Denitrification	250	500	750
Soil process	Nitrification	337	505	1010
Soil process	N-mineralisation	447	1095	1342
Soil process	Respiration	655	982	1964
Soil process	Substrate induced respiration	1733	2600	5200
Plants				
Radish	*Raphanus sativus*	100	500	300
Oat	*A. sativa*	100	500	300
Barley	*H. vulgare*	50	250	1270
Red spruce	*Picea rubens*	141	212	1228
Loblolly pine	*Pinus taeda*	546	819	659
Lettuce	*Latuca sativa*	125	188	174
Wheat	*T. aestivum*	250	500	750
Maize	*Z. mays*	100	150	300

8.4 Normalisation relationships

Only two normalisation relationships have been developed for Pb. One models the uptake of Pb by spring wheat (*T. aesitivum*) (Nan et al. 2002) while the other models Pb toxicity to lettuce (*L. sativa*) (Hamon et al. 2003). The toxicity normalisation relationship is presented below:

EC_{50} = 23 pH + 171 clay content (%) - 40 (r^2 = 0.84) *(equation 8)*

However, while the above relationship is based on ten toxicity data sets, they were only tested in five soils. This, combined with the fact that the relationship was not validated, severely limits its applicability. The EU ecological risk assessment for Pb (LDA 2008) stated that there is no relationship between soil pH and Pb toxicity. However, it did not make any statement on whether there are relationships between Pb toxicity and other soil physicochemical properties. This was examined as part of this body of work. Relationships between the logarithm of NOEC and/or EC_{10} data and soil pH, log organic matter content (%), log organic carbon content (%), log clay content (%) and log cation exchange capacity (CEC) for all toxicity data combined, for plants only, for invertebrates only and for soil microbial processes only were determined (data not shown). Normalisation relationships were only derived using NOEC and EC_{10} data as there was considerably more of this data than LOEC and EC_{30} or EC_{50} data. Only the relationship between logarithm of Pb toxicity to plants and the logarithm of the organic carbon content was able to explain more than 50% of the variation in toxicity data (r^2 = 0.56).

Normalisation relationships that explain such a low percentage of the variation (that is, <60%) are not usually used to normalise toxicity data as they do not account for enough of the variability caused by the soil (Warne et al. 2008b). The majority of the relationships derived explained less than 10% of the variation in toxicity data and only three could explain more than 10%. Thus there are no useful normalisation relationships available for Pb, so the toxicity data was not normalised to the Australian reference soil, nor were soil-specific SQGs derived.

8.5 Sensitivity of organisms to lead

The SSD for the Pb NOEC toxicity data is presented in Figure 8. There was only toxicity data for 19 different species/microbial processes and the available data has not been normalised; therefore, the distribution reflects the variability in sensitivity of the organisms and the effect of soil properties. There was insufficient data to make a robust assessment of the relative sensitivity of the groups of organisms. However, the distributions of all three types of organisms overlap, so it was considered appropriate to use all the toxicity data to derive the SQGs.

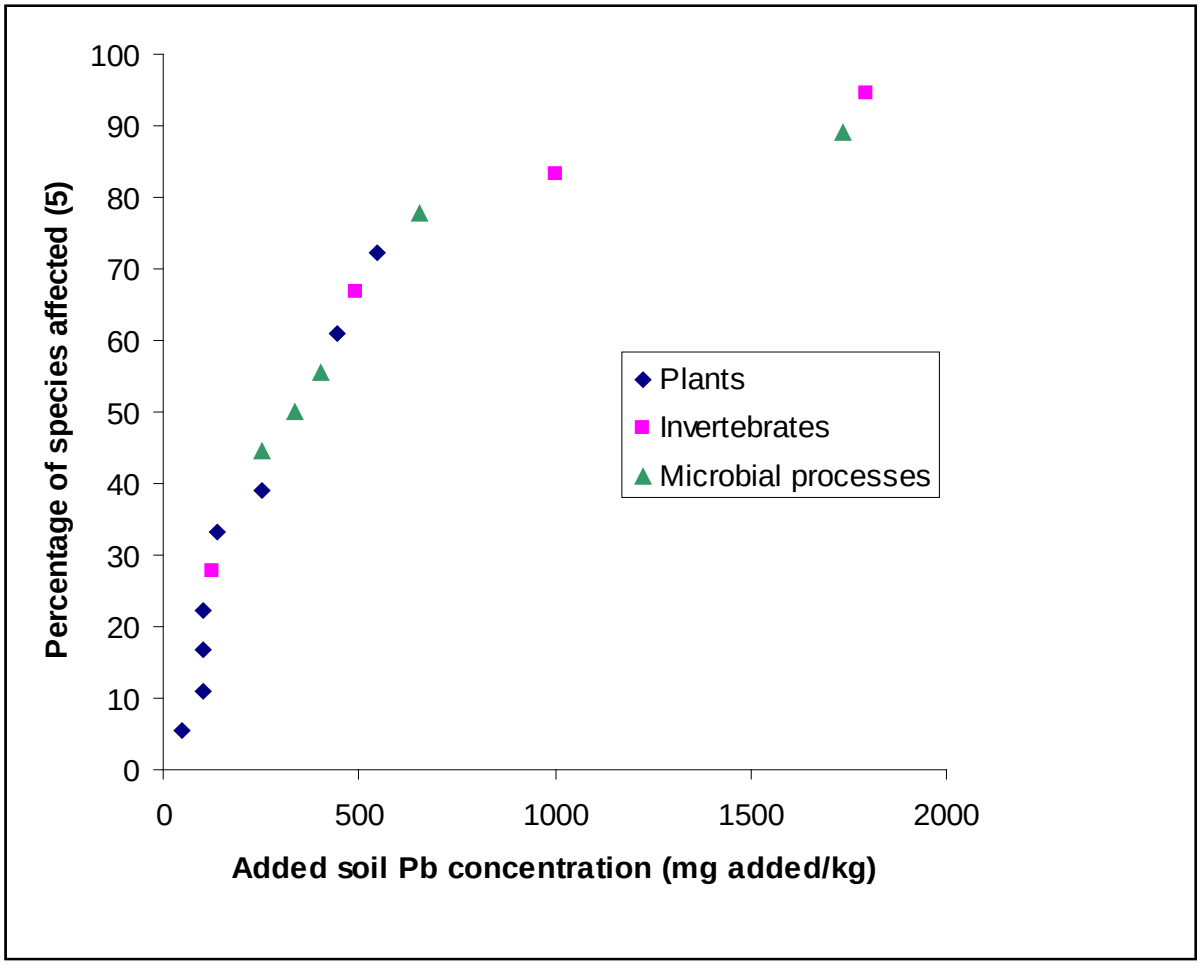


Figure 8. The species sensitivity distribution of fresh lead (Pb) contamination (plotted as a cumulative frequency of the Pb NOEC toxicity data against soil Pb concentration) for soil invertebrates, plants and microbial processes.

8.6 Calculation of soil quality guidelines for fresh lead contamination

There was NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} Pb toxicity data so ACLs and SQGs could be derived using each of these datasets. These were generated using the same general methods as for Cu.

8.6.1 Calculation of soil quality guidelines for fresh lead contamination based on NOEC and 10% effect concentration toxicity data

8.6.1.1 Calculation of soil-specific added contaminant limits

There were no normalisation relationships available for Pb and therefore the NOEC and EC_{10} toxicity data was not normalised, nor could soil-specific ACL values be derived. The single numerical output from the SSD analysis for each land use became the generic (not soil-specific) ACL for that land use and these are presented in Table 63.

Table 63. Generic ACL (mg/kg) values based on NOEC and 10% effect concentration toxicity data (EC_{10}) for fresh lead (Pb) contamination in soil with various land uses.

Land use	$ACL_{(NOEC\ \&\ EC10)}$ (mg/kg)
Areas of ecological significance	40
Urban residential/public open space	130
Commercial/industrial	220

8.6.1.2 Calculation of ambient background concentration values

For sites with no history of contamination, the method of Hamon et al. (2004) is recommended to estimate the ABC. The equation to predict the Pb ABC is

$$\log \text{Pb conc (mg/kg)} = 1.039 \log \text{Fe content (\%)} + 0.118 \qquad \textit{(equation 9)}$$

Examples of the ABC values predicted by this equation are presented in Table 64. Predicted ABC values for Pb range from approximately 0.1 to 30 mg/kg in soils with iron concentrations between 0.1 and 20%.

Table 64. Lead (Pb) ABCs predicted using the method of Hamon et al. (2004) (see equation 9 above).

Fe content (%)	Predicted ABC (mg/kg)
0.1	0.1
0.5	0.6
1	1
2	3
5	7
10	15
15	20
20	30

8.6.1.3 Examples of soil quality guidelines for fresh lead contamination in Australian soils based on no observed effect concentration and 10% effect concentration data

The ABC values for Pb vary with the iron content of the soil. Therefore, it is not possible to present a specific set of $SQGs_{(NOEC \& EC10)}$, but rather two examples of the range of SQGs that will be encountered in urban settings are presented.

Example 1	
Site descriptors – urban residential land/public open space use in a new suburb (i.e. fresh contamination).	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.	
The resulting $ACL_{(NOEC \& EC10)}$, ABC and $SQG_{(NOEC \& EC10)}$ values are:	
$ACL_{(NOEC \& EC10)}$:	130 mg/kg
ABC:	1 mg/kg
$SQG_{(NOEC \& EC10)}$:	131 mg/kg, which would be rounded off to 130 mg/kg.

Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.	
The resulting $ACL_{(NOEC \& EC10)}$, ABC and $SQG_{(NOEC \& EC10)}$ values are:	
$ACL_{(NOEC \& EC10)}$:	220 mg/kg
ABC:	15 mg/kg
$SQG_{(NOEC \& EC10)}$:	235 mg/kg, which would be rounded off to 230 mg/kg.

8.6.2 Calculation of soil quality guidelines for fresh lead contamination based on LOEC and 30% effect concentration toxicity data and on 50% effect concentration data

8.6.2.1 Calculation of soil-specific added contaminant limits

ACLs based on LOEC and EC_{30} toxicity data ($ACL_{(LOEC\ \&\ EC30)}$) and based on EC_{50} data ($ACL_{(EC50)}$) were calculated using the method used to derive the ACL values based on NOEC and EC_{10} data, the one exception being that in order to maximise the amount of LOEC and EC_{30} and EC_{50} data, actual measured NOEC data was used to estimate LOEC, EC_{30} and EC_{50} data. This was done using the conversion factors derived by Heemsbergen et al. (2008) and presented in Table 17. The geometric means of the LOEC and EC_{30} data and of the EC_{50} data for the various species/microbial processes that were used to derive the $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ are presented in Table 62.

The resulting $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(\mathbf{EC50})}$ values for the three land uses are presented in Table 65. As expected, these values are larger than the corresponding $ACL_{(NOEC\ \&\ EC10)}$ values. The $ACL_{(EC50)}$ values are also generally larger than the $ACL_{(LOEC\ \&\ EC30)}$ values, with the exception of the values for areas of ecological significance. This occurs because the slope of the SSD for the LOEC and EC_{30} data is less than that of the EC_{50} data, the SSDs intersect and the LOEC and EC_{30} data ends up having larger toxicity values.

Table 65. Generic ACLs (mg/kg) based on LOEC and 30% effect concentration data (EC30) and based on 50% effect concentration data (EC50) values for fresh lead (Pb) contamination in soil with various land uses.

Land use	$ACL_{(LOEC\ \&\ EC30)}$ (mg/kg)	$ACL_{(EC50)}$ (mg/kg)
Areas of ecological significance	110	60
Urban residential/public open space	270	490
Commercial/industrial	440	890

8.6.2.2 Calculation of ambient background concentration values

The ABC values for Pb were calculated using the Hamon et al. (2004) method as outlined previously.

8.6.2.3 Examples of soil quality guidelines for fresh lead contamination in Australian soils based on lowest observed effect concentration and 30% effect concentration data and on 50% effect concentration data

As stated previously, the ABC values for Pb vary with the iron content of the soil. Therefore it is not possible to present a specific set of $SQG_{(LOEC\ \&\ EC30)}$ or $SQG_{(EC50)}$ values. Four examples of SQGs that would apply to aged Pb contamination that represent the range (but not the extremes) of SQGs that would apply to urban residential/public open space and commercial/industrial land uses are presented below.

$SQG_{(LOEC\ \&\ EC30)}$ Example 1	
Site descriptors – urban residential land/public open space use in a new suburb (that is, fresh contamination).	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	270 mg/kg
ABC:	1 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	271 mg/kg, which would be rounded off to 270 mg/kg.

$SQG_{(LOEC \& EC30)}$ Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.	
The resulting $ACL_{(LOEC \& EC30)}$, ABC and $SQG_{(LOEC \& EC30)}$ values are:	
$ACL_{(LOEC \& EC30)}$:	440 mg/kg
ABC:	15 mg/kg
$SQG_{(LOEC \& EC30)}$:	455 mg/kg, which would be rounded off to 450 mg/kg.

$SQG_{(EC50)}$ Example 1	
Site descriptors – urban residential land/public open space use in a new suburb (that is, fresh contamination).	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	490 mg/kg
ABC:	1 mg/kg
$SQG_{(EC50)}$:	491 mg/kg, which would be rounded off to 490 mg/kg.

SQG(EC50) Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	890 mg/kg
ABC:	15 mg/kg
$SQG_{(EC50)}$:	905 mg/kg, which would be rounded off to 900 mg/kg.

8.7 Calculation of soil quality guidelines for aged lead contamination

8.7.1 Calculation of an ageing and leaching factor

Smolders et al. (2009) examined the literature and developed ALFs for Pb for a range of different organisms. The resulting ALFs ranged from 1.1 to 43 with a median of 4.2. The value of 4.2, recommended by Smolders et al. (2009), was adopted and used in the EU ecological risk assessment of Pb (LDA 2008). Leaching factors for Pb have been developed for five Australian soils from South Australia, which ranged from 0.92 to 2.98 and a median and geometric mean of 1.66 and 1.61 respectively (Stevens et al. 2003).

Given the values of Stevens et al. (2003) only account for leaching and not ageing, it is likely any ALFs for Australian soils would be larger and therefore are likely to be consistent with the ALF of Smolders et al. (2009). An ALF of 4.2 was adopted in this project to calculate the SQGs for aged Pb contamination.

8.7.2 Calculation of soil quality guidelines for aged lead contamination based on NOEC and 10% effect concentration toxicity data

8.7.2.1 Calculation of soil-specific added contaminant limits

The ACL values for aged contamination were calculated in exactly the same manner as those for fresh contamination except that the NOEC and EC_{10} toxicity data was corrected using the Smolders et al. (2009) ALF of 4.2. The resulting ACL values are presented in Table 66.

Table 66. Generic ACLs (mg/kg) based on NOEC data and 10% effect concentration data (EC_{10}) for aged lead (Pb) contamination in soil with various land uses.

Land use	$ACL_{(NOEC \& EC10)}$ (mg/kg)
Areas of ecological significance	170
Urban residential/public open space	530
Commercial/industrial	940

8.7.2.2 Calculation of ambient background concentration values

For aged contaminated sites (that is, the contamination has been in place for at least 2 years), the methodology (Schedule B5b) recommends using the 25th percentiles of the ABC data for the 'old suburbs' from Olszowy et al. (1995) (see Table 67).

Table 67: Lead (Pb) ABCs based on the 25th percentiles of Pb concentrations in 'old suburbs' (i.e. >2 years old) from various states of Australia (Olszowy et al. 1995).

Suburb type	25th percentile of Pb ABC values (mg/kg)			
	NSW	QLD	SA	VIC
Old suburb, low traffic	100	30	30	35
Old suburb, high traffic	160	150	90	70

8.7.2.3 Examples of soil quality guidelines for aged lead contamination in Australian soils based on no observed effect concentration and 10% effect concentration data.

As the ABC values for Pb vary with the geographical location of the site it is not possible to present a single set of $SQG_{(NOEC \& EC10)}$ values. Instead, two examples of the range of SQGs that will be encountered in urban settings are presented below.

Example 1
Site descriptors – urban residential land/public open space use in an old South Australian suburb (that is, contamination is >2 years old), with low traffic volume.
Soil descriptors – these are not relevant as soil properties are not considered in determining the ACL for Pb.
The resulting $ACL_{(NOEC \& EC10)}$, ABC and $SQG_{(NOEC \& EC10)}$ values are:
$ACL_{(NOEC \& EC10)}$: 530 mg/kg
ABC: 30 mg/kg
$SQG_{(NOEC \& EC10)}$: 560 mg/kg

Example 2	
Site descriptors – commercial/industrial land use in an old Queensland suburb (that is, contamination is >2 years old), with high traffic volume.	
Soil descriptors – these are not relevant as soil properties are not considered in determining the ACL for Pb.	
The resulting $ACL_{(NOEC \& EC10)}$, ABC and $SQG_{(NOEC \& EC10)}$ values are:	
$ACL_{(NOEC \& EC10)}$:	940 mg/kg
ABC:	150 mg/kg
$SQG_{(NOEC \& EC10)}$:	1090 mg/kg, which would be rounded off to 1100 mg/kg.

8.7.3 Calculation of soil quality guidelines for aged lead contamination based on LOEC and 30% effect concentration toxicity data and on 50% effect concentration data

8.7.3.1 Calculation of added contaminant limits

The $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ values for aged Pb contamination were calculated using the method explained earlier, except that the data was multiplied by an ALF of 4.2 (Smolders et al. 2009). The resulting $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ values for aged Pb contamination in the three land uses are presented in Table 68. As expected, these values are larger than the corresponding ACLs for fresh Pb contamination (Table 65).

Table 68: Generic ACLs based on LOEC and 30% effect concentration (EC30) toxicity data and based on 50% effect concentration toxicity data (EC50) values for aged lead (Pb) contamination in soil with various land uses.

Land use	$ACL_{(LOEC \& EC30)}$ (mg/kg)	$ACL_{(EC50)}$ (mg/kg)
Areas of ecological significance	470	250
Urban residential/public open space	1100	2000
Commercial/industrial	1800	3700

8.7.3.2 Calculation of ambient background concentration values

The ABC values for aged Pb contamination were calculated using the method described earlier in this Schedule.

8.7.3.3 Examples of soil quality guidelines for aged lead contamination in Australian soils based on lowest observed effect concentration and 10% effect concentration data and on 50% effect concentration data.

Four examples of SQGs that would apply to aged Pb contamination that represent the range (but not the extremes) of SQGs that would apply to urban residential/public open space and commercial/industrial land uses are presented below.

$SQG_{(LOEC\ \&\ EC30)}$ Example 1
Site descriptors – urban residential land/public open space use in an old South Australian (that is, contamination is >2 years old), with low traffic volume.
Soil descriptors – these are not relevant as soil properties are not considered in determining the ACL for Pb.
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:
$ACL_{(LOEC\ \&\ EC30)}$: 1100 mg/kg
ABC: 150 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$: 1250 mg/kg, which would be rounded off to 1,200 mg/kg.

$SQG_{(LOEC\ \&\ EC30)}$ Example 2
Site descriptors – commercial/industrial land use in an old Queensland suburb (that is, contamination is >2 years old), with high traffic volume..
Soil descriptors – these are not relevant as soil properties are not considered in determining the ACL for Pb.
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:
$ACL_{(LOEC\ \&\ EC30)}$: 1800 mg/kg
ABC: 150 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$: 1950 mg/kg, which would be rounded off to 1900 mg/kg,

$SQG_{(EC50)}$ Example 1
Site descriptors – urban residential land/public open space use in an old South Australian (that is, contamination is >2 years old), with low traffic volume.
Soil descriptors – these are not relevant as soil properties are not considered in determining the ACL for Pb.
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:
$ACL_{(EC50)}$: 2000 mg/kg
ABC: 30 mg/kg
$SQG_{(EC50)}$: 2030 mg/kg, which would be rounded off to 2000 mg/kg.

SQG(EC50) Example 2
Site descriptors – commercial/industrial land use in an old Queensland suburb (that is, contamination is >2 years old), with high traffic volume.
Soil descriptors – these are not relevant as soil properties are not considered in determining the ACL for Pb.
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:
$ACL_{(EC50)}$: 3700 mg/kg
ABC: 150 mg/kg
$SQG_{(EC50)}$: 3850 mg/kg, which would be rounded off to 3800 mg/kg.

8.8 Reliability of the soil quality guidelines

The Pb toxicity data set met the minimum data requirements to use the SSD method but there were no suitable normalisation relationships available to account for soil characteristics. Based on the criteria for assessing the reliability of SQGs (Schedule B5b), this means that the Pb SQGs were considered to be of moderate reliability.

8.9 Comparison with other guidelines

A compilation of SQGs for Pb in a number of jurisdictions is presented in Table 69. These SQGs have a variety of purposes and levels of protection and therefore comparison of the values is problematic. The superseded interim urban EIL for Pb was 600 mg/kg total.

The urban residential/public open space ACLs for fresh Pb contamination (irrespective of the type of toxicity data on which they were based) are all lower than the superceded interim urban EIL.

The aged $ACL_{(NOEC\ \&\ EC10)}$ for urban residential land/public open space land use, at 530 mg/kg added, is lower than the superseded interim urban EIL, while the aged $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ are considerably larger (1100 and 2000 mg/kg respectively). The $ACL_{(NOEC\ \&\ EC10)}$ for fresh Pb contamination is similar to the Canadian residential SQG and the plant Eco-SSL (Table 69).

The fresh $ACL_{(NOEC\ \&\ EC10)}$, $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ for urban residential land/public open space land use correspond to the minimal, warning and potential risk values for residential land use of the EU. The fresh $ACL_{(NOEC\ \&\ EC10)}$ is about 50% larger than the highest minimal risk SQG, but the $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ lie within the range of values for the corresponding EU SQGs.

The best comparison (in terms of the way in which the SQGs were derived) with the ACLs is with the limit derived by the EU ecological risk assessment for Pb (LDA 2008), which also corrected laboratory toxicity data for ageing and leaching. The EU derived a concentration that should protect 95% of terrestrial species of 333 mg/kg added Pb (LDA 2008). If the data and method that were used here (Schedule B5b) were used to calculate the concentration that should protect 95% of species, the value would be 275 mg/kg added Pb—this is slightly more conservative than the EU value.

Table 69. Soil quality guidelines for lead (Pb) in a number of international jurisdictions.

Name of the Pb soil quality guideline	Value of the guidelines (mg/kg)
Canadian SQG (residential)[1]	140 (total Pb)
Canadian SQG (commercial)[1]	260 (total Pb)
Canadian SQG (industrial)[1]	600 (total Pb)
Eco-SSL plants[3]	120 (total Pb)
Eco-SSL soil invertebrates[3]	1700 (total Pb)
Eco-SSL avian[3]	11 (total Pb)
Eco-SSL mammalian[3]	56 (total Pb)
Netherlands (target value)	85 (added Pb)
Netherlands (intervention value)	530 (added Pb)
EU minimal risk values (residential)[2]	25–85 (added Pb)
EU warning risk values (residential)[2]	40–700 (added Pb)
EU potential risk values (residential)[2]	100–700 (added Pb)
EC Pb ecological risk assessment (aged HC_5)[4]	333 (added Pb)

1 = CCME 1999f, 2006 and http://ceqg-rcqe.ccme.ca/

2 = Carlon 2007

3 = <http://www.epa.gov/ecotox/ecossl/>

4 = LDA 2008.

9 Nickel

9.1 Nickel compounds considered

The following salts were considered in deriving SQGs for nickel (Ni):

- nickel metal (CAS No. 7440-02-0)
- nickel sulphate (CAS No. 7786-81-4)
- nickel carbonate (CAS No. 3333-67-3)
- nickel chloride (CAS No. 7718-54-9)
- nickel dinitrate (CAS No. 13138-45-9).

9.2 Exposure pathway assessment

For the leaching to groundwater pathway, adsorption (K_d) is the critical parameter. If the logarithm of the K_d (log K_d) of an inorganic contaminant is less than 3 then it is considered to have the potential to leach to groundwater (Schedule B5b). The log K_d reported by Commentuijn et al. (2000) for Ni was 2.08 L/kg, therefore there is some potential for Ni to leach to groundwater. If this exposure pathway was considered important for a given site, the methodology for SQG derivation advocates that this be addressed on a site-specific basis as appropriate (Schedule B5b).

The literature assessing the potential for Ni to biomagnify is limited, particularly for terrestrial ecosystems. However, all the available literature suggests that Ni does not biomagnify (Outridge & Schuehammer 1993; Torres & Johnson 2001; Campbell et al. 2005; Muir et al. 2005; Lapointe & Couture 2006). The EU ecological risk assessment for Ni also concluded that Ni did not biomagnify (EC 2008b). Therefore only direct toxic effects were considered in deriving the SQGs for Ni.

9.3 Toxicity data

The raw toxicity data available for Ni is presented in Appendix H. There was a total of 338 toxicity measures for Ni. There was toxicity data for 11 plants species, 6 species of invertebrates and 26 microbial processes. The lowest geometric means of the toxicity data for each species and soil process are presented in Tables 70 and 71 respectively. This data exceeded the minimum data requirements to use the BurrliOZ software (Campbell et al. 2000) that is recommended in Schedule B5b. Therefore the SSD approach was used to derive the SQGs for Ni.

Table 70. The lowest geometric mean values of the normalised nickel (Ni) toxicity data for soil invertebrate and plant species.

Test species		Geometric means (mg/kg)		
Common name	**Scientific name**	**NOEC or EC_{10}**	**LOEC or EC_{30}**	**EC_{50}**
Invertebrates				
Earthworm	*E. fetida*	162	245	474
Earthworm	*Eisenia veneta*	103	365	409
Earthworm	*L. rubellus*	407	523	575
Potworm	*Enchytraeus albidus*	134	239	205
Springtail	*F. fimetaria*	210	315	631
Springtail	*F. candida*	235	359	680
Plants				
Alfalfa	*Medicago sativa*	36.4	80.8	87.1
Barley	*H. vulgare*	166.7	250	409

Fenugreek	*Trigonella poenumgraceum*	68.6	109	144
Lettuce	*L. sativa*	52.6	125	154
Maize	*Z. mays*	49.4	94.8	127
Oats	*A. sativa*	55.3	83.9	122
Onion	*Allium cepa*	37.6	59.7	84.5
Perennial ryegrass	*L. perenne*	40.9	50.2	57.1
Radish	*R. sativus*	57.5	65.5	66.8
Spinach	*Spinacia oleracea*	26.9	41.1	47.2
Tomato	*L. esculentum*	94.8	142	238

Table 71. The lowest geometric mean values of the normalised nickel (Ni) toxicity data for soil microbial processes.

Microbial process	Geometric means (mg/kg)		
	NOEC or EC_{10}	LOEC or EC_{30}	EC_{50}
Arylsulfatase	784	1176	1191
Aspergillus clavatus (hyphal growth)	14.9	45.9	91.0
Aspergillus flavus (hyphal growth)	451	586	689
Aspergillus flavipes (hyphal growth)	398	444	475
Aspergillus niger (hyphal growth)	459	545	606
ATP content	75.5	113	392
Gliocladium sp. (hyphal growth)	230	560	1036
Bacillus cereus (colony count)	327	1010	1958
Dehydrogenase	6.8	20.8	85.5
Glucose respiration	79.5	119	238
Glutamate respiration	44.5	191	381
Maize residue respiration	134	201	402
Nitrification	81.3	122	244
N-mineralisation	95.8	144	287
Nocardia rhodochrous (colony count)	203	662	943
Penicillium vermiculatum (hyphal growth)	117	271	460
Phosphatase	524	1347	5715
Protease	75.5	113	392
Proteus vulgaris (colony count)	17.2	88.8	249
Respiration (CO_2 release)	102	2583	4593
Rhizopus stolonifer (hyphal growth)	331	404	459
Rhodotorula rubra (colony count)	283	837	1796
Sacharase	75.5	113	392

Serratia marcescens (colony count)	178	337	395
Trichoderma viride (hyphal growth)	608	686	740
Urease	222	332	879

9.4 Normalisation relationships

Normalisation relationships relating the toxicity of Ni to three soil microbial processes (nitrification, glucose-induced respiration and maize residue mineralisation) were developed by Oorts et al. (2006b). Two normalisation relationships have also been developed for crops (tomato and barley) by Rooney et al. (2007). In addition, the EU Ni ecological risk assessment (EC 2008b) reported Ni normalisation relationships for two soil invertebrates (*F. candida and E. fetida*). All of these relationships were developed for both fresh and aged contamination and are presented in Table 72. No Ni normalisation relationships have been developed for Australian species and/or soils.

The normalisation relationships presented in Table 72 all model EC_{50} toxicity data, with the exception of the maize residue mineralisation which models EC_{20} data. Relationships between the logarithm of Ni NOEC and EC_{10} data and logarithm of CEC were developed as part of this project. Normalisation relationships were developed for (a) all organisms, (b) each group of organisms separately, and (c) each species or microbial process separately. Only CEC was used to develop the normalisation relationships as in all the published relationships for Ni the CEC was the best parameter (Oorts et al. 2006b; Rooney et al. 2007; EC 2008b). Only six normalisation relationships could explain more than 50% of the variation in the toxicity data (i.e. $r^2 > 0.5$) and these are presented in Table 73. The majority of the normalisation relationships had r^2 values of <0.1.

Normalisation relationships are available for a variety of biological end points based on both NOEC and EC_{10} data and on EC_{50} data. The relationships used to normalise the data in the current study were relationships 1, 5 and 9 from Table 72 for glucose-induced respiration, nitrification and tomato, and relationships 2, 3, 5, 6 from Table 73 for barley, all invertebrates, maize residue mineralisation and respiration. The relationships with the lowest gradients for each species were selected. The exception to this was the relationship for invertebrates. This was selected as it was based on all invertebrate species and its gradient was only marginally higher than the invertebrate relationship with the lowest gradient. For the species that did not have normalisation relationships, the relationship for the most closely related species was used, or in the case where there were relationships for several related species, the relationship with the lowest gradient was used. Thus, all plant species (apart from tomato) were normalised with the EC_{10} relationship for barley and all the microbial processes without a relationship were normalised with the EC_{10} relationship for maize residue mineralisation.

Table 72. Normalisation relationships between soil CEC and the toxicity of nickel (Ni) to a variety of soil plant and invertebrate species and soil microbial processes for both fresh and aged contamination. The relationships used to normalise the toxicity data in this project are in bold.

Eqn no.	Species/soil process	Y parameter	X parameter(s)	Reference
Northern hemisphere relationships[a]				
1	Glucose induced respiration	log EC_{50} (fresh)	**0.95 log CEC + 1.51** (r^2 = 0.82)	Oorts et al. 2006b
2		log EC_{50} (aged)	1.34 log CEC + 1.38 (r^2 = 0.92)	Oorts et al. 2006b
3	Maize residue mineralisation	log EC_{20} (fresh)	0.86 log CEC + 1.48 (r^2 = 0.55)	Oorts et al. 2006b

4		log EC_{20} (aged)	1.22 log CEC + 1.37 (r^2 = 0.72)	Oorts et al. 2006b
5	Nitrification	log EC_{50} (fresh)	**0.79 log CEC + 1.44** (r^2 = 0.69)	Oorts et al. 2006b
6		log EC_{50} (aged)	1.00 log CEC + 1.42 (r^2 = 0.60)	Oorts et al. 2006b
7	Barley root elongation	log EC_{50} (fresh)	0.90 log CEC + 1.60 (r^2 = 0.92)	Rooney et al. 2007
8		log EC_{50} (aged)	1.12 log CEC + 1.57 (r^2 = 0.83)	Rooney et al. 2007
9	Tomato shoot yield	log EC_{50} (fresh)	**1.06 log CEC + 1.09** (r^2 = 0.77)	Rooney et al. 2007
10		log EC_{50} (aged)	1.27 log CEC + 1.06 (r^2 = 0.67)	Rooney et al. 2007
11	*F. candida* (collembola)	log EC_{50} (fresh)	0.97 log CEC + 1.71 (r^2 = 0.84)	EC 2008b
12		log EC_{50} (aged)	1.17 log CEC + 1.70 (r^2 = 0.71)	EC 2008b
13	*Eisenia. fetida* (earthworm)	log EC_{50} (fresh)	0.72 log CEC + 1.79 (r^2 = 0.74)	EC 2008b
14		log EC_{50} (aged)	0.95 log CEC + 1.76 (r^2 = 0.72)	EC 2008b

a = all the CEC measurements were made using the silver thiourea method (Chhabra et al. 1975).

Table 73. The normalisation relationships for nickel (Ni) that could explain more than 50% of the variation in the NOEC and 10% effect concentration (EC_{10}) data. The x and y parameters in each equation are the logarithms of the CEC and of the NOEC or EC_{10} toxicity data, respectively. The relationships used to normalise the toxicity data in this project are in bold.

Eqn no.	Species and end point	X parameter(s)[a]
1	Tomato (shoot yield)	1.068 x + 0.908 (r^2 = 0.76)
2	Barley (root elongation)	**0.87 x + 1.35** (r^2 = 0.86**)**
3	All invertebrates (mixed endpoints)	**0.78 x + 1.51** (r^2 = 0.56)
4	Glucose respiration	1.42 x – 0.38 (r^2 = 0.58)
5	Maize residue mineralisation	**0.67 x + 1.45** (r^2 = 0.53)
6	Respiration	**2.37 x – 0.36** (r^2 = 0.92)

a = all CEC measurements were made using the silver thiourea method (Chhabra et al. 1975).

9.5 Sensitivity of organisms to nickel

Figure 9 shows the SSD (that is, the cumulative distribution of the geometric means of normalised NOEC and EC_{10} toxicity values) for the species used to derive the Ni SQGs. While there is an abundance of terrestrial toxicity data for Ni, the majority of data is for microbial processes and microbial enzymes, with only small amounts of data for plants and invertebrates. There does not appear to be any difference in the sensitivity of microbial processes and both plants and invertebrates. However, the distributions of the sensitivities of the plants and invertebrates only just overlap.

Nonetheless, there are no marked differences in the sensitivity of the three groups of organisms and therefore all the available toxicity data was used to derive the Ni SQGs.

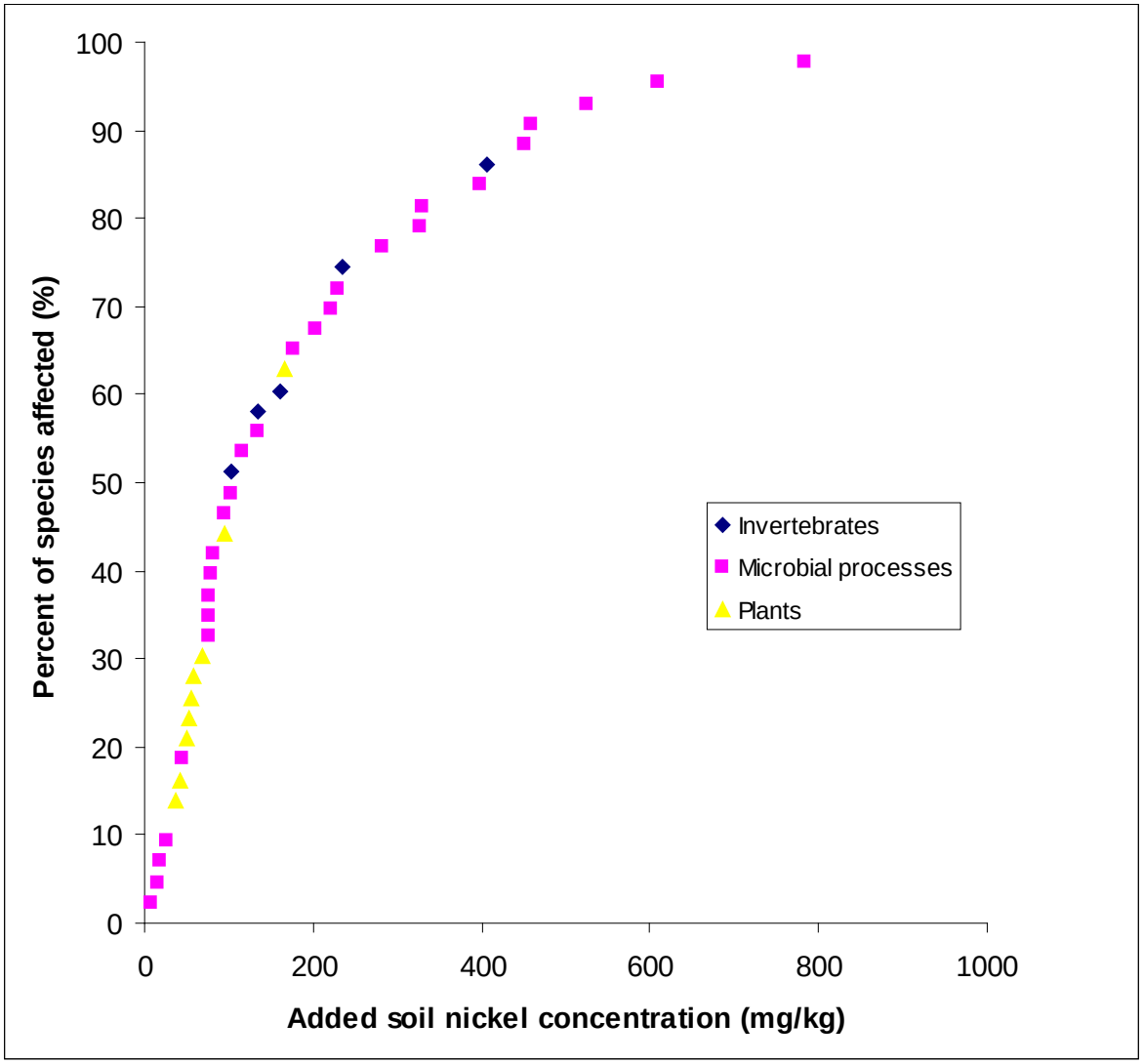


Figure 9. The SSD of normalised NOEC and 10% effect concentration (EC_{10}) toxicity data for fresh nickel (Ni) contamination against soil Ni concentration for soil invertebrates, plants and microbial processes.

9.6 Calculation of soil quality guidelines for fresh nickel contamination

Soil quality guidelines were derived using three different sets of toxicity data (that is, NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data) as part of this study.

9.6.1 Calculation of soil quality guidelines for fresh nickel contamination based on no observed effect concentration and 10% effect concentration toxicity data

9.6.1.1 Calculation of soil-specific added contaminant limits

All the toxicity data was normalised as set out earlier. The generic $ACL_{(NOEC\ \&\ EC10)}$ values generated for fresh Ni contamination for the three land uses are presented in Table 74.

Table 74. Generic ACLS for fresh nickel (Ni) contamination based on NOEC and 10% effect concentration (EC_{10}) toxicity data for various land uses.

Land use	Generic added contaminant limit (mg added/kg)
Areas of ecological significance	6
Residential urban/public open space	50
Commercial/industrial	95

The normalisation equations were then used to calculate soil-specific ACL values at a range of CEC values. Then the lowest ACL at each CEC value was adopted as the soil-specific ACL (Table 75).

Table 75. The soil-specific ACLs (mg/kg) at a range of cation exchange capacities for fresh nickel (Ni) contamination based on NOEC and 10% effect concentration (EC10) toxicity data.

Land use	Cation exchange capacities ($cmol_c/kg$)[a]					
	5	10	20	30	40	60
Areas of ecological significance	1	6	9	10	15	20
Residential urban/public open space	10	50	80	110	130	170
Commercial/industrial	20	95	150	200	240	310

a = all CEC measurements were made using the silver thiourea method (Chhabra et al. 1975).

9.6.1.2 Calculation of ambient background concentration values

For sites with no history of Ni contamination, the method of Hamon et al. (2004) is recommended in Schedule B5b to estimate the ABC. The equation to predict the ABC for Ni is

$$\log \text{Ni conc (mg/kg)} = 0.702 \log \text{Fe content (\%)} + 0.834 \qquad \textit{(equation 10)}$$

Examples of the ABC values predicted by this equation are presented in Table 76.

Table 76. ABCs for nickel (Ni) predicted using the equation from method of Hamon et al. (2004) (equation 10 above).

Fe content (%)	Predicted ABC (mg/kg)
0.1	1
0.5	4
1	7
2	10
5	20
10	35
15	45
20	55

Predicted ABC values for Ni range from approximately 1 to 55 mg/kg in soils with iron contents between 0.1 and 20%.

9.6.1.3 Examples of soil quality guidelines for fresh nickel contamination in Australian soils based on no observed effect concentration and 10% effect concentration data

To calculate the Ni $SQG_{(NOEC\ \&\ EC10)}$ values, the ABC value is added to the $ACL_{(NOEC\ \&\ EC10)}$. ABC values vary with soil type. Therefore, it is not possible to present a single set of $SQG_{(NOEC\ \&\ EC10)}$ values. Thus, two examples of Ni $SQG_{(NOEC\ \&\ EC10)}$ values for urban contaminated soils are provided below. These examples would be at the low and high end of the range of SQG values (but not the extreme values) generated for Australian soils.

Example 1	
Site descriptors – urban residential land/public open space use in a new suburb (that is, fresh contamination).	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.	
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:	
$ACL_{(NOEC\ \&\ EC10)}$:	50 mg/kg
ABC:	7 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$:	57 mg/kg, which would be rounded off to 55 mg/kg.

Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.	
The resulting $ACL_{(NOEC\ \&\ EC10)}$ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:	
$ACL_{(NOEC\ \&\ EC10)}$:	240 mg/kg
ABC:	35 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$:	275 mg/kg, which would be rounded off to 270 mg/kg.

9.6.2 Calculation of soil quality guidelines for fresh nickel contamination based on LOEC and 30% effect concentration toxicity data, and on 50% effect concentration data

9.6.2.1 Calculation of soil-specific added contaminant limits

To maximise the data available to generate the $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50),}$ the available toxicity data was converted to the appropriate measure of toxicity using the conversion factors recommended in Schedule B5b and presented in Table 17. As there were normalisation equations available, soil-specific ACLs could be generated. The $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ values were calculated using the same method as that for the corresponding values for Cu and Pb and are presented in Table 77.

Table 77. The soil-specific ACLs (mg/kg) at a range of cation exchange capacities for fresh nickel (Ni) contamination based on LOEC and 30% effect concentration (EC_{30}) toxicity data, and based on 50% effect concentration (EC_{50}) toxicity data.

Land use	Cation exchange capacities ($cmol_c/kg$)					
	5	**10**	**20**	**30**	**40**	**60**
	Based on LOEC and EC_{30} data					
Areas of ecological significance	1	7	10	15	15	25
Residential urban/public open space	10	50	85	110	130	170
Commercial/industrial	20	100	170	220	260	350
	Based on EC_{50} data					
Areas of ecological significance	5	25	40	55	65	90
Residential urban/public open space	30	160	250	330	400	520
Commercial/industrial	55	280	450	590	710	940

9.6.2.2 Calculation of ambient background concentration values

The ABC values for Ni were calculated using the method previously set out, and the values presented in Table 76.

9.6.2.3 Examples of soil quality guidelines for fresh nickel contamination in Australian soils based on lowest observed effect concentration and 30% effect concentration data, and based on 50% data

To calculate the Ni $SQG_{(LOEC \& EC30)}$ and the $SQG_{(EC50)}$ values, the ABC value is added to the corresponding ACL values. ABC values and Ni ACL values vary with soil type. Therefore it is not possible to present a single set of $SQG_{(LOEC \& EC30)}$ or $SQG_{(EC50)}$ values. Thus, two examples of Ni $SQG_{(LOEC \& EC30)}$ and two examples for Ni $SQG_{(EC50)}$ are provided below. These examples would be at the low and high end of the range of SQG values (but not the extreme values) generated for Australian soils.

$SQG_{(LOEC\ \&\ EC30)}$ Example 1	
Site descriptors – urban residential land/public open space use in a new suburb (that is, fresh contamination).	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	50 mg/kg
ABC:	7 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	57 mg/kg, which would be rounded off to 55 mg/kg.

$SQG_{(LOEC\ \&\ EC30)}$ Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	260 mg/kg
ABC:	35 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	295 mg/kg, which would be rounded off to 290 mg/kg.

$SQG_{(EC50)}$ Example 1	
Site descriptors – urban residential land/public open space use in a new suburb (that is, fresh contamination).	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	160 mg/kg
ABC:	7 mg/kg
$SQG_{(EC50)}$:	167 mg/kg, which would be rounded off to 170 mg/kg

$SQG_{(EC50)}$ Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	710 mg/kg
ABC:	35 mg/kg
$SQG_{(EC50)}$:	745 mg/kg, which would be rounded off to 750 mg/kg.

9.7 Calculation of soil quality guidelines for aged nickel contamination

9.7.1 Calculation of ageing and leaching factors for nickel

Smolders et al. (2009) state that, based on an extensive review of the literature, the ALF for Ni is a function of soil pH (measured in 0.01 M calcium chloride solution) and ranges between 1 and 3.5. Further detail on this relationship is provided in the EU ecological risk assessment report for Ni (EC 2008b). The relationship between the ALF and soil pH is:

$$ALF = 1 + \exp(1.4(\text{soil pH} - 7.0) \qquad \textit{(equation 11)}$$

However, using this equation indicates that the ALF will rapidly increase after a soil pH of 7.5 to values considerably higher than 3.5 (Table 78).

Table 78. ALF values for nickel (Ni) at various soil pH values. The ALF values were derived using the relationship from the European Union ecological risk assessment for Ni (EC 2008b).

Soil pH ($CaCl_2$)	ALF
5	1.07
6	1.25
7	2.00
7.5	3.01
8	5.06
8.5	9.17
9.0	17.45

The above ALF values were calculated after a maximum of 1.5 years ageing in the field, therefore in most 'aged' Australian sites the ALFs would be larger. However, there is no information available that would permit estimates of how much larger the ALFs would be and therefore the above ALF values were used to calculate the Ni SQGs.

9.7.2 Use of ageing and leaching factors in the methodology

There are two possible approaches to incorporating the relationship between ALF and soil pH into the methodology for deriving SQGs. In the first, a soil pH that is reasonably representative or protective of the majority of Australian soils is selected and the corresponding ALF is then used to calculate the aged SQGs. The resulting SQGs would be protective of all aged soils with a pH higher than the selected pH, but would not provide the same level of protection to soils with lower soil pH. Such soils would have to proceed to further desktop analysis by using the ALF–pH relationship to determine the appropriate ALF for that soil and then apply that to the fresh contamination SQGs. To maximise the utility of this approach and minimise the number of sites that would require the additional analysis, the selected soil pH would have to be low, perhaps as low as 5. This would result in an ALF of 1.07 and with such a small increase in the resulting aged SQGs, it is doubtful that it would be of any real benefit.

The second approach would be to fully adopt the ALF–pH relationship into the methodology for deriving SQGs, where the pH of the site would need to be determined and then the appropriate ALF calculated for the site and applied to the toxicity data to generate the aged contamination ACLs and thence the aged SQGs. While the latter is more complex, the benefits of having the most scientifically defensible ACLs and SQGs outweigh this. It is recommended that SQGs are derived by multiplying fresh (non-aged and non-leached) toxicity data by the ALF determined using the ALF–pH relationship (see equation 11).

9.7.3 Calculation of soil quality guidelines for aged nickel contamination based NOEC and 10% effect concentration toxicity data

9.7.3.1 Calculation of soil-specific added contaminant limits

The aged $SQG_{(NOEC\ \&\ EC10)}$ values for Ni were calculated using the same methodology as that used for the $SQG_{(NOEC\ \&\ EC10)}$ values for fresh Ni contamination, with two exceptions. These were (i) that the 'fresh' toxicity data was corrected using the Ni ALFs (equation 11) and (ii) the ABCs were the 25th percentile values for old suburbs from Olszowy et al. (1995). The resulting $ACL_{(NOEC\ \&\ EC10)}$ values for aged Ni contamination are presented in Table 79.

Table 79. The soil-specific ACLs (mg/kg) at a range of cation exchange capacities for aged nickel (Ni) contamination based on NOEC and 10% effect concentration (EC_{10}) toxicity data.

Land use	Cation exchange capacities ($cmol_c/kg$)					
	5	10	20	30	40	60
Areas of ecological significance	2	9	15	20	20	30
Residential urban/public open space	15	85	140	180	220	290
Commercial/industrial	30	160	250	330	400	530

9.7.3.2 Calculation of ambient background concentration values

For aged contaminated sites (that is, the contamination has been in place for at least 2 years) Heemsbergen et al. (2008) recommends using the 25th percentiles of the ABC data for 'old suburbs' in Olszowy et al. (1995) (see Table 80). The Olszowy et al. (1995) data is derived from soils low in geogenic Ni and, by using low ABCs, could create low SQGs in some areas with naturally high background Ni concentrations. This problem could be overcome in areas with elevated soil Ni by using measured ABC values or using the method of Hamon et al. (2004).

Table 80. Nickel (Ni) ABCs based on the 25 percentiles of Ni concentrations in 'old suburbs' (i.e. >2 years old) from various states of Australia (Olszowy et al. 1995).

Suburb type	25th percentile of Ni ABC values (mg/kg)			
	NSW	QLD	SA	VIC
Old suburb, low traffic	5	5	6	5
Old suburb, high traffic	5	4	6	10

9.7.3.3 Examples of soil quality guidelines for aged nickel contamination in Australian soils based on no observed effect concentration and 10% effect concentration data

To calculate the aged Ni $SQG_{(NOEC\ \&\ EC10)}$ values , the ABC value is added to the ACL. Ambient background concentration values vary with soil type, region and history of exposure to contamination. Therefore, it is not possible to present a single set of $SQG_{(NOEC\ \&\ EC10)}$ values. Thus, two examples of Ni $SQG_{(NOEC\ \&\ EC10)}$ values are presented below. These examples would be at the low and high end of the range of SQG values (but not the extreme values) generated for Australian soils.

Example 1
Site descriptors – urban residential land/public open space use in an old Queensland suburb (that is, aged contamination), with low traffic volume.
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.
The resulting $ACL_{(NOEC \& EC10)}$, ABC and $SQG_{(NOEC \& EC10)}$ values are:
$ACL_{(NOEC \& EC10)}$: 85 mg/kg
ABC: 5 mg/kg
$SQG_{(NOEC \& EC10)}$: 90 mg/kg

Example 2
Site descriptors – commercial/industrial land use in an old Victorian suburb (that is, aged contamination), with high traffic volume.
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.
The resulting $ACL_{(NOEC \& EC10)}$, ABC and $SQG_{(NOEC \& EC10)}$ values are:
$ACL_{(NOEC \& EC10)}$: 400 mg/kg
ABC: 10 mg/kg
$SQG_{(NOEC \& EC10)}$: 410 mg/kg

9.7.4 Calculation of soil quality guidelines for aged nickel contamination based on LOEC and 30% effect concentration toxicity data, and on 50% effect concentration data

9.7.4.1 Calculation of soil-specific added contaminant limits

Soil-specific aged Ni ACL values based on LOEC and EC_{30} and on EC_{50} data were calculated using the method previously set out, except the type of toxicity data used was different. The resulting ACLs are presented in Table 81.

Table 81. The soil-specific ACLs at a range of cation exchange capacities for aged nickel (Ni) contamination based on lowest observed effect concentration (LOEC) and 30% effect concentration (EC_{30}) toxicity data, and based on 50% effect concentration (EC_{50}) toxicity data.

Land use	Cation exchange capacities ($cmol_c/kg$)					
	5	**10**	**20**	**30**	**40**	**60**
	Based on LOEC and EC_{30} data					
Areas of ecological significance	5	30	45	60	70	95
Urban residential/public open space	30	170	270	350	420	560
Commercial/industrial	55	290	460	600	730	960
	Based on EC_{50} data					
Areas of ecological significance	10	65	100	130	160	210
Urban residential/public open space	55	270	440	570	700	910
Commercial/industrial	90	460	730	960	1200	1500

9.7.4.2 Calculation of ambient background concentration values

The ABC values used for aged Ni were obtained from Table 80.

9.7.4.3 Examples of soil quality guidelines for fresh nickel contamination in Australian soils based on lowest observed effect concentration and 30% effect concentration data, and based on 50% effect concentration data

Ambient background concentration values for Ni vary with soil type as do the Ni ACL values. Therefore, it is not possible to present a single set of $SQG_{(LOEC\ \&\ EC30)}$ or $SQG_{(EC50)}$ values. Thus, two examples of Ni $SQG_{(LOEC\ \&\ EC30)}$ values and two examples for Ni $SQG_{(EC50)}$ values are provided below. These examples would be at the low and high end of the range of SQG values (but not the extreme values) generated for Australian soils.

$SQG_{(LOEC\ \&\ EC30)}$ Example 1	
Site descriptors – urban residential land/public open space use in an old Queensland suburb (that is, aged contamination), with high traffic volume.	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	170 mg/kg
ABC:	4 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	174 mg/kg, which would be rounded off to 170 mg/kg.

$SQG_{(LOEC\ \&\ EC30)}$ Example 2	
Site descriptors – commercial/industrial land use in an old Victorian suburb, with high traffic volume.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	730 mg/kg
ABC:	10 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	740 mg/kg

$SQG_{(EC50)}$ Example 1	
Site descriptors – urban residential land/public open space use in an old Queensland suburb (that is, aged contamination), with high traffic volume.	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10) with 1% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	270 mg/kg
ABC:	4 mg/kg
$SQG_{(EC50)}$:	274 mg/kg, which would be rounded off to 270 mg/kg.

$SQG_{(EC50)}$ Example 2

Site descriptors – commercial/industrial land use in an old Victorian suburb, with high traffic volume.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40) with 10% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	1200 mg/kg
ABC:	10 mg/kg
$SQG_{(EC50)}$:	1210 mg/kg, which would be rounded off to 1200 mg/kg.

9.8 Reliability of the soil quality guidelines

The SQGs for Ni were considered to be of high reliability, as the toxicity data set met the minimum data requirements to use an SSD method and there were normalisation relationships available to account for soil characteristics (Schedule B5b).

9.9 Comparison with other guidelines

Soil quality guidelines for Ni in a number of international jurisdictions are presented in Table 82. These SQGs have a variety of purposes and levels of protection and therefore a comparison of the values is problematic. The SQGs for Ni range from 24 to 500 mg/kg added and total Ni, with both of these values coming from countries within the EU. The superseded interim urban EIL for Ni (NEPC 1999) was 60 mg/kg total Ni.

There are also four health-based investigation level (HIL) values that range from 400 to 4000 mg/kg total Ni (see Schedule B1). The urban residential/public open space ACLs based on NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data for fresh Ni contamination range from 10–170, 10–170, and 30 to 520 mg/kg added Ni respectively. These correspond to the 'minimal risk', 'warning risk' and the 'potential risk' values of EU member countries and the values are very similar. The urban residential/public open space ACLs based on NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data for aged Ni contamination range from 15–290, 30–560, and 55–910 mg/kg added Ni respectively. These limits permit higher concentrations than in any of the other jurisdictions, but this is not suprising as the other jurisdictions do not account for ageing or leaching, nor do they take into account the bioavailability in different soils.

The most meaningful comparisons can be made between the SQGs and the concentrations that would protect 95% of species based on NOEC and EC_{10} data that was derived in the EU ecological risk assessment for Ni (EC 2008b). These values ranged from 8.3 to 188.7 mg/kg added Ni for soils with CEC values ranging from 2.4 to 36 $cmol_c$/kg (EC 2008b). SQGs that protected 95% of species were not derived, but rather the SQGs were derived that protect 99, 80 and 60% of species. The SQGs that aim to protect 99% of species based on NOEC and EC_{10} data ranged from 1–20 mg/kg added Ni. The SQGs that aim to protect 80% of species based on NOEC and EC_{10} data ranged from 10–170mg/kg added Ni. These comparisons indicate that the SQGs derived in this project are slightly more conservative than the EU values, but overall the values are similar.

Table 82. Soil quality guidelines for nickel (Ni) in a number of international jurisdictions.

Name of the Ni soil quality guideline	Value of the guideline (mg/kg Ni)
Dutch target values[1]	35 (added Ni)
Dutch intervention value[1]	210 (added Ni)
Canadian SQG (residential, commercial and industrial)[2]	50 (total Ni)
Eco-SSL plants[3]	38 (total Ni)

Eco-SSL soil invertebrates[3]	280 (total Ni)
Eco-SSL avian[3]	210 (total Ni)
Eco-SSL mammalian[3]	130 (total Ni)
EU minimal risk values (residential)[4]	24–60 (added & total Ni)
EU warning risk values (residential)	30–180 (added & total Ni)
EU potential risk values (residential)[4]	30–500 (added & total Ni)
EU Ni ecological risk assessment (conc that should protect 95% of species)[5]	8.3–188.7 (added & total Ni)

1 = VROM 2000

2 = CCME 1999g 2006 and http://ceqg-rcqe.ccme.ca/

3 = http://www.epa.gov/ecotox/ecossl/

4 = Carlon 2007

5 = EC 2008b.

10 Trivalent chromium

10.1 Chromium (III) compounds considered

Chromium occurs in a number of oxidation states: II, III, IV, V and VI. The two dominant states in soils are trivalent (III) and hexavalent (VI) Cr. The only forms of Cr (III) for which there was toxicity data were chromium chloride, chromium nitrate and chromium sulphate.

10.2 Exposure pathway assessment

Chromium is the seventh most abundant element (McGrath & Smith 1990). It is also an essential element for humans and for some groups of organisms (Crommentuijn et al. 2000), yet the hexavalent form is generally considered to be highly toxic and a carcinogen.

The two key considerations in determining the most important exposure pathways for inorganic contaminants, such as Cr (III), are whether they biomagnify and whether they have the potential to leach to groundwater. A surrogate measure of the potential for a contaminant to leach is its water–soil partition coefficient (K_d). If the logarithm of the K_d (log K_d) of an inorganic contaminant is less than 3 then it is considered to have the potential to leach to groundwater (Schedule B5b). The log K_d reported by Commentuijn et al. (2000) for Cr (with the oxidation state not identified) was 2.04 L/kg; therefore, Cr has the potential in some soils to leach to groundwater. However, the ability of Cr to migrate from soil to either groundwater or surface water depends greatly on its oxidation state. Hexavalent Cr is highly water-soluble whereas trivalent Cr is almost insoluble in water and immobile in soil (Bartlett & James 1988; Cervantes et al. 2001). Therefore, Cr (III) is unlikely to pose an environmental risk by leaching. In addition, Cr (III) cannot cross most cells (Cervantes et al. 2001). In contrast, Cr (VI) is actively transported across cell membranes (Dreyfuss, 1964; Wiegand et al. 1985). Chromium (III) is not known to biomagnify (Scott-Fordsmand & Pedersen 1995; Heemsbergen et al. [2008]) and therefore only direct toxicity routes of exposure were considered in deriving the SQGs for Cr (III).

10.3 Toxicity data

Unlike the preceding elements, there is a lack of ecotoxicity data for Cr (III). This is reflected by the fact that the US EPA (US EPA 2008) could not derive Eco-SSL values (which require toxicity data for species belonging to three different types of organisms) for Cr (either as III or VI) for soil invertebrates and plants. Also, neither the Canadians (CCME 1999h,) nor the Dutch (Crommentuijn et al. 2000) have SQGs for Cr (III) but simply total Cr.

Extensive searches of the available scientific literature were conducted on ISI web of knowledge, the US EPA ECOTOX database (http://cfpub.epa.gov/ecotox), the Dutch RIVM e-toxbase database (http://www.e-toxbase.com – this is not publicly available), the database of the French National Institute of Industrial Environment and Risk (INERIS, www.ineris.fr), and the Australasian Ecotoxicology Database (Warne et al. 1998; Warne & Westbury 1999; Markich et al. 2002; Langdon et al. 2009). There were a number of publications (Bonet et al. 1991; Scoccianti et al. 2006) which presented toxicity data for Cr (III) that were not included in the derivation of SQGs in this guideline. This was because these were based on exposing plants solely via aqueous media (that is, hydroponics) or the growth medium was agar and this is vastly different from exposure via soil.

The raw toxicity data for Cr (III) is presented in Appendix I. The toxicity data (geometric means for each species) used to calculate the SQGs is presented in Table 83. There was toxicity data for a total of 21 species or soil microbial processes. There was data for 2 soil invertebrate species, 12 species of plants and 7 soil microbial processes. This data meets the minimum data requirements recommended in Schedule B5b to use the BurrliOZ SSD method (Campbell et al. 2000). The toxicity data for nitrogenase was not used as it was all ‘less than’ values and the lowest concentration tested (that is, 50 mg/kg) caused an effect considerably larger than 50%. It should be noted that the toxicity data for the enzyme catalase was markedly lower (that is, more than one order of magnitude) than all the other toxicity data. Given this and the fact that the toxicity data was quantified using nominal (not measured) concentrations, there is uncertainty in the reliability of this data. Therefore the catalase toxicity data was not used to derive the SQGs.

Table 83. The lowest geometric mean values of normalised (invertebrate) and non-normalised (all other species and microbial processes) trivalent chromium (Cr (III)) toxicity data, expressed in terms of added Cr (III) for soil invertebrate species, plant species, and soil microbial processes.

Test species		Geometric mean (mg/kg)		
Common name	**Scientific name**	**EC_{10} or NOEC**	**EC_{30} or LOEC**	**EC_{50}**
Arylsulfatase		121	181	321
Barley	*H. vulgare*	200	300	600
Beans		200	500	600
Bent grass	*Agrostis tenius*	3333	5000	10000
Bush bean	*Phaseolus vulgaris*	41	70.7	141
Catalase		0.19	0.88	2.32
Corn	*Z. mays*	294	611	1233
Earthworm	*Eisenia fetida*	467	700	1400
Earthworm	*E. Andrei*	25.4	79.5	159
Glutamic acid decomposition		55	400	800
Grass		200	500	600
Indian mustard	*Brassica juncea*	500	750	1100
Lettuce	*L. sativa*	500	387	775
Nitrogenase		<<50	<<50	<<50
Nitrogen mineralisation		172	302	626
Nitrogenate formation		50	200	500
Oat	*A. sativa*	339	508	1016
Perennial ryegrass	*L. perenne*	3333	5000	10000
Radish	*R. sativus*	500	387	775
Respiration		36.3	114	139
Rye	*Secale cereale*	233	350	700
Urease		71.2	122	205

In order to maximise the use of the available toxicity data, conversion factors provided in Schedule B5b were used to permit the inter-conversion of NOEC, LOEC, EC_{50}, EC_{30} and EC_{10} data. The conversion factors used are presented in Table 17.

10.4 Normalisation relationships

There are only three published normalisation relationships for Cr (III) toxicity (Sivakumar & Subbhuraam 2005). They all relate the toxicity of Cr (III) to survival of *E. fetida* and are presented in Table 84. These are all based on clay content. The logarithmic form of normalisation relationship 1 was used to normalise the *E. fetida and E. andrei* toxicity data. This relationship was not applied to the toxicity data of the other species/microbial processes as they do not belong to the same organism type (that is, soft-bodied invertebrate) as the earthworm. This approach is consistent with the method

recommended in Schedule B5b and adopted in the various EU ecological risk assessments that have been conducted for metals (EC 2008a; EC 2008b; LDA 2008).

Table 84. Normalisation relationships for the toxicity of trivalent chromium (Cr (III)) to soil invertebrates. The relationship used to normalise the toxicity data is in bold. All equations from Sivakumar & Subbhuraam (2005).

Species/soil process	Y Parameter	X parameter(s)
E. fetida	log EC_{50}	**-5.46 clay content + 1905.93** (r^2 = 0.92)
		-5.75 clay content – 10.62 pH + 1980.46 (r^2 = 0.92)
		-3.59 clay content + 4.16 pH + 65.83 soil N + 1748.22 (r^2 = 0.95)

10.5 Sensitivity of organisms to trivalent chromium

Figure 10 shows the SSD (that is, the cumulative distribution of the geometric means of species sensitivities to Cr (III)) for all species for which Cr (III) toxicity data was available). Due to the limited amount of Cr (III) toxicity data and the fact that the data was not normalised (and thus soil properties affect the values), it is difficult to draw conclusions regarding the relative sensitivity of plants, invertebrates and soil processes to Cr (III). Given the lack of data and the overlaps in the sensitivity of the organism types, all the Cr (III) toxicity data was used to derive the SQGs.

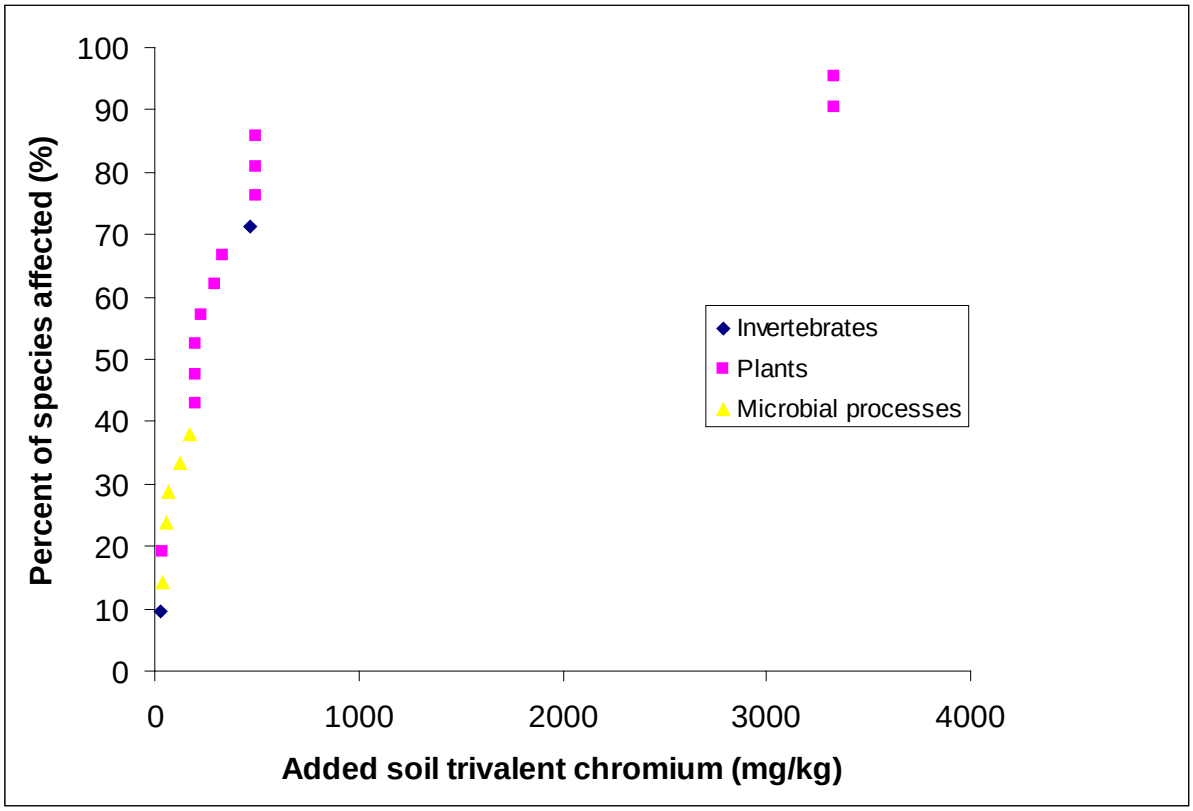


Figure 10. The SSD (plotted as a cumulative frequency against added trivalent chromium (Cr (III)) concentration) of Cr (III) for soil invertebrate species, plant species and soil microbial processes.

10.6 Calculation of soil quality guidelines for fresh trivalent chromium contamination

10.6.1 Calculation of added contaminant limits for fresh trivalent chromium contamination

Only the Cr (III) toxicity data for *E. fetida* and *E. andrei* could be normalised to the Australian reference soil. Thus, a set of generic ACLs and a set of soil-specific ACLs were derived (for the earthworms). The soil-specific ACL values below a clay content of 10% were smaller than the generic ACL values. The soil-specific ACL at a clay content of 10% equalled the generic ACL, and all soil-specific ACLs for soils with a clay content greater than 10% were larger than the generic ACLs. The lower of the soil-specific ACL values and the generic ACL values were adopted as the final ACLs for Cr (III). Thus, the situation was simplified to the soil-specific ACLs only applying up to a clay content of 10% at which point the generic ACL values apply. The generated ACLs for the three land uses and the three types of toxicity data (that is, NOEC and EC_{10}, LOEC and EC_{30}, EC_{50}) are presented in Table 85.

The range between the largest and smallest ACL values generated was approximately 4.0 to 470 mg added Cr (III)/kg. The residential/urban ACLs based on NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data ranged from 35–75, 75–160, and 110–230 mg added Cr (III)/kg respectively.

Table 85. The ACLs based on NOEC and 10% effect concentration (EC_{10}) data, LOEC and 30% effect concentration (EC_{30}), and 50% effect concentration (EC_{50}) toxicity data for trivalent chromium (Cr (III)) for various land uses. These are based on all the Cr (III) toxicity data, except the catalase and nitrogenase enzyme activity data.

Data type	Land use	Clay content			
		1	2.5	5	≥10
NOEC	**AES**	4	6	7	9
	UR	35	45	60	75
	C/I	65	90	110	140
LOEC	**AES**	25	30	40	50
	UR	75	100	130	160
	C/I	120	170	210	270
EC_{50}	**AES**	9	10	15	20
	UR	110	150	190	230
	C/I	220	300	375	470

AES = Areas of ecological significance
UR = urban residential/public open space
C/I = commercial/industrial land uses.

10.6.2 Calculation of ambient background concentration values for fresh trivalent chromium contamination

For sites with no history of Cr (III) contamination, the method of Hamon et al. (2004) is recommended to estimate the Cr ABC. Technically this method predicts total Cr but under aerobic soil conditions the vast majority of Cr will be present as Cr (III). It is therefore appropriate to use the Hamon et al (2004) method to estimate Cr (III) ABC values. The equation to predict the Cr ABC is:

$$\log \text{Cr conc (mg/kg)} = 0.75 \log \text{Fe content (\%)} + 1.242 \qquad \textit{(equation 12)}$$

Examples of the ABC values predicted by this equation are presented in Table 86. Predicted ABC values for Cr (III) range from approximately 3 to 160 mg/kg in soils with iron concentrations between 0.1 and 20%.

Table 86. ABCs for chromium (Cr) predicted using the method of Hamon et al. (2004) (equation 12 above).

Fe content (%)	Predicted Cr ABC (mg/kg)
0.1	3
0.5	10
1	15
2	30
5	60
10	100
15	130
20	160

10.6.3 Examples of soil quality guidelines for fresh trivalent chromium contamination in Australian soils

ABC values for Cr (III) vary with soil type (Table 86). Therefore, it is not possible to present a single set of SQG values. Thus, two examples of each of Cr (III) $SQG_{(NOEC\ \&\ EC10)}$ values, $SQG_{(LOEC\ \&\ EC30)}$ values and $SQG_{(EC50)}$ values are provided below. These examples would be at the low and high end of the range of SQG values (but not the extreme values) generated for Australian soils.

$SQG_{(NOEC\ \&\ EC10)}$ Example 1
Site descriptors – urban residential land/public open space use in a new suburb.
Soil descriptors – a sandy acidic soil (pH 5, CEC 10, clay content 2.5%) with 1% iron content.
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:
$ACL_{(NOEC\ \&\ EC10)}$: 45 mg/kg
ABC: 15 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$: 60 mg/kg

$SQG_{(NOEC\ \&\ EC10)}$ Example 2
Site descriptors – commercial/industrial land use in a new suburb.
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40, clay content 20%) with 10% iron content.
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:
$ACL_{(NOEC\ \&\ EC10)}$: 140 mg/kg
ABC: 100 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$: 240 mg/kg

$SQG_{(LOEC\ \&\ EC30)}$ Example 1	
Site descriptors – urban residential land /public open space use in a new suburb.	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10, clay content 2.5%) with 1% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	100 mg/kg
ABC:	15 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	115 mg/kg, which would be rounded off to 110 mg/kg.

$SQG_{(LOEC\ \&\ EC30)}$ Example 2	
Site descriptors – commercial/industrial land use/public open space in a new suburb.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40, clay content 20%) with 10% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	270 mg/kg
ABC:	100 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	370 mg/kg

$SQG_{(EC50)}$ Example 1	
Site descriptors – urban residential land/public open space use in a new suburb.	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10, clay content 2.5%) with 1% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	150 mg/kg
ABC:	15 mg/kg
$SQG_{(EC50)}$:	165 mg/kg, which would be rounded off to 160 mg/kg.

$SQG_{(EC50)}$ Example 2	
Site descriptors – commercial/industrial land use in a new suburb.	
Soil descriptors – an alkaline clay soil (clay content 20%) with 10% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	470 mg/kg
ABC:	100 mg/kg
$SQG_{(EC50)}$:	570 mg/kg

10.7 Calculation of soil quality guidelines for aged trivalent chromium contamination

10.7.1 Calculation of an ageing and leaching factor for trivalent chromium

There are no ALFs available for Cr (III) nor data available to derive ALFs. Therefore, as an interim measure, the mean of the ALF values available for other cations (that is, Cd, Cu, Co, Ni, Pb and Zn)

from Smolders et al. (2009) was determined. This resulted in a value of 2.35[4], which was rounded off to 2.5.

10.7.2 Calculation of added contaminant limits for aged trivalent chromium contamination

All the Cr (III) toxicity data was multiplied by the ALF of 2.5. Therefore, the aged SQG(NOEC & EC_{10}), SQG(LOEC & EC_{30}) and SQG(EC_{50}) values are exactly 2.5 times the corresponding fresh SQGs for Cr (III). The resulting aged SQG(NOEC & EC_{10}), SQG(LOEC & EC_{30}) and SQG(EC_{50}) values are presented in Table 87.

10.7.3 Calculation of ambient background concentration values

For aged contaminated sites (that is, the contamination has been in place for at least 2 years, Schedule B5b) the methodology recommends using the 25th percentiles of the ABC data for the 'old suburbs' of Olszowy et al. (1995) (see Table 88). Chromium concentrations in old suburbs are higher than those for new suburbs (Olszowy et al. 1995); therefore, it is appropriate to use the ABC values for aged suburbs. The Cr concentrations reported by Olszowy et al (1995) are for total Cr; however, as was the case with the Hamon et al. (2004) method, the majority of the Cr measured will be Cr (III) and thus the data can be used to estimate ABC values for Cr (III). The Olszowy et al. (1995) data was derived from soils low in geogenic Cr and, by using low ABCs, could create low SQGs in some areas with naturally high background Cr concentrations. This problem could be overcome in areas of high natural Cr (III) by using measured ABC values or using the Hamon et al. (2004) method.

Table 87. The ACLs based on NOEC and 10% effect concentration (EC_{10}) data, LOEC and 30% effect concentration (EC_{30}), and 50% effect concentration (EC_{50}) toxicity data for trivalent chromium (Cr (III)) for various land uses. These are based on all the Cr (III) toxicity data, except the catalase and nitrogenase enzyme activity data.

Data type	Land use	Clay content			
		1	2.5	5	≥10
NOEC	AES	10	15	20	20
	UR	85	120	150	190
	C/I	170	230	280	360
LOEC	AES	60	80	100	130
	UR	190	250	310	400
	C/I	310	420	530	660
EC_{50}	AES	25	30	40	50
	UR	275	370	460	580
	C/I	550	750	940	1200

AES = Areas of ecological significance, UR = urban residential/public open space, C/I = commercial/industrial land uses.

Table 88. Chromium ABCs based on the 25th percentiles of Cr concentrations in 'old suburbs' (that is, >2 years old) from various states of Australia (Olszowy et al. 1995).

Suburb type	25th percentile of Cr ABC values (mg/kg)			
	NSW	QLD	SA	VIC
Old suburb, low traffic	8	15	15	10
Old suburb, high traffic	15	7	15	10

[4] For cations with a single ALF, these were used to calculate the mean ALF. For cations with a range of values, both the lowest and highest values were used to calculate the mean. Therefore the value of 2.35 was the mean of 3, 2, 1, 1, 3, 1.1, 3.5, 4.2, 1.

10.7.4 Examples of soil quality guidelines for aged trivalent chromium contamination in Australian soils

ABC values for Cr (III) vary with soil type and location (Table 88). Therefore, it is not possible to present a single set of SQG values. Thus, two examples of each of Cr (III) $SQG_{(NOEC\ \&\ EC10)}$ values, $SQG_{(LOEC\ \&\ EC30)}$ values and $SQG_{(EC50)}$ values for aged Cr (III) contamination are provided below. These examples would be at the low and high end of the range of SQG values (but not the extreme values) generated for Australian soils.

$SQG_{(NOEC\ \&\ EC10)}$ Example 1	
Site descriptors – urban residential land /public open space use in an old Victorian suburb with low traffic volume.	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10, clay content 2.5%) with 1% iron content.	
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:	
$ACL_{(NOEC\ \&\ EC10)}$:	120 mg/kg
ABC:	10 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$:	130 mg/kg

$SQG_{(NOEC\ \&\ EC10)}$ Example 2	
Site descriptors – commercial/industrial land use in an old NSW suburb with high traffic volume.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40, clay content 20%) with 10% iron content.	
The resulting $ACL_{(NOEC\ \&\ EC10)}$, ABC and $SQG_{(NOEC\ \&\ EC10)}$ values are:	
$ACL_{(NOEC\ \&\ EC10)}$:	360 mg/kg
ABC:	15 mg/kg
$SQG_{(NOEC\ \&\ EC10)}$:	375 mg/kg, which would be rounded off to 370 mg/kg.

$SQG_{(LOEC\ \&\ EC30)}$ Example 1	
Site descriptors – urban residential land/public open space use in an old Victorian suburb with low traffic volume.	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10, clay content 2.5%) with 1% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	250 mg/kg
ABC:	10 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	260 mg/kg

$SQG_{(LOEC\ \&\ EC30)}$ Example 2	
Site descriptors – commercial/industrial land use in an old NSW suburb with high traffic volume.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40, clay content 20%) with 10% iron content.	
The resulting $ACL_{(LOEC\ \&\ EC30)}$, ABC and $SQG_{(LOEC\ \&\ EC30)}$ values are:	
$ACL_{(LOEC\ \&\ EC30)}$:	660 mg/kg
ABC:	15 mg/kg
$SQG_{(LOEC\ \&\ EC30)}$:	675 mg/kg, which would be rounded off to 670 mg/kg.

$SQG_{(EC50)}$ Example 1	
Site descriptors – urban residential land/public open space use in an old Victorian suburb with low traffic volume.	
Soil descriptors – a sandy acidic soil (pH 5, CEC 10, clay content 2.5%) with 1% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	370 mg/kg
ABC:	10 mg/kg
$SQG_{(EC50)}$:	380 mg/kg

$SQG_{(EC50)}$ Example 2	
Site descriptors – commercial/industrial land use in an old NSW suburb with high traffic volume.	
Soil descriptors – an alkaline clay soil (pH 7.5, CEC 40, clay content 20%) with 10% iron content.	
The resulting $ACL_{(EC50)}$, ABC and $SQG_{(EC50)}$ values are:	
$ACL_{(EC50)}$:	1200 mg/kg
ABC:	15 mg/kg
$SQG_{(EC50)}$:	1215 mg/kg, which would be rounded off to 1200 mg/kg.

10.8 Reliability of the soil quality guidelines

The Cr (III) toxicity data set met the minimum data requirements to use the SSD method but there was only one normalisation relationship available (for the earthworm *Eisenia fetida*) to account for soil characteristics. Based on the criteria for assessing the reliability of SQGs in Schedule B5b, this means that the Cr (III) SQGs were considered to be of moderate reliability.

10.9 Comparison with other guidelines

A compilation of SQGs for Cr (III), Cr (VI) and total Cr from a number of international jurisdictions is presented in Table 89. These guidelines have a variety of purposes and levels of protection and therefore comparison of the values is problematic. The SQGs for Cr (III) range from 26–50 mg/kg (total Cr (III)). The majority of jurisdictions do not have SQGs for Cr (III), more typically they have SQGs for total Cr. Carlon (2007), in his review of the SQGs of members of the EU, did not identify whether the SQGs were for added or total Cr, nonetheless they range from 34–1000 mg/kg. Hexavalent Cr is typically considered to be more toxic than Cr (III) and this is reflected by it having lower SQGs (Table 89).

The ACLs for fresh Cr (III) contamination that apply to urban residential land/public open space land use based on NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data ranged from 35–75, 75–160 and 100–230 mg added Cr (III)/kg respectively. The SQGs based on NOEC and EC_{10} data are closest to the existing international SQGs for Cr (III). It should be noted that all of the ACLs for urban residential land/public open space land use (irrespective of what data was used to generate them) are considerably smaller than the superseded interim urban EIL of 400 mg total Cr/kg (NEPC 1999). However, the ACLs are consistent with the available Cr (III) toxicity data where there are 6 species/microbial processes that have EC_{50} values below the superseded interim urban EIL and there are 12 and 16 species/microbial processes that have LOEC and EC_{30} or NOEC and EC_{10} data respectively, below the superseded interim urban EIL. The species/microbial processes with toxicity values below the superseded interim urban EIL can be indentified by referring to Table 83.

The ACLs for aged Cr (III) contamination that apply to urban residential land/public open space land use based on NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data ranged from 85–190, 175–400 and 270–580 mg added Cr (III)/kg respectively. None of the ACLs based on NOEC & EC_{10} and LOEC & EC_{30} toxicity data were larger than the current interim EIL. However, once the clay content was 5% or above, the ACL values based on EC_{50} data were larger than the superseded interim EIL. All of the ACLs for aged Cr (III) contamination are considerably larger than the collated international Cr (III) SQGs.

Table 89. Soil quality guidelines (mg/kg) for total chromium, trivalent chromium (Cr (III)) and hexavalent chromium (Cr (VI)) from international jurisdictions.

Name of chromium soil quality guideline	Total chromium	Trivalent chromium	Hexavalent chromium
Canadian SQG (residential)[1]			0.4 (total)
Canadian SQG (commercial and industrial)[1]			1.4 (total)
Danish soil quality guideline[2]		50 (total)	2 (total)
Dutch target value[3]	100 (added Cr)		
Dutch maximum permissible addition[3]	380 (added Cr)		
Eco-SSL plants[4]		ID	ID
Eco-SSL soil invertebrates[4]		ID	ID
Eco-SSL avian[4]		26 (total)	ID
Eco-SSL mammalian[4]		34 (total)	130 (total)
EU minimal risk values (residential)[5]	34–130 (added & total)		2.5 (added & total)
EU warning risk values (residential)[5]	50–450 (added & total)		4.2–20 (added & total)
EU potential risk values (residential)[5]	100–1000 (added & total)		

1 = CCME 1999h and 2006 and http://ceqg-rcqe.ccme.ca/
2 = Scott-Fordsmand and Pedersen 1995
3 = VROM 2000
4 = http://www.epa.gov/ecotox/ecossl/
5 = Carlon 2007
ID = insufficient data.

11 Summary

The methodology for deriving SQGs, detailed in Schedule B5b, was implemented to calculate SQGs based on different types of toxicity data for eight contaminants (arsenic, chromium, copper, DDT, lead, naphthalene, nickel, zinc). These eight chemicals were selected as they have a variety of physicochemical properties and, as a result, would behave differently in the environment. They are frequently found in urban Australian contaminated sites. The results of this process are summarised below for each contaminant. Some contaminants have the potential to leach from the contaminated site and thus may cause deleterious effects on groundwater and surface water ecosystems. The fact that contaminants can leach can be taken into account in deriving SQGs. This was done for zinc and arsenic, to illustrate the process and to illustrate the effect that it can have on the resulting SQG.

There was a considerable amount of toxicity data available for the essential element zinc. Zinc does not biomagnify but has the potential to leach from contaminated soil to groundwater. The minimum data requirements to use the SSD method were exceeded, there were multiple normalisation relationships, and there was an ageing/leaching factor. The toxicity data could be expressed in terms of added Zn concentrations; therefore, high reliability soil-specific Zn $ACL_{(NOEC\ \&\ EC10)}$, $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ values and corresponding SQG values could be derived for:

- fresh contamination
- aged contamination
- protection of aquatic ecosystems
- areas of ecological significance, urban residential/public open space, and commercial/industrial land uses.

Soil-specific ACLs could be derived, so a suite of values were generated. For example, the $ACL_{(NOEC\ \&\ EC10)}$ values for urban residential/public open space sites freshly contaminated with Zn ranged from 20 (at a cation exchange capacity of 5 and a soil pH of 4) to 330 mg/kg (at a cation exchange capacity of 60 and a soil pH of 7.5). The range of ACL values reflects the ability of different soils to modify the bioavailability and toxicity of Zn. Correcting for ageing led to a marked increase in the ACL values. The corresponding $ACL_{(NOEC\ \&\ EC10)}$ values for aged Zn contamination range from 45–800 mg/kg. As such, correcting for the ageing of Zn led to a more than doubling of the recommended ACL values. The $ACL_{(LOEC\ \&\ EC30)}$ and $ACL_{(EC50)}$ values were approximately 1.25–2 and 1.5–2 times larger, respectively, than the corresponding $ACL_{(NOEC\ \&\ EC10)}$ values. The lowest of the Zn ACLs for urban residential land/public open space (20 mg/kg) are essentially identical to the lowest corresponding international SQGs, while the higher Zn ACLs are considerably larger than any international SQG.

Arsenic does not biomagnify in oxidised soils but has the potential to leach from contaminated soil to groundwater. Therefore, only the direct toxicity route of exposure needs to be considered in deriving the SQGs. The minimum data requirements to use the SSD method were exceeded, there were no normalisation relationships, and an ageing/leaching factor was available.

The toxicity data could only be expressed in terms of total As concentrations, therefore moderate reliability generic (not soil-specific) As $SQG_{(NOEC\ \&\ EC10)}$, $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values could be derived for:

- fresh contamination
- aged contamination
- protection of aquatic ecosystems
- areas of ecological significance, urban residential/public open space, and commercial/industrial land uses.

The generic As $SQG_{(NOEC\ \&\ EC10)}$ value for soils with areas of ecological significance, urban residential/public open space and commercial/industrial land uses were 8, 20 and 30 mg/kg (total As) respectively. The $SQG_{(LOEC\ \&\ EC30)}$ and $SQG_{(EC50)}$ values were approximately 2.5–5 and 3.75–5 times larger, respectively, than the corresponding $SQG_{(NOEC\ \&\ EC10)}$ values. The As $SQG_{(NOEC\ \&\ EC10)}$ for urban

residential/public open space soils is identical to the superseded interim urban EIL of 20 mg/kg (NEPC1999). Both the As $SQG_{(NOEC \& EC10)}$ and the superseded EIL lie in the lower portion of the range of international As SQGs. The $SQG_{(NOEC \& EC10)}$ for aged contamination, at 40 mg/kg, was twice the superseded interim urban EIL for As. The aged As $SQG_{(LOEC \& EC30)}$ for urban residential/public open space soils lies in the upper part of the range of international SQGs while the aged As $SQG_{(EC50)}$ value for urban residential/public open space soils is markedly larger than any other international SQG.

Naphthalene does not biomagnify and has only a moderate potential to leach to groundwater. Therefore, only the direct toxicity exposure route was considered in deriving the SQGs. The minimum data requirements to use the SSD method were exceeded, there were no normalisation relationships, and there was no ageing/leaching factor. The toxicity data could only be expressed as total naphthalene concentrations. Therefore, moderate reliability generic (not soil-specific) naphthalene $SQG_{(NOEC \& EC10)}$, $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values could be derived for:

- fresh contamination
- areas of ecological significance, urban residential/public open space and commercial/industrial land uses.

The generic naphthalene $SQG_{(NOEC \& EC10)}$ values for soils with areas of ecological significance, urban residential/public open space and commercial/industrial land uses were 5, 70 and 150 mg/kg (total naphthalene) respectively. The $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values were approximately 2–2.5 and 5 times larger, respectively, than the corresponding $SQG_{(NOEC \& EC10)}$ values. There is only a very limited number of international SQGs for naphthalene, which differ markedly (that is, from 0.6 to 125). The $SQG_{(NOEC \& EC10)}$ for urban residential/public open space soils of 70 mg/kg is very similar to the top of the EU range of SQGs and in the middle of the range of collated international SQGs.

DDT biomagnifies and has a very low potential to leach to groundwater. Therefore, only the biomagnification and direct toxicity exposure pathways were assessed in deriving SQGs. The minimum data requirements to use the SSD method were exceeded, there were no normalisation relationships, and there was no ageing/leaching factor. The toxicity data could only be expressed as total DDT concentrations. Therefore, moderate reliability generic (not soil-specific) DDT $SQG_{(NOEC \& EC10)}$, $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ could be derived for:

- fresh contamination
- areas of ecological significance, urban residential/public open space, and commercial/industrial land uses.

The generic DDT $SQG_{(NOEC \& EC10)}$ values for soils with areas of ecological significance, urban residential/public open space and commercial/industrial land uses were 1, 70 and 250 mg/kg (total DDT) respectively. The $SQG_{(LOEC \& EC30)}$ and $SQG_{(EC50)}$ values were approximately 2.6– 2 and 5–6 times larger, respectively, than the corresponding $SQG_{(NOEC \& EC10)}$ values. The international SQGs for DDT range from 0.01 to 4 mg/kg. The $SQG_{(NOEC \& EC10)}$ value for freshly contaminated urban residential/public open space soil is thus considerably larger than the international guidelines but is considerably smaller than the HILs, which range from 260 to 4000 mg/kg (see Schedule B1).

Copper is an essential element. It has a low potential to leach to groundwater. Copper does not biomagnify and therefore only direct toxic effects were considered. There was an extensive toxicity data set for Cu (39 species or soil microbial processes). There were normalisation relationships available for plants, invertebrates and soil microbial processes. An ageing/leaching factor was also available. Therefore high reliability soil-specific ACLs could be derived using NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data for:

- fresh contamination
- aged contamination
- areas of ecological significance, urban residential/public open space, and commercial/industrial land uses.

The $ACL_{(NOEC\ and\ EC10)}$ values for urban residential/public open space sites freshly contaminated with Cu ranged from approximately 20 (at a soil pH of 4.5) to 70 mg added Cu/kg (at a soil pH of 8). Correcting for ageing led to a marked increase in the ACL values. The corresponding ACL values for aged Cu contamination range from 30–120 mg added Cu/kg. The range of ACL values reflects the ability of different soils to modify the bioavailability and toxicity of Cu. The ACLs based on LOEC and EC_{30} data and based on EC_{50} data were approximately 1.5–2 and 2.5–3 times larger, respectively, than the corresponding SQGs based on NOEC and EC_{10} data. All of the Cu ACLs for residential land use lie within the range of international SQGs for Cu (14–1000 mg/kg). The superseded interim urban EIL for Cu was 100 mg/kg (total Cu). Therefore the superseded interim EIL for Cu falls within the range of values of all of the SQGs for urban residential land/public open space land uses. The SQGs will permit both considerably less and considerably more Cu in urban residential/public open space soils, depending on the properties of the soils.

Lead is not an essential element but it does not biomagnify in terrestrial ecosystems, nor does it have any significant potential to leach to groundwater. There was toxicity data for 19 species and soil microbial processes which included plants, invertebrates and soil microbial processes. There were no useful normalisation relationships. An ageing/leaching factor has been published in the literature. Therefore moderate reliability generic (not soil-specific) Pb SQGs could be derived using NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data for:

- fresh contamination
- aged contamination
- areas of ecological significance, urban residential/public open space, and commercial/industrial land uses.

The generic Pb ACL for urban residential/public open space land use that was calculated using NOEC and EC_{10} data was 130 mg added Pb/kg. The equivalent SQG for aged Pb contamination was 530 mg added Pb/kg. The corresponding ACLs calculated using LOEC and EC_{30} and using EC_{50} data were approximately 2 and 4 times larger than the NOEC and EC_{10} derived ACL values. All the Pb ACLs for urban residential/public open space soils fell within the range of SQGs that have been adopted in other international jurisdictions (25–700 mg/kg).

The superseded interim urban EIL was 600 mg/kg (total Pb). All of the Pb SQGs for fresh contamination are lower than the superseded interim urban EIL. The aged SQGs based on NOEC and EC_{10} are slightly smaller than the superseded interim urban EIL, while the SQGs based on LOEC and EC_{30} and based on EC_{50} data are considerably higher.

Nickel does not biomagnify so only the direct toxicity exposure route was considered in deriving the SQGs. Nickel, however, does have the potential to leach to groundwater. There was toxicity data for a total of 53 plant and animal species or soil microbial processes. In addition, there were normalisation relationships available for invertebrates, plants and soil microbial processes. A soil pH-modified ageing/leaching factor was available. The minimum data requirements to use the SSD method were exceeded, there were no normalisation relationships, and there was no ageing/leaching factor. Therefore high reliability soil-specific ACLs could be derived using NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data for:

- fresh contamination
- aged contamination
- areas of ecological significance, urban residential/public open space, and commercial/industrial land uses.

The soil-specific Ni ACLs based on NOEC and EC_{10} data for urban residential/public open space soils ranged from 10–170 mg added Ni/kg for soils with a CEC ranging from 5 to 60 $cmol_c$/kg. The corresponding ACL values for aged Ni contamination ranged from 15–290 mg added Ni/kg. The ACL values based on LOEC and EC_{30} data and based on EC_{50} data were essentially identical and approximately 3 times larger than the NOEC and EC_{10}-based ACL values. The range of international

SQGs for Ni is 24–500 mg/kg. Thus, only the urban residential/public open space ACLs for soils with a CEC above 40 $cmol_c$/kg lie outside the range of internationally adopted SQGs. The superseded interim urban EIL for Ni was 60 mg/kg (total Ni). All of the SQGs would permit both lower and higher concentrations than the superseded interim urban EIL. In soils with a low Ni bioavailability, the maximum recommended concentration of Ni that can be added is 15 times the superseded interim urban EIL.

Trivalent chromium is an essential element for humans and animals but not for plants. It does not pose a potential environmental problem due to leaching (unless it is oxidised to hexavalent chromium), nor does it biomagnify. Toxicity data was available for a total of 21 invertebrate and plant species and soil microbial processes. There were only normalisation relationships available for earthworms. There was no ageing/leaching factor available for Cr (III). Therefore moderate reliability soil-specific ACLs could be derived using NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} data for:

- fresh contamination
- areas of ecological significance, urban residential/public open space and commercial/industrial land uses.

The soil-specific Cr (III) ACL values based on NOEC and EC_{10} data for urban residential/ public open space land uses ranged from 35–75 mg added Cr (III)/kg for soils with a clay content from 1 to greater than 10%. The ACL values based on LOEC and EC_{30} and based on EC_{50} data were approximately 2 and 3 times larger than the NOEC-based ACLs. The ACLs for aged Cr (III) contamination were approximately 2.5 times larger than the corresponding ACLs for fresh contamination. The ACLs for Cr (III) based on NOEC and EC_{10} data are consistent with other internationally adopted Cr (III) SQGs. The ACL values based on LOEC and EC_{30} and on EC_{50} data are larger than the current international Cr (III) SQGs.

The superseded interim urban EIL for total Cr was 400 mg/kg. This is considerably higher than any of the SQGs for fresh Cr (III) by a factor of at least 2.6. The aged ACLs are essentially 2.5 times larger than the corresponding fresh ACLs.

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13 Appendices

13.1 Appendix A: Raw toxicity data for zinc

There are three tables in this appendix (Tables A1 to A3).

Table A1: Raw toxicity data for zinc to soil microbial processes with the corresponding toxicity values when they were normalised to the Australian reference soil, the corresponding values when corrected for ageing and leaching, and the source of the data.

Geographical location	Soil process	Soil pH	Delta pH	EC_{10} or NOEC	Log EC_{10} or NOEC	Log normalised EC_{10} or NOEC	Normalised EC_{10} or NOEC	Age corrected normalised EC_{10} or NOEC	Source
Europe	Acetate decomposition	7.4	-1.4	303	2.48	2.27	187	560	Vanbeelen et al. 1994
Europe	Amidase	7.4	-1.4	200	2.3	2.09	123	370	Hemida et al. 1997
Europe	Amidase	7.5	-1.5	200	2.3	2.08	119	357	Hemida et al. 1997
Europe	Ammonification	7.1	-1.1	1000	3	2.84	684	2052	Premi & Cornfield 1969
Europe	Arylsulphatase	6.2	-0.2	820	2.91	2.88	765	2296	Al-Khafaji & Tabatabai 1979
Europe	Arylsulphatase	7.8	-1.8	140	2.15	1.88	75	226	Al-Khafaji & Tabatabai 1979
Europe	Arylsulphatase	5.8	0.2	164	2.21	2.24	176	527	Al-Khafaji & Tabatabai 1979
Europe	Arylsulphatase	7.4	-1.4	820	2.91	2.7	506	1517	Al-Khafaji & Tabatabai 1979
Europe	Arylsulphatase	5.1	0.9	728	2.86	3	993	2980	Haanstra & Doelman 1991
Europe	Arylsulphatase	7.7	-1.7	105	2.02	1.77	58.4	175	Haanstra & Doelman 1991
Europe	Arylsulphatase	6.8	-0.8	2353	3.37	3.25	1785	5355	Haanstra & Doelman 1991
Europe	Arylsulphatase	7.4	-1.4	151	2.18	1.97	93	279	Haanstra & Doelman 1991
Europe	Denitrification	6.8	-0.8	100	2	1.88	76	228	Bollag & Barabasz 1979
Europe	Nitrate reductase	7.4	-1.4	67	1.83	1.62	41	124	Hemida et al. 1997
Europe	N-mineralisation	6.9	-0.9	100	2	1.87	73	220	Chang & Broadbent 1982
Europe	N-mineralisation	5.8	0.2	164	2.21	2.24	176	527	Liang & Tabatabai 1977
Europe	N-mineralisation	6.6	-0.6	164	2.21	2.12	133	400	Liang & Tabatabai 1977
Europe	N-mineralisation	7.8	-1.8	164	2.21	1.94	88	264	Liang & Tabatabai 1977
Europe	N-mineralisation	7.4	-1.4	164	2.21	2	101	303	Liang & Tabatabai 1977
Europe	N-mineralisation	3.4	2.6	233	2.37	2.76	572	1716	Necker & Kunze 1986
Europe	Phosphatase	5.1	0.9	1341	3.13	3.26	1830	5490	Doelman & Haanstra 1989

Geographical location	Soil process	Soil pH	Delta pH	EC_{10} or NOEC	Log EC_{10} or NOEC	Log normalised EC_{10} or NOEC	Normalised EC_{10} or NOEC	Age corrected normalised EC_{10} or NOEC	Source
Europe	Phosphatase	6.8	-0.8	160	2.2	2.08	121	364	Doelman & Haanstra 1989
Europe	Phosphatase	7.4	-1.4	2623	3.42	3.21	1617	4852	Doelman & Haanstra 1989
Europe	Phosphatase	5.8	0.2	164	2.21	2.24	176	527	Juma & Tabatabai 1977
Europe	Phosphatase	7.4	-1.4	164	2.21	2	101	303	Juma & Tabatabai 1977
Europe	Phosphatase	4.7	1.3	508	2.71	2.9	796	2388	Svenson 1986
Europe	Phytase	4.7	1.3	590	2.77	2.97	924	2773	Svenson 1986
Europe	Py-phosphatase	4.6	1.4	1640	3.21	3.42	2660	7979	Stott et al. 1985
Europe	Py-phosphatase	6.2	-0.2	1640	3.21	3.18	1531	4592	Stott et al. 1985
Europe	Py-phosphatase	7.4	-1.4	1640	3.21	3	1011	3034	Stott et al. 1985
Europe	Respiration	6.9	-0.9	17	1.23	1.1	12	37	Chang & Broadbent 1981
Europe	Respiration	6.7	-0.7	110	2.04	1.94	86	259	Lighthart et al. 1983
Europe	Respiration	7	-1	165	2.22	2.07	117	350	Lighthart et al. 1983
Europe	Respiration	7.2	-1.2	110	2.04	1.86	73	218	Lighthart et al. 1983
Europe	Respiration	8.2	-2.2	17	1.23	0.9	8	24	Lighthart et al. 1983
Europe	Respiration	5.2	0.8	50	1.7	1.82	66	198	Saviozzi et al. 1997
Europe	Respiration	3	3	120	2.08	2.53	338	1015	Smolders et al, 2003
Europe	Respiration	4.8	1.2	469	2.67	2.85	710	2130	Smolders et al, 2003
Europe	Respiration	5.1	0.9	50	1.7	1.83	68	205	Smolders et al. 2003
Europe	Respiration	5.7	0.3	1400	3.15	3.19	1553	4659	Smolders et al. 2003
Europe	Respiration	6.8	-0.8	38	1.58	1.46	29	86	Smolders et al. 2003
Europe	Respiration	7.4	-1.4	150	2.18	1.97	92	277	Smolders et al. 2003
Europe	Respiration	7.4	-1.4	600	2.78	2.57	370	1110	Smolders et al. 2003
Europe	Respiration	7.5	-1.5	150	2.18	1.95	89	268	Smolders et al. 2003
Europe	Respiration	7.5	-1.5	300	2.48	2.25	179	536	Smolders et al. 2003
Australia	SIN[1]	5.42	0.58	209	2.32	2.52	328	328	NBRP unpublished data[2]
Australia	SIN	4.52	1.48	63	1.8	2.3	200	200	NBRP unpublished data
Australia	SIN	7.26	-1.26	1181	3.07	2.64	440	440	NBRP unpublished data
Australia	SIN	4.89	1.12	346	2.54	2.92	829	829	NBRP unpublished data
Australia	SIN	3.96	2.04	10	1.01	1.7	50	50	NBRP unpublished data

Geographical location	Soil process	Soil pH	Delta pH	EC_{10} or NOEC	Log EC_{10} or NOEC	Log normalised EC_{10} or NOEC	Normalised EC_{10} or NOEC	Age corrected normalised EC_{10} or NOEC	Source
Australia	SIN	4.39	1.61	70	1.84	2.39	247	247	NBRP unpublished data
Australia	SIN	5.03	0.97	270	2.43	2.76	577	577	NBRP unpublished data
Australia	SIN	5.13	0.87	901	2.95	3.25	1782	1782	NBRP unpublished data
Australia	SIN	6.32	-0.32	919	2.96	2.85	716	716	NBRP unpublished data
Australia	SIN	6.33	-0.33	462	2.66	2.55	357	356	NBRP unpublished data
Australia	SIN	4.8	1.2	188	2.27	2.68	482	482	NBRP unpublished data
Australia	SIN	7.63	-1.63	7538	3.88	3.32	2110	2110	NBRP unpublished data
Australia	SIR[3]	5.42	0.58	158	2.2	2.4	249	249	NBRP unpublished data
Australia	SIR	4.52	1.48	369	2.57	3.07	1176	1176	NBRP unpublished data
Australia	SIR	7.26	-1.26	187	2.27	1.84	70	70	NBRP unpublished data
Australia	SIR	4.89	1.12	462	2.66	3.04	1105	1105	NBRP unpublished data
Australia	SIR	4.39	1.61	73	1.86	2.41	257	257	NBRP unpublished data
Australia	SIR	5.03	0.97	499	2.7	3.03	1064	1064	NBRP unpublished data
Australia	SIR	5.13	0.87	281	2.45	2.74	555	555	NBRP unpublished data
Australia	SIR	6.32	-0.32	25	1.41	1.3	20	20	NBRP unpublished data
Australia	SIR	6.33	-0.33	268	2.43	2.32	207	207	NBRP unpublished data
Australia	SIR	4.8	1.2	345	2.54	2.95	885	885	NBRP unpublished data
Australia	SIR	7.63	-1.63	190	2.28	1.73	53	53	NBRP unpublished data
Europe	Urease	5.1	0.9	30	1.48	1.61	41	123	Doelman & Haanstra 1986
Europe	Urease	7.7	-1.7	70	1.85	1.59	39	117	Doelman & Haanstra 1986
Europe	Urease	6.8	-0.8	460	2.66	2.54	349	1047	Doelman & Haanstra 1986
Europe	Urease	7.4	-1.4	30	1.48	1.27	19	55	Doelman & Haanstra 1986
Europe	Urease	7.4	-1.4	64	1.81	1.6	39	118	Tabatabai 1977
Europe	Urease	7.8	-1.8	52	1.72	1.45	28	84	Tabatabai 1977
Europe	Urease	5.8	0.2	109	2.04	2.07	117	350	Tabatabai 1977

1 SIN = substrate induced nitrification

2 = This EC_{10} data has not been published but was determined using the same biological response and soil concentration data as the EC_{50} values published in Broos et al. (2007)

3 SIR = substrate induced respiration.

Table A2: Raw toxicity data for zinc to soil invertebrates with the corresponding toxicity values when they were normalised to the Australian reference soil, the corresponding values when corrected for ageing and leaching, and the source of the data.

Scientific name	Toxicity end point	CEC[1]	Log CEC	Delta log CEC	EC_{10} or NOEC	Log EC_{10} or NOEC	Log normalised EC_{10}	Normalised EC_{10}	Aged normalised EC_{10}	Source
Acrobeloides sp.		3.6	0.56	0.44	99	1.99	2.34	221	663	Korthals et al. 1996
A. rosea[2]	survival	15	1.18	-0.18	538	2.73	2.59	391	1172	Spurgeon & Hopkin 1996
A. caliginosa	reproduction	9.2	0.97	0.03	210	2.32	2.35	223	669	Spurgeon et al. 2000
C. elegans[3]		2.4	0.38	0.62	112	2.05	2.54	345	1035	Boyd & Williams 2003
C. elegans		7.2	0.86	0.14	118	2.07	2.18	153	458	Boyd & Williams 2003
C. elegans		28.4	1.45	-0.45	383	2.58	2.22	168	504	Boyd & Williams 2003
C. elegans		10.0	1	0	25	1.4	1.4	25	76	Jonker et al. 2004
C. elegans[4]		3.6	0.56	0.44	308	2.49	2.84	689	2068	Korthals et al. 1996
E. andrei[5]	reproduction	26	1.41	-0.41	320	2.51	2.18	152	456	van Gestel et al. 1993
E. fetida[5]	reproduction	26	1.41	-0.41	350	2.54	2.22	166	499	Spurgeon et al. 1997
E. fetida	reproduction	26	1.41	-0.41	350	2.54	2.22	166	499	Spurgeon et al. 1997
E. fetida	reproduction	15	1.18	-0.18	237	2.37	2.24	172	516	Spurgeon & Hopkin 1996
E. fetida	reproduction	15	1.18	-0.18	199	2.3	2.16	144	433	Spurgeon et al. 1994
E. fetida	reproduction	26	1.41	-0.41	553	2.74	2.42	263	788	Spurgeon & Hopkin 1996
E. fetida	reproduction	18	1.27	-0.27	97	1.99	1.78	60	179	Spurgeon & Hopkin 1996
E. fetida	reproduction	33	1.52	-0.52	484	2.68	2.28	189	568	Spurgeon & Hopkin 1996
E. fetida	reproduction	16	1.21	-0.21	85	1.93	1.77	58	175	Spurgeon & Hopkin 1996
E. fetida	reproduction	22	1.34	-0.34	183	2.26	2	99	297	Spurgeon & Hopkin 1996
E. fetida	reproduction	27	1.44	-0.44	414	2.62	2.27	186	559	Spurgeon & Hopkin 1996
E. fetida	reproduction	14	1.14	-0.14	115	2.06	1.95	90	269	Spurgeon & Hopkin 1996
E. fetida	reproduction	18	1.25	-0.25	161	2.21	2.01	101	304	Spurgeon & Hopkin 1996
E. fetida	reproduction	22	1.35	-0.35	223	2.35	2.08	119	357	Spurgeon & Hopkin 1996
E. fetida	reproduction	5.8	0.76	0.24	180	2.26	2.44	277	830	Smolders et al. 2003
E. fetida	reproduction	1.9	0.28	0.72	100	2	2.57	371	1114	Smolders et al. 2003
E. fetida	reproduction	13.3	1.12	-0.12	320	2.51	2.41	255	766	Smolders et al. 2003
E. fetida	reproduction	11.2	1.05	-0.05	560	2.75	2.71	512	1536	Smolders et al. 2003

Scientific name	Toxicity end point	CEC[1]	Log CEC	Delta log CEC	EC_{10} or NOEC	Log EC_{10} or NOEC	Log normalised EC_{10}	Normalised EC_{10}	Aged normalised EC_{10}	Source
E. fetida	reproduction	4.7	0.67	0.33	320	2.51	2.76	581	1743	Smolders et al. 2003
E. fetida	reproduction	21.1	1.32	-0.32	1000	3	2.74	554	1663	Smolders et al. 2003
E. fetida	reproduction	23.4	1.37	-0.37	560	2.75	2.46	286	858	Smolders et al. 2003
E. fetida	reproduction	8.9	0.95	0.05	180	2.26	2.3	197	592	Smolders et al. 2003
E. fetida	reproduction	20.1	1.3	-0.3	180	2.26	2.02	104	311	Smolders et al. 2003
E. fetida	reproduction	16.9	1.23	-0.23	350	2.54	2.36	231	694	Smolders et al. 2003
E. fetida	reproduction	15	1.18	-0.18	572	2.76	2.62	415	1246	Spurgeon & Hopkin 1996
E. fetida	reproduction	9.2	0.97	0.03	792	2.9	2.93	843	2530	Spurgeon et al. 2000
E. albidus[6]		15	1.18	-0.18	262	2.42	2.28	190	571	Lock & Janssen 2001
E. albidus		15	1.18	-0.18	132	2.12	1.98	96	287	Lock & Janssen 2001
E. albidus		15	1.18	-0.18	180	2.26	2.12	131	392	Lock & Janssen 2001
E. albidus		11.5	1.06	-0.06	100	2	1.95	90	269	Lock & Janssen 2001
E. crypticus[6]		15	1.18	-0.18	380	2.58	2.44	276	828	Lock & Janssen 2001
Eucephalobus sp.		3.6	0.56	0.44	60	1.78	2.13	134	403	Korthals et al. 1996
F. candida[7]	reproduction	26	1.41	-0.41	366	2.56	2.1	125	375	Smit & van Gestel 1998
F. candida	reproduction	26	1.41	-0.41	620	2.79	2.33	212	636	Sandifer & Hopkin 1996
F. candida	reproduction	26	1.41	-0.41	399	2.6	2.13	136	409	van Gestel & Hensbergen 1997
F. candida	reproduction	5	0.66	0.34	275	2.44	2.83	680	2040	Smit & van Gestel 1998
F. candida	reproduction	5	0.66	0.34	314	2.5	2.89	776	2329	Smit & van Gestel 1998
F. candida	reproduction	22	1.34	-0.34	300	2.48	2.09	123	370	Sandifer & Hopkin 1996
F. candida	reproduction	20	1.3	-0.3	300	2.48	2.14	137	411	Sandifer & Hopkin 1996
F. candida	reproduction	26	1.41	-0.41	300	2.48	2.01	103	308	Sandifer & Hopkin 1997
F. candida	reproduction	1.9	0.28	0.72	32	1.51	2.33	213	638	Smolders et al. 2003
F. candida	reproduction	13.3	1.12	-0.12	320	2.51	2.36	231	694	Smolders et al. 2003
F. candida	reproduction	11.2	1.05	-0.05	100	2	1.94	88	264	Smolders et al, 2003
F. candida	reproduction	22.6	1.35	-0.35	320	2.51	2.1	126	379	Smolders et al. 2003
F. candida	reproduction	21.1	1.32	-0.32	320	2.51	2.14	137	410	Smolders et al. 2003
F. candida	reproduction	20	1.3	-0.3	560	2.75	2.41	254	762	Smolders et al. 2003

Scientific name	Toxicity end point	CEC[1]	Log CEC	Delta log CEC	EC_{10} or NOEC	Log EC_{10} or NOEC	Log normalised EC_{10}	Normalised EC_{10}	Aged normalised EC_{10}	Source
F. candida	reproduction	36.3	1.56	-0.56	1000	3	2.36	230	690	Smolders et al. 2003
F. candida	reproduction	16.9	1.23	-0.23	320	2.51	2.25	176	528	Smolders et al. 2003
L. rubellus[8]	reproduction	15	1.18	-0.18	121	2.08	1.94	88	264	Spurgeon & Hopkin 1996
L. rubellus	reproduction	9.2	0.97	0.03	517	2.71	2.74	550	1649	Spurgeon et al. 2000
L. rubellus	reproduction	9.2	0.97	0.03	325	2.51	2.54	346	1039	Spurgeon & Hopkin 1999
L. rubellus	reproduction	9.2	0.97	0.03	648	2.81	2.84	690	2069	Spurgeon & Hopkin 1999
L. rubellus	reproduction	9.2	0.97	0.03	470	2.67	2.7	500	1501	Spurgeon & Hopkin 1999
L. terrestris[8]	reproduction	9.2	0.97	0.03	998	3	3.03	1062	3187	Spurgeon et al. 2000
Nematode community		5.1	0.7	0.3	560	2.75	2.98	961	2882	Smit et al. 2002
Nematode community		5.1	0.7	0.3	180	2.26	2.49	309	926	Smit et al. 2002
Nematode community		5.1	0.7	0.3	180	2.26	2.49	309	926	Smit et al. 2002
Nematode community		5.1	0.7	0.3	56	1.75	1.98	96	288	Smit et al. 2002
Plectus sp.		3.6	0.56	0.44	10	1.02	1.37	23	70	Korthals et al. 1996
Rhabditidae sp.		3.6	0.56	0.44	89	1.95	2.3	199	597	Korthals et al. 1996

[1] CEC = cation exchange capacity [2] A. = *Aporrectodea* [3] C. = *Caenorhabditis* [4]. dauer larval stage [5] E. = *Eisenia* [6] E. = *Enchytraeus* [7] F. = *Folsomia* [8] L. = *Lumbriculus*.

Table A3: Raw toxicity data for zinc to plant species with the corresponding toxicity values when they were normalised to the Australian reference soil, the corresponding values when corrected for ageing and leaching, and the source of the data. The wheat toxicity was sourced from Warne et al. (2008a), all other Australian data is unpublished data from the Australian National Biosolids Research Program.

Site	Plant species	Scientific name	CEC	Log CEC	Delta CEC	pH	Delta pH	EC_{10}	Log EC_{10}	Log normalised EC_{10}	Normalised EC_{10}	Aged normalised EC_{10}
Europe[1]	Alfalfa	*Medicago sativa*				7.50	-1.50	300.00	2.48	2.30	198.21	594.62
Australia	Barley	*Hordeum vulgare*	9.95	1.00	0.00	7.63	-1.63	56.36	1.75	1.31	20.49	20.49
Australia	Barley	*H. vulgare*	17.71	1.25	-0.25	6.32	-0.32	490.45	2.69	2.43	268.91	268.91
Australia	Barley	*H. vulgare*	10.29	1.01	-0.01	6.33	-0.33	486.69	2.69	2.59	387.88	387.88
Europe[1]	Barley	*H. vulgare*				7.50	-1.50	100.00	2.00	1.82		
Europe[2]	Barley	*H. vulgare*	17.64	1.25	-0.25	5.60	0.40	33.30	1.52	1.35	22.44	67.31
Europe[3]	Barley	*H. vulgare*				7.80	-1.80	215.00	2.33	2.12		
Europe[1]	Beet	*Beta vulgaris*				7.50	-1.50	300.00	2.48	2.30	198.21	594.62
Europe[4]	Black or white lentil	*Vigna mungo L.*				6.20	-0.20	100.00	2.00	1.98	94.62	283.87
Australia	Canola	*Brassica napus*	10.29	1.01	-0.01	6.33	-0.33	178.84	2.25	2.15	142.53	142.53
Australia	Canola	*B. napus*	3.16	0.50	0.50	5.42	0.58	139.13	2.14	2.65	448.08	448.08
Australia	Canola	*B. napus*	4.95	0.69	0.31	4.80	1.20	52.26	1.72	2.26	181.45	181.45
Australia	Canola	*B. napus*	12.99	1.11	-0.11	4.89	1.12	144.60	2.16	2.38	241.34	241.34
Europe[5]	Common vetch	*Vicia sativa*	12.46	1.10		5.00	1.00	32.00	1.51	1.63	42.18	126.55
Australia	Cotton	*Gossypium sp*	60.97	1.79	-0.79	7.26	-1.26	2127.60	3.33	2.44	272.44	272.44
Europe[6]	Fenugreek	*Trigonella foenum graceum*	17.02	1.23		8.30	-2.30	200.00	2.30	2.03	105.93	317.80
Europe[1]	Lettuce	*Lactuca sativa*				7.50	-1.50	400.00	2.60	2.42	264.28	792.83
Australia	Maize	*Zea mays*	16.51	1.22	-0.22	5.03	0.97	500.53	2.70	2.81	644.29	644.29
Europe[7]	Maize	*Z. mays*	11.58	1.06	-0.06	4.90	1.10	83.00	1.92	1.99	98.72	296.17
Europe[1]	Maize	*Z. mays*				7.50	-1.50	300.00	2.48	2.30	198.21	594.62
Europe[1]	Maize	*Z. mays*				7.50	-1.50	200.00	2.30	2.12	132.14	396.42
Australia	Millet	*Panicum milaceum*	16.51	1.22	-0.22	5.03	0.97	419.12	2.62	2.73	539.50	539.50
Europe[8]	Oats	*Avena sativa*	9.19	0.96	0.04	5.60	0.40	100.00	2.00	2.08	120.38	361.14

Site	Plant species	Scientific name	CEC	Log CEC	Delta CEC	pH	Delta pH	EC_{10}	Log EC_{10}	Log normalised EC_{10}	Normalised EC_{10}	Aged normalised EC_{10}
Europe[8]	Oats	*A. sativa*	24.02	1.38	-0.38	5.40	0.60	200.00	2.30	2.03	108.22	324.66
Europe[8]	Oats	*A. sativa*	5.50	0.74	0.26	5.00	1.00	200.00	2.30	2.65	448.99	1346.96
Europe[8]	Oats	*A. sativa*	11.50	1.06	-0.06	5.40	0.60	400.00	2.60	2.62	417.04	1251.11
Europe[6]	Onion	*Allium cepa*	17.02	1.23	-0.23	8.30	-2.30	200.00	2.30	1.82	65.97	197.92
Europe[1]	Pea	*Pisum sativum (perfection)*				7.50	-1.50	400.00	2.60	2.42	264.28	792.83
Australia	Peanuts	*Arachis hypogaea*	16.51	1.22	-0.22	5.03	0.97	227.06	2.36	2.47	292.27	292.27
Australia	Peanuts	*A. hypogaea*	4.94	0.69	0.31	4.52	1.48	16.29	1.21	1.83	67.27	67.27
Europe[5]	Red clover	*Trifolium pratense*	26.42	1.42		6.20	-6.20	100.00	2.00	1.26	18.03	54.09
Europe[5]	Red clover	*T. pratense*	26.42	1.42		6.20	-0.20	84.00	1.92	1.90	79.48	238.45
Europe[5]	Red clover	*T. pratense*	12.46	1.10		5.00	1.00	32.00	1.51	1.63	42.18	126.55
Europe[5]	Red clover	*T. pratense*	3.52	0.55		5.30	0.70	32.00	1.51	1.59	38.83	116.49
Europe[9]	Red clover	*T. pratense*	3.52	0.55		5.30	0.70	32.00	1.51	1.59	38.83	116.49
Europe[9]	Red clover	*T. pratense*	3.52	0.55		5.30	0.70	32.00	1.51	1.59	38.83	116.49
Europe[1]	Spinach	*Spinacia oleracea*				7.50	-1.50	200.00	2.30	2.12	132.14	396.42
Australia	Sorghum	*Sorghum spp*	60.97	1.79	-0.79	7.26	-1.26	1660.64	3.22	2.33	212.64	212.64
Europe[1]	Sorghum	*S. bicolor var RS-626)*				7.50	-1.50	200.00	2.30	2.12	132.14	396.42
Europe[1]	Sorghum	*S. bicolor var XK-125)*				7.50	-1.50	100.00	2.00	1.82	66.07	198.21
Australia	Sugar cane	*Saccharum*	4.94	0.69	0.31	4.52	1.48	780.00	2.89	3.51	3220.34	3220.34
Europe[1]	Tomato	*Lycopersicon esculentum*				7.50	-1.50	400.00	2.60	2.42	264.28	792.83
Australia	Triticale	*Tritosecale*	11.58	1.06	-0.06	3.96	2.04	310.18	2.49	3.00	998.11	998.11
Australia	Wheat	*Triticum aestivum*	9.95	1.00	0.00	7.63	-1.63	4764.45	3.68	3.24	1732.26	1732.26
Australia	Wheat	*T. aestivum*	3.16	0.50	0.50	5.42	0.58	91.05	1.96	2.47	293.23	293.23
Australia	Wheat	*T. aestivum*	7.82	0.89	0.11	4.39	1.61	373.62	2.57	3.08	1215.42	1215.42
Australia	Wheat	*T. aestivum*	17.71	1.25	-0.25	6.32	-0.32	1216.50	3.09	2.82	667.01	667.01

Site	Plant species	Scientific name	CEC	Log CEC	Delta CEC	pH	Delta pH	EC_{10}	Log EC_{10}	Log normalised EC_{10}	Normalised EC_{10}	Aged normalised EC_{10}
Australia	Wheat	*T. aestivum*	17.41	1.24	-0.24	5.13	0.87	1312.80	3.12	3.19	1532.36	1532.36
Australia	Wheat	*T. aestivum*	10.29	1.01	-0.01	6.33	-0.33	688.94	2.84	2.74	549.07	549.07
Australia	Wheat	*T. aestivum*	4.95	0.69	0.31	4.80	1.20	101.93	2.01	2.55	353.88	353.88
Australia	Wheat	*T. aestivum*	16.51	1.22	-0.22	5.03	0.97	262.46	2.42	2.53	337.84	337.84
Australia	Wheat	*T. aestivum*	60.97	1.79	-0.79	7.26	-1.26	2351.09	3.37	2.48	301.05	301.05
Australia	Wheat	*T. aestivum*	12.99	1.11	-0.11	4.89	1.12	428.96	2.63	2.85	715.97	715.97
Australia	Wheat	*T. aestivum*	11.58	1.06	-0.06	3.96	2.04	255.16	2.41	2.91	821.05	821.05

[1] Boawn and Rasmussen 1971; [2] Luo and Rimmer 1995; [3] Aery and Jagatiya 1997; [4] Kalyanaraman and Sivagurunathan 1993; [5] van der Hoeven & Henzen 1994; [6] Dang et al. 1990; [7] MacLean 1974; [8] De Haan et al. 1985; [9] Hooftman and Henzen 1996.

13.2 Appendix B. Raw toxicity data for arsenic

There are two tables in this appendix (Tables B1 and B2).

Table B1: Raw toxicity data for arsenic to plants with the corresponding toxicity values when they were converted to NOEC values.

Crop	Toxic concentration soil (mg/kg)		Reported toxic effect (%)	Interpreted toxic effect	Est. NOEC (mg/kg)	Source
	Range	Value or mean of range				
Barley		283	lower yield	LOEC	113.2	Cooper et al. 1931
Barley			90	NOEC		Davis et al. 1978
Bean	0–10	5	58–95	LOEC	2.07	Woolson 1973
Bean	<25		86	NOEC		Stewart & Smith 1922
Bean		25	lower yield	LOEC	10	Walsh & Keeney 1975
Bean		25	lower yield	LOEC	10	Sandberg & Allen 1975
Bean	0–45	22.5	89	NOEC	22.5	Jacobs and Keeney 1970
Bean		140	77 (NS)	NOEC	140	Chisholm & MacPhee 1972
Bean		140	40	EC_{50}	28	MacPhee et al. 1960
Bean		414	71	LOEC	414	Clements & Munson 1947
Blueberry		44	lower yield	LOEC	17.6	Walsh & Keeney 1975
Blueberry		70	78	LOEC	70	Anastasia & Kender 1973
Corn	10–100	55	55	EC_{50}	11	Woolson et al. 1971
Corn		20	70	LOEC	8	Jacobs & Keeney 1970
Corn		20	90	NOEC	20	Jacobs & Keeney 1970
Corn		50	lower yield	LOEC	20	Sandberg & Allen 1975
Corn		67	24–73	EC_{50}	13.4	Woolson et al. 1971
Corn		80	40	EC_{50}	16	Jacobs & Keeney 1970
Corn		90	91	NOEC	90	Jacobs et al. 1970
Corn		100	86	NOEC	100	Woolson 1972
Corn		125	lower yield	LOEC	50	Sandberg & Allen 1975
Cotton		25	48	EC_{50}	5	Deuel & Swoboda 1972
Cotton		50	lower yield	LOEC	20	Ray 1975
Cotton		50	lower yield	LOEC	20	Ray 1975

Crop	Toxic concentration soil (mg/kg)		Reported toxic effect (%)	Interpreted toxic effect	Est. NOEC (mg/kg)	Source
	Range	Value or mean of range				
Cotton		125	60	EC_{50}	25	Deuel & Swoboda 1972
Cotton		196	lower yield	LOEC	78.4	Ray 1975
Grass		3.2	5	EC_{95}		Millhollon 1970
Grass		45	0–25	LOEC	18	Weaver et al. 1984
Grass		90	50	EC_{50}	18	Weaver et al. 1984
Grass		104	88	NOEC	104	Clements & Munson 1947
Oat	0–10	5	78	NOEC	5	Woolson et al. 1971
Oat	0–10	5	94	NOEC	5	Woolson et al. 1971
Oat		100	2	EC_{98}		Jacobs et al. 1970
Oat	40–290	165	5	EC_{95}		Rosenfels & Crafts 1940
Oat		50	90	NOEC	50	Sandberg & Allen 1975
Oat	160–340	250	5	EC_{95}		Rosenfels & Crafts 1940
Oat		188	lower yield	LOEC	75.2	Cooper et al. 1931
Oat	280–590	435	5	EC_{95}		Rosenfels & Crafts 1940
Oat	540–850	695	5	EC_{95}		Rosenfels & Crafts 1940
Pea	11–14	12.5	90	NOEC	12.5	Steevens et al. 1972
Pea		25	lower yield	LOEC	10	Walsh & Keeney 1975
Pea	25–75	50	85	NOEC	50	Stewart & Smith 1922
Pea	0–45	22.5	90	NOEC	22.5	Jacobs & Keeney 1970
Pea		140	50	EC_{50}	28	MacPhee et al. 1960
Pine	>200	200	lethal	NOEC	200	Sheppard et al. 1985
Pine	>250	250	lethal	NOEC	250	Sheppard et al. 1985
Pine	>500	500	no effect	NOEC	500	Sheppard et al. 1985
Potato	45–73	59	85	NOEC	59	Sheppard et al. 1985
Potato		68	lower yield	LOEC	27.2	Walsh & Keeney 1975
Potato		75	33	EC^{50}	15	Stewart & Smith 1922
Potato		180	79	LOEC	72	Jacobs & Keeney 1970

Crop	Toxic concentration soil (mg/kg)		Reported toxic effect (%)	Interpreted toxic effect	Est. NOEC (mg/kg)	Source
	Range	Value or mean of range				
Radish		2.5	lower yield	LOEC	6.33	Hiltbold 1975
Radish	10–100	55	23–93	EC_{50}	11	Woolson 1973
Radish		15	89	NOEC	15	Sheppard et al. 1985
Radish		36	52	EC_{50}	7.2	Woolson & Isensee 1981
Radish		390	82	NOEC	390	Sheppard et al. 1982
Radish		500	86	NOEC	500	Stewart & Smith 1922
Sedge		1.8	lower yield	LOEC	0.72	Hiltbold 1975
Soyabean		12.5	55	EC_{50}	2.5	Deuel & Swoboda 1972
Soyabean		34	lower yield	LOEC	13.6	Raab 1972a, 1972b
Soyabean		37	65	LOEC	14.8	Woolson & Isensee 1981
Soyabean		50	61	EC_{40}	10	Sandberg & Allen 1975
Soyabean		84	60	EC_{40}	16.8	Deuel & Swoboda 1972
Tomato	0–10	5	77–94	NOEC	8.47	Woolson 1973
Tomato		140	76	LOEC	56	MacPhee et al. 1960
Tomato		514	90	NOEC	514	Clements & Munson 1947
Wheat		94	lower yield	LOEC	37.6	Cooper et al. 1931
Wheat		250	63	LOEC	100	Stewart & Smith 1922

NS= not statistically significant (P>0.05)

Table B2: Raw toxicity data for arsenic to soil invertebrates and terrestrial mammals with the corresponding toxicity values when they were converted to NOEC values.

Common name	Scientific name	Measure of toxicity	Toxicity data (mg/kg)	Est. EC_{10}	Source
Common rat	*Rattus norvegicus*	NOEC	10	10	US EPA 2007
Deer mouse	*Peromyscus maniculatus*	EC_{50}	1600	320	US EPA 2007
Earthworm	*Eisenia fetida*	EC_{50}	100	20	Langdon et al. 2003
Earthworm	*Lumbriculus rubellus*	EC_{50}	1510	302	Langdon et al. 2001
Earthworm	*L. rubellus*	EC_{50}	96	19.2	Langdon et al. 2001
Earthworm	*L. terrestris*	NOEC	100	100	Meharg et al. 1998
Earthworm	*L. terrestris*	NOEC	100	100	Meharg et al. 1998
Fulvous whistling duck	*Dendrocygna bicolor*	EC_{50}	1145	229	Kegley et al. 2008
Northern bobwhite	*Colinus virginianus*	EC_{50}	168.5	33.7	Kegley et al. 2008
Northern bobwhite	*C. virginianus*	EC_{50}	432	86.4	Kegley et al. 2008
Sheep	*Ovis aries*	NOEC	25	25	US EPA 2007

13.3 Appendix C: Raw toxicity data for naphthalene

There are two tables in this appendix (Tables C1 and C2).

Table C1. Raw data for naphthalene where the toxicity was expressed in terms of mg/kg.

Test species		Measure of toxicity	Toxic conc. (mg/kg)	Source
Common name	**Scientific name**			
Common rat	*Rattus norvegicus*	NOEC	1000	US EPA 2007
Earthworm	*Eisenia fetida*	EC_{25}	54	CCME 1999b
European rabbit	*Oryctolagus cuniculus*	NOEC	2000	US EPA 2007
House mouse	*Mus musculus*	LD_{10}	320	US EPA 2007
House mouse	*M. musculus*	LD_{10}	518	US EPA 2007
Lettuce	*Lactuca sativa*	NOEC	100	Adema & Henzen 2001
Lettuce	*L. sativa*	NOEC	32	Adema & Henzen 2001
Lettuce	*L. sativa*	NOEC	100	Adema & Henzen 2001
Lettuce	*L. sativa*	NOEC	3.2	Adema & Henzen 2001
Lettuce	*L. sativa*	NOEC	32	Adema & Henzen 2001
Lettuce	*L. sativa*	EC_{25}	3	CCME 1999b
Northern bobwhite	*Colinus virginianus*	NOEC	1000	US EPA 2007
Northern bobwhite	*C. virginianus*	NOEC	1000	US EPA 2007
Northern bobwhite	*C. virginianus*	LD_{50}	538	US EPA 2007
Radish	*Raphanus sativa*	EC_{25}	61	CCME 1999b
Springtail	*Folsomia fimetaria*	EC_{10}	20	Sverdrup et al. 2002

LD_{10} = dose lethal to 10% of organisms.

Table C2: Raw toxicity data for naphthalene that caused a 50% effect (EC_{50}) and was expressed in terms of g/m², the corresponding value expressed in terms of mg/kg, the corresponding EC_{10} or NOEC values, and the source of the original data.

Test species		EC_{50} (g/m^2)	EC_{50} (mg/kg)	Estimated NOEC or EC_{10} (mg/kg)	Source
Common name	Scientific name				
Mite	*Acari sp.*	13	1000	200	Best et al. 1978
Mite	*Acari sp.*	11	846	169	Best et al. 1978
Mite	*Acari sp.*	24	1846	369	Best et al. 1978
Mite	*Mesostigmata sp.*	10	769	154	Best et al. 1978
Mite	*Mesostigmata sp.*	16	1231	246	Best et al. 1978
Mite	*Oribatida sp.*	10	769	153	Best et al. 1978
Mite	*Oribatida sp.*	24	1846	369	Best et al. 1978
Mite	*Oribatida sp.*	12	923	185	Best et al. 1978
Spider	*Grammonota inornata*	9	692	138	Best et al. 1978
Spider	*G. inornata*	17	1308	262	Best et al. 1978
Spider	*G. inornata*	10	769	154	Best et al. 1978
Springtail	*Collembola sp.*	8	615	123	Best et al. 1978
Springtail	*Collembola sp.*	21	1615	323	Best et al. 1978
Springtail	*Collembola sp.*	16	1231	246	Best et al. 1978
Springtail	*Poduromorpha sp.*	18	1385	277	Best et al. 1978
Springtail	*Poduromorpha sp.*	16	1231	246	Best et al. 1978
Springtail	*Poduromorpha sp.*	8	615	123	Best et al. 1978

13.4 Appendix D: Raw toxicity data for DDT

Table D1:The raw toxicity data for DDT that measured a variety of toxic effects, the estimated NOEC or EC_{10} value, and the source.

Test species		Measure of toxicity	Toxic conc. (mg/kg)	Est. NOEC or EC_{10} (mg/kg)	Source
Common name	Scientific name				
Earthworm	*Eisenia fetida*	EC_{10}	47.7	47.7	Hund-Rindke & Simon 2005
Earthworm	*E. fetida*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Earthworm	*E. fetida*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Field mustard	*Brassica rapa*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Field mustard	*B. rapa*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Field mustard	*B. rapa*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Helmeted guineafowl	*Numida meleagris*	LOEC	75	30	US EPA 2007
House sparrow	*Passer domesticus*	LOEC	1500	600	US EPA 2007
Japanese quail	*Coturnix japonica*	LOEC	200	80	US EPA 2007
Mallard duck	*Anas platyrhynchos*	LOEC	59.5	23.8	US EPA 2007
Northern bobwhite	*Colinus virginianus*	NOEC	50	50	US EPA 2007
Northern bobwhite	*C. virginianus*	LOEC	232	92.8	US EPA 2007
Oats	*Avena sativa*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Oats	*A. sativa*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Oats	*A. sativa*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Ring-necked pheasant	*Phasianus colchicus*	LC_{50}	522	104	US EPA 2007
Soil process	Ammonification	EC_{12}	1250	1250	CCME 1999a
Soil process	Nitrification	EC_{36}	1000	400	CCME 1999a
Soil process	Nitrification	EC_{31}	12.5	5	CCME1999a
Soil process	Nitrification	EC_{24}	50	50	CCME 1999a
Soil process	Nitrification	EC_{22}	100	100	CCME 1999a
Soil process	Potential ammonium oxidation	NOEC	1000	1000	Hund-Rindke & Simon 2005
Soil process	Potential ammonium oxidation	NOEC	1000	1000	Hund-Rindke & Simon 2005
Soil process	Potential ammonium oxidation	NOEC	1000	1000	Hund-Rindke & Simon 2005
Soil process	Respiration	NOEC	1000	1000	Hund-Rindke & Simon 2005
Soil process	Respiration	NOEC	1000	1000	Hund-Rindke & Simon 2005

Test species		Measure of toxicity	Toxic conc. (mg/kg)	Est. NOEC or EC_{10} (mg/kg)	Source
Common name	Scientific name				
Soil process	Respiration	NOEC	1000	1000	Hund-Rindke & Simon 2005
Soil process	SIR	NOEC	1000	1000	Hund-Rindke & Simon 2005
Soil process	SIR	NOEC	1000	1000	Hund-Rindke & Simon 2005
Soil process	SIR	NOEC	1000	1000	Hund-Rindke & Simon 2005
Springtail	*Folsomia candida*	EC_{10}	99.9	99.9	Hund-Rindke & Simon 2005
Springtail	*F. candida*	NOEC	1000	1000	Hund-Rindke & Simon 2005
Springtail	*F. candida*	NOEC	1000	1000	Hund-Rindke & Simon 2005

LC_{50} = the concentration that is lethal to 50% of the organisms.

13.5 Appendix E: Raw toxicity data for copper

Table E1: The raw toxicity data for copper and the ageing/leaching factors that were used in the derivation of the soil quality guidelines derived in this project, and the source of the toxicity data.

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Andryala integrifolia	mortality	76	106	130	2	Brun et al. 2003
Andryala integrifolia	seedling emergence	78	106	128	2	Brun et al. 2003
Arachis hypogaea	grain yield	398		467	1	Barry & Bell 2006
Arachis hypogaea	grain yield	197		516	1	Barry & Bell 2006
Avena sativa	grain yield	200	300	600	2	De Haan et al. 1985
Avena sativa	grain yield	200	300	600	2	De Haan et al. 1985
Avena sativa	grain yield	200	300	600	2	De Haan et al. 1985
Avena sativa	grain yield	200	300	600	2	De Haan et al. 1985
Avena sativa	grain yield	200	300	600	2	De Haan et al. 1985
Brassica napus	grain yield	1310	1965	1370	1	Heemsbergen et al. 2007
Brassica napus	grain yield	926	1136	1566	1	NBRP unpublished data
Brassica napus	grain yield	315	473	452	1	Butler et al. 2007
Gossypium sp.	crop yield	1451	2177	1757	1	Barry & Bell 2006
Hordeum vulgare	grain yield	77	116	720	1	Heemsbergen et al. 2007
Hordeum vulgare	grain yield	313	470	1300	1	Heemsbergen et al. 2007
Hordeum vulgare	grain yield	222	333	645	1	Heemsbergen et al. 2007
Hordeum vulgare	grain yield	49	74	515	1	Butler et al. 2007
Hordeum vulgare	grain yield	28	41	227	1	Butler et al. 2007

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Hordeum vulgare	seedling emergence	112	305	335	2	Ali et al. 2004
Hordeum vulgare	shoot weight	305	>304.8	914	2	Ali et al. 2004
Hordeum vulgare	root weight	3	11	305	2	Ali et al. 2004
Hordeum vulgare	root yield	58	87	137	2	Rooney et al. 2006
Hordeum vulgare	root yield	16	24	36	2	Rooney et al. 2006
Hordeum vulgare	root yield	85	128	173	2	Rooney et al. 2006
Hordeum vulgare	root yield	80	120	233	2	Rooney et al. 2006
Hordeum vulgare	root yield	45	68	536	2	Rooney et al. 2006
Hordeum vulgare	root yield	14	21	40	2	Rooney et al. 2006
Hordeum vulgare	root yield	83	125	161	2	Rooney et al. 2006
Hordeum vulgare	root yield	20	30	56	2	Rooney et al. 2006
Hordeum vulgare	root yield	35	53	129	2	Rooney et al. 2006
Hordeum vulgare	root yield	144	216	376	2	Rooney et al. 2006
Hordeum vulgare	root yield	69	104	187	2	Rooney et al. 2006
Hordeum vulgare	root yield	53	80	359	2	Rooney et al. 2006
Hordeum vulgare	root yield	77	116	252	2	Rooney et al. 2006
Hordeum vulgare	root yield	120	180	405	2	Rooney et al. 2006
Hordeum vulgare	root yield	96	144	344	2	Rooney et al. 2006
Hordeum vulgare	root yield	111	167	326	2	Rooney et al. 2006
Hordeum vulgare	root yield	98	147	375	2	Rooney et al. 2006
Hordeum vulgare	root yield	26	39	114	2	Rooney et al. 2006
Hypochoeris radicata	mortality	99	165	227	2	Brun et al. 2003
Hypochoeris radicata	reproduction	157	173	187	2	Brun et al. 2003
Hypochoeris radicata	seedling emergence	175	187	195	2	Brun et al. 2003

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Lolium perenne	shoot yield	95	513	1036	2	Jarvis 1978
Lolium perenne	root yield	95	831	947	2	Jarvis 1978
Lycopersicon esculentum	shoot yield	46	69	130	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	159	239	427	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	370	555	829	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	48	72	115	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	29	44	61	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	89	134	237	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	179	269	281	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	598	897	851	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	252	378	351	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	311	467	933	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	481	722	795	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	212	318	771	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	212	318	659	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	251	377	444	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	116	174	429	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	70	105	325	2	Rooney et al. 2006
Lycopersicon esculentum	shoot yield	175	300	600	2	Rhoads et al. 1989
Lycopersicon esculentum	shoot yield	350	700	1400	2	Rhoads et al. 1989
Lycopersicon esculentum	shoot yield	350	700	1400	2	Rhoads et al. 1989
Panicum milaceum	yield	206	309	389	1	Barry & Bell 2006
Poa annua	mortality	200	389	418	2	Brun et al. 2003

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Poa annua	reproduction	200	216	262	2	Brun et al. 2003
Poa annua	seedling emergence	100	91	141	2	Brun et al. 2003
Polygonum convolvulus	yield (total dm)	188	237	276	2	Kjær & Elmegaard 1996
Polygonum convolvulus	yield (total dm)	188	301	309	2	Kjær & Elmegaard 1996
Polygonum convolvulus	reproductive dry matter	188	222	251	2	Kjær & Elmegaard 1996
Polygonum convolvulus	reproductive dry matter	188	247	287	2	Kjær & Elmegaard 1996
Polygonum convolvulus	seed biomass	188	303	327	2	Kjær & Elmegaard 1996
Polygonum convolvulus	mortality	113	211	257	2	Kjær & Elmegaard 1996
Polygonum convolvulus	mortality	113	188	387	2	Kjær & Elmegaard 1996
Polygonum convolvulus	shoot yield	200	300	259	2	Pedersen et al. 2000
Polygonum convolvulus	root yield	200	300	291	2	Pedersen et al. 2000
Sacharum sp.	yield	203	305	342	1	Barry & Bell 2006
Senecio vulgaris	mortality	78	150	228	2	Brun et al. 2003
Senecio vulgaris	reproduction	156	173	184	2	Brun et al. 2003
Senecio vulgaris	seedling emergence	28	57	88	2	Brun et al. 2003

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Sorghum sp.	yield	598	897	1433	1	Barry & Bell 2006
Sorghum sp.	yield	206	309	318	1	Barry & Bell 2006
Triticum aestivum	grain yield	1133	1139	1147	1	Warne et al. 2008a
Triticum aestivum	grain yield	132	176	286	1	Warne et al. 2008a
Triticum aestivum	grain yield	731	1561	5705	1	Warne et al. 2008a
Triticum aestivum	grain yield	148	228	476	1	Warne et al. 2008a
Triticum aestivum	grain yield	284	385	649	1	Warne et al. 2008a
Triticum aestivum	grain yield	130	157	212	1	Warne et al. 2008a
Triticum aestivum	grain yield	209	242	310	1	Warne et al. 2008a
Triticum aestivum	grain yield	787	1316	3170	1	Warne et al. 2008a
Triticum aestivum	grain yield	586	603	632	1	Warne et al. 2008a
Triticum aestivum	grain yield	622	752	1040	1	Warne et al. 2008a
Triticum aestivum	grain yield	473	768	1760	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	3	36	2070	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	351	360	375	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	635	792	1154	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	117	168	315	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	193	220	272	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	144	233	526	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	40	75	223	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	1100	1128	1183	1	Warne et al. 2008a
Triticum aestivum	8wk plant biomass	52	102	330	1	Warne et al. 2008a
Tritosecale sp.	yield	481	1020	2040	1	Butler et al. 2007
Zea mays	yield	274		363	1	Barry & Bell 2006

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Cognettia sphagnetorum	growth	20	50	91	2	Augustsson & Rundgren 1998
Cognettia sphagnetorum	growth	63	85	167	2	Augustsson & Rundgren 1998
Cognettia sphagnetorum	growth	441	502	605	2	Augustsson & Rundgren 1998
Cognettia sphagnetorum	growth	312	435	557	2	Augustsson & Rundgren 1998
Cognettia sphagnetorum	fragmentation	455	538	676	2	Augustsson & Rundgren 1998
Cognettia sphagnetorum	fragmentation	23	82		2	Augustsson & Rundgren 1998
Eisenia andrei	growth	56	84	168	2	van Dis et al. 1988
Eisenia andrei	growth	56	84	168	2	van Gestel et al. 1991
Eisenia andrei	reproduction	120	180	360	2	van Gestel et al. 1989
Eisenia andrei	reproduction	100	223	327	2	Kula & Larink 1997
Eisenia andrei	reproduction	100	168	240	2	Kula & Larink 1997
Eisenia andrei	reproduction	3	45	79	2	Kula & Larink 1997
Eisenia andrei	reproduction	154			2	Criel et al. 2008
Eisenia andrei	reproduction	88	188	264	2	Svendsen & Weeks 1997a
Eisenia andrei	mortality	188	335	564	2	Svendsen & Weeks 1997a
Eisenia fetida	mortality	208	311	555	2	Spurgeon et al. 1994
Eisenia fetida	mortality	293	440	836	2	Spurgeon & Hopkin 1995
Eisenia fetida	growth	725	1088	601	2	Spurgeon & Hopkin 1995
Eisenia fetida	growth	700	1000		2	Scott-Fordsmand et al. 2000
Eisenia fetida	reproduction	30	44	51	2	Spurgeon et al. 1994
Eisenia fetida	reproduction	29	44	87	2	Spurgeon & Hopkin 1995
Eisenia fetida	reproduction	10	132	174	2	Kula & Larink 1997

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Eisenia fetida	reproduction	32	72	108	2	Kula & Larink 1997
Eisenia fetida	reproduction	2	13	42	2	Kula & Larink 1997
Eisenia fetida	reproduction	0	3	10	2	Kula & Larink 1997
Eisenia fetida	reproduction	100	300	210	2	Scott-Fordsmand et al. 2000
Eisenia fetida	reproduction	161	243	190	2	Criel et al. 2008
Eisenia fetida	reproduction	84	172	211	2	Criel et al. 2008
Eisenia fetida	reproduction	120	92	708	2	Criel et al. 2008
Eisenia fetida	reproduction	86	100	171	2	Criel et al. 2008
Eisenia fetida	reproduction	88	289	296	2	Criel et al. 2008
Eisenia fetida	reproduction	67	165	198	2	Criel et al. 2008
Eisenia fetida	reproduction	31	94	67	2	Criel et al. 2008
Eisenia fetida	reproduction	213	464	329	2	Criel et al. 2008
Eisenia fetida	reproduction	195	237	230	2	Criel et al. 2008
Eisenia fetida	reproduction	279	538	487	2	Criel et al. 2008
Eisenia fetida	reproduction	151	501	267	2	Criel et al. 2008
Eisenia fetida	reproduction	346	501	407	2	Criel et al. 2008
Eisenia fetida	reproduction	148	281	309	2	Criel et al. 2008
Eisenia fetida	reproduction	454	258	731	2	Criel et al. 2008
Eisenia fetida	reproduction	188	160	358	2	Criel et al. 2008
Eisenia fetida	reproduction	69	153	149	2	Criel et al. 2008
Eisenia fetida	reproduction	223	361	347	2	Criel et al. 2008
Lumbricus rubellus	mortality	150	224	486	2	Svendsen & Weeks 1997b
Lumbricus rubellus	mortality	117	344	393	2	Ma 1984
Lumbricus rubellus	mortality	123	359	408	2	Ma 1984
Lumbricus rubellus	mortality	150		459	2	Ma 1982
Lumbricus rubellus	mortality	447	521	1384	2	Spurgeon et al. 2004
Lumbricus rubellus	litter breakdown	40	123	162	2	Ma 1984

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Lumbricus rubellus	litter breakdown	50	168	189	2	Ma 1984
Lumbricus rubellus	growth	117	358	393	2	Ma 1984
Lumbricus rubellus	growth	73	150	228	2	Svendsen & Weeks 1997b
Lumbricus rubellus	growth	140	642	462	2	Spurgeon et al. 2004
Lumbricus rubellus	reproduction	40	97	162	2	Ma 1984
Plectus acuminatus	reproduction	32	100	300	2	Kammenga et al. 1996
Folsomia candida	reproduction	190	299	260	2	Criel et al. 2008
Folsomia candida	reproduction	10	49	43	2	Criel et al. 2008
Folsomia candida	reproduction	417	530	952	2	Criel et al. 2008
Folsomia candida	reproduction	1380	2070	2200	2	Criel et al. 2008
Folsomia candida	reproduction	50	75	166	2	Criel et al. 2008
Folsomia candida	reproduction	51	85	112	2	Criel et al. 2008
Folsomia candida	reproduction	206	314	325	2	Criel et al. 2008
Folsomia candida	reproduction	186	489	325	2	Criel et al. 2008
Folsomia candida	reproduction	618	551	1238	2	Criel et al. 2008
Folsomia candida	reproduction	195	285	510	2	Criel et al. 2008
Folsomia candida	reproduction	659	803	862	2	Criel et al. 2008
Folsomia candida	reproduction	80	291	434	2	Criel et al. 2008
Folsomia candida	reproduction	1186	1666	1626	2	Criel et al. 2008
Folsomia candida	reproduction	550	707	845	2	Criel et al. 2008
Folsomia candida	reproduction	200	311	640	2	Criel et al. 2008
Folsomia candida	reproduction	683	1629	1199	2	Criel et al. 2008
Folsomia candida	reproduction	686	919	835	2	Criel et al. 2008
Folsomia candida	reproduction	227	1049	632	2	Criel et al. 2008
Folsomia candida	reproduction	16	37	73	2	Criel et al. 2008

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Folsomia candida	reproduction	797		813	2	Herbert et al. 2004
Folsomia candida	reproduction	198	411	650	2	Sandifer & Hopkin 1996
Folsomia candida	reproduction	231	486	774	2	Sandifer & Hopkin 1996
Folsomia candida	reproduction	920	1083	1200	2	Sandifer & Hopkin 1996
Folsomia candida	reproduction	200	300	700	2	Sandifer & Hopkin 1997
Folsomia candida	reproduction	200	300	640	2	Sandifer & Hopkin 1997
Folsomia candida	reproduction	400	600	1200	2	Rundgren & van Gestel 1988
Folsomia candida	reproduction	400	600	1200	2	Rundgren & van Gestel 1988
Folsomia candida	mortality	1281	1821	2271	2	Sandifer & Hopkin 1997
Folsomia candida	mortality	387	981	1761	2	Sandifer & Hopkin 1997
Folsomia candida	mortality	135	676	1859	2	Sandifer & Hopkin 1997
Folsomia candida	mortality	135	676		2	Sandifer & Hopkin 1996
Folsomia candida	mortality	561	1586		2	Sandifer & Hopkin 1996
Folsomia candida	mortality	2657	2978		2	Sandifer & Hopkin 1996
Folsomia candida	growth	800	1200	2400	2	Rundgren & van Gestel 1988
Folsomia candida	growth	200	300	600	2	Rundgren & van Gestel 1988
Folsomia fimetaria	mortality	878	1000	2000	2	Scott-Fordsmand et al. 1997
Folsomia fimetaria	mortality	1000	>1000	3000	2	Scott-Fordsmand et al. 1997
Folsomia fimetaria	mortality	1000	>1000	3000	2	Scott-Fordsmand et al. 1997
Folsomia fimetaria	growth	542	400	800	2	Scott-Fordsmand et al. 1997
Folsomia fimetaria	growth	845	800	1600	2	Scott-Fordsmand et al. 1997
Folsomia fimetaria	growth	527	600	1200	2	Scott-Fordsmand et al. 1997
Folsomia fimetaria	reproduction	38	57	113	2	Scott-Fordsmand et al. 1997
Folsomia fimetaria	reproduction	122	183	638	2	Pedersen et al. 2000

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Folsomia fimetaria	reproduction	698	1047	1225	2	Pedersen et al. 2001a
Folsomia fimetaria	reproduction	776	1164	1635	2	Pedersen et al. 2001a
Folsomia fimetaria	reproduction	888	1332	1674	2	Pedersen et al. 2001a
Folsomia fimetaria	reproduction	648	972	1259	2	Pedersen et al. 2001a
Folsomia fimetaria	reproduction	688	1032	1395	2	Pedersen et al. 2001a
Hypoaspis aculeifer	reproduction	174	261	522	2	Krogh & Axelsen 1998
Isotoma viridis	growth	50	75	150	2	Rundgren & van Gestel 1988
Isotoma viridis	growth	400	600	1200	2	Rundgren & van Gestel 1988
Platynothrus peltifer	reproduction	63	95	189	2	van Gestel & Doornekamp 1998
Platynothrus peltifer	reproduction	63	95	189	2	van Gestel & Doornekamp 1998
Platynothrus peltifer	reproduction	63	95	189	2	van Gestel & Doornekamp 1998
Soil microbial process	microbial biomass C	118	268	354	2	Khan & Scullion 2002
Soil microbial process	microbial biomass C	118	268	354	2	Khan & Scullion 2002
Soil microbial process	microbial biomass N	468	768	1404	2	Khan & Scullion 2002
Soil microbial process	microbial biomass N	<118	118	236	2	Khan & Scullion 2002
Soil microbial process	SIR[1]	635	953	1905	2	Speir et al. 1999
Soil microbial process	SIR	635	953	1905	2	Speir et al. 1999
Soil microbial process	SIR	1200	1800	3600	2	University of Leuven 2004
Soil microbial process	SIR	150	225	450	2	University of Leuven 2004
Soil microbial process	SIR	50	75	150	2	University of Leuven 2004
Soil microbial process	SIR	600	900	1800	2	University of Leuven 2004
Soil microbial process	SIR	100	150	300	2	University of Leuven 2004
Soil microbial process	SIR	25	38	75	2	University of Leuven 2004

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Soil microbial process	SIR	100	150	300	2	University of Leuven 2004
Soil microbial process	SIR	50	75	150	2	University of Leuven 2004
Soil microbial process	SIR	25	38	75	2	University of Leuven 2004
Soil microbial process	SIR	400	600	1200	2	University of Leuven 2004
Soil microbial process	SIR	300	450	900	2	University of Leuven 2004
Soil microbial process	SIR	50	75	150	2	University of Leuven 2004
Soil microbial process	SIR	102	153	306	2	University of Leuven 2004
Soil microbial process	SIR	200	300	600	2	University of Leuven 2004
Soil microbial process	SIR	89	134	267	2	University of Leuven 2004
Soil microbial process	SIR	23	35	69	2	University of Leuven 2004
Soil microbial process	SIR	300	450	900	2	University of Leuven 2004
Soil microbial process	SIR	200	300	600	2	University of Leuven 2004
Soil microbial process	SIR	50	75	150	2	University of Leuven 2004
Soil microbial process	SIR	170	255	510	2	University of Leuven 2004
Soil microbial process	SIR	12	18	36	2	University of Leuven 2004
Soil microbial process	SIR	25	38	75	2	University of Leuven 2004
Soil microbial process	SIR	100	150	300	2	University of Leuven 2004
Soil microbial process	SIR	27	41	81	2	University of Leuven 2004
Soil microbial process	SIR	185	345	1000	1	Broos et al. 2007
Soil microbial process	SIR	3	31	1078	1	Broos et al. 2007
Soil microbial process	SIR	326	450	555	1	Broos et al. 2007
Soil microbial process	SIR	230	496	1842	1	Broos et al. 2007
Soil microbial process	SIR	255	503	1606	1	Broos et al. 2007
Soil microbial process	SIR	48	134	784	1	Broos et al. 2007
Soil microbial process	SIR	39	111	662	1	Broos et al. 2007
Soil microbial process	SIR	222	559	2321	1	Broos et al. 2007
Soil microbial process	SIR	202	421	1478	1	Broos et al. 2007
Soil microbial process	SIR	26	73	431	1	Broos et al. 2007
Soil microbial process	SIR	134	259	795	1	Broos et al. 2007

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Soil microbial process	SIR	25	97	940	1	Broos et al. 2007
Soil microbial process	GAD[2]	55	400	800	1	Haanstra & Doelman 1984
Soil microbial process	GAD	55	400	800	1	Haanstra & Doelman 1984
Soil microbial process	GAD	400	1000	2000	1	Haanstra & Doelman 1984
Soil microbial process	MRR[3]	2400	3600	7200	2	University of Leuven 2004
Soil microbial process	MRR	1200	1800	3600	2	University of Leuven 2004
Soil microbial process	MRR	1200	1800	3600	2	University of Leuven 2004
Soil microbial process	MRR	300	450	900	2	University of Leuven 2004
Soil microbial process	MRR	50	75	150	2	University of Leuven 2004
Soil microbial process	MRR	200	300	600	2	University of Leuven 2004
Soil microbial process	MRR	100	150	300	2	University of Leuven 2004
Soil microbial process	MRR	50	75	150	2	University of Leuven 2004
Soil microbial process	MRR	400	600	1200	2	University of Leuven 2004
Soil microbial process	MRR	150	225	450	2	University of Leuven 2004
Soil microbial process	MRR	50	75	150	2	University of Leuven 2004
Soil microbial process	MRR	400	600	1200	2	University of Leuven 2004
Soil microbial process	MRR	600	900	1800	2	University of Leuven 2004
Soil microbial process	MRR	150	225	450	2	University of Leuven 2004
Soil microbial process	MRR	150	225	450	2	University of Leuven 2004
Soil microbial process	MRR	51	77	153	2	University of Leuven 2004
Soil microbial process	MRR	83	125	249	2	University of Leuven 2004
Soil microbial process	MRR	100	150	300	2	University of Leuven 2004
Soil microbial process	MRR		144	288	2	Oorts et al. 2006a
Soil microbial process	MRR		348	696	2	Oorts et al. 2006a
Soil microbial process	MRR		802	1604	2	Oorts et al. 2006a
Soil microbial process	respiration	89	1402	7932	1	Doelman & Haanstra 1984

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Soil microbial process	respiration	400	600	1200	1	Doelman & Haanstra 1984
Soil microbial process	respiration	493	4097	15477	1	Doelman & Haanstra 1984
Soil microbial process	respiration	32	219	730	1	Doelman & Haanstra 1984
Soil microbial process	PNR[4]	200	300	400	2	University of Leuven 2004
Soil microbial process	PNR	1200	1800	2400	2	University of Leuven 2004
Soil microbial process	PNR	25	38	50	2	University of Leuven 2004
Soil microbial process	PNR	25	38	50	2	University of Leuven 2004
Soil microbial process	PNR	50	75	100	2	University of Leuven 2004
Soil microbial process	PNR	100	150	200	2	University of Leuven 2004
Soil microbial process	PNR	300	450	600	2	University of Leuven 2004
Soil microbial process	PNR	200	300	400	2	University of Leuven 2004
Soil microbial process	PNR	800	1200	1600	2	University of Leuven 2004
Soil microbial process	PNR	400	600	800	2	University of Leuven 2004
Soil microbial process	PNR	600	900	1200	2	University of Leuven 2004
Soil microbial process	PNR	800	1200	1600	2	University of Leuven 2004
Soil microbial process	PNR	300	450	600	2	University of Leuven 2004
Soil microbial process	PNR	400	600	800	2	University of Leuven 2004
Soil microbial process	PNR	52	78	104	2	University of Leuven 2004
Soil microbial process	PNR	127	191	254	2	University of Leuven 2004
Soil microbial process	PNR	65	98	130	2	University of Leuven 2004
Soil microbial process	PNR	100	150	200	2	University of Leuven 2004
Soil microbial process	PNR	50	75	100	2	University of Leuven 2004
Soil microbial process	PNR			771	2	Oorts et al. 2006a
Soil microbial process	PNR			677	2	Oorts et al. 2006a

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Soil microbial process	SIN[6]	100	150	200	2	Quraishi & Cornfield 1973
Soil microbial process	SIN	100	150	200	2	Quraishi & Cornfield 1973
Soil microbial process	SIN	1000	1500	2000	2	Premi & Cornfield 1969
Soil microbial process	SIN	2594	2594	2594	1	Broos et al. 2007
Soil microbial process	SIN	34	254	1078	1	Broos et al. 2007
Soil microbial process	SIN	206	208	211	1	Broos et al. 2007
Soil microbial process	SIN	1271	1451	1821	1	Broos et al. 2007
Soil microbial process	SIN	175	228	355	1	Broos et al. 2007
Soil microbial process	SIN	1	5	59	1	Broos et al. 2007
Soil microbial process	SIN	47	70	140	1	Broos et al. 2007
Soil microbial process	SIN	383	502	797	1	Broos et al. 2007
Soil microbial process	SIN	887	914	964	1	Broos et al. 2007
Soil microbial process	SIN	919	932	953	1	Broos et al. 2007
Soil microbial process	SIN	502	571	712	1	Broos et al. 2007
Soil microbial process	SIN	141	225	497	1	Broos et al. 2007

Species	End point	NOEC or EC_{10} added (mg/kg)	LOEC and EC_{30} (mg/kg)	EC_{50} added (mg/kg)	ALF	Reference
Soil microbial process	N-mineralisation	100	150	300	2	Quraishi & Cornfield 1973
Soil microbial process	N-mineralisation	268	465	804	2	Khan & Scullion 2002
Soil microbial process	N-mineralisation		115	230	2	Khan & Scullion 2002
Soil microbial process	ammonification	1000	1500	3000	2	Premi & Cornfield 1969
Soil microbial process	denitrification	100	250	300	2	Bollag & Barabasz 1979

[1] SIR = substrate induced nitrification, [2] GAD = glutamic acid decomposition, [3] MRR = maize residue respiration, [4] PNR = potential nitrification rate, [5] SIN = substrate induced respiration.

13.6 Appendix F: Explanation of the selection of the soil properties that control the added contaminant limits for copper

A total of ten normalisation relationships were used to normalise the Cu toxicity data. The same ten normalisation relationships were used to generate the soil-specific ACLs. The generated soil-specific ACLs are the concentrations for each species/soil process that correspond to the desired level of protection (for example, 80% for urban residential land/public open space land use). Therefore, in order to provide the desired level of protection, the lowest ACL at each soil property value must be adopted as the final ACL.

For Cu there were six normalisation relationships based on CEC. These were for *H. vulgare, L. escultentum, E. fetida, F. candida, F. fimetaria* and PNR. Of these, PNR always generated the lowest ACL when the CEC was less than 10 $cmol_c/kg$. At all higher CEC values the *H. vulgare* normalisation relationship always resulted in the lowest ACL. Therefore, one set of soil-specific ACLs was generated by for *H. vulgare* and another for PNR with the lowest of the two at each CEC being adopted as the CEC-based ACL values for Cu.

In addition, there was one normalisation relationship based on a combination of soil pH and organic carbon content (OC)—for *T. aestivum*. There were also two normalisation relationships for SIN and MRM that were based on soil pH and one for SIR based on OC. The MRM normalisation relationship was not used as it had a negative relationship with toxicity, which was inconsistent with all the other normalisation relationships for Cu and all other elements. The SIN normalisation relationship always generated ACL values lower than those generated by the *T. aestivum* relationship at soil pH values up to 5.5. At higher soil pH values the situation was reversed. In addition, the ACLs generated by the SIR relationship (based on OC) were lower than all the ACLs generated by the *T. aestivum* relationship except when the OC was set at 1 in the *T. aestivum* relationship. Therefore one set of soil-specific ACLs was generated for *T. aestivum* and another for SIN with the lowest of the two at each pH being adopted as the CEC-pH-based ACL values for Cu.

The pH and CEC-based ACLs for Cu were presented in tables in this Schedule. The actual ACL values that apply for Cu are the lowest of either the pH-based ACLs or the CEC-based ACLs, depending on the properties of the soil in question.

13.7 Appendix G. Raw toxicity data for lead

Table G1: The raw toxicity data for lead and the ageing/leaching factors that were used in the derivation of the soil quality guidelines derived in this project, and the source of the toxicity data.

Species	End point	NOEC or EC_{10} (added)	LOEC and EC_{30} (added)	EC_{50} (added)	ALF	References
Avena sativa	root yield	100	500	300	4.2	Khan & Frankland 1984
Hordeum vulgare	shoot yield	50	250	1270	4.2	Aery & Jagetiya 1997
Lactuca sativa	shoot yield	432	648	2553	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield	1172	1758	107	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield	457	686	960	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield	5120	7680	7500	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			132	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			141	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			240	4.2	Stevens et al, 2003
Lactuca sativa	shoot yield			847	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			807	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			731	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			2290	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			2630	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			3090	4.2	Stevens et al. 2003
Lactuca sativa	shoot yield			3100	4.2	Stevens et al. 2003
Lactuca sativa	germination	125	188	174	4.2	Vaughan & Greenslade 1998
Picea rubens	net photosynthesis	141	212	1228	4.2	Seiler & Paganelli 1987
Pinus taeda	root yield	546	819	659	4.2	Seiler & Paganelli 1987
Raphanus sativus	root yield	100	500	1800	4.2	Khan & Frankland 1983

Species	End point	NOEC or EC_{10} (added)	LOEC and EC_{30} (added)	EC_{50} (added)	ALF	References
Raphanus sativus	chlorophyll	100	500	300	4.2	Zaman & Zereen 1998
Triticum aestivum	net photosynthesis	1138	1707	5613	4.2	Waegeneers et al. 2004
Triticum aestivum	net photosynthesis	2064	3096	5037	4.2	Waegeneers et al. 2004
Triticum aestivum	net photosynthesis	1614	2421	5200	4.2	Waegeneers et al. 2004
Triticum aestivum	root yield	250	500	750	4.2	Khan & Frankland 1984
Zea mays	root length	100	150	300	4.2	LDA 2008
Dendrobaena rubida	hatching success	129	194	387	4.2	Bengtsson et al. 1986
Eisenia andrei	survival	1000	1500	3410	4.2	Vaughan & Greenslade 1998
Eisenia fetida	reproduction	608	912	1629	4.2	Spurgeon & Hopkin 1995
Eisenia fetida	reproduction	1810	2715	3760	4.2	Spurgeon et al. 1994
Eisenia fetida	reproduction	400	600	1200	4.2	Davies et al. 2003a
Eisenia fetida	reproduction	3000	4500	9000	4.2	Davies et al. 2003b
Folsomia candida	reproduction	2000	5000	1360	4.2	Sandifer & Hopkin 1996
Folsomia candida	reproduction	400	2000	2970	4.2	Sandifer & Hopkin 1996
Folsomia candida	reproduction	2000	3000	3160	4.2	Sandifer & Hopkin 1996
Folsomia candida	reproduction	400	2000	1570	4.2	Sandifer & Hopkin 1997
Folsomia candida	reproduction			2970	4.2	Sandifer & Hopkin 1997
Folsomia candida	reproduction	1300	1950	1900	4.2	Bongers et al. 2004
Folsomia candida	reproduction	1138	1707	3414	4.2	Waegeneers et al. 2004
Folsomia candida	reproduction	2064	3096	6192	4.2	Waegeneers et al. 2004
Folsomia candida	reproduction	1614	2421	4842	4.2	Waegeneers et al. 2004
Folsomia candida	reproduction			2560	4.2	Waegeneers et al. 2004

Species	End point	NOEC or EC_{10} (added)	LOEC and EC_{30} (added)	EC_{50} (added)	ALF	References
Lumbriculus rubellus	growth	1000	1500	3000	4.2	Ma, 1982
Denitrification		250	500	750	4.2	Bollag & Barabasz 1979
Nitrification		448	672	1344	4.2	Waegeneers et al. 2004
Nitrification		2064	3096	6192	4.2	Waegeneers et al. 2004
Nitrification		253	380	759	4.2	Waegeneers et al. 2004
N-mineralisation		200	300	600	4.2	Chang & Broadbent 1982
N-mineralisation		1000	4000	3000	4.2	Wilke 1989
Respiration		188	282	564	4.2	Doelman & Haanstra 1979
Respiration		1500	2250	4500	4.2	Doelman & Haanstra 1979
Respiration		750	1125	2250	4.2	Doelman & Haanstra 1979
Respiration		1000	1500	3000	4.2	Doelman & Haanstra 1984
Respiration		150	225	450	4.2	Doelman & Haanstra 1984
Respiration		400	600	1200	4.2	Doelman & Haanstra 1984
Respiration		93	140	400	4.2	Chang & Broadbent 1981
Respiration		100	150	300	4.2	Saviozzi et al. 1997
Respiration		4144	6216	12432	4.2	Speir et al. 1999
Respiration		2279	3419	6838	4.2	Frostegård et al. 1993
Substrate-induced respiration		2072	3108	6216	4.2	Speir et al. 1999
Substrate-induced respiration		1450	2175	4350	4.2	Speir et al. 1999
ATP				3108	4.2	Frostegård et al. 1993

13.8 Appendix H: Raw toxicity data for nickel

Table H1: The raw toxicity data for nickel and the ageing/leaching factors that were used in the derivation of the soil quality guidelines derived in this project, and the source of the toxicity data.

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Lycopersicon esculentum	shoot yield	21	31.5	63	1.01	Rothamsted 2005
Lycopersicon esculentum	shoot yield	599	898.5	1797	1.02	Rothamsted 2005
Lycopersicon esculentum	shoot yield	16	24	48	1.02	Rothamsted 2005
Lycopersicon esculentum	shoot yield	125	187.5	375	1.02	Rothamsted 2005
Lycopersicon esculentum	shoot yield	10	15	30	1.03	Rothamsted 2005
Lycopersicon esculentum	shoot yield	42	63	126	1.07	Rothamsted 2005
Lycopersicon esculentum	shoot yield	52	78	156	1.14	Rothamsted 2005
Lycopersicon esculentum	shoot yield	150	225	450	1.28	Rothamsted 2005
Lycopersicon esculentum	shoot yield	118	177	354	1.66	Rothamsted 2005
Lycopersicon esculentum	shoot yield	250	375	750	2.00	Rothamsted 2005
Lycopersicon esculentum	shoot yield	200	300	600	3.32	Rothamsted 2005
Lycopersicon esculentum	shoot yield	504	756	1512	3.01	Rothamsted 2005
Lycopersicon esculentum	shoot yield	224	336	672	3.32	Rothamsted 2005
Lycopersicon esculentum	shoot yield	144	216	432	3.32	Rothamsted 2005
Lycopersicon esculentum	shoot yield	189	283.5	567	3.66	Rothamsted 2005
Hordeum vulgare	root yield	31	46.5	93	1.01	Rothamsted 2005
Hordeum vulgare	root yield	1101	1651.5	3303	1.02	Rothamsted 2005
Hordeum vulgare	root yield	90	135	270	1.02	Rothamsted 2005
Hordeum vulgare	root yield	249	373.5	747	1.02	Rothamsted2005
Hordeum vulgare	root yield	46	69	138	1.03	Rothamsted 2005
Hordeum vulgare	root yield	123	184.5	369	1.07	Rothamsted 2005
Hordeum vulgare	root yield	261	391.5	783	1.14	Rothamsted 2005

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Hordeum vulgare	root yield	128	192	384	1.14	Rothamsted 2005
Hordeum vulgare	root yield	398	597	1194	1.28	Rothamsted 2005
Hordeum vulgare	root yield	106	159	318	1.66	Rothamsted 2005
Hordeum vulgare	root yield	211	316.5	633	2.00	Rothamsted 2005
Hordeum vulgare	root yield	268	402	804	3.32	Rothamsted 2005
Hordeum vulgare	root yield	289	433.5	867	3.01	Rothamsted 2005
Hordeum vulgare	root yield	587	880.5	1761	3.32	Rothamsted 2005
Hordeum vulgare	root yield	96	144	288	3.32	Rothamsted 2005
Hordeum vulgare	root yield	304	456	912	3.66	Rothamsted 2005
Spinach	yield	10	21.7	32.7	1.03	Willaert & Verloo 1988
Spinach	yield	100	40	40	5.66	Willaert & Verloo 1988
Spinach	yield		200	200	5.66	Willaert & Verloo 1988
Avena sativa	grain yield	500	750	1500	2.32	Halstead et al. 1969
Avena sativa	grain yield	20	51	56.2	1.12	Halstead et al. 1969
Avena sativa	grain yield	50	75.7	100	1.12	Halstead et al. 1969
Avena sativa	grain yield	50	55.4	63.1	1.38	Halstead et al. 1969
Avena sativa	grain yield	50	82.2	100	1.33	Halstead et al. 1969
Avena sativa	grain yield	100	144	159	1.08	Halstead et al. 1969
Avena sativa	grain yield	100	144	159	1.07	Halstead et al. 1969
Avena sativa	grain yield	100	144	159	1.43	Halstead et al. 1969
Avena sativa	grain yield	100	144	159	1.28	Halstead et al. 1969
Avena sativa	grain yield	66	99	198	1.14	De Haan et al. 1985
Avena sativa	grain yield	45	67.5	135	1.11	De Haan et al. 1985
Avena sativa	grain yield	47	70.5	141	1.08	De Haan et al. 1985
Avena sativa	grain yield	16	24	48	1.06	De Haan et al. 1985
Avena sativa	grain yield	40	60	120	1.11	De Haan et al. 1985

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Avena sativa	yield	80	171	241	3.01	Liang & Schoenau 1995
Avena sativa	yield	>160	160	160	3.01	Liang & Schoenau 1995
Medicago sativa	$EC_{10}y(t)$	100	366	404	3.32	Halstead et al. 1969
Medicago sativa	$EC_{10}y(t)$	100	389	423	2.32	Halstead et al. 1969
Medicago sativa	$EC_{10}y(t)$	20	19.1	20.9	1.12	Halstead et al. 1969
Medicago sativa	$EC_{10}y(t)$	20	47.6	49.9	1.38	Halstead et al. 1969
Medicago sativa	$EC_{10}y(t)$	20	40.5	42.3	1.33	Halstead et al. 1969
Medicago sativa	$EC_{10}y(t)$	20	43.5	45.5	1.08	Halstead et al. 1969
Medicago sativa	$EC_{10}y(t)$	50	101	106	1.07	Halstead et al. 1969
Medicago sativa	$EC_{10}y(t)$	20	45.6	48.2	1.43	Halstead et al. 1969
Medicago sativa	$EC_{10}y(t)$	50	100	118	1.28	Halstead et al. 1969
Raphanus sativus	yield	80	100.8	115	3.01	Liang & Schoenau 1995
Raphanus sativus	yield	>160	160	160		Liang & Schoenau 1995
Allium cepa	yield	46	73.1	103.4	7.17	Dang et al. 1990
Trigonella poenumgraceum	yield	84	132.8	176.6	7.17	Dang et al. 1990
Lolium perenne	yield	110	134.8	153.3	1.25	Frossard et al. 1989
Lactuca sativa	leaf yield	13	41	50.1	1.05	Gupta et al. 1987
Lactuca sativa	leaf yield	155	260	316	1.14	Gupta et al. 1987
Lactuca sativa	leaf yield	230	412	501	3.66	Gupta et al. 1987
Lactuca sativa	leaf yield	334	653	794	1.57	Gupta et al. 1987

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Lactuca sativa	yield	40	77.5	99.5	3.01	Liang & Schoenau 1995
Zea mays	yield	120	164	200	4.53	Metwally & Rabie 1989
Zea mays	yield	40	107	158	6.37	Metwally & Rabie 1989
Folsomia candida	reproduction	36.4	54.6	109.2	1.01	University of Ghent/Euras 2005
Folsomia candida	reproduction	558	837	1674	1.02	University of Ghent/Euras 2005
Folsomia candida	reproduction	120	180	360	1.02	University of Ghent/Euras 2005
Folsomia candida	reproduction	527	790.5	1581	1.02	University of Ghent/Euras 2005
Folsomia candida	reproduction	104	156	312	1.03	University of Ghent/Euras 2005
Folsomia candida	reproduction	101	151.5	303	1.14	University of Ghent/Euras 2005
Folsomia candida	reproduction	180	270	540	1.14	University of Ghent/Euras 2005
Folsomia candida	reproduction	622	933	1866	1.28	University of Ghent/Euras 2005
Folsomia candida	reproduction	269	403.5	807	1.66	University of Ghent/Euras 2005
Folsomia candida	reproduction	384	576	1152	2.00	University of Ghent/Euras 2005
Folsomia candida	reproduction	662	993	1986	3.32	University of Ghent/Euras 2005
Folsomia candida	reproduction	828	1242	2484	3.01	University of Ghent/Euras 2005
Folsomia candida	reproduction	1100	1650	3300	3.32	University of Ghent/Euras 2005
Folsomia candida	reproduction	61.7	92.55	185.1	3.32	University of Ghent/Euras 2005
Folsomia candida	reproduction	562	843	1686	3.66	University of Ghent/Euras 2005
Folsomia candida	reproduction	320	560	476	1.25	Lock & Janssen 2002
Folsomia candida	mortality		1000	1000	1.25	Lock & Janssen 2002
Folsomia fimetaria	reproduction	173	259.5	519	1.12	Scott-Fordsmand et al. 1998
Eisenia fetida	reproduction	49.8	74.7	149.4	1.01	University of Ghent/Euras 2005
Eisenia fetida	reproduction	1110	1665	3330	1.02	University of Ghent/Euras 2005

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Eisenia fetida	reproduction	54.5	81.75	163.5	1.02	University of Ghent/Euras 2005
Eisenia fetida	reproduction	362	543	1086	1.02	University of Ghent/Euras 2005
Eisenia fetida	reproduction	46.5	69.75	139.5	1.03	University of Ghent/Euras 2005
Eisenia fetida	reproduction	182	273	546	1.07	University of Ghent/Euras 2005
Eisenia fetida	reproduction	230	345	690	1.14	University of Ghent/Euras 2005
Eisenia fetida	reproduction	66.1	99.15	198.3	1.14	University of Ghent/Euras 2005
Eisenia fetida	reproduction	151	226.5	453	1.28	University of Ghent/Euras 2005
Eisenia fetida	reproduction	172	258	516	1.66	University of Ghent/Euras 2005
Eisenia fetida	reproduction	297	445.5	891	2.00	University of Ghent/Euras 2005
Eisenia fetida	reproduction	233	349.5	699	3.32	University of Ghent/Euras 2005
Eisenia fetida	reproduction	239	358.5	717	3.01	University of Ghent/Euras 2005
Eisenia fetida	reproduction	490	735	1470	3.32	University of Ghent/Euras 2005
Eisenia fetida	reproduction	186	279	558	3.32	University of Ghent/Euras 2005
Eisenia fetida	reproduction	198	297	594	3.66	University of Ghent/Euras 2005
Eisenia fetida	reproduction	180	320	362	1.25	Lock & Janssen 2002
Eisenia fetida	mortality		1000	1000	1.25	Lock & Janssen 2002
Enchytraeus albidus	reproduction	180	320	275	1.25	Lock & Janssen 2002
Enchytraeus albidus	mortality		127.5	510	1.25	Lock & Janssen 2002
Eisenia veneta	reproduction	85	300	300	1.12	Scott-Fordsmand et al. 1998
Lumbricus rubellus	mortality	842	1080	1190	2.52	Ma 1982
Microbial process	nitrification	170	255	510	1.02	University of Leuven 2005
Microbial process	nitrification	111	166.5	333	1.02	University of Leuven 2005

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Microbial process	nitrification	44	66	132	1.14	University of Leuven 2005
Microbial process	nitrification	137	205.5	411	1.14	University of Leuven 2005
Microbial process	nitrification	67	100.5	201	1.66	University of Leuven 2005
Microbial process	nitrification	214	321	642	2.00	University of Leuven 2005
Microbial process	nitrification	439	658.5	1317	3.01	University of Leuven 2005
Microbial process	nitrification	169	253.5	507	3.32	University of Leuven 2005
Microbial process	nitrification	53	79.5	159	3.32	University of Leuven 2005
Microbial process	nitrification	67	100.5	201	3.66	University of Leuven 2005
Microbial process	N-mineralisation	257	385.5	771	2.00	Smolders 2000
Microbial process	N-mineralisation	20	30	60	2.00	Smolders 2000
Microbial process	Glucose respiration	22	33	66	1.02	University of Leuven 2005
Microbial process	Glucose respiration	254	381	762	1.14	University of Leuven 2005
Microbial process	Glucose respiration	376	564	1128	1.28	University of Leuven 2005
Microbial process	Glucose respiration	45	67.5	135	1.66	University of Leuven 2005
Microbial process	Glucose respiration	242	363	726	2.00	University of Leuven 2005
Microbial process	Glucose respiration	116	174	348	3.32	University of Leuven 2005
Microbial process	Glucose respiration	302	453	906	3.01	University of Leuven 2005
Microbial process	Glucose respiration	167	250.5	501	3.32	University of Leuven 2005
Microbial process	Glucose respiration	140	210	420	3.32	University of Leuven 2005
Microbial process	Glucose respiration	56	84	168	3.66	University of Leuven 2005
Microbial process	MRR	42	63	126	1.01	University of Leuven 2005
Microbial process	MRR	343	514.5	1029	1.02	University of Leuven 2005
Microbial process	MRR	55	82.5	165	1.14	University of Leuven 2005
Microbial process	MRR	121	181.5	363	1.28	University of Leuven 2005
Microbial process	MRR	88	132	264	2.00	University of Leuven 2005

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Microbial process	MRR	203	304.5	609	3.01	University of Leuven 2005
Microbial process	MRR	446	669	1338	3.32	University of Leuven 2005
Microbial process	MRR	370	555	1110	3.66	University of Leuven 2005
Aspergillus flavipes	hyphal growth	347	386.9	414.2	1.05	Babich & Stotzky 1982
Aspergillus flavus	hyphal growth	393	510.2	600.8	1.05	Babich & Stotzky 1982
Aspergillus clavatus	hyphal growth	13	40	79.3	1.05	Babich & Stotzky 1982
Aspergillus niger	hyphal growth	400	474.5	527.8	1.05	Babich & Stotzky 1982
Penicillium vermiculatum	hyphal growth	102	235.9	400.4	1.05	Babich & Stotzky 1982
Rhizopus stolonifer	hyphal growth	288	352.2	399.8	1.05	Babich & Stotzky 1982
Trichoderma viride	hyphal growth	530	597.9	644.8	1.05	Babich & Stotzky 1982
Gliocladium sp.	hyphal growth	200	505	902.4	1.05	Babich & Stotzky 1982
Serratia marcescens	colony count	155	293.3	344.1	1.05	Babich & Stotzky 1982
Proteus vulgaris	colony count	15	77.4	216.6	1.05	Babich & Stotzky 1982
Bacillus cereus	colony count	285	880.4	1706	1.05	Babich & Stotzky 1982
Nocardia rhodochrous	colony count	177	577.2	821.6	1.05	Babich & Stotzky 1982

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Rhodotorula rubra	colony count	247	729.3	1565	1.05	Babich & Stotzky 1982
Microbial process	Respiration	400	8000	8000	2.00	Doelman & Haanstra 1984
Microbial process	Respiration		8000	8000	2.00	Doelman & Haanstra 1984
Microbial process	Respiration	2542	8000	8000	1.25	Doelman & Haanstra 1984
Microbial process	Respiration		1370	7292	1.25	Doelman & Haanstra 1984
Microbial process	Respiration	291	8000	8000	3.66	Doelman & Haanstra, 1984
Microbial process	Respiration		8000	8000	3.66	Doelman & Haanstra 1984
Microbial process	Respiration		8000	8000	3.01	Doelman & Haanstra 1984
Microbial process	Respiration		8000	8000	3.01	Doelman & Haanstra 1984
Microbial process	Respiration		3585	12 072	1.03	Doelman & Haanstra 1984
Microbial process	Respiration	27	93.9	1655	1.08	Saviozzi et al. 1997
Microbial process	Glutamate respiration	55	400	800	2.00	Haanstra & Doelman 1984
Microbial process	Glutamate respiration	55	400	800	1.03	Haanstra & Doelman 1984
Microbial process	Glutamate respiration	55	400	800	3.01	Haanstra & Doelman 1984
Microbial process	Glutamate respiration		55	110	3.66	Haanstra & Doelman 1984
Enzyme	ATP content	77	115.5	400	1.25	Wilke 1988
Enzyme activity	urease	120	180	410	2.00	Doelman & Haanstra 1986
Enzyme activity	urease				2.00	Doelman & Haanstra 1986
Enzyme activity	urease	2300	3450	2790	1.25	Doelman & Haanstra 1986
Enzyme activity	urease				1.25	Doelman & Haanstra 1986
Enzyme activity	urease	130	195	1740	3.66	Doelman & Haanstra 1986
Enzyme activity	urease				3.66	Doelman & Haanstra 1986
Enzyme activity	urease	90	135	370	3.01	Doelman & Haanstra 1986
Enzyme activity	urease				3.01	Doelman & Haanstra 1986

Species	Endpoint	NOEC & EC10 added (mg/kg)	Collated LOEC & EC30 added (mg/kg)	Collated EC50 added (mg/kg)	ALF	References
Enzyme activity	urease	540	810	2320	1.03	Doelman & Haanstra 1986
Enzyme activity	urease				1.03	Doelman & Haanstra 1986
Enzyme activity	phosphatase	7021	10531.5	10071	2.00	Doelman & Haanstra 1989
Enzyme activity	phosphatase	251	376.5	8040	1.25	Doelman & Haanstra 1989
Enzyme activity	phosphatase	380	570	2130	3.66	Doelman & Haanstra 1989
Enzyme activity	phosphatase			6514	3.01	Doelman & Haanstra 1989
Enzyme activity	arylsulfatase	372	558	2119	2.00	Haanstra & Doelman 1991
Enzyme activity	arylsulfatase			98.6	2.00	Haanstra & Doelman 1991
Enzyme activity	arylsulfatase	610	915	2347	1.25	Haanstra & Doelman 1991
Enzyme activity	arylsulfatase	2207	3310.5	5399	3.66	Haanstra & Doelman 1991
Enzyme activity	arylsulfatase			92.1	3.66	Haanstra & Doelman 1991
Enzyme activity	arylsulfatase	272	408	5658	3.01	Haanstra & Doelman 1991
Enzyme activity	arylsulfatase			2436	3.01	Haanstra & Doelman 1991
Enzyme activity	arylsulfatase	7080	10620	8099	1.03	Haanstra & Doelman 1991
Enzyme activity	dehydrogenase	7.9	24.3	100	2.03	Welp 1999
Enzyme activity	saccharase	77	115.5	400	1.25	Wilke 1988
Enzyme activity	protease	77	115.5	400	1.25	Wilke 1988

MRR = maize residue respiration.

13.9 Appendix I: Raw toxicity data for trivalent chromium

Table I1:The raw toxicity data for trivalent chromium that was used in the derivation of the soil quality guidelines derived in this project, and the source of the toxicity data.

Species	Endpoint	NOEC or EC10 added	LOEC or EC30 added	EC50 added	Reference
Agrostis tenuis	growth	3333	5000	10000	Beeze 1973
Avena sativa	growth	400	600	1200	De Haan et al. 1985
Avena sativa	growth	200	300	600	De Haan et al. 1985
Avena sativa	growth	200	300	600	De Haan et al. 1985
Avena sativa	growth	400	600	1200	De Haan et al. 1985
Avena sativa	growth	200	300	600	De Haan et al. 1985
Avena sativa	growth	800	1200	2400	De Haan et al. 1985
Avena sativa	growth	500	750	1500	McGrath 1982
Beans	growth	200	500	600	Sykes et al. 1981
Brassica juncea	biomass	500	750	1100	Han et al. 2004
Grass	growth	200	500	600	Sykes et al. 1981
Grass	growth				
H. vulgare	growth	200	300	600	Patterson 1971
H. vulgare	growth	200	300	600	Patterson 1971
H. vulgare	growth	200	300	600	Patterson 1971
L. sativa	growth	500	750	1500	Sykes et al. 1981
L. sativa	growth	133	200	400	Sykes et al. 1981
Lollium perenne	growth	3333	5000	10000	Beeze 1973

Species	Endpoint	NOEC or EC10 added	LOEC or EC30 added	EC50 added	Reference
Phaseoleus vulgaris	growth	50	100	200.0	Wallace et al. 1976
Phaseoleus vulgaris	growth	33.3	50	100	Wallace et al. 1976
R. sativus	growth	500	750	1500	Sykes et al. 1981
R. sativus	growth	133	200	400	Sykes et al. 1981
Secale cereale	growth	233	350	700	Cunningham et al. 1975
Secale cereale	growth	233	350	700	Cunningham et al, 1975
Z. mays	growth	233	350	700	Cunningham et al. 1975
Z. mays	growth	80	320	640	Mortveldt & Giordano 1975
Z. mays	growth	1360	2040	4080	Mortveldt & Giordano 1975
E. andrei	reproduction	167	250	500.0	Molnar et al. 1989
E. andrei	reproduction	32	100	200	van Gestel et al. 1993
E. andrei	growth	320	1000	2000	van Gestel et al. 1992
E. andrei	juveniles per adult	32	100	200	van Gestel et al. 1992
E. andrei	fertility	320	1000	2000	van Gestel et al. 1992
E. andrei	fecundity	320	1000	2000	van Gestel et al. 1992
E. fetida	survival	589	883	1767	Sivakumar & Subbhuraam 2005
E. fetida	survival	552	828	1657	Sivakumar & Subbhuraam 2005
E. fetida	survival	598	897	1793	Sivakumar & Subbhuraam 2005
E. fetida	survival	609	914	1828	Sivakumar & Subbhuraam 2005
E. fetida	survival	619	928	1856	Sivakumar & Subbhuraam 2005
E. fetida	survival	567	851	1702	Sivakumar & Subbhuraam 2005

Species	Endpoint	NOEC or EC10 added	LOEC or EC30 added	EC50 added	Reference
E. fetida	survival	630	946	1891	Sivakumar & Subbhuraam 2005
E. fetida	survival	549	823	1646	Sivakumar & Subbhuraam 2005
E. fetida	survival	587	880	1761	Sivakumar & Subbhuraam 2005
E. fetida	survival	585	878	1756	Sivakumar & Subbhuraam 2005
microbial process	arylsulfatase	87	130	260	Al-khafaji & Tabatabai 1979
microbial process	arylsulfatase	867	1300	2600	Al-khafaji & Tabatabai 1979
microbial process	arylsulfatase	37	55	56	Haanstra & Doelman 1991
microbial process	arylsulfatase	37	55	203	Haanstra & Doelman 1991
microbial process	arylsulfatase	55	83	235	Haanstra & Doelman 1991
microbial process	arylsulfatase	37	55	87	Haanstra & Doelman 1991
microbial process	arylsulfatase	1819	2729	2205	Haanstra & Doelman,1991
microbial process	catalase	0.11	0.67	2.08	Stępniewska et al. 2009
microbial process	catalase	0.19	0.95	2.67	Stępniewska et al. 2009
microbial process	catalase	0.18	0.798	2.03	Stępniewska et al. 2009
microbial process	catalase	0.04	0.219	0.644	Stępniewska et al. 2009
microbial process	catalase	0.72	2.33	4.88	Stępniewska et al. 2009
microbial process	catalase	0.43	1.79	4.4	Stępniewska et al. 2009
microbial process	glutamic acid decomposition	55	400	800	Haanstra & Doelman 1984
microbial process	glutamic acid decomposition	55	400	800	Haanstra & Doelman 1984
microbial process	N-mineralisation	50	200	500	Skujins et al. 1986
microbial process	N-mineralisation	4.28	18.8	47.8	Chang & Broadbent,1982
microbial process	N-mineralisation	400	600	1200	Doelman & Haanstra 1983
microbial process	N-mineralisation	423	634	1268	Doelman & Haanstra 1983
microbial process	N-mineralisation	324	486	972	Doelman & Haanstra 1983

Species	Endpoint	NOEC or EC10 added	LOEC or EC30 added	EC50 added	Reference
microbial process	N-mineralisation	123	184	368	Doelman & Haanstra 1983
microbial process	N-mineralisation	8.00	12	24	Doelman & Haanstra 1983
microbial process	N-mineralisation	296	444	888	Doelman & Haanstra 1983
microbial process	N-mineralisation	431	646	1292	Doelman & Haanstra 1983
microbial process	N-mineralisation	1853	2780	5560	Doelman & Haanstra 1983
microbial process	N-mineralisation	2823	4234	8468	Doelman & Haanstra 1983
microbial process	N-mineralisation	86.7	130	260	Fu & Tabatabai 1989
microbial process	N-mineralisation	173	260	520	Liang & Tabatabai 1977
microbial process	nitrogenase	<<50	<<50	<<50	Skujins et al. 1986
microbial process	respiration	50.0	200	500	Skujins et al. 1986
microbial process	respiration	33.3	50	100	Chang & Broadbent 1981
microbial process	respiration	32.1	219	730	Doelman & Haanstra 1984
microbial process	respiration	2099	7514	>8000	Doelman & Haanstra 1984
microbial process	respiration	66.7	100	200	Ross et al. 1981
microbial process	respiration	66.7	100	200	Ross et al. 1981
microbial process	respiration	0.3	5.3	10.6	Stadelmann & Santschi-Fuhriman 1987
microbial process	respiration	21.3	32	64	Stadelmann & Santschi-Fuhriman 1987
microbial process	urease	50	200	1000.0	Skujins et al. 1986
microbial process	urease	0.093	0.25	0.4	Samborska et al. 2004
microbial process	urease	50	75	150	Bremner & Douglas 1971
microbial process	urease	390	585	630	Doelman & Haanstra, 1986
microbial process	urease	890	1335	1110	Doelman & Haanstra 1986
microbial process	urease	350	525	420	Doelman & Haanstra 1986
microbial process	urease	369	554	1360	Doelman & Haanstra 1986
microbial process	urease	173	260	520	Tabatabai 1977
microbial process	urease	26	26	52	Tabatabai 1977

14 Glossary

ACL (EC50) is the added contaminant limit calculated using 50% effect concentration (EC_{50}) toxicity data.
ACL (LOEC & EC_{30}) is the added contaminant limit calculated using lowest observed effect concentration (LOEC) and 30% effect concentration (EC_{30}) toxicity data.
ACL (NOEC & EC_{10}) is the added contaminant limit calculated using no observed effect concentration (NOEC) and 10% effect concentration (EC_{10}) toxicity data.
Adaptation is (1) change in an organism, in response to changing conditions of the environment (specifically chemical), which occurs without any irreversible disruption of the given biological system and without exceeding the normal (homeostatic) capacities of its response, and (2) a process by which an organism stabilises its physiological condition after an environmental change.
Added contaminant limit (ACL) is the added concentration of a contaminant above which further appropriate investigation and evaluation of the impact on ecological values will be required. ACL values are generated in the process of deriving the three sets of SQGs (calculated using NOEC and EC_{10}, LOEC and EC_{30}, and EC_{50} toxicity data). ACL values denote which toxicity data was used in their derivation by using subscripts. Thus, $ACL_{(NOEC \&EC10)}$, $ACL_{(LOEC \& EC30)}$ and $ACL_{(EC50)}$ are calculated using NOEC & EC_{10}, LOEC & EC_{30}, and EC_{50} data respectively.
Adsorption is the adhesion of molecules to surfaces of solids.
Ambient background concentration (ABC) of a contaminant is the soil concentration in a specified locality that is the sum of the naturally occurring background and the contaminant levels that have been introduced from diffuse or non-point sources by general anthropogenic activity not attributed to industrial, commercial, or agricultural activities.
An **area of ecological significance** is one where the planning provisions or land-use designation is for the primary intention of conserving and protecting the natural environment. This would include national parks, state parks, and wilderness areas and designated conservation areas.
Bioaccumulation factor (BAF) is a partition coefficient for the distribution of a chemical between an organism exposed through all possible routes and an environmental compartment or food.
Bioaccumulation is the net result of the uptake, distribution and elimination of a substance due to all routes of exposure; that is, exposure to air, water, soil/sediment and food.
Bioavailability is the ability of substances to interact with the biological system of an organism. Systemic bioavailability will depend on the chemical or physical reactivity of the substance and its ability to be absorbed through the gastrointestinal tract, respiratory tract or skin. It may be locally bioavailable at all these sites.
Bioconcentration factor (BCF) is a quantitative measure of a chemical's tendency to be taken up from the ambient environment (for example, water for aquatic organisms and soil or soil pore water for soil organisms). The BCF is the ratio of the concentration of the chemical in tissue (or a specific organ) and the concentration in the ambient environment.
Bioconcentration is the net result of the uptake, distribution and elimination of a substance due to exposure in the ambient environment (for example, water for aquatic organisms and soil or soil pore water for soil organisms).
Biological half life is the time needed to reduce the concentration of a test chemical in the environmental compartment or organisms to half the initial concentration, by transport processes, (for example, diffusive elimination), transformation processes (for example, biodegradation or metabolism) or growth.
Biomagnification factor (BMF) is a quantitative measure of a chemical's tendency to be taken up through the food web.
Biomagnification is the accumulation and transfer of chemicals via the food web due to ingestion, resulting in an increase of the internal concentration in organisms at the succeeding trophic levels.
Chronic is extended or long-term exposure to a stressor, conventionally taken to include at least a tenth of the life-span of a species.

Default conversion factors are numerical values that are used to convert a measure of toxicity to another measure of toxicity (for example, EC_{50} to a NOEC) when no experimentally determined values are available.

Ecological investigation level (EIL) is the concentration of a contaminant above which further appropriate investigation and evaluation of the impact on ecological values will be required. The EILs are calculated using EC_{30} or LOEC toxicity data. EILs are the sum of the added contaminant limit (ACL) and the ambient background concentration (ABC) and the level is expressed in terms of total concentration.

ECx is effective concentration; the concentration which affects X% of a test population after a specified exposure time.

Environmental fate is the destiny of a chemical or biological pollutant after release into the natural environment.

Generic soil quality guidelines describe a single concentration-based value that applies to all Australian soils that have a particular land use. These are derived when normalisation relationships are not available. Compare these with soil-specific soil quality guidelines.

K_d (see **water–soil partition coefficient**).

K_{oc} (see **organic carbon–water partition coefficient**).

K_{ow} (see **octanol–water partition coefficient**).

Leaching is the dissolving of contaminants in soil and subsequent downward transport to groundwater or surface water bodies.

Leachate is water that has percolated through a column of soil.

LOEC is the lowest observed effect concentration; the lowest concentration of a material used in a test that has a statistically significant effect on the exposed population of test organisms compared to the control.

NOEC is no observed effect concentration; the highest concentration of a test substance to which organisms are exposed that does not cause any observed and statistically significant adverse effects on the organisms compared to the controls.

Normalisation relationships are empirical, generally linear, relationships that can predict the toxicity of a contaminant to an organism using soil physicochemical properties. These are used in the EIL derivation methodology to generate soil-specific soil quality guidelines.

Octanol–water partitioning (K_{ow}) is the ratio of a chemical's solubility in n-octanol and water at equilibrium. This is widely used as a surrogate for the ability of a contaminant to accumulate in organisms and to biomagnify. These are often expressed in the logarithmic form (that is, log K_{ow}). Chemicals with a log K_{ow} value ≥ 4 is considered to have the potential to biomagnify. There is a linear relationship between log K_{ow} and log K_{oc} values. Thus, K_{ow} can also be used to indicate the ability of chemical to leach to groundwater. A log K_{ow} value <2 indicates a chemical has the potential to leach to groundwater.

Organic carbon–water partition coefficient (K_{oc}) is the ratio of a chemical's solubility in organic carbon and water at equilibrium. This is widely used as a surrogate for the ability of a contaminant to accumulate in soils and conversely to leach to groundwater or to be removed by surface run-off. These are often expressed in the logarithmic form (that is, log K_{oc}). Chemicals with a log K_{oc} <2.4 were considered to be mobile and therefore have the ability in some soils to leach to groundwater.

Precautionary principle is the general principle by which all that can reasonably be expected is done to prevent unnecessary risks.

Reference site is a relatively unpolluted site used for comparison with polluted sites in environmental monitoring studies or used for the assessment of ambient background concentrations of contaminants.

Soil quality guidelines (SQGs) are any concentration-based limits for contaminants in soils. Ecological investigation levels are a type of SQG.

Soil-specific soil quality guidelines is a suite of concentration-based values, where each value applies to a soil with different physicochemical properties. These values take into account properties of soils that modify the bioavailability and toxicity of contaminants. These can only be derived if normalisation relationships are available. Compare these to generic SQGs.

Speciation is the exact chemical form of contaminant in which an element occurs in a sample.
Statistically significant effects are effects (responses) in the exposed population which are different from those in the controls at a statistical probability level of $p < 0.05$.

Steady state is the non-equilibrium state of a system in which matter flows in and out at equal rates so that all of the components remain at constant concentrations (dynamic equilibrium).
Water–soil partition coefficient (K_d) is the ratio of the concentration of a contaminant in soil pore water to that in the solid phase of soil at equilibrium. The units are L/kg. This contaminant property is affected by physicochemical properties of the contaminant and the soil. This property is usually expressed as a logarithm (that is, log K_d). A chemical with log $K_d < 3$ is considered to have the potential to leach.

15 Shortened forms

ABC	ambient background concentration
ACL	added contaminant limit
AF	assessment factor
ALF	ageing and leaching factor
ANZECC	Australia and New Zealand Environment and Conservation Council
ARMCANZ	Agriculture and Resource Management Council of Australia and New Zealand
BAF	bioaccumulation factor
BCF	bioconcentration factor
BMF	biomagnification factor
CCME	Canadian Council of Ministers of the Environment
CEC	cation exchange capacity
DAF	dilution and attenuation factor
EC	European cCommission
EC10	10% effect concentration
EC30	30% effect concentration
EC50	50% effect concentration
Eco-SSL	ecological soil screening level
EIL	ecological investigation level
ERA	ecological risk assessment
EQG	environmental quality guideline
EU	European Union
HIL	health-based investigation level
LD_{10}	The dose that is lethal to 10% of organisms
LC_{10}	The concentration that is lethal to 10% of organisms
LOEC	lowest observed effect concentration
MATC	maximum acceptable toxicant concentration
MRM	maize residue mineralisation
NA	not available

N/A	not applicable
NBRP	National Biosolids Research Program
NEPC	National Environment Protection Council
NEPM	National Environment Protection Measure
NOEC	no observed effect concentration
NS	Not statistically significant (P>0.05)
OC	organic carbon
OECD	Organisation for Economic Cooperation and Development
PNEC	predicted no-effect concentration
PNR	potential nitrification rate
SIN	substrate induced nitrification
SIR	substrate induced respiration
SQG	soil quality guideline
SSD	species sensitivity distribution
US EPA	United States Environmental Protection Agency
TRV	toxicity reference value
TV	trigger value
VROM	Ministry of Housing, Spatial Planning, and the Environment (The Netherlands)

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

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Schedule B6

GUIDELINE ON

The Framework for Risk-Based Assessment of Groundwater Contamination

Explanatory note

The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

The original Schedule B6 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document.

The National Environment Protection Council (NEPC) acknowledges the contribution of the Western Australian Department of Environment and Conservation and the New South Wales Environment Protection Authority to the development of this Schedule.

Contents
The Framework for Risk Based Assessment of Groundwater Contamination

1 Introduction

1.1 Background

This Schedule provides a framework for the risk-based assessment of groundwater that has been affected, or may have been affected, by site contamination. The general process outlined for the assessment of contaminated groundwater is compatible with the policy framework (outlined in the Measure) and the site assessment process shown in Schedule A. The aim of this process is to minimise the risk of adverse human health and environmental impacts arising from contaminated groundwater and to ensure that the quality of groundwater is appropriate for its environmental values.

The framework is applicable to the assessment of groundwater quality arising from point-source contamination (for example, leaks from fuel storage depots, sheep and cattle dips). This Schedule is not intended to address all aspects of the management of groundwater quality, or to replace the regulatory requirements of individual jurisdictions for the assessment and management of groundwater contamination. Nor is it generally applicable to other broadscale groundwater issues associated with agriculture, catchment management or salinity. These resource management issues are administered by jurisdictions through various regulatory processes.

This framework is intended to be used in conjunction with Schedule B2, which includes guidance on related matters such as groundwater monitoring, the characterisation of groundwater contamination and the application of contaminant fate and transport modelling.

Detailed groundwater investigations should only be undertaken by appropriately qualified and experienced groundwater professionals. Their advice should be sought early in the assessment process to avoid remobilisation of assessment personnel and associated additional costs.

1.2 Relationship with other national guidelines

The framework recognises nationally developed approaches, policies and water quality criteria developed to protect groundwater, surface water and sources of public drinking water supply. This includes guidelines developed under the National Water Quality Management Strategy to protect marine and freshwater quality (ANZECC & ARMCANZ 2000) and for managing risks in recreational waters (NHMRC 2008) and, in addition, the Australian Drinking Water Guidelines (NHMRC & NRMMC 2011).
The framework for risk-based assessment of groundwater applies these national guidelines to the specific issue of assessing the quality of groundwater impacted by site contamination.

1.3 Definition of contaminated groundwater

In the context of a contaminated site assessment, groundwater is considered to be contaminated when its quality is such that it is not suitable for the current or realistic future use or presents the likelihood of causing an unacceptable environmental or human health impact in the discharge environment. This differs from the National Water Quality Management Strategy definition in the *Guidelines for groundwater protection in Australia* (ARMCANZ & ANZECC 1995). The latter considers groundwater to be contaminated whenever there is a change in water quality that produces a noticeable or detectable change in its characteristics. Therefore, if a change in groundwater quality is detected or is reasonably suspected, relevant jurisdictional policies should also be taken into account regarding groundwater protection.

2 Site assessment process and terminology

2.1 Site assessment process

The site assessment process is shown in Schedule A. Once the need for an assessment is triggered, a preliminary site investigation (PSI) should be conducted using the guidance outlined in Schedule B2. The scope of the PSI should be sufficient to identify the potential contaminants of concern and the environmental media that are potentially affected by these contaminants.

A detailed site investigation (DSI) is required when the results of the PSI indicate that contamination is present or is likely to be present and there is insufficient information to delineate the extent of contamination and to enable site management strategies to be devised. Monitoring of groundwater conditions is an important part of the site assessment process to determine seasonal, and where appropriate, longer-term trends. The detailed investigation stage should identify the nature of the contamination and delineate its lateral and vertical extent to a sufficient degree that an appropriate level of risk assessment may be undertaken and, if necessary, to provide the basis for the development of an appropriate remediation or management strategy.

This more detailed investigation should result in an estimation of the current and projected contaminant concentrations in the receiving environment at the points of existing and realistic future use. Contaminant fate and transport modelling may be required to estimate the contaminant concentrations at these points. The investigation process should consider:

- all potential exposure pathways
- the properties of the contaminants such as persistence and bioavailability
- the likely temporal variability in contaminant concentrations
- the physicochemical and biochemical transformations that occur between the contamination source and the point of current or future realistic use.

Further guidance on related matters such as groundwater monitoring, the characterisation of groundwater contamination and the application of contaminant fate and transport modelling can be found in Schedule B2.

2.2 Groundwater investigation levels

Groundwater investigation levels (GILs) are defined as 'the concentration of a contaminant in groundwater above which further investigation (point of extraction) or a response (point of use) is required'. Selected GILs are tabulated in Table 1C of Schedule B1 and are sourced from the:

- Australian water quality guidelines for fresh and marine water (AWQG) (ANZECC & ARMCANZ 2000)
- Australian drinking water guidelines (ADWG) (NHMRC & NRMMC 2011)
- Guidelines for managing risk in recreational water (GMRRW) (NHMRC 2008).

The GILs are designed to avoid unacceptable impact to exposed populations or ecosystems under a range of circumstances. For example, the GILs for protection of freshwater and marine water ecosystems were derived using a statistical distribution method and were calculated at four different protection levels, where the data permitted, and are applied according to the ecosystem condition. The aquatic ecosystem protection GILs presented in Table 1C of Schedule B1 are applicable to 'slightly – moderately disturbed' ecosystems. The AWQG should be consulted for additional values for protection of disturbed ecosystems and pristine ecosystems.

For guidance on the selection of relevant GILs—see Section 3.

Levels marginally in excess of the GILs do not imply unacceptability or that a significant human health or ecosystem impact is likely to be present. Subject to an appropriate investigation and assessment process, a decision not to take further action or to take further action may be justifiable based on the findings.

GILs are not intended to be clean-up levels. The decision on whether clean-up is required (and to what extent), should be based on site-specific assessment. Risk assessment is one aspect of making the decision; however, other considerations such as practicality, timescale, effectiveness, community acceptance, cost and durability are also important.

The referenced source documents should be consulted for information on how to develop site-specific criteria where generic guidelines are not available or applicable.

2.3 Conceptual site model

In order to commence an effective risk-based assessment of a site, a preliminary understanding of the potential site issues is necessary. The understanding of the site is referred to as a conceptual site model (CSM) and describes the source(s) of contamination, the pathway(s) by which contaminants may migrate through the various environmental media, and the populations (human and/or ecological) that may be exposed. For further information on the issues to be considered in the development of a CSM, refer to Schedules B2 (Sections 4 and 8) and B4.

2.4 The tiered approach

The risk assessment process for contaminated sites is usually undertaken in stages or 'tiers' involving progressively more detailed levels of data collection and analysis. In this guidance, the tiers are referred to as Tier 1, Tier 2 and Tier 3. The approach provides for assessment at a level of complexity that is appropriate for the problem under consideration. As the amount of data and assessment detail increases, and the CSM is refined and data gaps are filled, the level of uncertainty decreases. In turn, the level of uncertainty in the risk assessment process is reduced.

2.5 The basis for groundwater risk assessment

Groundwater should be assessed on the basis of its environmental values and the risk that the current (or realistic future) use may pose to human health and/or the environment. With regard to realistic future uses, consideration should be given to the quality and yield of the aquifer, the likely demand for water resources in the vicinity of the site, and technological practicalities.

The assessment process for groundwater contamination differs from that for land contamination in that there is greater emphasis on suitability for current and realistic future uses, compared with the emphasis on current and intended uses with soil assessment. The focus on the protection of environmental values and realistic future uses (based on the inherent capacity of the aquifer to support those uses) is derived from the following considerations:

- groundwater contamination may be persistent and difficult to contain or to remediate within a short timeframe
- some groundwater contamination may persist beyond current planning horizons, affecting future uses that today are not considered likely
- the stress on Australia's water resources is expected to increase, highlighting the importance of protecting groundwater resources for the future.

With soil assessment, land use and the level of contact between the most sensitive human or ecological receptor and the contaminated soil primarily determine the level of protection required. In the case of groundwater, consideration may include a combination of several different exposure scenarios and multiple potential receptors, for example, groundwater may be used for irrigation purposes, pass

beneath a freshwater lake and then go on to discharge into the marine environment. Potential receptors will differ in each scenario and acceptable contaminant levels may well be different for each receptor. An assessment of groundwater contamination should consider the sensitivity of receptors in each exposure scenario.

3 Framework for applying water quality guidelines in the risk-based assessment of contaminated groundwater

3.1 Introduction

This section provides a framework for the use of the following guidelines in the risk-based assessment of contaminated groundwater:

- *Australian water quality guidelines for fresh and marine water* (AWQG) (ANZECC & ARMCANZ 2000)
- *Australian drinking water guidelines* (ADWG) (NHMRC & NRMMC 2011)
- *Guidelines for managing risk in recreational water* (GMRRW) (NHMRC 2008).

These guidelines present criteria for potential contaminants of concern. These criteria are adopted as GILs in this NEPM and form the basis for the assessment of contaminated groundwater and associated risks. The GILs are trigger levels which, if exceeded, have the potential to cause a problem and so trigger further investigation or management action.
The criteria defined within the ADWG apply at the point of use, for example, at the tap, and are applicable to any water, including bore water, where that water is intended for potable use. In this Schedule, the ADWG criteria are used as investigation levels for comparison with groundwater quality monitoring data (Tier 1 and 2) and, for example, the results of contaminant fate and transport modelling (Tier 3). A management response should be considered if the ADWG are (or are likely to be) exceeded at the point of use.

The assessment framework is based on identifying the receptors (human and/or ecological) for groundwater that is contaminated and determining the level of protection required by referring to the appropriate set of guidelines within the AWQG, the ADWG and the GMRRW.

Schedule B1 of this NEPM introduces Health Screening Levels (HSLs) for groundwater, for protection of human health from petroleum hydrocarbon vapours. Schedule B1 and references therein should be consulted for details of the application of the groundwater HSLs.

3.2 Groundwater environmental values

Environmental values are values or uses of the environment that are conducive to public benefit, welfare, safety or health and that require protection from the effects of pollution, waste discharge and deposits. The AWQG, ADWG and GMRRW set out criteria for water quality relating to a number of environmental values:

- ecosystem protection
- aquaculture and human consumers of food
- agricultural water (irrigation and stock water)
- recreation and aesthetics
- drinking water
- industrial water.

For each environmental value, a set of guideline criteria is presented for potential contaminants of concern.

Ecosystem protection, in this context, refers to aquatic ecosystems which depend at least in part on groundwater to maintain ecosystem health (groundwater-dependent ecosystems). Depending on the site setting, this may include surface water bodies such as wetlands, streams and rivers reliant on groundwater base flow, some estuarine and near-shore marine systems, as well as aquifer and cave

ecosystems. Consideration of the water body/groundwater characteristics will determine whether the freshwater or marine water GILs are the most appropriate to apply.

Table 1. Summary of relevant guidelines to protect environmental values of groundwater.

Environmental value	Relevant guideline
Ecosystems	AWQG (fresh and marine ecosystem guidelines)
Drinking water	ADWG
Recreational use*	GMRRW
Industrial use • Agricultural use (irrigation and stock watering) • Aquaculture	 • AWQG (irrigation and stock watering guidelines) • AWQG (aquaculture)

* The recreational and aesthetics sections of the AWQG have been superseded by the GMRRW (NHMRC 2008).

3.3 Background groundwater quality

The application of the policy framework includes consideration of background groundwater quality. Background groundwater quality is considered to be the sum of both ambient and natural sources in the local area of a site. Very few organic chemicals would be expected to have elevated natural background levels in groundwater. However, in the case of metals, metalloids and some inorganic substances, background concentrations may be elevated due to both natural and ambient background contributions.

The assessment of background water quality should be undertaken at an area(s) that is not affected by the activities that have contributed to the contamination present at the site. For example, it may be appropriate to collect samples unaffected by the contamination upgradient of the impacted area; otherwise a suitable area in the vicinity of the site, which is unlikely to have been impacted by the potential contaminants of concern, should be selected. Further information may be found in SA EPA (2008).

In addition to the potential contaminants of concern identified in the conceptual site model, it is recommended that the sampling and analysis plan should include the analysis of major ions to assist with differentiation between contaminated and non-contaminated groundwater.

Sufficient sampling in terms of both areal and temporal considerations should be undertaken to establish the natural variation in groundwater quality due to seasonal effects. Where sufficient concentration data is available for statistical analysis, the 80th percentile of the background concentration data may be used for comparison with the site data (ANZECC & ARMCANZ 2000).

3.4 Fundamentals of the tiered approach

3.4.1 Tier 1

A Tier 1 assessment is the first stage of assessment and provides an initial screening of the site data. This includes:

- reviewing site contamination history, identifying all past and present contaminating activities and associated potential contamination
- reviewing available information about local and regional geology and hydrogeology
- identifying aquifers and confining layers, groundwater flow domain, potential receptors

- identifying natural geochemistry of the groundwater system
- sampling of site groundwater monitoring wells and identifying seasonal trends in groundwater quality
- comparing site data with relevant GILS.

The purpose of Tier 1 assessment is to determine whether further assessment is required. It includes a comparison of monitoring data from the site with relevant GILs (such as those listed in Table 1C Groundwater Investigation Levels in Schedule B1). The relevant GILs should be selected on the basis of the environmental values identified in the conceptual site model.
Exceedence of Tier 1 criteria is generally used to determine whether there is a need to collect more data and/or progress to a Tier 2 assessment. An assessment of the significance of exceedences may be necessary where they are marginal or present over a limited area. Under some circumstances further assessment of contaminants exceeding Tier 1 criteria may not be conducted (e.g. where the extent of the exceedence and cost of remediation is small and further assessment is not cost-effective). Where further assessment of contaminants exceeding Tier 1 criteria is not proposed, a clear and transparent explanation should be provided.

This means that a groundwater sample from a monitoring well with contaminant levels above the GILs will trigger further investigation rather than initiate remedial action. However, site-specific consideration should be given to water quality impacts that cause variations from ambient water quality even when GILs are not exceeded. This is because individual jurisdictions may operate protective strategies for groundwater that require action at levels below the GILs or whenever levels of contaminants above ambient background are detected. Such issues are the responsibility of jurisdictions.

3.4.2 Tier 2

A Tier 2 assessment is typically required when one or more contaminants are present at the site at levels that exceed Tier 1 guidance criteria, or if there are no appropriate Tier 1 criteria, or if there are unresolved and significant uncertainties identified in the Tier 1 assessment.

Tier 2 assessment includes consideration of the site-specific conditions and the modification of Tier 1 generic GILs according to the site conditions, including actual exposure. For example, the toxicity of some metals (Cd, Cr III, Cu, Pb, Ni and Zn) to freshwater biota is known to reduce with increasing water hardness (ANZECC & ARMCANZ 2000). The AWQG are conservatively presented on the basis of low hardness (30 mg/L $CaCO_3$) and the relevant GILs may be modified for increased levels of hardness according to the algorithm presented in ANZECC & ARMCANZ 2000.

Exceedence of Tier 2 criteria may result in a need for a Tier 3 assessment. As with Tier 1 exceedences, an assessment of the significance of exceedences may be necessary where they are marginal or present over a limited area. If Tier 2 criteria are exceeded, but further assessment (or action) is not proposed, the information and logic used to inform the decision should be documented clearly and transparently.

If no modification of the Tier 1 criteria is applicable, then the risk assessor may decide to proceed directly to Tier 3.

3.4.3 Tier 3

A Tier 3 assessment may be required where exceedence of Tier 2 site-specific target levels is judged to represent a potentially unacceptable risk to human health and/or the environment. The Tier 3 assessment typically focuses on the risk-driving contaminants in more detail and generally requires additional site investigation to reduce critical uncertainties in the risk assessment.

Tier 3 risk assessments compare groundwater contaminant concentrations at the point of exposure (point of use) with existing generic GILs or can incorporate additional information such as ecosystem/environmental variability and exposure to derive modified, site-specific response levels.

The relevant jurisdictional policy should be consulted when modifying GILs at the point of use. For example, when determining criteria for groundwater discharging to a surface water body, these should be determined on a site-specific basis, as some jurisdictions allow for a mixing zone or water treatment, whereas others apply the GILs at the point of discharge without mixing in order to protect benthic organisms.

Further information is available in Schedule B2 and ANZECC & ARMCANZ (2000).
Example Tier 3 activities include:

- contaminant fate and transport modelling to predict groundwater quality at existing (and realistic future) receptors using a range of aquifer conditions to assess the significance of the site contamination at the point of exposure/use – refer Schedule B2
- consideration of metal speciation (speciation modelling or chemical measurement) – refer AWQG
- biological effects testing (for example direct toxicity testing) – refer AWQG.

3.5 Risk management

At the point of use or exposure, GILs may be considered as response levels: the response may include further investigation or management as appropriate.

Contaminant levels marginally in excess of the GILs do not imply unacceptability or that a significant human health or ecosystem risk is likely to be present. The decision on whether clean-up is required (and, if so, to what extent) should be based on site-specific assessment. Risk assessment is one aspect of making the decision though other considerations such as practicality, timescale, effectiveness, cost, durability, relevant regulatory policy, and community acceptance are also important.

A management plan for unacceptable levels of contamination may include one or more of the following:

- work plan
- determination of site-specific clean-up criteria
- development of site management options
- determination of clean-up methods
- implementation plan of remedial actions
- water treatment at the point of use
- restriction on the use of the aquifer
- provision of alternative water supply
- future monitoring and information provisions.

These management issues are beyond the scope of the NEPM and are matters administered by jurisdictions.

4 Bibliography

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SA EPA 2008, Site contamination – Determination of background concentrations, South Australia Environmental Protection Authority, Adelaide, Australia.

5 Glossary

Ambient background means the condition of groundwater representative of the area surrounding the site not attributable to an identifiable point source(s). The impacts of widespread diffuse sources of groundwater contamination are included within 'ambient background'.

Aquifer is a rock or sediment in a geological formation, group of formations or part of a formation which is capable of being permeated permanently or intermittently and can thereby transmit water.

Background groundwater quality means the condition of groundwater in the vicinity of a site which is the sum of the ambient and natural background.

Bioavailability is a general term meaning the amount of a contaminant that is absorbed into the body following dermal contact, ingestion or inhalation.

Contaminated groundwater means groundwater that has contamination at such a level that the condition of groundwater is such that it is not suitable for the current or realistic future use or presents the likelihood of causing an unacceptable environmental or human health impact in the discharge environment.

Contamination means the condition of land or water where any chemical substance or waste has been added as a direct or indirect result of human activity at above background level and represents, or potentially represents, an adverse health or environmental impact.

Discharge area means an area in which there are upward components of hydraulic head in the aquifer. Groundwater flowing toward the land surface in a discharge area may escape as a spring, leading to a discharge, seep or base flow, or by evaporation and transpiration.

Environmental value is a value or use of the environment which is conducive to public benefit, welfare, safety or health and which requires protection from the effects of pollution, waste discharge and deposits.

Groundwater-dependent ecosystem means an ecosystem that is wholly or partially dependent on groundwater for ecosystem health. For groundwater risk assessment this may include surface water bodies such as wetlands and rivers with groundwater base flow, some estuarine and near-shore marine systems, as well as aquifer and cave ecosystems.

Groundwater investigation level (GIL) is the concentration of a groundwater parameter at which further investigation (point of extraction) or a response (point of use) is required. Includes Australian water quality guidelines/drinking water guidelines/guidelines for managing risk in recreational water criteria and site-specific derived criteria.

Groundwater means all waters occurring below the land surface.

Natural background means the condition of groundwater derived/originating from natural processes in the environment as close as possible to natural conditions, exclusive

of specific anthropogenic activities or sources.

Point source means a source of contamination which comes from a contaminating activity at a particular site.

Receptor is the entity (organism, population, community, or set of ecological processes) that may be adversely affected by contact with, or exposure to, a contaminant of concern.

Response level means the concentration of a contaminant at a specific site, based on a site assessment, for which some form of response is required to provide an adequate margin of safety to protect public health and/or the environment.

Risk assessment is a process intended to calculate or estimate the risk to a given target organism, system, or sub-population, including the identification of attendant uncertainties, following exposure to a particular contaminant, taking into account the inherent characteristics of the agent of concern as well as the characteristics of the specific target system.

Risk is the probability of an adverse effect in an organism, system or sub-population caused under specific circumstances by exposure to a contaminant.

Risk management is a decision-making process involving consideration of political, social, economic, and technical factors with relevant risk assessment information relating to a hazard to determine an appropriate course of action.

Tier 1 assessment is a risk-based analysis comparing site data with generic published screening criteria (Tier 1 criteria) for various environmental values.

Tier 2 assessment is a site-specific assessment in which risks to potentially exposed populations are assessed using site-specific data on pathways, and the characteristics of the exposed populations. In Tier 2, site data is compared with generic criteria modified for site-specific conditions.

Tier 3 assessment is a further step from a Tier 2 evaluation and examines the specific risk-driving factors in more detail. This often involves additional data collection and may incorporate more sophisticated modelling techniques. In Tier 3, site data is compared with site-specific target levels.

Well is a hole drilled into an aquifer for the purpose of monitoring or extracting groundwater. This generic term includes bores, water wells and tubewells.

6 Shortened forms

ADWG	Australian Drinking Water Guidelines
ANZECC	Australian and New Zealand Environment and Conservation Council
AWQG	Australian Water Quality Guidelines
CSM	conceptual site model
DSI	detailed site investigation
HIL	health investigation level
HSL	health screening level
GIL	groundwater investigation level
GMRRW	Guidelines for managing risk in recreational waters
NEPM	National Environment Protection Measure
NHMRC	National Health and Medical Research Council
NRMMC	National Resource Management Ministerial Council
PSI	preliminary site investigation

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Schedule B7

APPENDIX A2

The Derivation of HILs for PAHs and Phenols

Contents
The derivation of HILs for PAHs and phenols

1 Benzo(a)pyrene

1.1 General

Several comprehensive reviews of polycyclic aromatic hydrocarbons (PAHs) and benzo(a)pyrene (BaP) in the environment and their toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 1995; WHO 1998; CCME 2008). The following provides a summary of the key aspects of these compounds that are relevant to the derivation of a soil HIL.

PAHs are a large group of organic compounds with two or more fused aromatic rings made up of carbon and hydrogen atoms. PAHs are formed from incomplete combustion of organic materials such as the processing of coal, crude oil, combustion of natural gas, refuse, vehicle emissions, heating, cooking and tobacco smoking, as well as natural processes including carbonisation. The natural background level is due to PAH production in plant species. Because of such widespread sources, PAHs are present almost everywhere. Food is considered to be the major source of human exposure to PAH, due to the formation of PAH during cooking or from atmospheric deposition of PAHs on grains, fruits and vegetables (WHO 1998).

There are several hundred PAHs, including derivatives of PAHs. The best known (and studied) is BaP. While there are hundreds of PAHs, typically only 16 individual PAHs are analysed in site contamination investigations. These individual PAHs address a broad range of the equivalent carbon spectrum and are therefore more commonly reported and assessed (where there is more data available on these PAHs).

The major sources of PAHs to soils at any given location invariably contribute a mixture of PAHs, not just single compounds. Various PAH source types can be distinguished based on the characteristic compositions of PAH mixtures and information on the site history, but the contaminated soil matrix is nonetheless challenging from an environmental risk assessment perspective, since in a PAH-contaminated soil there is likely to be a diverse compositional range of non-carcinogenic and carcinogenic PAHs of varying potency.

The major approach advocated by regulatory agencies such as the NEPC (NEPC 1999; Fitzgerald 1991; Fitzgerald 1998), California EPA (OEHHA), Netherlands (RIVM 2001), England and Wales (DEFRA & EA 2002), Canada (CCME 2008) and US EPA (2010 draft) for assessing the human health risks of PAH-containing mixtures involves the use of toxicity equivalence factors (TEFs). This approach relates the toxicity of other (potentially carcinogenic) individual PAHs relative to that of BaP, the most widely studied PAH.

There are more than a dozen sets of equivalency numbers that have been proposed over the last two decades. The most recent (published final) review of TEF and their basis, presented by CCME (2008), suggests the use of TEF recommended by the World Health Organization (WHO 1998), with minor modifications. This is a scheme based on the order-of-magnitude cancer potency.

Any finer-scale assertions about relative potency for more generic application are hard to justify given the current state of knowledge and confounding influences such as the route of exposure or non-additive effects in complex PAH mixtures. It is not currently possible to develop different relative potency schemes across different exposure routes (oral, dermal, inhalation), owing to a lack of data. Hence the TEFs adopted have been applied for all routes of exposure for the carcinogenic PAHs assessed. Application of the TEFs is relevant to the assessment of PAHs that are considered to be carcinogenic. Other PAHs that are not carcinogenic should be assessed separately on an individual basis.

The following table presents a summary of the TEFs adopted for the assessment of carcinogenic PAHs (CCME 2008):

PAH	IARC Classification	US EPA Classification	TEF
Benzo(a)anthracene	2B	B2	0.1
Benzo(a)pyrene	1	B2	1
Benzo(b+j)fluoranthene	2B	B2	0.1
Benzo(k)fluoranthene	2B	B2	0.1
Benzo(g,h,i)perylene*	3	D	0.01
Chrysene	2B	B2	0.01
Dibenz(a,h)anthracene	2A	B2	1
Indeno(1,2,3-cd)pyrene	2B	B2	0.1

Notes: 1/A= Human Carcinogen, 2A/B2= Probable Human Carcinogen, 2B/C=Possible Human Carcinogen, 3/D= Not classifiable.

* Benzo(g,h,i)perylene included due to positive findings in genotoxicity studies (WHO 1998). Note there is insufficient data available to determine carcinogenicity.

The toxic effects of different PAH compounds in a mixture are additive. Experimental evidence suggests that this is a fair assumption (Fitzgerald 1991; Fitzgerald 1998; CCME 2008).

The following relates to the approach used to assess BaP in the derivation of HILs (which can be used for the assessment of BaP alone or for carcinogenic PAHs using the above TEFs).

1.2 Previous HIL

The derivation of the previous HIL (HIL A = 1 mg/kg) for BaP is presented by Fitzgerald (1991) and NEPC (1999). In summary, the HIL was derived on the basis of the following:

- Intakes associated with daily exposure by children and adults living near or on soil containing 1 mg/kg BaP were assessed on the basis of:
    - Dermal absorption, with 1% BaP absorbed via the skin
    - Ingestion, with 100% bioavailability assumed
    - Inhalation, over 24 hours, with 100% bioavailability assumed.
- In comparison to background intakes of BaP, intakes from soil at 1 mg/kg are low but higher intakes may be nearing a significant contribution. Adoption of 1 mg/kg was considered appropriate also due to the potential for further review by S EPA where reference values for BaP may change.

Further review of BaP (and PAHs using TEFs) by Fitzgerald (1998) and Fitzgerald et al. (2004), on the basis of a derived modified benchmark dose, calculated a value of 5 mg/kg on the basis of soil ingestion only.

1.3 Significance of Exposure Pathways

1.3.1 Oral Bioavailability

A study by Hansen et al. (2007) demonstrated bioavailability of PAHs in three different soil samples ranging from 14– 40% using an in vitro bioavailability model that simulates gastric digestion. In addition, the Massachusetts DEP uses a relative absorption fraction of 28% for PAHs (MADEP 2008) in its risk assessment program. In addition it is noted that BaP (and PAHs) present in bitumen

fragments are largely immobile and typically have a low bioavailability. However, as bioavailability is highly site- and source-specific, insufficient data is available to adequately define a value that differs from the default approach of 100% oral bioavailability. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

1.3.2 Dermal absorption

Review of dermal absorption of BaP has been conducted by MfE (2011). This review has identified the following, based on studies on animals and humans (rather than modelled as presented by CCME (2008)):

- As BaP is actively metabolised in the skin, it is relevant to include both the amount that passes through the skin and that which remains bound to the skin to estimate dermal uptake.
- US EPA (2004) recommends a dermal absorption factor of 0.13 (13%), which is based on data from Wester et al. (1990). These authors indicate that 13.2% of BaP in soil was absorbed by rhesus monkeys over a 24-hour period. However, they also indicate that a reduced amount (1.4%) was absorbed into human skin from soil over the same time period, although no partitioning into human plasma occurred, i.e. the BaP remained bound to the skin.
- Another study on the dermal absorption of BaP from soils also showed that a minimal amount (0.1%) of BaP was absorbed through pig skin and 1.7% and 3.5% remained bound to the skin when BaP respectively in aged sandy and clay soils was applied to the skin (Abdel-Rahman et al. 2002). A higher amount (3.3% and 8.3% in clay and sandy soils, respectively) was absorbed when non-aged soil (i.e. freshly spiked) was applied to the skin.
- A more recent study with human skin showed greater absorption through the skin, with approximately 7% of BaP passing through when applied as freshly spiked soil (Moody et al. 2007). A further 7% remained bound to the skin.
- As ageing soils decrease the bioavailability of BaP, the dermal absorption data from freshly spiked soils can provide a worst-case estimate of dermal absorption. The geometric mean of dermal absorption using freshly spiked soils from the above studies (including in vivo studies) is 6%, while using data for aged soils yields a geometric mean of 2.6% (Abdel-Rahman et al. 2002).

Review by MfE (2011) resulted in the adoption of a dermal absorption factor of 2.6%, the arithmetic mean of data from aged soil (Abdel-Rahman et al. 2002). In the derivation of soil HILs in this review, the higher arithmetic mean value of 6% (based on data from freshly spiked soil and noted by MfE (2011) as a worst-case value that is supported by studies from Wester et al. (1990), Abdel-Rahman et al. (2002) and Moody et al. (2007)) has been adopted and is considered relevant for all source types.

1.3.3 Inhalation of Dust

BaP (and other carcinogenic PAHS) are not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. Exposure via inhalation of dust is estimated to be less than 1% of the total exposure.

1.3.4 Plant Uptake

CCME (2008) notes that concentrations of PAHs in uncooked produce depend principally on its source. Plants grown on PAH-contaminated soils, however, have only a limited ability to take in through the roots and translocate anthropogenic PAHs to the aboveground plant biomass—especially for higher molecular weight PAHs. One mode of plant contamination is via the deposition of PAH-containing fine particulates onto plant surfaces.

PAHs may be bound within soils (via lignification), mineralised (ultimately to CO_2 and water) or metabolised outside or within the plant (CCME 2008). Higher molecular weight PAHs such as BaP (and other carcinogenic PAHs) are considered persistent and are strongly absorbed to the soil. Lipophilic organic compounds such as PAHs (and BaP), with a low solubility in water, high Henry's law constant and high Kow($>10^4$), are bound strongly to the root surface and/or soils and are not readily translocated within plants (Schnoor 1997). These generally tend to partition into the epidermis or outer layers of the root tissue (or peel) and remain there bound to lipids in cell walls; transfer into the inner root or xylem is very slow or non-existent. CCME (2008) notes that the general consensus in the literature is that the root uptake pathway of organic contaminants such as hydrocarbons and PAH constituents from the soil by plants is extremely limited, particularly for the heavier PAHs such as BaP.

On the basis of the above, plant uptake has not been considered in the derivation of HIL A. However it is noted that if plant uptake were considered (using the equations presented in Appendix B), intakes derived from this source are low and do not significantly contribute to the HIL (<1%).

1.3.5 Intakes from Other Sources – Background

Intakes of BaP from sources other than soil have been considered by Fitzgerald (1991) to range from 0.166–1.6 µg/day (US EPA 1980) with intakes derived from food identified as the most significant. While more detailed reviews are available on potential intakes of BaP (CCME 2008), background intakes are not considered in the derivation of an HIL for BaP, as a non-threshold approach has been adopted.

1.4 Identification of Toxicity Reference Values

1.4.1 Classification

The International Agency for Research on Cancer (IARC 2010) has classified BaP as 1—human carcinogen.
The US EPA has classified BaP as B2—probable human carcinogen.

1.4.2 Review of Available Values/Information

BaP has been shown to be carcinogenic via all routes of exposure. BaP is an indirect carcinogen, that is, its carcinogenicity results from its metabolites, primarily various epoxides, as opposed to BaP itself. Several different types of tumours have been observed as a result of exposure to BaP, although tumour development is closely related to route of administration, i.e. dermal application induces skin tumours and oral administration induces gastric tumours. Exposure to BaP causes disruption to cellular genetic material, in particular DNA adducts are formed as a result of exposure and BaP is considered to be a genotoxic carcinogen (WHO 1998).

In addition BaP has been demonstrated to be a skin irritant and dermal sensitiser (WHO 1998).

US EPA (2005) has concluded that BaP (and carcinogenic PAHs assessed on the basis of a TEF) acts via a mutagenic mode of action and recommends that susceptibility associated with early lifetime exposures be addressed. No non-threshold values available for BaP have been derived to specifically address early lifetime susceptibility and hence these issues may need to be addressed when characterising exposure to BaP.
On this basis, a peer-reviewed non-threshold reference value is recommended for BaP. The following non-threshold values are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC	Not available	Current guideline of 0.00001 mg/L established in ADWG (NHMRC 2011) is based on the consideration of health

Source	Value	Basis/Comments
2011)		effects in relation to the limit of determination for analysis. The assessment provided by the WHO is noted.
OCS (2012)	No evaluation available	
International		
WHO (2011)	SF = 0.5 $(mg/kg/day)^{-1}$ UR =$8.7x10^{-5}$ $(ng/m^3)^{-1}$	WHO (2011) derived a drinking water guideline of 0.0007 mg/L on the basis of an excess lifetime cancer risk of10^{-5} from an oral carcinogenicity study (Neal & Rigdon 1967) and a two-stage birth–death mutation model. Slope factor has been calculated on the basis of a 70 kg adult and consumption of 2 L water per day. Inhalation UR derived (WHO 2000 and 2010) based on observations in coke oven workers to mixtures of PAHs. It is noted that the composition of PAHs to which coke oven workers are exposed may differ from that present in ambient air, or derived from soil contamination. It is noted that an inhalation UR is in the same order of magnitude as that derived using a linear multistage model associated with lung tumours in a rat inhalation study of coal tar/pitch condensation aerosols.
MfE (2011)	SF = 0.233 $(mg/kg/day)^{-1}$	Review of the carcinogenic reference values available for oral intakes by MfE (2011) considered the range of values available and differences in approaches adopted for low dose extrapolation. The application of cross-species scaling appeared to be the most significant factor affecting the cancer potency estimates. It was recommended that cross-species scaling should not be applied, consistent with the approach outlined in NHMRC (1999). Review of available studies (14 risk estimates using 4 databases) resulted in the calculation of a geometric mean based on data without scaling that was recommended for use in the derivation of a soil guideline value.
EA (2002)	Derived index doses from WHO evaluations	Oral index dose derived on the basis of WHO approach and a lifetime cancer risk of 10^{-5}. Inhalation index dose based on WHO approach and adopting an air guideline of 0.25 ng/m^3. The air guideline is equivalent to a lifetime cancer risk of $4x10^{-5}$.
RIVM (2001)	SF = 0.2 $(mg/kg/day)^{-1}$	Oral SF derived by RIVM based on a chronic oral carcinogenic rat study and linear multistage model. The study considered was more recent than that considered by WHO. No inhalation assessment is provided by RIVM.
CCME (2008)	SF = 2.3 $(mg/kg/day)^{-1}$	Oral SF derived from a less than lifetime diet study on inbred CFW-Swiss mice associated with incidence of papillomas and squamous cell carcinomas and linear extrapolation. This is the same study as used by US EPA in the derivation of its oral slope factor. The CCME review also noted that dermal exposures and primary oral exposures result in different kinds of cancers. Health Canada is currently reviewing data with respect to the derivation of a dermal cancer slope factor, which may require consideration when peer-reviewed and published. The oral slope factor has been used to derive a soil

Source	Value	Basis/Comments
		guideline associated with exposures via oral, dermal and inhalation exposures.
OEHHA (CEPA 1999)	SF = 11.5 $(mg/kg/day)^{-1}$ UR =0.0011 to0.0033 $(ug/m^3)^{-1}$	Oral SF derived using the same model and study as reported by US EPA (IRIS 2012) and CCME (2008), with the upper end of the range of values adopted by OEHHA. Inhalation UR derived on the basis of respiratory tract tumours in an inhalation study in hamsters and a linearised multistage model.
US EPA (IRIS 2012)	SF = 7.3 $(mg/kg/day)^{-1}$	Oral SF (last reviewed in 1994) derived on the basis of the same study considered by CCME (above) where a range of slope factors was derived (4.5 to 11.7 $(mg/kg/day)^{-1}$). The geometric mean was adopted as the recommended slope factor for derivation of a drinking water guideline. No assessment of inhalation toxicity is available.

There is a wide range of non-threshold reference values available for oral intakes of BaP. The most recent review, where the methodology used for low dose extrapolation was reviewed, was conducted by MfE (2011). The evaluation presented considered all the available and relevant studies noted in the above tables and identified an oral reference value based on the geometric mean. This value is considered appropriate for the derivation of HILs. However it is noted that the reference document remains a draft at the time of preparation of this evaluation, hence additional consideration of a finalised peer-reviewed reference value has also been presented.

Based on the available published peer-reviewed sources, the oral reference value presented in the WHO DWG (2011) can also be considered (remains current and relevant) in the derivation of soil HILs. The WHO oral reference value is similar to the value derived by RIVM (2001) and has been adopted by EA (2002).

The data available on inhalation exposures is dominated by occupational studies associated with exposure to coke oven emissions or coal tar pitch aerosols. BaP is not volatile and hence the relevance of these studies to the assessment of dust issues derived from contaminated sites is not clear. It is therefore recommended that the WHO oral reference value be considered for the assessment of all pathways of exposure.

1.4.2.1 Note on Dermal Exposures

BaP is suggested to act largely as a point-of-contact carcinogen (Knafla et al. 2006), as opposed to systemically, hence it is more appropriate to derive soil guideline values for the dermal route of exposure using a route-specific slope factor, as opposed to consideration on the basis of systemic absorption and use of the oral slope factor.

For most compounds such data is not available but for BaP, Knafla et al. (2011) have derived a dermal slope factor, normalised to a per unit skin surface area basis, that is relevant to the assessment of BaP in soil in skin. The dermal slope factor of 3.5 $(\mu g/cm^2/day)^{-1}$ was derived by Knafla et al. (2011) and appropriate methods and parameters have been suggested for the use of this factor in the assessment of soil exposures. The dermal slope factor is an extension of previous work published by Knafla et al. (2006), where a dermal slope factor was derived on the basis of skin carcinogenicity from skin painting studies with mice. The revised dermal slope factor (Knafla et al. 2011) considered various factors for interspecies extrapolation, particularly in relation to sensitivity (to tumour development) and differences in epidermal (target tissue) thickness. This dermal slope factor has not yet been adopted for use by other international agencies, however CCME (2008) indicate that Health Canada may consider the revised dermal slope factor once published (as occurred in 2011).

The dermal slope factor as proposed by Knafla et al. (2011) has been considered in the derivation of the HIL for BaP, in addition to the use of the oral TRV. The calculations have been conducted for garden soil using default values presented by Knafla et al. (2011) for loading rates and epidermal thickness.

1.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for BaP in the derivation of HILs:

Recommendation for BaP and carcinogenic PAHs as BaP TEF

Oral TRV (TRV_O) = 0.233 $(mg/kg/day)^{-1}$ (MfE 2011) for all routes of exposure

Value has been compared with TRV_O = 0.5 $(mg/kg/day)^{-1}$ (WHO 2011) for all routes of exposure

Dermal absorption factor (DAF) = 0.06 (or 6%) (MfE 2011)

BaP equivalents to be determined for carcinogenic and potential genotoxic PAHs only using TEFs presented by CCME (2008)

Note: early lifetime exposures to BaP may need to be addressed in the quantification of exposure as per US EPA (2005).

1.5 Calculated HILs for BaP and Carcinogenic PAHs (as BaP TEF)

It is noted that the discussion above has identified that further consideration of early lifetime exposures to BaP may need to be considered in the quantification of exposure (calculated as per US EPA 2006). Other uncertainties have also been noted in the above discussion, particularly in relation to the selection of the oral TRV (where the value from MfE (2011) may also be considered, although it is a draft) and dermal exposures.

With respect to the derivation of HIL A, the following can be noted:

- HIL A = 20 mg/kg on the basis of the recommended oral TRV from MfE (2011) (also adopted for dermal exposures) and no additional consideration of early-lifetime exposures.
- HIL A = 8 mg/kg on the basis of the oral TRV from WHO (2011) (also adopted for dermal exposures) and no additional consideration of early-lifetime exposures.
- HIL A = 6 mg/kg on the basis of the recommended oral TRV from MfE (2011) (also adopted for dermal exposures) and consideration of early-lifetime exposures[1];
- HIL A = 3 mg/kg on the basis of the oral TRV from WHO (2011) and consideration of early-lifetime exposures[1].
- HIL A = 0.3 mg/kg on the basis of the recommended oral TRV from MfE (2011), but consideration of the dermal slope factor presented by Knafla et al. (2011) and no consideration of early lifetime exposures. Note that the HIL is lower (0.1 mg/kg) if early lifetime exposures are assessed for oral intakes.

[1] Based on guidance available from US EPA (2005), early lifetime exposures have been accounted for by the application of adjustment factors (ADAFs) to calculate the risk for different life stages: risk during the first 2 years of life (ADAF = 10); risk for ages 2 through to less than 16 years (ADAF = 3); and the risk for ages 16 through to 70 years (ADAF = 1). The total calculated risk for a lifetime is the sum of risk over all life stages.

With consideration of the uncertainties (particularly in relation to the assessment of dermal exposures) identified and the effect of these on the derived HIL A value (noted above), it is recommended that the lower value derived on the basis of the WHO (2011) oral TRV (also adopted for dermal exposures) with consideration of early-lifetime exposures (for HILs A, B and C only), that results in the calculation of HIL A = 3 mg/kg, be adopted.

It is noted that while the approach adopted for the derivation of the HILs has not directly incorporated the dermal approach outlined by Knafla et al. (2011), individual jurisdictions may require consideration of these issues in a site-specific assessment, particularly where people may come into direct contact with coal tar.

On this basis, the following HILs are recommended for BaP and carcinogenic PAHs (assessed as BaP TEF) (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL* (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	3	46	--	54	<1
Residential B	4	17	--	83	<1
Recreational C	3	29	--	71	<1
Commercial D	40	18	--	82	<1

-- Pathway not included in derivation of HIL

* Noted that as the dermal absorption pathway dominates the derivation of HILs A, B and C and the exposure assumptions differ little between these scenarios, the HIL remains essentially unchanged. Note derived HILs to 2 significant figures presented in brackets.

Elevated levels of BaP in relatively immobile sources, such as bitumen fragments, do not represent a significant health risk.

1.6 Calculated HILs for Total PAHs

The derived HILs above relate to BaP and carcinogenic PAHs calculated on the basis of a BaP TEF (refer to Section 2.2 of Schedule B(7)). However, there are several hundred PAHs, including derivatives of PAHs of which typically only 16 individual PAHs are analysed in site contamination investigations. These individual PAHs have been identified as the most significant based on: the amount of information available on each individual PAH; the toxicity (suspected to be more harmful than other PAHs), there is a greater chance of being exposed to these PAHs; and of all the PAHs analysed, the 16 selected are the most commonly reported at contaminated sites.

Hence to assist in the assessment of contaminated sites it is relevant to also consider total PAHs. Of the PAHs reported these will comprise BaP and carcinogenic PAHs and other non-carcinogenic PAHs where the following can be noted with respect to the derivation of HILs:

- BaP and carcinogenic PAHs assessed as BaP TEF should be assessed on the basis of the above HILs.
- Naphthalene is the most significant volatile PAH and therefore the assessment of this compound should address all significant pathways of concern, including vapour inhalation (not addressed in the HIL for total PAHs). The presence of this compound in soil should be assessed on the basis of relevant guidelines such as the Health Screening Levels (HSLs) (Friebel & Nadebaum 2011).
- The remaining PAHs are considered non-carcinogenic and include acenaphthene, acenaphthylene, anthracene, fluoranthene, fluorene, phenanthrene and pyrene. Rather than review the toxicity of each individual non-carcinogenic PAH, the published potencies to BaP (or TEFs) available for these PAHs (WHO 1998 and CCME 2008) suggest that individual non-carcinogenic PAHs are at least 100 to 1000 times less toxic/potent

than BaP. On this basis a factor of 100 has been applied to the calculated BaP HILs to establish HILs for total PAHs. Review of soil guidelines developed by US EPA (Regional Screening Levels, 2010) indicates that based on consideration of the same pathways of exposure (soil ingestion, dermal contact and inhalation of particulates), health-based guidelines for non-carcinogenic PAHs are at least 10,000 times higher than the BaP guideline. Hence the adoption of a factor of 100 as an additive total for other non-carcinogenic PAHs is considered reasonable.

- The HILs for total PAHs are only relevant provided carcinogenic PAHs meet the BaP HILs and naphthalene also meets the relevant HSLs.

On the basis of the above, the following HILs are recommended for total PAHs (provided carcinogenic PAHs meet the BAP HIL and naphthalene meets the relevant HSL):

HIL Scenario	HIL (mg/kg)
Residential A	300
Residential B	400
Recreational C	300
Commercial D	4000

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2 Phenol

2.1 General

Several comprehensive reviews of phenol in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 2008; WHO 1994; Health Canada, 2000; UK EA 2009). The following provides a summary of the key aspects of phenol that is relevant to the derivation of a soil HIL.

Phenol is a colourless to white to pale pink crystalline solid at room temperature and ambient pressure. Phenol has a distinctive aromatic, somewhat 'sickening', sweet and acrid odour. Phenol is soluble in water and miscible with most organic solvents (e.g. acetone and benzene) (ATSDR 2008). Many substituted phenols exist, for example dimethyl and trimethylphenols. These have different toxicities from phenol (ATSDR 2008). The widely varying toxicities and difficulty of making a generic assumption on the likely composition of phenol mixtures mean presenting an HIL representing 'total phenols' is considered impractical.

Therefore if substituted phenols may be present, these should be analysed and assessed as separate compounds, rather than on the basis of the phenol HIL.

Phenol can occur naturally in the environment as a product of organic matter decomposition and combustion of wood. Phenol is manufactured for use in phenolic resins, disinfectant and antiseptic and as an intermediate in organic synthesis (ATSDR 2008). Anthropogenic sources of phenol in the environment include vehicle exhaust and waste streams associated with its manufacture. Predominantly, phenol is released as an air emission resulting from venting. Phenol can also be released in the metabolic processes in which it occurs as an intermediate. For example, phenol can be produced from the degradation of organic wastes containing benzene, an organic compound found extensively in the environment. Its primary occurrence as a soil contaminant is in former gas works and coking works sites (ATSDR 2008).

2.2 Previous HIL

The derivation of the previous HIL (HIL A = 8500 mg/kg) for phenol is presented by Turczynowicz (1993) and NEPC (1999). In summary, the HIL was derived on the basis of the following:

- Background intakes were considered in the derivation of the previous HIL with the intakes from food, water and ambient air considered, where available. Due to the lack of available data, the quantification of intakes was limited, hence intakes from contaminated soil were taken to be 25% of the adopted ADI to address these limitations.
- An RfD of 0.6 mg/kg/day referenced from US EPA, based on a NOAEL of 60 mg/kg/day and uncertainty factor of 100 was considered.
- Dermal absorption of phenol was considered to be 12%.
- Oral bioavailability of phenol was considered to be 100%.

Based on intakes derived from soil (ingestion, dermal absorption and dust inhalation) an HIL of 8500 mg/kg was calculated.

2.3 Significance of Exposure Pathways

2.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of phenol in the range of contaminated sites that may need to be considered in Australia. On this basis, a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

2.3.2 Dermal absorption

ATSDR (2008) notes that phenol is readily absorbed through the skin, and the skin is considered the primary route of entry during occupational exposure (when considered as a product rather than in soil). Dermal absorption of phenol from soil has been shown and maximum phenol penetration was within 2 and 4 hours after application.

No compound-specific dermal absorption value is available for phenol and hence the default value of 0.1 (10%) for semi-volatile compounds available from US EPA (2004) has been adopted.

It is noted that phenol is a skin irritant and skin necrosis has been produced by contact with 1% solutions (UK EA 2009).

2.3.3 Inhalation of Dust

Phenol is not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

2.3.4 Plant Uptake

Phenols occur naturally in plants and soils. Since phenol and phenolics are relatively water-soluble, they are present in the soil solution and are easily taken up by plants via root absorption and stored in different parts of the plant (CCME 1999). Although it has been shown that plants readily take up phenol, bioaccumulation does not take place, due to a high rate of respiratory decomposition of phenol to CO_2. The potential for the uptake of phenol into home-grown produce has been considered in the derivation of HIL A. This has been undertaken on the basis of the equations presented in Appendix B with the following parameters and plant uptake factors estimated:

Parameter	Value	Reference/Comment
Parameters		
Koc	187 (cm^3/g)	RAIS (2010)
log Kow	1.46	RAIS (2010)
Diffusivity in water	$1.03x10^{-5}$ (cm^2/s)	RAIS (2010)
Calculated Plant Uptake Factors (mg/kg produce fresh weight per mg/kg soil)		
Green vegetables	0.204	calculated
Root vegetables	0.307	calculated
Tuber vegetables	0.244	calculated
Tree fruit	0.00098	calculated

It is noted that plants can metabolise phenol readily, hence exposure through eating food derived from plants grown in phenol-containing soil is probably minimal and the above is likely to be conservative.

2.3.5 Intakes from Other Sources – Background

Background intakes of phenol were estimated in the supporting documentation for the current HIL (Turczynowicz, 1993). Due to the lack of available data, the quantification of intakes was limited, hence intake from contaminated soil was taken to be 25% of the adopted ADI to address these limitations.

No data is available on potential intakes of phenol in Australia from food, water, consumer products and air. Estimates of background intakes by RIVM (2001) suggest intake may be dominated by inhalation exposures and background intakes may comprise 1 µg/kg/day. A more detailed review of background intakes by UK (UK EA 2009) considered intakes from food (dominated by the use of phenol as a flavouring additive), water (insignificant compared with food intakes), air and consumer products where the total intake was estimated to be approximately 390 µg/day (350 µg/day from oral

sources and 40 µg/day from inhalation sources) or 5.5 µg/kg/day for a 70 kg adult. These are higher than estimated by Health Canada (2000) where intakes by young children (0.5–4 years) were estimated to be 0.27–0.66 µg/kg/day; these are more consistent with intakes estimated by RIVM (2001).

If the more conservative estimates of background intakes available from the UK (UK EA 2009) were considered, for a child these would comprise approximately 10% of the recommended oral TRV and 25% of the recommended inhalation TRV. A conservative assumption that background intakes comprise approximately 30% (with rounding) of the TRV can be assumed.

2.4 Identification of Toxicity Reference Values

2.4.1 Classification

The International Agency for Research on Cancer (IARC 1999) has classified phenol as Group 3—not classifiable as to its carcinogenicity.

It is also noted that US EPA (last reviewed in 2002) has classified phenol as Group D—not classifiable as to its carcinogenicity.

2.4.2 Review of Available Values/Information

Notwithstanding the above, data on carcinogenicity of phenol is inconclusive. For example, RIVM (2001) report that studies in experimental animals suggest phenol can act as a tumour promoter. Further, ATSDR (2008) noted that 'under certain conditions, especially at high doses, phenol has the potential to be genotoxic. However at the exposure levels likely to occur near hazardous waste sites, phenol is not anticipated to be genotoxic.' Hence phenol (at least at concentrations expected at contaminated site) is not considered genotoxic. On the basis of the available information, it is considered appropriate that a threshold dose–response approach be adopted for phenol.

Few quantitative toxicity values are available; however the following threshold values are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG	No evaluation available	
OCS (2012)	No evaluation available	
International		
WHO (2011)	No evaluation available	
WHO (1994)	TDI = 0.06–0.2 mg/kg/day	Based on the range of NOAEL values associated with kidney and developmental effects in rats with the application of an uncertainty factor of 200 to get a range which is the recommended upper limit of the TDI. Some uncertainty is noted with respect to genotoxic potential and hence the evaluation provided is recommended to be periodically reviewed.
RIVM (2001)	TDI = 0.04 mg/kg/day TC = 0.02 mg/m^3	TDI based on a NOAEL of 40 mg/kg/day associated developmental effects in rats, and an uncertainty factor of 900 (and the TDI rounded). TC is provisional (due to the poor database) and based on a NOAEC of 20 mg/m^3 associated with adverse effects in

Source	Value	Basis/Comments
		various experimental animals after sub-chronic inhalation exposure, and an uncertainty factor of 1000.
Health Canada (2000)	TDI = 0.12 mg/kg/day	TDI based on review of the available database and consideration that developmental effects are the most sensitive end points (noting other end points have limited data). Value derived on the basis of a NOAEL of 12 mg/kg/day for kidney effects (noted to be lower than that from developmental effects) in rats, and an uncertainty factor of 100. Value derived is considered conservative.
EC (2006)	No ADI/TDI derived	No ADI/TDI derived however critical data points were identified for systemic toxicity where an oral LOAEL of 1.8 mg/kg/day (based on reduced blood cell count in mice), inhalation LOAEL of 21 mg/m^3 (based on possible liver injury in exposed workers) and a dermal NOAEL of 1.18% (equivalent to 130 mg/kg/day) were identified. A NOAEL for developmental toxicity of 93 mg/kg/day was identified from a 2-generation rat study.
UK (UK EA 2009)	TDI = 0.7 mg/kg/day TC = 0.035 mg/m^3	TDI based on review of current studies and evaluations. The TDI is based on a NOAEL of 70 mg/kg/day associated with a 2-generation drinking water rat study, and an uncertainty factor of 100. The study chosen is considered more appropriate that that considered by US EPA, WHO and RIVM as it was of longer duration and associated with drinking water administration (note that phenol exhibited a higher degree of toxicity when given by stomach tube/gavage than when administered via drinking water). Inhalation value derived on the basis of a LOAEL of 21 mg/m^3 (same as identified by EC 2006) associated with potential liver effects in occupationally exposed workersand an uncertainty factor of 600. It is note that the review undertaken considers that the critical effect associated with inhalation exposures to phenol is likely to be its mutagenic potential, and a non-threshold approach may be appropriate, however no evaluations are available. Also noted that despite significant limitations in the available data, it appears that phenol has more toxicity potential via inhalation than when ingested.
ATSDR (2008)	No chronic MRL derived	Oral MRL based on a LOAEL of 1 mg/kg/day associated thyroid effects in mink, and an uncertainty factor of 1000 (same study as considered by RIVM).
US EPA (2002)	RfD = 0.3 mg/kg/day	RfD (last reviewed in 2002) based on a benchmark dose approach where a BMDL of 93 mg/kg/day associated with decreased maternal weight gain in a short duration developmental rat study was derived, and an uncertainty factor of 300 considered. The previous evaluation by the US EPA considered an oral RfD of 0.6 mg/kg/day, adopted in the derivation of the current HIL (Turczynowicz 1993).

While a number of limitations have been identified by the UK review of the available data with respect to the quantification of phenol toxicity (UK EA 2009), the oral value recommended is based on the most recent review where a number of the database deficiencies have been more fully reviewed. This value has been adopted in the derivation of soil HILs.

Few inhalation values are available, and hence the threshold value derived by the UK (UK EA 2009) is recommended as it is based on a more recent review. As inhalation exposures appear to be more toxic than oral exposures the consideration of separate toxicity values for oral and inhalation routes of exposure (even if the inhalation route of exposure is not as significant for the characterisation of contaminated soil issues) is appropriate.

2.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for phenol in the derivation of HILs:

Recommendation for Phenol

Oral TRV (TRV_O) = 0.7 mg/kg/day (UK EA 2009) relevant to oral and dermal routes of exposure

Dermal absorption factor (DAF) = 0.1 (or 10%) (US EPA 2004)

Inhalation TRV (TRV_I) = 0.035 mg/m^3 (UK EA 2009) relevant to inhalation routes of exposure

Background intakes from other sources (as % of TRV):

BI_O = 70% for oral and dermal intakes

BIi = 70% for inhalation

Uptake in home-grown produce considered in derivation of HIL A.

2.5 Calculated HILs

On the basis of the above the following HILs have been derived for phenol (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	3000	4	91	5	<1
Residential B	45 000	15	--	83	2
Recreational C	40 000	27	--	73	<1
Commercial D	240 000	11	--	87	2

-- Pathway not included in derivation of HIL

2.6 References

ATSDR 2008, *Toxicological profile for phenol,* US Department of Health and Human Services, Agency for Toxic Substances and Disease Registry, Atlanta, GA, available at: http://www.atsdr.cdc.gov/toxprofiles/tp115.pdf.

CCME 1999, *Phenol. Canadian Soil Quality Guidelines for the Protection of Environmental and Human Health*, Canadian Council of Ministers for the Environment.

EC 2006. *European Union Risk Assessment Report: phenol. CAS No: 108-95-2, EINECS No: 203-632-7.* 1st Priority List, Volume 64. EUR 22522 EN/1, Revised edn, European Commission, Brussels, available at: http://ecb.jrc.it/Documents/Existing-Chemicals/RISK_ASSESSMENT/REPORT/phenolreport060.pdf.

IARC 1999, *Re-evaluation of some organic chemicals, hydrazine and hydrogen peroxide. Phenol. IARC Monographs on the Evaluation of Carcinogenic Risks to Humans,* vol. 71 (Part 2), International Agency for Research on Cancer, Lyon, France.

Health Canada 2000, *Priority Substances List Assessment Report, Phenol*, Health Canada.

NEPC 1999, *Schedule B (7a), Guideline on Health-Based Investigation Levels, National Environment Protection (Assessment of Site Contamination) Measure*, National Environment Protection Council.

RAIS 2010, Risk Information Assessment System, database maintained by the Oak Ridge Operations Office, available from http:rais.ornl.gov/.

RIVM 2001, *Re-evaluation of human-toxicological Maximum Permissible Risk levels*, National Institute of Public Health and the Environment, Bilthoven, Netherlands, available from: http://www.rivm.nl/bibliotheek/rapporten/711701025.html.

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UK EA 2009, *Contaminants in soil: updated collation of toxicological data and intake values for humans, Phenol*, Science report: Sc050021/TOX 9, UK Environment Agency, Bristol, UK.

US EPA 2002, *Toxicological review of Phenol (CAS No. 108-95-2)*, in support of summary information on the Integrated Risk Information System (IRIS), (Appendix A to US EPA, 2002a), US Environmental Protection Agency, Washington, DC, available at: http://www.epa.gov/iris/toxreviews/0088-tr.pdf#page=132.

US EPA 2004, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment), Final,* EPA/540/R-99/005, OSWER 9285.7-02EP.

WHO 1994, *Environmental Health Criteria Document 161. Phenol,* International Programme on Chemical Safety, Geneva, available at: http://www.inchem.org/documents/ehc/ehc/ehc161.htm.

3 Pentachlorophenol (PCP)

3.1 General

Several comprehensive reviews of pentachlorophenol (PCP) in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 2001; WHO 1987). The following provides a summary of the key aspects of PCP that are relevant to the derivation of a soil HIL.

Pure pentachlorophenol is a colourless, white or light tan crystalline solid (WHO 1987; ATSDR 2001). It has a characteristic phenolic odour at high temperatures but it is relatively odourless at room temperature. Pentachlorophenol is moderately volatile at ambient temperature and insoluble in water (WHO 1987; ATSDR 2001). Technical grade pentachlorophenol is typically 86% pure and is dark grey to brown in colour as a result of the polychlorinated phenol impurities. It is typically manufactured in the form of dust, beads or flakes (ATSDR 2001).

Pentachlorophenol is an effective biocide and had wide applications in the commercial and agricultural industries as an insecticide (termiticide), fungicide, herbicide, molluscicide and algicide. The primary use of the compound was for wood preservation. In the United States, the use of wood products treated with pentachlorophenol in domestic settings was banned but the compound is still used to preserve power line poles, railroad sleepers, wharf pilings, cross arms and fence posts (ATSDR 2001). Pentachlorophenol was also historically used as a disinfectant, as an ingredient in antifouling paint, as an insecticide or herbicide in domestic environments, in the textile industry, leather industry, in mineral oil and in glue (WHO 1987; ATSDR 2001).

Pentachlorophenol is no longer registered as the active ingredient in any chemical in Australia.

Review of the toxicity of PCP is complicated by the relatively large database on the toxicity of technical-grade PCP and the comparatively small database on pure PCP. Technical-grade PCP has been shown to contain a large number of impurities, including tetrachlorophenols and, to a much lesser extent, polychloro-dibenzodioxins, polychlorodibenzofurans, polychlorodiphenyl ethers, polychloro-phenoxy phenols and chlorinated hydrocarbons. These impurities, in particular the polychloro-dibenzodioxins and furans, are indicated to be responsible for at least some of the observed toxicity of the technical-grade PCP (MfE 2011). Notwithstanding, specific haematopoietic cancer risks are observed with PCP exposure and which are not likely to be due to dioxins or other chlorophenol contaminants (Cooper & Jones 2008).

3.2 Previous HIL

No previous HIL has been derived for PCP (NEPC 1999).

3.3 Significance of Exposure Pathways

3.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of PCP in the range of contaminated sites that may need to be considered in Australia. On this basis, a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

3.3.2 Dermal absorption

PCP is rapidly absorbed across the skin, and therefore dermal exposure potentially represents a significant route of exposure. The US EPA (2004) has identified a dermal absorption fraction of 0.25 (25%), based on a study by Wester et al. (1993) for PCP in soil. The study found that in vivo

absorption in monkeys of PCP in soil was similar to PCP in acetone, with 24% of PCP absorbed over a 24-hour period.

Few other studies are available with quantitative values and hence the dermal absorption value of 0.24 (24%) from Wester et al. (1993) has been used in the derivation of HILs for PCP.

3.3.3 Inhalation of Dust

PCP is not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

3.3.4 Plant Uptake

In a review paper, McAllister et al. (1996) reported that available data on the plant uptake and transformation of PCP is inconsistent among studies and is inconclusive with regard to the abilities of specific plants to take up the compound. It was observed that the biodegradation of PCP by microorganisms and its adsorption to soil limit the availability of the compound for plant uptake (ATSDR 2001).

Further review by MfE (2011) considered that plant uptake of PCP is not a significant pathway of exposure given that PCP is known to be metabolised by plants (resulting in an over-prediction of plant uptake by the models available), bioconcentration factors relevant to plant uptake are low, and recent papers relating to PCP and plants where uptake is noted are associated with phytoremediation through enhanced microbial activity at plant roots.

On the basis of the above, plant uptake of PCP is not considered significant. In addition, the application of general plant uptake equations is not considered appropriate.

3.3.5 Intakes from Other Sources – Background

Limited information is available on background exposures to PCP by the general population (PCP intakes have not been addressed in the Australian Total Diet Surveys). PCP is no longer used in Australia and while it is persistent, background levels are expected to be low. Dietary intakes are expected to be the most significant background source (ATSDR 2001). Total intakes of PCP (dominated by food intakes) have been estimated to be between 0.1 and 6 µg/day (equal to 1.4–80 ng/kg/day) (WHO 1987) and 5–35 µg/day (70–500 ng/kg/day) (WHO 2011), though these estimates are based on older data.

ATSDR (2001) notes that intakes estimated from a US total diet survey (1982–1984) suggested intakes for 2-year-old children were up to 48.5 ng/kg/day (about 0.6 µg/day). Estimates from a later total diet survey (1986–1991) suggested lower intakes by children aged 2 years of 1.4 ng/kg/day (about 20 ng/day). Intakes from the later study are consistent with background intakes estimated by RIVM (2001). These intakes are essentially negligible compared with the recommended oral TRV. Hence intakes from other sources have been considered to be negligible.

3.4 Identification of Toxicity Reference Values

3.4.1 Classification

The International Agency for Research on Cancer (IARC 1991) has classified PCP as Group 2B—possibly carcinogenic to humans.

It is also noted that US EPA has classified PCP as Group B2—probable human carcinogen.

3.4.2 Review of Available Values/Information

Studies on experimental animals have shown some carcinogenic potential associated with oral exposures to technical grade and mixtures of PCP. However PCP has not demonstrated genotoxicity in in vitro and in vivo test systems and in occupationally exposed humans (RIVM 2001 and NHMRC 2010). Review by ATSDR (2001) and IARC (1991) suggests PCP may exhibit weak clastogenic effects.

Review by MfE (2011) suggested that the data on the genotoxicity of PCP is equivocal, with the strongest indication of genotoxicity (chromosomal effects) occurring in assays with rat microsomal protein (S9). The primary rodent metabolite, tetrahydrochloroquinone (TeHQ), is unambiguously genotoxic. TeHQ does not appear to be a major metabolite of PCP in humans. Furthermore, the majority of PCP appears to be excreted unchanged (ATSDR 2001).

On the basis of the available information, it is considered appropriate that a threshold dose–response approach be adopted for PCP.

Few quantitative toxicity values are available; however the following threshold values are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	TDI = 0.003 mg/kg/day	The current ADWG (NHMRC 2011) has derived a health-based guideline of 0.01 mg/L, based on a TDI of 0.003 mg/kg/day, noted to be based on a NOEL of 3 mg/kg/day from a 2-year rat study, and an uncertainty factor of 1000 (10 for interspecies extrapolation, 10 for intraspecies variability extrapolation and an additional safety factor of 10 due to the limitations of the toxicological data available at the time the ADI was set).
OCS (2012)	No evaluation available	
International		
WHO (2011)	No threshold value set	The current WHO DWG (2011) has derived a provisional guideline of 0.009 mg/L based on a US NTP study and a linear multistage model associated with tumour increases and an excess lifetime risk of 10^{-5} (review unchanged since 1993). It is noted that pentachlorophenol is included in the rolling revisions to the DWG, with no revisions currently available.
WHO (1987)	ADI = 0.003 mg/kg/day	References an ADI derived by the National Academy of Sciences which is based on a NOEL of 3 mg/kg/day from a long-term feeding study in rats, and an uncertainty factor of 1000 (same study as considered in the ADWG).
RIVM (2001)	TDI = 0.003 mg/kg/day	TDI based on a LOAEL of 1 mg/kg/day associated with thyroid effects in mink, and an uncertainty factor of 300.
ATSDR (2001)	MRL = 0.001 mg/kg/day	Oral MRL based on a LOAEL of 1 mg/kg/day associated with thyroid effects in mink, and an uncertainty factor of 1000 (same study as considered by RIVM.

Source	Value	Basis/Comments
US EPA (IRIS 2012)	RfD = 0.005 mg/kg/day	RfD (reviewed in 2010) based on a LOAEL of 1.5 mg/kg/day associated hepatotoxicity, including dose-related increases in incidence and severity of hepatocellular pigmentation, cytoplasmic vacuolation, and chronic inflammation, and significant increases in relative liver weight and increases in absolute liver weight (significant in females), observed in a chronic oral study in dogs (Mecler 1996). An uncertainty factor of 300 was applied. US EPA has also derived a non-threshold oral slope factor not considered relevant here.

While different key studies were considered by the various agencies noted above, use of these studies has largely resulted in the derivation of oral toxicity reference values that are essentially the same (ranging from 0.001 to 0.005 mg/kg/day). Hence the threshold reference value adopted in the ADWG (NHMRC 2011), which is consistent with that derived by all other agencies, including ATSDR, US EPA and RIVM, is recommended.

No dermal or inhalation specific studies or data are available. For the presence of PCP in soil it is considered appropriate to consider use of the available TDI for all pathways of exposures.

3.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for PCP in the derivation of HILs:

Recommendation for Pentachlorophenol

Oral TRV (TRV_O) = 0.003 mg/kg/day (NHMRC 2011) relevant to all pathways of exposure

Dermal absorption factor (DAF) = 0.24 (or 24%) (Wester et al. 1993)

Background intakes from other sources (as % of TRV):

BI_O = 0% for oral and dermal intakes

BIi = 0% for inhalation

3.5 Calculated HILs

On the basis of the above, the following HILs have been derived for PCP (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	100	24	--	76	<1
Residential B	130	7	--	93	<1
Recreational C	120	13	--	87	<1
Commercial D	660	5	--	95	<1

-- Pathway not included in derivation of HIL

3.6 References

ATSDR 2001, *Toxicological profile for Pentachlorophenol*, Agency for Toxic Substances and Disease Registry, US Department of Health and Human Services, Atlanta, Georgia, USA.

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NEPC 1999, *Schedule B (7a), Guideline on Health-Based Investigation Levels, National Environment Protection (Assessment of Site Contamination) Measure*, National Environment Protection Council, Australia.

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NHMRC 2011, *National water quality management strategy. Australian drinking water guidelines*, National Health and Medical Research Council, Australia.

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US EPA (IRIS 2012), Data and information available from the *Integrated Risk Information System*, an online database, available from http://www.epa.gov/iris/.

US EPA 2004, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment), Final*, EPA/540/R-99/005, OSWER 9285.7-02EP.

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WHO 1987, *Environmental Health Criteria No 71, Pentachlorophenol*. World Health Organisation, Geneva.

WHO 2011, *Guidelines for drinking-water quality, 4th edition*, World Health Organisation, Geneva, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html.

4 Total Cresols

4.1 General

Several comprehensive reviews of cresols in the environment and their toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 2008; WHO 1995). The following provides a summary of the key aspects of cresols that are relevant to the derivation of a soil HIL.

Cresols are a group of isomers comprising a single benzene ring, a hydroxyl group and a methyl group (C_7H_8O). There are three structural isomers, including m-cresol (2-methylphenol), p-cresol (3-methylphenol), and o-cresol (4-methylphenol). These isomers may occur separately or as a mixture (ATSDR 2008). In their pure form, cresols are colourless solids, while mixtures are more commonly liquids. Cresols are semi-volatile compounds with moderate solubility in water and a medicinal-type odour (ATSDR 2008). The abundance of p-cresols in the environment is significantly greater than that of the alternative isomers, as is the abundance of o-cresol relative to that of m-cresols. However, there is a greater amount of information and studies surrounding the health effects associated with m- and o-cresols. It should be noted that the behaviour of all three isomers in the environment is considered to be similar.

Cresols are both a naturally occurring and manufactured group of chemicals that may be used as solvents, disinfectants, deodorisers, wood preservatives and to make other chemicals (ATSDR 2008). O-cresol is used in the manufacture of several dye intermediates (ATSDR 2008). P-cresol is predominantly used in the manufacture of anti-oxidants, synthetic food flavours and fragrances, and m-cresol is used in the synthesis of many herbicides and insecticides (ATSDR 2008). Cresols occur in various plant oils including peppermint, sandalwood, jasmine, Easter lily, ylang ylang, eucalyptus and camphor.

4.2 Previous HIL

No previous HIL is available for cresols (NEPC 1999).

4.3 Significance of Exposure Pathways

4.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of cresols in the range of contaminated sites that may need to be considered in Australia. On this basis, a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

4.3.2 Dermal absorption

Insufficient data is available on the dermal absorption of cresols from soil. Hence the default values of 0.1 (10%) suggested by US EPA (2004) for semi-volatiles has been adopted in the derivation of HILs.

4.3.3 Inhalation of Dust

Cresols are not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

4.3.4 Plant Uptake

No data is available on the potential for the uptake of cresols into edible fruit and vegetable crops. Limited data is also available on the potential or cresols to bioaccumulate. Cresols are soluble in water

and, based on Koc values referenced by OECD SIDS (2003), there is a low sorption potential for cresols. Hence, while specific data is lacking, there is the potential for cresols to be available in soil water to be taken up by plants.

Hence a conservative approach has been taken to consider the potential for the uptake of cresols into home-grown produce in the derivation of HIL A. This has been undertaken on the basis of the equations presented in Appendix B, with the following parameters and plant uptake factors estimated:

Parameter	Value	Reference/Comment
Parameters		
Koc	307 (cm^3/g)	RAIS (2010)
log Kow	1.95	RAIS (2010)
Diffusivity in water	$9.78x10^{-6}$ (cm^2/s)	RAIS (2010)
Calculated Plant Uptake Factors (mg/kg produce fresh weight per mg/kg soil)		
Green vegetables	0.18	calculated
Root vegetables	0.255	calculated
Tuber vegetables	0.152	calculated
Tree fruit	0.00044	calculated

4.3.5 Intakes from Other Sources – Background

Limited information is available on background exposures to cresols by the general population. Available reviews by ATSDR (2008), OECD SIDS (2003) and RIVM (2001) have not been able to quantify background intakes due to a lack of data. As data is lacking for background intakes of cresols, an estimate or default value can be assumed. Cresols are expected to be widely present in the environment and hence a value of 50% may be relevant where data are not available.

4.4 Identification of Toxicity Reference Values

4.4.1 Classification

The International Agency for Research on Cancer (IARC) has not classified cresol with respect to human carcinogenicity.
US EPA has classified cresols as Group C–possible human carcinogen.

4.4.2 Review of Available Values/Information

There is no adequate data available to assess carcinogenicity of cresols. One study suggests cresols may promote skin tumours. Genotoxicity of cresols has been evaluated (ATSDR 2008) and the weight of evidence suggests that ‘cresols do not pose a genotoxic threat to humans under normal environmental exposure conditions’. On the basis of the available information, it is considered appropriate that a threshold dose-response approach be adopted for cresols.

Few quantitative toxicity values are available, however the following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG	No evaluation available	
OCS (2012)	No evaluation available	
International		
WHO	ADI = 0.17	ADI derived by WHO (1995) on a NOAEL of 50

Source	Value	Basis/Comments
(1995)	mg/kg/day	mg/kg/day from a sub-chronic study and a 300-fold uncertainty factor (which included an additional 10 fold factor to address the lack of chronic studies and possible genotoxic and promoting activity).
RIVM (2001)	TDI = 0.05 mg/kg/day TC = 0.17 mg/m^3	TDI based on a 90-day sub-chronic oral study. TC based on route extrapolation from oral data.
OEHHA (2009)	REL = 0.6 mg/m^3	Chronic REL based on route extrapolation of the LOAEL and NOAEL derived from the study used to derive the current US EPA RfD for 2- and 3-methylphenol.
ATSDR (2008)	MRL = 0.1 mg/kg/day	Oral MRL based on a LOAEL associated with increased incidences of bronchiole hyperplasia of the lung and follicular degeneration of the thyroid gland from a 2-year dietary study in female mice (NTP 2008).
US EPA (IRIS 2012)	RfD = 0.05 mg/kg/day	RfD (last reviewed in 1988) derived for 2- and 3-methylphenol based on decreased body weights and neurotoxicity in a 90-day sub-chronic study in rats.

The threshold value derived by ATSDR (2008) is based on a chronic study not available at the time when the WHO (1995), RIVM (2001) or US EPA conducted their review (where threshold values were derived on the basis of sub-chronic studies). On this basis, the oral value (taken as an ADI) available from ATSDR (2008) is considered the most current and robust value for deriving a soil HIL.

No dermal or inhalation specific studies or data are available. For the presence of cresols in soil, it is considered appropriate to consider use of the available ADI for all pathways of exposures.

4.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for cresols (as sum of all isomers) in the derivation of HILs:

Recommendation for Cresols

Oral TRV (TRV_O) = 0.1 mg/kg/day (ATSDR 2008) relevant to all pathways of exposure

Dermal absorption factor (DAF) = 0.1 (or 10%) (US EPA 2004)

Background intakes from other sources (as % of TRV):

BI_O = 50% for oral and dermal intakes

BIi = 50% for inhalation

Uptake in home-grown produce considered in derivation of HIL A.

4.5 Calculated HILs

On the basis of the above, the following HILs have been derived for cresols (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	400	5	89	6	<1
Residential B	4700	16	--	84	<1

Recreational C	4000	27	--	73	<1
Commercial D	25 000	12	--	88	<1

-- Pathway not included in derivation of HIL

4.6 References

ATSDR 2008, *Toxicological profile for Cresols*, Agency for Toxic Substances and Disease Registry U.S. Department of Health and Human Services, Atlanta, Georgia, USA, available from http://www.atsdr.cdc.gov/ToxProfiles/tp.asp?id=946&tid=196.

NEPC 1999, *Schedule B (7a), Guideline on Health-Based Investigation Levels, National Environment Protection (Assessment of Site Contamination) Measure*, National Environment Protection Council, Australia.

NHMRC 2011, *National water quality management strategy, Australian drinking water guidelines*, National Health and Medical Research Council, Australia.

NTP 2008, *Toxicology and carcinogenesis studies of cresols (CAS No. 1319-77-3) in male F344/N rats and female B6C3F1 mice (feed studies)*, National Toxicology Program, TR-550, Draft technical report, Research Triangle Park, NC.,

OECD SIDS 2003, *M/P-Cresol Category. SIDS Initial Assessment Report,* for SIAM 16.

OEHHA 2009, *Air Toxicology and Epidemiology, summary of current Reference Exposure Levels and Cancer Potency Factors*, available from Office of Environmental Health Hazard Assessment (OEHHA), http://www.oehha.org/air/toxic_contaminants/index.html.

RAIS (n.d.), *Risk Assessment Information System*, website and database maintained by the Oak Ridge Operations Office, available from: http://rais.ornl.gov/.

RIVM 2001, *Re-evaluation of human-toxicological Maximum Permissible Risk levels*, National Institute of Public Health and the Environment, Bilthoven, Netherlands, .available from: http://www.rivm.nl/bibliotheek/rapporten/711701025.html.

US EPA (IRIS 2012), Data and information available from the *Integrated Risk Information System*, an online database, available from http://www.epa.gov/iris/.

US EPA 2004, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment), Final*, EPA/540/R-99/005, OSWER 9285.7-02EP.

WHO 1995, *Environmental Health Criteria 168, Cresols*. International Programme on Chemical Safety, World Health Organization, Geneva.

5 Shortened forms

ADI	acceptable daily intake
ADAF	adjustment factor
ADWG	Australian Drinking Water Guidelines
AI	adequate intake
ANZECC	Australia and New Zealand Environment and Conservation Council
APVMA	Australian Pesticides and Veterinary Medicines Authority
ATDS	Australian Total Diet Survey
ATSDR	Agency for Toxic Substances and Disease Registry
BA	bioavailability
BAP	benzo(a)pyrene
BI	background intake
BMD	benchmark dose
BMDL	Benchmark dose lower confidence limit
CCME	Canadian Council of Ministers of the Environment
CICAD	Concise International Chemicals Assessment Document
CNS	central nervous system
DAF	dermal absorption factor
DW	dry weight
DWG	drinking water guidelines
EA	Environment Agency (England and Wales)
EHC	Environmental Health Criteria
EPA	Environment Protection Authority
FSANZ	Food Standards Australia New Zealand
GAF	gastrointestinal absorption factor
HEC	human equivalent concentration
HED	human equivalent dose
HIARC	Hazard Identification Assessment Review Committee
HIL	health investigation level
HSDB	Hazardous Substances Data Bank
HSL	health screening level

IARC	International Agency for Research on Cancer
IEUBK	Integrated exposure uptake biokinetic model
IRIS	Integrated Risk Information System
JECFA	Joint FAO/WHO Expert Committee on Food Additives
JMPR	WHO/FAO Joint Meeting on Pesticide Residues
LOAEL	lowest observed adverse effect level
LOEL	lowest observed effect level
MF	modifying factor
MOA	mode (or mechanism) of action
MRL	minimal risk level
NEPC	National Environment Protection Council
NEPM	National Environment Protection Measure
NHMRC	National Health and Medical Research Council
NOAEL	no observable adverse effect level
NOEL	no observable effect level
NSW DECC	New South Wales Department of Environment and Climate Change
OCS	Office of Chemical Safety
OEHHA	Office of Environmental Health Hazard Assessment
PAH	polycyclic aromatic hydrocarbon
PCP	pentachlorophenol
PTDI	provisional tolerable daily intake
PTMI	provisional tolerable monthly intake
PTWI	provisional tolerable weekly intake
RAIS	Risk Assessment Information System
RDI	recommended daily intake
REL	reference exposure level
RfC	reference concentration
RfD	reference dose
RME	reasonable maximum exposure
SF	slope factor
TC	tolerable concentration

TDI	tolerable daily intake
TEF	toxicity equivalence factor
TRV	toxicity reference value
UF	uncertainty factor
UL	upper limit
UR	unit risk
US EPA	United States Environmental Protection Agency
WHO	World Health Organization
WHO DWG	World Health Organization Drinking Water Guidelines

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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This compilation has been split into 22 volumes

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Prepared by the Office of Parliamentary Counsel, Canberra

Volume 10: Schedule B7 - Appendix 1
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About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

Schedule B7

APPENDIX A4

The Derivation of HILs for Herbicides and Other Pesticides

Contents
The derivation of HILs for herbicides and other pesticides

1 2,4,5-T

1.1 General

2,4,5-T is the common name for 2,4,5-trichlorophenoxyacetic acid (or 2,4,5-triphenoxyacetic acid), a chlorophenoxy herbicide.

Several comprehensive reviews of 2,4,5-T in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (OCS 2004; HSDB 2010). The following provides a summary of the key aspects of 2,4,5-T that are relevant to the derivation of a soil HIL.

The herbicide was also commercially produced as an amine salt, alkali metal salt and ester derivative of 2,4,5-T. Pure 2,4,5-T is a white to light tan solid. It is slightly soluble in water whereas the amine and alkali metal salt derivatives are highly soluble. The ester, however, is insoluble in water. 2,3,7,8-tetrachlorodibenzo*p*-dioxin (TCDD), a known human carcinogen, was a common contaminant in the manufacture of 2,4,5-T and its derivatives and was typically present in the low mg/kg to high mg/kg level (OCS 2004). 2,4,5-T with TCDD contamination is now controlled in international trade through the 'Rotterdam Convention' (Joint FAO/UNEP 2005). It is noted that 2,4,5-T is not expected to persist in the environment for any significant period of time but TCDD will remain and should be considered in a site-specific assessment where a 2,4,5-T source may have been present.

2,4,5-T and its derivatives were introduced in the 1960s and were used as herbicides for broad-leaved wood plants such as blackberries. 2,4,5-T was also combined with the compound 2,4-D to form the 'agent orange' herbicide which was widely used by the US military in the Vietnam war (OCS 2004). 2,4,5-T and its derivatives were withdrawn from use in the late 1980s and are no longer approved for use or marketed in Australia.

1.2 Previous HIL

No previous HIL is available for 2,4,5-T (NEPC 1999).

1.3 Significance of Exposure Pathways

1.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of 2,4,5-T, hence a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

1.3.2 Dermal absorption

Insufficient data is available on the dermal absorption of 2,4,5-T from soil. Hence the default value of 0.1 (10%) suggested by US EPA (1995) for pesticides has been adopted in the derivation of HILs.

1.3.3 Inhalation of Dust

2,4,5-T is not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

1.3.4 Plant Uptake

Most chlorophenoxy herbicides are toxic to plants and, as such, will be phytotoxic to almost all broadleaf crops including tomatoes, grapes and fruit trees, well before plant uptake into edible portions

of fruit and vegetable crops is of significance. Hence the uptake of these compounds into home-grown produce has not been considered in the derivation of HIL A.

Note that the phytotoxic effects of these compounds may need to be addressed on a site-specific basis if detected in soil.

1.3.5 Intakes from Other Sources – Background

Review of available publications suggests that very little data is available for Australia. Based on the available information on 2,4,5-T and 2,4-D in the environment, it is likely that background intakes by the general public will be similar to those considered for 2,4-D, which can be considered to be essentially negligible (0%).

1.4 Identification of Toxicity Reference Values

1.4.1 Classification

The International Agency for Research on Cancer (IARC 1987) has classified chlorophenoxy herbicides as Group 2B—possibly carcinogenic to humans.

US EPA has not classified 2,4,5-T.

1.4.2 Review of Available Values/Information

Limited data is available on the assessment of carcinogenicity and genotoxicity for 2,4,5-T. Available information on 2,4,5-T is often confounded with the presence of dioxin (TCDD) which was a common contaminant in 2,4,5-T herbicides. 2,4,5-T alone has not been found to be carcinogenic (Joint FAO/UNEP 2005).

On the basis of the available information, it is considered appropriate that a threshold dose-response approach be adopted for 2,4-5-T. The following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	TDI = 0.03 mg/kg/day	Current drinking water guideline of 0.1 mg/L based on 10% intake from drinking water. Based on equations presented in the ADWG (NHMRC 2011), the TDI considered in this derivation is equal to 0.029 mg/kg/day, essentially equivalent to the ADI available from the Joint FAO/WHO. No further information on the basis for this value is available.
OCS (2012)	Deleted from current list in 2003. Prior to this, the ADI was listed as 0.03 mg/kg/day.	Previous ADI referenced from Joint FAO/WHO evaluation from 1981.
International		
WHO (1981)	Temporary ADI of 0-0.03 mg/kg/day	Temporary ADI based on a NOEL of 3 mg/kg/day from a rat carcinogenicity study with 2,4,5-T containing 0.05 ppm TCDD.

Source	Value	Basis/Comments
WHO (2011)	TDI = 0.003 mg/kg/day	2,4,5-T has been reviewed in the WHO DWG (originally reviewed and established in 1996,) with a TDI of 0.003 mg/kg/day derived based on a NOAEL for reduced body weight gain, increased liver and kidney weights and renal toxicity in a 2 -year rat study. The same NOAEL was derived for reproductive effects from a three-generation rat study. It is noted that the derivation of the TDI included an additional 10 fold factor to address a suggested association between 2,4,5-T and soft-tissue sarcoma and non-Hodgkin lymphoma (not noted in other reviews available). 2,4,5-T is included in the WHO plan for rolling revisions to the drinking water guidelines. No reviews with respect to this chemical are currently available.
ATSDR	No evaluation available	
US EPA (IRIS 2012)	RfD = 0.01 mg/kg/day	The US EPA evaluation was established in 1982 and last reviewed in 1988 and provides an oral RfD of 0.01 mg/kg/day, based on a NOAEL of 3 mg/kg/day based on kidney effects in rats, and a 300-fold uncertainty factor. The value derived is considered protective of reproductive end points.

The available information from all the above sources is dated. There are some issues with the temporary ADI derived by the Joint FAO/WHO (1981) in that the study considered for the derivation of the ADI included the dioxin (TCDD) contaminant and addressed an end point not associated with 2,4,5-T alone. This value has subsequently been adopted in the derivation of the current ADWG without further review.

The value has been deleted from the current ADI list (OCS 2012). The TDI available in the current WHO DWG (2011) is based on the same studies as considered in 1981, though an additional uncertainty factor has been incorporated to address uncertainties in the database, including potential carcinogenic effects. The basis for this additional factor is not clear, as the carcinogenic effects noted have not been identified in other studies. On this basis, the most appropriate threshold reference value for 2,4,5-T is from US EPA, which is similar to the previous ADI from WHO (and is considered in the current ADWG (NHMRC 2011)).

No dermal or inhalation specific studies or data are available. For the presence of 2,4,5-T in soil, it is considered appropriate to consider use of the available US EPA RfD as a TRV for all pathways of exposures.

1.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for 2,4,5-T in the derivation of HILs:

Recommendation for 2,4,5-T

Oral TRV (TRV_O) = 0.01 mg/kg/day (RfD from US EPA (IRIS 2012) relevant to all pathways of exposure

Dermal absorption factor (DAF) = 0.1 (or 10%) (US EPA 1995)

Background intakes from other sources (as % of TRV):

BI_O = 0% for oral and dermal intakes

BIi = 0% for inhalation

1.5 Calculated HILs

On the basis of the above, the following HILs have been derived for 2,4,5-T (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	600	43	--	57	<1
Residential B	900	16	--	84	<1
Recreational C	800	27	--	73	<1
Commercial D	5000	12	--	88	<1

-- Pathway not included in derivation of HIL

1.6 References

HSDB 2010, *Hazardous Substances Data Bank*, online database available from: http://toxnet.nlm.nih.gov/cgi-bin/sis/htmlgen?HSDB.

IARC 1987, *Summaries and Evaluations, Chlorophenoxy herbicides,* Supplement 7: (1987), p.256, International Agency for Research on Cancer.

NEPC 1999, *Schedule B (7a), Guideline on Health-Based Investigation Levels, National Environment Protection (Assessment of Site Contamination) Measure,* National Environment Protection Council, Australia.

NHMRC 2011, *National water quality management strategy, Australian drinking water guidelines*, National Health and Medical Research Council, Australia.

OCS 2004, *Human Health Risk Assessment of Dioxins in Australia, National Dioxins Program, Technical Report No. 12*, Department of the Environment and Heritage, Australian Government, Canberra, Australia.

OCS 2012, *ADI List, Acceptable Daily Intakes for Agricultural and Veterinary Chemicals*, current to 31 March 2012, Australian Government, Department of Health and Ageing, Office of Chemical Safety (OCS), available from: http://www.health.gov.au/internet/main/publishing.nsf/content/E8F4D2F95D616584CA2573D700770C2A/$File/ADI-apr12.pdf.

Joint FAO/UNEP 2005, Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade, 2005, *Decision Guidance Document: 2,4,5-T and its Salts and Esters*, Joint Food and Agriculture Organisation of the United Nations and United Nations Environment Programme for the Operation of the Operation of the Prior Informed Consent, Geneva.

US EPA 1995, *Technical Guidance Manual, Assessing Dermal Exposure from Soil*, US EPA Region 3, December 1995, available from: http://www.epa.gov/reg3hwmd/risk/human/info/solabsg2.htm.

US EPA (IRIS 2012), data and information available from the *Integrated Risk Information System*, an online database, available from http://www.epa.gov/iris/.

WHO 1981, *Pesticide Residues in Food, Evaluations 1981*, FAO Plant Production and Protection Paper 42, Joint FAO/WHO review.

WHO 2011, *Guidelines for drinking-water quality, 4th edn*, World Health Organization, Geneva, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html.

2 2,4-D

2.1 General

2,4-D is the common name for the chlorophenoxy herbicide 2,4-dichlophenoxy acetic acid.

Several comprehensive reviews of 2,4-D in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (APVMA 2006; WHO 1984; WHO 1987). The following provides a summary of the key aspects of 2,4-D that are relevant to the derivation of a soil HIL.

The herbicide is also formulated as an amine salt, alkali metal salt and ester derivative of 2,4-D (WHO 1984). Pure 2,4-D is a white to off-white crystalline powder with a slight phenolic odour (APVMA 2006). The commercial grade herbicide is often combined with solvents or surfactants and sold as granules, dust, emulsions and liquid concentrates (WHO 1984). 2,4-D is slightly soluble in water whereas the amine and alkali metal salt derivatives are highly soluble. The ester derivate is insoluble in water (WHO 1984). 2,4-D esters with short chain alcohols are highly volatile whereas 2,4-D and its salt and amine derivatives have a low volatility (APVMA 2006).

Some chlorinated by-products produced during manufacture of 2,4-D and its derivatives such as 2,7-dichlorodibenzo-p-dioxin, 1,3,6,8- and 1,3,7,9-tetrachlorodibenzo-p-dioxins and 1,3,7-trichlorodibenzo-p-dioxin have been associated with enhanced toxicity findings (WHO 1984).

2,4-D and its derivatives are systemic herbicides commonly used in Australia to control broadleaf and aquatic weeds (NHMRC 2004). At least 122 separate products containing these compounds were registered in Australia in 2003 (APVMA 2006). They were registered to control weeds in agricultural crops such as cereals, sugar cane and rice and in pastures and turf. 2,4-D herbicides were also applied at very low application rates to citrus and pears to reduce premature fruit drop and increase fruit storage life (WHO 1984; APVMA 2006). In addition, 2,4-D is used to increase the proportion of medium-sized potato tubers and the intensity of colour in red-skinned varieties (APVMA 2006). In 2006, the Australian Pesticides and Veterinary Medicines Authority conducted a review of the environmental fate and ecotoxicity of volatile 2,4-D esters and concluded that the registration of these compounds should be suspended (APVMA 2006). This review process is ongoing and the APVMA website (www.apvma.gov.au) should be checked for any updates on which products are currently registered.

2.2 Previous HIL

No previous HIL is available for 2,4-D (NEPC 1999).

2.3 Significance of Exposure Pathways

2.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of 2,4-D, hence a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

2.3.2 Dermal absorption

A dermal absorption value of 0.05 (5%) is available from US EPA (2004) based on a study by Wester et al. (1996). This study evaluated potential dermal absorption of 2,4-D from soil, where absorption over time changed over time (noted to be not-linear). Data from the study showed low absorption over 8 hours (0.03-0.05%) with slightly higher absorption over 16 hours (2.2%). Limited other data is available on the dermal absorption of 2,4-D from soil, hence the value of 0.05 (5%) has been adopted.

2.3.3 Inhalation of Dust

2,4-D is not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

2.3.4 Plant Uptake

Most chlorophenoxy herbicides are toxic to plants and, as such, will be phytotoxic to almost all broadleaf crops including tomatoes, grapes and fruit trees well before plant uptake into edible portions of fruit and vegetable crops is of significance. Hence the uptake of these compounds into home-grown produce has not been considered in the derivation of an HIL A.

Note that the phytotoxic effects of these compounds may need to be addressed on a site-specific basis if detected in soil.

2.3.5 Intakes from Other Sources – Background

Exposure concentrations provided by WHO (1984, 1987) (as well as noted in APVMA (2006)) are derived from areas where 2,4-D is used and is not expected from the presence of 2,4-D contamination in soil. The intakes, however, may be of concern if the HILs were being applied to an area where products containing 2,4-D are used (or have been used in the recent past).

With respect to background intakes of 2,4-D, the following is noted from WHO (1987):

- It is expected background intakes for the general population will be associated with the presence of residues in food and water.
- Intakes from air is considered negligible.
- Where 2,4-D is not used, intakes by the general population are considered negligible.
- In areas where 2,4-D is used, background intakes from air, food and water are estimated to be 0.3–2 μg/kg/day.

FSANZ (2011) has estimated that the 90th percentile intake of 2,4-D by young children aged 2–5 years (most sensitive) is 0.014 μg/kg/day or 0.000014 mg/kg/day. This intake is negligible in comparison with the adopted TRV of 0.01 mg/kg/day.

On the basis of the above, background intakes of 2,4-D have been assumed to be essentially negligible (where 2,4-D is not used).

2.4 Identification of Toxicity Reference Values

2.4.1 Classification

The International Agency for Research on Cancer (IARC 1987) has classified chlorophenoxy herbicides as Group 2B—possibly carcinogenic to humans.

US EPA has not classified 2,4-D.

2.4.2 Review of Available Values/Information

There is limited information on the assessment of carcinogenicity and genotoxicity for 2,4-D from IARC and US EPA. Ibrahim et al. (1991) provided a summary of a review of carcinogenicity of 2,4-D following review by a panel of 13 scientists. Based on a weight-of-evidence approach 2,4-D was considered unlikely to be a genotoxic carcinogen because it has not been shown to be mutagenic in most in vitro and in vivo systems. The predominant opinion from the panel was that the weight of evidence indicates that it is possible that exposure to 2,4-D may cause cancer in humans.

On the basis of the available information, it is considered appropriate that a threshold dose-response approach be adopted for 2,4-D. The following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	TDI = 0.01 mg/kg/day	Current ADWG (NHMRC 2011) of 0.03 mg/L based on 10% intake from drinking water. Based on equations presented in the ADWG, the TDI considered in this derivation is equal to 0.009 mg/kg/day, which can be rounded to 0.01 mg/kg/day, essentially equivalent to the ADI available from the OCS.
OCS (2012)	ADI = 0.01 mg/kg/day	The ADI is noted to have been last reviewed in June 2006 and is based on a NOEL of 1 mg/kg/day associated with abnormal renal morphology in a 2-year rat study, supported by the same NOELs (based on kidney effects) in a 2-year mouse and 1-year dog study.
International		
WHO (2011)	ADI = 0.01 mg/kg/day	ADI, used in the derivation of the current WHO DWG (2011), was established by JMPR (FAO/WHO 1997) for 2,4-D and its salts and esters on the basis of a NOAEL of 1 mg/kg/day in a 1-year toxicity study in dogs and 2-year study in rats, and an uncertainty factor of 100-fold.
ATSDR	No evaluation available	
US EPA (IRIS 2012)	RfD = 0.01 mg/kg/day	US EPA has derived an oral RfD of 0.01 mg/kg/day. The value was last reviewed in 1986 and is derived based on a LOAEL of 1 mg/kg/day associated with abnormal renal morphology from a 90-day rat bioassay and a 1-year interim report from a 2 year rat study, and an uncertainty factor of 100.

Based on the available data above, there is general agreement from Australian and international sources on the consideration of an oral toxicity reference value of 0.01 mg/kg/day.

No dermal or inhalation specific studies or data are available. For the presence of 2,4-D in soil (not during use), it is considered appropriate to consider the use of the available ADI for all pathways of exposures.

2.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for 2,4-D in the derivation of HILs:

Recommendation for 2,4-D

Oral TRV (TRV_O) = 0.01 mg/kg/day (OCS 2012) for all pathways of exposure

Dermal absorption factor (DAF) = 0.05 (or 5%) (US EPA 2004)

Background intakes from other sources (as % of TRV):

BI_O = 0% for oral and dermal intakes

BIi = 0% for inhalation

2.5 Calculated HILs

On the basis of the above, the following HILs have been derived for 2,4-D (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	900	59	--	41	<1
Residential B	1600	27	--	73	<1
Recreational C	1300	43	--	57	<1
Commercial D	9000	21	--	79	<1

-- Pathway not included in derivation of HIL

2.6 References

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WHO 1987, Health and Safety Guide No. 5, 2,4-Dichlorophenoxyacetic (2,4-D), IPCS International Programme on Chemical Safety.

WHO 2011, Guidelines for drinking-water quality, 4th edn, World Health Organization, Geneva, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html.

3 MCPA, MCPB and Mecoprop

3.1 General

The following information on MCPA (4-chloro-2-methylphenoxyacetic acid), MCPB (4-(2-methyl-4-chlorophenoxy)butyric acid) and mecoprop (also referenced as MCPP) are grouped together as they are structurally similar chlorophenoxy herbicides.

While limited data is available, reviews of these compounds in the environment and their toxicity to humans are available and should be consulted for more detailed information not presented in this summary (WHO 2011; HSDB 2010). The following provides a summary of the key aspects of these compounds that are relevant to the derivation of a soil HIL.

In their pure form the three compounds are white crystalline solids, though technical grade products can be white to light brown crystal powders or liquids. The compounds are often formulated as salts (e.g. potassium or diethylamine salts) or esters (e.g. iso-octyl esters). The three compounds are the active ingredients in post emergence herbicides used to control annual and perennial weeds in agricultural, commercial/industrial and domestic environments. In Australia all three compounds are registered for agricultural application on wheat, barley, oats, sorghum, rice, linseed, peas, grass pastures, turf, clover, corn/maize and oilseed poppies, and for the home garden to control broadleaf weeds (WHO 2011).

3.2 Previous HIL

No previous HIL is available for MCPA, MCPB or mecoprop (NEPC 1999).

3.3 Significance of Exposure Pathways

3.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of MCPA, MCPB or mecoprop, hence a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

3.3.2 Dermal absorption

Insufficient data is available on the dermal absorption of MCPA, MCPB or mecoprop from soil. Hence the default value of 0.1 (10%) suggested by US EPA (1995) for pesticides has been adopted in the derivation of HILs.

3.3.3 Inhalation of Dust

MCPA, MCPB and mecoprop are not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

3.3.4 Plant Uptake

Most chlorophenoxy herbicides are toxic to plants and, as such, will be phytotoxic to almost all broadleaf crops including tomatoes, grapes and fruit trees well before plant uptake into edible portions of fruit and vegetable crops is of significance. Hence the uptake of these compounds into home-grown produce has not been considered in the derivation of HIL A.

Note that the phytotoxic effects of these compounds may need to be addressed on a site-specific basis if detected in soil.

3.3.5 Intakes from Other Sources – Background

Limited data is available for the assessment of background intakes of MCPA, MCPB and mecoprop. These compounds are currently registered for use in Australia (while some areas are only allowed controlled use of MCPA) and they are generally not considered persistent in the environment. The compounds are not included in the Australian Total Diet Surveys (FSANZ 2003; FSANZ 2011) and there is no data regarding concentrations in drinking water or air in Australia. Away from areas where these herbicides are used, exposure by the general public is expected to be low. In the USA, MCPA was detected up to 0.54 µg/L in surface waters and up to 5.5 µg/L in groundwater (WHO 2011). Background intakes may be similar to those considered for 2,4-D, which is essentially negligible (where these products are not used).

3.4 Identification of Toxicity Reference Values

3.4.1 Classification

The International Agency for Research on Cancer (IARC 1987) has classified chlorophenoxy herbicides as Group 2B—possibly carcinogenic to humans. Information provided in the IARC evaluation relates more specifically to MCPA and mecoprop. No evaluation is available for MCPB.

US EPA has not classified MCPA, MCPB or mecoprop.

3.4.2 Review of Available Values/Information

There is limited information on the assessment of carcinogenicity and genotoxicity for these compounds. WHO (2011) notes that recent studies on rats and mice do not indicate that MCPA was carcinogenic and there is only limited and inconclusive data on the genotoxicity of MCPA. Limited studies available on MCPB and mecoprop were negative with respect to genotoxicity. On the basis of the available information, it is considered appropriate that a threshold dose–response approach be adopted for these herbicides. The following are available from Level 1 Australian and International sources:

3.4.2.1 MCPA

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	TDI = 0.011 mg/kg/day	MCPA has been assessed with a health-based guideline of 0.04 mg/L based on a TDI of 0.011 mg/kg/day based on a NOEL of 1.1 mg/kg/day from a 2-year study in rats, and an uncertainty factor of 100
OCS (2012)	ADI = 0.01 mg/kg/day	The ADI is noted to have been set in April 1994 and is based on a NOEL of 1.1 mg/kg/day (as considered in the ADWG (NHMRC 2011)).
International		
WHO (2011)	TDI = 0.0005 mg/kg/day	The TDI was derived on the basis of a NOAEL of 0.15 mg/kg/day associated with renal and liver toxicity observed in a 1-year feeding study in dogs, and an uncertainty factor of 300. It is noted that the current guideline has remained unchanged since first derived in 1993. MCPA is included in the rolling revisions to the WHO DWG (2011) with no significant revisions issued to date.
ATSDR	No evaluation available	
US EPA (IRIS 2012)	RfD = 0.0005 mg/kg/day	The RfD (last reviewed in 1987) is derived based on the same study and evaluation provided in the WHO DWG (2011).

3.4.2.2 MCPB

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	No evaluation available	
OCS (2012)	ADI = 0.01 mg/kg/day	The ADI is noted to have been set in May 1994 and is based on a NOEL of 1.1 mg/kg/day.
International		
WHO (2011)	No quantitative value available	Insufficient data was available to establish a guideline value for MCPB in drinking water.
ATSDR	No evaluation available	
US EPA (IRIS 2012)	RfD = 0.01 mg/kg/day	The RfD (last reviewed in 1991) is derived based on a NOEL of 12 mg/kg/day associated with reproductive effects in a 13-week feeding study with dogs, and an uncertainty factor of 1000.

3.4.2.3 Mecoprop (MCPP)

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	No evaluation available	
OCS (2012)	ADI = 0.01 mg/kg/day	The ADI is noted to have been set in July 1998 and is based on a NOEL of 1 mg/kg/day, and an uncertainty factor of 100.
International		
WHO (2011)	TDI = 0.0033 mg/kg/day	The TDI was derived on the basis of a NOAEL of 1 mg/kg/day associated with kidney effects in 1- and 2-year studies in rats, and an uncertainty factor of 300. It is noted that the current guideline has remained unchanged since first published in 1996. Mecoprop is included in the rolling revisions to the WHO DWG (2011) with no significant revisions issued to date.
ATSDR	No evaluation available	
US EPA (IRIS 2012)	RfD = 0.001 mg/kg/day	The RfD (last reviewed in 1990) is derived based on a NOEL of 3 mg/kg/day associated with kidney effects in a 90-day rat feeding study, and an uncertainty factor of 3000.

The available evaluations in relation to MCPA, MCPB and mecoprop are all dated (none more recent than 1996) and are based on limited databases of studies. In relation to MCPB, the evaluations available from OCS (2012) and US EPA are consistent. In relation to MCPA and mecoprop, the critical studies identified for the determination of the point of departure differ between the OCS and WHO/US EPA evaluations. The subsequent application of uncertainty factors (with WHO/US EPA more conservative) also differs. Insufficient data is available to support any one evaluation, hence preference has been given to the Australian values adopted by OCS (2012), which have also been adopted in the derivation of the Australian Drinking Water Guidelines (NHMRC 2011). On this basis, the current Australian ADIs (as presented by OCS (2012)) have been adopted for the derivation of soil HILs.

No dermal or inhalation-specific studies or data are available. For the presence of MCPA, MCPB and mecoprop in soil (not during use) it is considered appropriate to consider use of the available ADI for all pathways of exposures.

3.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for MCPA, MCPB and mecoprop in the derivation of HILs:

Recommendation for MCPA, MCPB and Mecoprop

Oral TRV (TRV_O) = 0.01 mg/kg/day (OCS 2012) for each compound, for all pathways of exposure

Dermal absorption factor (DAF) = 0.1 (or 10%) (US EPA 1995)

Background intakes from other sources (as % of TRV):

BI_O = 0% for oral and dermal intakes

BIi = 0% for inhalation

Note that background intakes in areas where herbicides are used need to be considered on a site-specific basis.

3.5 Calculated HILs

On the basis of the above, the following HILs have been derived for MCPA, MCPB and mecoprop (as individual compounds) (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	600	43	--	57	<1
Residential B	900	16	--	84	<1
Recreational C	800	27	--	73	<1
Commercial D	5000	12	--	88	<1

-- Pathway not included in derivation of HIL

3.6 References

FSANZ 2003, *The 20th Australian Total Diet Survey*, a total diet survey of pesticide residues and contaminants, website: *http://www.anzfa.gov.au/*.

FSANZ 2011, *The 23rd Australian Total Diet Study*, Food Standards Australia and New Zealand.

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WHO 2011, *Guidelines for drinking-water quality, 4th edition*, World Health Organization, Geneva, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html

4 Picloram

4.1 General

Limited data is available on picloram, however reviews of this compound in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (Health Canada 1988; US EPA 1995a; OEHHA 1997). The following provides a summary of the key aspects of picloram that are relevant to the derivation of a soil HIL.

Picloram is a member of the pyridine carboxylic acid group and is manufactured in a number of forms. Picloram acid is only manufactured as an intermediate product in the production of herbicides whereas the amine salt, potassium salt and ester derivatives of picloram are produced as commercial herbicides. Technical grade picloram acid is an off-white to brown powder. It is slightly soluble in water and the amine and potassium salt derivatives are highly soluble. The ester derivative, however, is insoluble in water (US EPA 1995a).

Picloram acid and its derivatives have been used since the 1960s as a systemic herbicide to control woody plants and broadleaf weeds in rights of way, forestry, rangeland and pasture. In Australia, picloram derivatives are used to control weeds in winter cereals and linseed crops and to control a number of environmental and noxious weeds (APVMA 2009).

Picloram products are commonly contaminated with hexachlorobenzene (HCB). The presence of HCB in picloram affects the assessment of toxicity in a number of studies. Limited data is available for picloram alone. Available data also show that picloram is synergistic with several common herbicides (in particular 2,4-D, atrazine and alachlor) with respect to its toxicity to mammals and fish (NCAP 1998).

4.2 Previous HIL

No previous HIL is available for picloram (NEPC 1999).

4.3 Significance of Exposure Pathways

4.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of picloram, hence a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

4.3.2 Dermal absorption

Insufficient data is available on the dermal absorption of picloram from soil. Hence the default value of 0.1 (10%) suggested by US EPA (1995b) for pesticides has been adopted in the derivation of HILs.

4.3.3 Inhalation of Dust

Picloram is not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

4.3.4 Plant Uptake

Most carboxylic herbicides are toxic to plants and, as such, will be phytotoxic to almost all broadleaf crops including tomatoes, grapes and fruit trees well before plant uptake into edible portions of fruit and vegetable crops is of significance. Hence the uptake of these compounds into home-grown produce has not been considered in the derivation of HIL A.

Note that the phytotoxic effects of these compounds may need to be addressed on a site-specific basis if detected in soil.

4.3.5 Intakes from Other Sources – Background

Limited data is available for the assessment of background intakes of picloram. Picloram products are currently registered for use in Australia and the compound is considered persistent in the environment. Picloram is not included in the Australian Total Diet Surveys (FSANZ 2003; FSANZ 2011) and there is no data regarding concentrations in drinking water or air in Australia. Away from areas where picloram products are used, exposure by the general public is expected to be low. Review by US EPA (1995b) suggests that dietary intakes comprise only 0.5% of the threshold reference value (RfD) adopted (0.2 mg/kg/day) for most of the US population, with intakes from non-nursing infants highest at 1.9% of the RfD adopted. Review by Health Canada (1988) also noted the maximum dietary intake of picloram is estimated to be negligible, based on available data in Canada and the USA. On this basis, intakes from other sources have been assumed to be negligible in the derivation of HILs.

4.4 Identification of Toxicity Reference Values

4.4.1 Classification

The International Agency for Research on Cancer (IARC 1991) has classified picloram as Group 3—not classifiable.

US EPA has not classified picloram.

4.4.2 Review of Available Values/Information

Studies associated with the assessment of carcinogenicity of picloram are noted to be affected by the presence of HCB as a contaminant/impurity. Hence a number of reviews of carcinogenicity are conflicting. The review by IARC noted limited evidence of carcinogenicity for technical grade picloram in experimental animals. In general, the available data suggests the picloram is not genotoxic (Health Canada 1988; US EPA 1995) or at most weakly mutagenic (OEHHA 1997). On the basis of the limited available information, it is considered appropriate that a threshold dose–response approach be adopted for picloram. The following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	TDI = 0.07 mg/kg/day	The current ADWG (NHMRC 2011) derive a guideline of 0.3 mg/L derived from a NOEL of 7 mg/kg/day associated with increased liver weights in a short-term dietary study in rats, and an uncertainty factor of 100.
OCS (2012)	ADI = 0.07 mg/kg/day	The ADI is noted to have been set in February 1987 and is based on a NOEL of 7 mg/kg/day (as considered in the ADWG, noted above).
International		
WHO(2011)	No evaluation available	
ATSDR	No evaluation available	
Health Canada (1988)	NDI = 0.02 mg/kg/day	Negligible daily intake (NDI) derived on the basis of a NOAEL of 20 mg/kg/day associated with liver and kidney changes in rat and mouse studies, and an uncertainty factor of 1000.
US EPA	RfD = 0.07 mg/kg/day	The RfD (last reviewed in 1987) is derived based on the

Source	Value	Basis/Comments
(IRIS 2012)		same study and evaluation provided in the ADWG (NHMRC 2004). Value also derived by OEHHA (1997).
US EPA (1995)	RfD = 0.2 mg/kg/day	RfD calculated based on a NOEL of 20 mg/kg/day from a 2-year chronic rat feeding study, and an uncertainty factor of 100.

Limited quantitative data is available for picloram, however it is recommended that the current Australian ADI/TDI be adopted for the derivation of a soil HIL.

No dermal or inhalation-specific studies or data are available. For the presence of picloram in soil (not during use), it is considered appropriate to consider use of the available ADI for all pathways of exposures.

4.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for picloram in the derivation of HILs:

Recommendation for Picloram

Oral TRV (TRV_O) = 0.07 mg/kg/day (OCS 2008; NHMRC 2004; NHMRC 2009) for all pathways of exposure

Dermal absorption factor (DAF) = 0.1 (or 10%) (US EPA 1995)

Background intakes from other sources (as % of TRV):

BI_O = 0% for oral and dermal intakes

BIi = 0% for inhalation

Note that background intakes in areas where herbicides are used need to be considered on a site-specific basis.

4.5 Calculated HILs

On the basis of the above, the following HILs have been derived for picloram (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	4500	43	--	57	<1
Residential B	6600	16	--	84	<1
Recreational C	5700	27	--	73	<1
Commercial D	35 000	12	--	88	<1

-- Pathway not included in derivation of HIL

4.6 References

APVMA 2009, *Chemicals Nominated for Review*, last update unknown, accessed July 2009, Australian Pesticides and Veterinary Medicines Authority (APVMA), http://www.apvma.gov.au/chemrev/ChemRevProgram.shtml.

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OEHHA 1997, *Public Health Goal for Picloram in Drinking Water*, prepared by Pesticide and Environmental Toxicology Section, Office of Environmental Health Hazard Assessment, California Environmental Protection Agency, December 1997.

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US EPA 1995b, *Technical Guidance Manual, Assessing Dermal Exposure from Soil*, US EPA Region 3, December 1995, available from: http://www.epa.gov/reg3hwmd/risk/human/info/solabsg2.htm.

WHO 2011, *Guidelines for drinking-water quality, 4th edn*, World Health Organization, Geneva, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html.

5 Atrazine

5.1 General

Several comprehensive reviews of atrazine in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 2003; NRA 1997; APVMA 2008; IARC 1999). The following provides a summary of the key aspects of atrazine that are relevant to the derivation of a soil HIL.

Atrazine is the common name for the compound 6-chloro-N2-ethyl-N4-isopropyl-1,3,5-triazine-2,4-diamine which is an odourless white powder or colourless crystal (ATSDR 2003). Commercially manufactured atrazine is typically greater that 90% pure. Common impurities include dichlorotriazines, hydroxytriazines, tris(alkyl)aminotriazines, simazine, propazine and sodium chloride (ATSDR 2003). Atrazine is manufactured as a liquid, granules or wettable powder and can also be formulated in combination with other herbicides such as ametryn, amitrole, hexazinone, metalochlor, glyphosate and dicamba (NRA 1997).

Atrazine is one of the most widely used herbicides in Australian agriculture and has been used since the 1960s (NRA 1997). It is primarily used to control broadleaf weeds and some grasses between crops such as sorghum, maize, lupins, sugar cane and triazine-tolerant canola. Atrazine is also widely used to control weeds and some grasses by the forestry industry in pine and eucalyptus plantations (NRA 1997; NHMRC 2011). Non-agricultural uses in Australia such as the spraying of weeds along fence lines, irrigation channels, drains, driveways and footpaths were discontinued in 1995 (NRA 1997).

Regulatory actions (by the National Registration Authority for Agricultural and Veterinary Chemicals [APVMA]) undertaken in 1997 included cancellation of industrial and non-agricultural uses of atrazine (home garden uses and all commercial turf uses), deletion of use patterns and maximum residue limits (MRLs) for label claims for which there were no current use patterns (citrus, grapes and pineapples) and the introduction of a range of label instructions to reduce the risk of atrazine entering waterways. In addition, registrants were required to provide additional residue and monitoring data.

The APVMA has initiated a project to re-examine the possibility that the triazines (atrazine and related chemicals with a similar MoA) may have unintended harmful effects on humans, taking into account ongoing research into a newly hypothesised endocrine MoA. This project will take into account international reports, such as the work of the Joint Meeting on Pesticide Residues (JMPR).

Registrants who have a product whose label specifies a claim for weed control on triazine-tolerant canola will be required to either generate additional data or include an additional label restraint that specifies that atrazine must not be used post-emergence on triazine-tolerant canola grown on raised beds.

After consideration of the additional assessments completed after 1997, APVMA accepts the recommendations of OCS and the 2004 recommendations of DEWHA, and the following regulatory actions have been applied:

1. Active constituent approvals have been affirmed.

2. Existing label instructions have been deemed to be inadequate and the most recently approved labels have been amended as follows:

    - Labels have been amended to specify additional restraints to further reduce the risk of contamination of waterways.

    - Withholding period instructions have been amended.

- Herbicide resistance reporting details have been added to labels.

These variations to label instructions satisfy the requirements for continued registration of products; and so

3. Product registrations have been affirmed.

4. To ensure that all labels are in line with the recommendations of the 2008 report, any previously approved labels that do not contain the amended instructions have been cancelled.

As an associated outcome of the review, changes will be made to the MRL Standard to align entries in the standard with existing approved use patterns.

5.2 Previous HIL

No previous HIL is available for atrazine (NEPC 1999).

5.3 Significance of Exposure Pathways

5.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of atrazine hence a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

5.3.2 Dermal absorption

Insufficient data is available on the dermal absorption of atrazine from soil. Hence the default value of 0.1 (10%) suggested by US EPA (1995) for pesticides has been adopted in the derivation of HILs.

5.3.3 Inhalation of Dust

Atrazine is not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

5.3.4 Plant Uptake

Atrazine is used as a herbicide and, as such, is phytotoxic to almost all broadleaf weeds and plants. Some plants are more sensitive than others to residues of atrazine in the soil, however in general, phytotoxicity will occur well before plant uptake into edible portions of fruit and vegetable crops is of significance. Hence the uptake of these compounds into home-grown produce has not been considered in the derivation of an HIL A.

Note that the persistence of atrazine in soil and potential for phytotoxic effects may need to be addressed on a site-specific basis if detected in soil.

5.3.5 Intakes from Other Sources – Background

Reviews of potential intakes from sources other than soil (primarily food) by NRA (1997), NHMRC (2011) and RIVM (2001) suggested these intakes were essentially negligible. Further review of residue data by APVMA (2008) noted that, when atrazine was used in accordance with the revised label directions, residues were unlikely to pose a risk to human health. Potential exposures during application of atrazine products may require further consideration on a site-specific basis; however exposures by the general public (in areas away from application) are negligible.

5.4 Identification of Toxicity Reference Values

5.4.1 Classification

The International Agency for Research on Cancer (IARC 1999) has classified atrazine as Group 3—not classifiable. US EPA has not classified atrazine.

5.4.2 Review of Available Values/Information

The available data reviewed by JMPR (2007) and APVMA (2008) suggested that atrazine was not likely to pose a carcinogenic risk to humans. Review by JMPR (2007) and RIVM (2001) suggested that based on the weight of evidence, atrazine was not genotoxic. There is some evidence that it can induce mammary tumours in rats as a result of hormonal changes, but the mechanism is believed to be non-genotoxic. On the basis of the available information, it is considered appropriate that a threshold dose–response approach be adopted for atrazine.

The following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	ADI = 0.005 mg/kg/day	Current ADWG (NHMRC 11) of 0.04 mg/L based on 50% intake from drinking water and an ADI of 0.005 mg/kg/day as referenced from the TGA (NRA 1997).
OCS (2012)	ADI = 0.005 mg/kg/day	The ADI of 0.005 mg/kg/day is noted to be based on a NOEL of 10ppm associated with mammary tumours from a 24-month female rat study, and a 100-fold safety factor. This value was set in December 1996.
NRA (1997)	ADI = 0.005 mg/kg/day	The NRA (1997) review identified the relevance of adopting an ADI of 0.005 mg/kg/day for atrazine. This value has been reconfirmed in the update provided by APVMA (2008). However the review noted that APVMA has initiated a project to re-examine the possibility that the triazines may have harmful endocrine effects, including updates available from JMPR. APVMA also note that US EPA is currently reviewing atrazine.
International		
JMPR (2007)	ADI = 0.02 mg/kg/day	Review of atrazines by the Joint FAO/WHO Meeting on Pesticides Residues (JPMR, 2007) identified a group ADI (for atrazine, diethyl-atrazine, di-isopropyl-atrazine and diaminochlorotriazine) of 0–0.02 mg/kg/day based on oestrous cycle disruption.
WHO (2011)	ADI = 0.02 mg/kg/day	Group ADI for atrazine and its chloro-s-triazine metabolites (reviewed in 2011) is based on a NOAEL of 1.8 mg/kg/day identified on the basis of luteinizing hormone surge suppression and subsequent disruption of the oestrous cycle seen at 3.6 mg/kg body weight per day in a 6-month study in rats, using a safety factor of 100
RIVM (2001)	TDI = 0.005 mg/kg/day	TDI based on a NOAEL of 0.5 mg/kg/day associated with reproductive effects in rats, and a 100-fold uncertainty factor.
ATSDR	No evaluation available	
US EPA (IRIS 2012)	RfD = 0.035 mg/kg/day	The US EPA (available from IRIS) have derived an oral RfD of 0.035 mg/kg/day. The value was last reviewed in 1993 and is based on a NOAEL of 3.5 mg/kg/day associated with

Source	Value	Basis/Comments
		decreased body weight gain from a 2-year rat study, and an uncertainty factor of 100.

While the most recent review by WHO (2011) provides a less conservative ADI, the current Australian ADI of 0.005 mg/kg/day is considered relevant and appropriate for consideration in the derivation of a soil HIL.

No dermal or inhalation-specific studies or data are available. For the presence of atrazine in soil (not during use in herbicide products), it is considered appropriate to consider use of the available threshold ADI for all pathways of exposures.

5.4.3 Recommendation

On the basis of the discussion above the following toxicity reference values (TRVs) have been adopted for atrazine in the derivation of HILs:

Recommendation for Atrazine

Oral TRV (TRV_O) = 0.005 mg/kg/day (NHMRC 2011; OCS 2008; APVMA 2008) for all pathways of exposure

Dermal absorption factor (DAF) = 0.1 (or 10%) (US EPA 1995)

Background intakes from other sources (as % of TRV):

BI_O = 0% for oral and dermal intakes

BIi = 0% for inhalation

Note that background intakes in areas where herbicides are used need to be considered on a site-specific basis.

5.5 Calculated HILs

On the basis of the above, the following HILs have been derived for atrazine (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	320	43	--	57	<1
Residential B	470	16	--	84	<1
Recreational C	400	27	--	73	<1
Commercial D	2500	12	--	88	<1

-- Pathway not included in derivation of HIL

5.6 References

APVMA 2008, *Atrazine, Final Review Report and Regulatory Decision*, Australian Pesticides & Veterinary Medicines Authority, March 2008.

ATSDR 2003, *Toxicological Profile for Atrazine*, US Department of Health and Human Services, ATSDR, September 2003, available from http://www.atsdr.cdc.gov/ToxProfiles/tp.asp?id=338&tid=59.

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OCS 2012, *ADI List, Acceptable Daily Intakes for Agricultural and Veterinary Chemicals*, current to 31 March 2012, Australian Government, Department of Health and Ageing, Office of Chemical Safety (OCS), available from: http://www.health.gov.au/internet/main/publishing.nsf/content/E8F4D2F95D616584CA2573D700770C2A/$File/ADI-apr12.pdf.

RIVM 2001, *Re-evaluation of human-toxicological Maximum Permissible Risk levels*, National Institute of Public Health and the Environment, Bilthoven, Netherlands, available from: http://www.rivm.nl/bibliotheek/rapporten/711701025.html.

US EPA 1995, *Technical Guidance Manual, Assessing Dermal Exposure from Soil*, US EPA Region 3, December 1995, available from: http://www.epa.gov/reg3hwmd/risk/human/info/solabsg2.htm.

US EPA (IRIS 2012) data and information available from the *Integrated Risk Information System*, an online database, available from http://www.epa.gov/iris/.

WHO 2011, *Guidelines for drinking-water quality, 4th edn*, World Health Organization, Geneva, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html.

6 Chlorpyrifos

6.1 General

Several reviews of chlorpyrifos in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 1997; WHO 2004; NRAAVC 2000; APVMA 2009; Taylor & Di Marco 2003). The following provides a summary of the key aspects of chlorpyrifos that are relevant to the derivation of a soil HIL.

Chlorpyrifos is the common name for the organophosphorous insecticide *O*,*O*-diethyl *O*-3,5,6-trichloro-2-pyridyl phophorothioate. Pure chlorpyrifos is an odourless, white to colourless crystalline solid. The compound is non-polar and therefore has a low solubility in water and an affinity for organic substances. It is also thermally sensitive at temperatures over 50 °C and decomposes at 130 °C (NRAAVC 2000; WHO 2004).

Technical grade chlorpyrifos has a minimum purity of 940 to 990 g/kg. It is a white to light yellowish brown crystalline solid with a mild mercaptan odour. Commercial formulations of chlorpyrifos are generally produced as a concentrated emulsion, liquid, wettable powder, dust, solid bait or granules (NRAAVC 2000).

Chlorpyrifos has been widely used in the Australian agricultural industry since the mid-1960s as it is reportedly less harmful to beneficial insects and is a useful tool in insecticide resistance management programs (NRAAVC 2000). It is used to control insects in soil and on crop foliage including fruit (pome, stone and citrus fruit, strawberries, figs, pineapples, kiwifruit and bananas), nuts, vines, vegetables (potatoes, asparagus), grains (rice, cereals, maize, sorghum), cotton, mushrooms, sugar cane, turf and ornamental plants (NRAAVC 2000). In industrial/commercial and domestic buildings chlorpyrifos is used to control termites, cockroaches, spiders, ants, mosquitoes and fleas and is generally sprayed in the sub-floor region during construction or applied around the building. It is also registered for use in dog and cat flea collars, sprays and shampoos. While the number of products containing chlorpyrifos changes on a yearly basis[1], in 2000 there were 164 products registered in Australia that contained chlorpyrifos (NRAAVC 2000).

In contrast to Australia, the US banned all domestic use of chlorpyrifos in 2001.

Chlorpyrifos is persistent in the environment with a half-life in soil reported to range from 33–56 days for soil-incorporated applications (Tomlin 2003) to 462 days in Australian soil under conditions similar to the application of products on soil for termite control (Baskaran et al. 1999).

6.2 Previous HIL

No previous HIL is available for chlorpyrifos (NEPC 1999). It is noted, however that review of chlorpyrifos by Taylor & Di Marco (2003) derived a health-based soil investigation level (residential) of 80 mg/kg on the basis of a threshold toxicity reference value of 0.003 mg/kg/day (noted to be derived from US EPA), 100% oral bioavailability, soil ingestion only, and an assumption that exposures from soil contribute (by default) 20% of the reference value.

[1] Refer to APVMA Public Chemical Registration Information System (PUBCRIS) for current information on products that contain chlorpyrifos (http://services.apvma.gov.au/PubcrisWebClient/welcome.do)

6.3 Significance of Exposure Pathways

6.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of chlorpyrifos, hence a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

6.3.2 Dermal absorption

Limited data is available on dermal absorption of chlorpyrifos. Review by APVMA (2009) identified that in acute animal studies, dermal absorption has been shown to be low. In human volunteers, dermal absorption was estimated to be 1.35% of the applied dose (NRAAVC 2000). Dermal absorption of chlorpyrifos in soil (not in solution) is expected to be lower. The assessment of occupational exposures by NRAAVC (2000), as confirmed by APVMA (2009), has adopted a dermal absorption value of 3%. This has been adopted in the derivation of HILs.

6.3.3 Inhalation of Dust

The inhalation exposure pathway is expected to be of significance during and immediately after the application of products containing the product. In these cases chlorpyrifos may be present in the vapour phase as well as sorbed to particulates (ATSDR 1997). An Australian study by Beard et al. (1995) demonstrated that airborne exposures to pesticides in the community can be substantial and are largely related to residential use of pesticides rather than agricultural applications. These issues should be considered on a site-by-site basis.

For the assessment of chlorpyrifos as a soil contaminant (no product application considered), the compound is not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

6.3.4 Plant Uptake

Information relating to the potential for plant uptake of chlorpyrifos is mixed. ATSDR (1997) notes that some research has shown that only very small levels of chlorpyrifos are taken up by plant roots, translocated, or metabolised by plant tissues. However, other researchers have found that soil-applied doses of chlorpyrifos are transported to foliage. APVMA (2009) notes that absorption and translocation of foliar deposits of chlorpyrifos is very low, with the bulk dissipating through volatilisation. Absorption by roots from the soil is also poor. This is further supported by studies presented by JMPR (1972) that show that the uptake of chlorpyrifos or its degradation products is insignificant through the foliage or roots. Only through the use of specialised techniques has plant uptake of chlorpyrifos been significant.

Chlorpyrifos has the potential to strongly adsorb to soil and sediments (based on log K_{oc} of 3.73 from ATSDR (1997)) and has low water solubility. Hence the potential for chlorpyrifos to be present in soil solution, and subsequent uptake by plants, is considered to be low.

On the basis of the available information, plant uptake into edible fruit and vegetable crops is considered low and has not been considered in the derivation of soil HILs.

6.3.5 Intakes from Other Sources – Background

Background intakes were evaluated in more detail by Taylor & Di Marco (2003), where data (from Australia where relevant) for food, water and air were considered. Background intakes were estimated to range from 0.81 µg/kg/day for adults and infants to 1 µg/kg/day for toddlers. Dietary intakes of 0.63

µg/kg/day for toddlers (based on older surveys) were higher than currently reported. Current data on intakes from food and air (most significant pathways considered) include:

- Intakes of chlorpyrifos based on *The Australian Total Diet Survey* (FSANZ 2011) were 0.23 µg/kg/day for children aged 2–5 years (most significant). While it is accepted that there are limitations in the data provided in these studies, the data is consistent with information from studies conducted in the US (ATSDR 1997) and have been considered indicative of potential intakes from food.
- A range of air concentrations have been reported for chlorpyrifos, during or immediately after application, some period after application, and ambient concentrations. Mean concentrations of chlorpyrifos in homes treated with termiticide several years previously were 2.23 µg/m^3 (EA 2001). Intakes derived from these concentrations are estimated to be 1.4 µg/kg/day, significantly more than intakes derived from dietary sources.

Other sources of exposure may be associated with house dust, though as there is limited data available to quantify exposures related to the presence of chlorpyrifos in house dust, it has not been included in this evaluation. It is noted that the derivation of the soil HIL considers ingestion of both soil and dust.

Consideration of intakes derived from food and air suggests background intakes may be approximately 1.6 µg/kg/day, which comprise approximately 50% of the recommended TRV. Review of dietary intakes by APVMA (2009), based on a conservative estimate of chemical residues in food, indicated that intakes may comprise up to 55% of the TRV, similar to the estimate presented on the basis of the above.

As chlorpyrifos remains in use in Australia it is reasonable, based on the above, to consider background intakes to be more than negligible. Based on the estimates, intakes derived from dietary and atmospheric sources have been estimated to be approximately 1.6 µg/kg/day (50% of the TRV) and have been considered in the derivation of soil HILs.

6.4 Identification of Toxicity Reference Values

6.4.1 Classification

The International Agency for Research on Cancer (IARC) has not classified chlorpyrifos as to carcinogenicity and US EPA has classified it as Group D—not classified for carcinogenicity

6.4.2 Review of Available Values/Information

Limited data is available on the carcinogenicity of chlorpyrifos. However, chlorpyrifos has not been identified as carcinogenic in long-term animal studies, and was not genotoxic in a wide range of assays (NRAAVC 2000; APVMA 2009). On this basis, the assessment of exposures to chlorpyrifos on the basis of a threshold approach is appropriate.

The following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	ADI = 0.003 mg/kg/day	Current ADWG (NHMRC 2011, established in 1998) of 0.01 mg/L based on a NOEL of 0.03 mg/kg/day for plasma cholinesterase inhibition from a 28-day volunteer study in humans, and an uncertainty factor of 10.
OCS (2012)	ADI = 0.003 mg/kg/day	The ADI of 0.003 mg/kg/day (set in December 1998) is based on the same approach as noted in the ADWG above.
NRAAVC (2000) and APVMA (2009)	ADI = 0.003 mg/kg/day	The APVMA (2009) review provided an updated toxicology assessment for chlorpyrifos. The review considered the range of threshold values derived by different countries with respect to the selection of relevant end points and other factors (including sensitive sub-populations such as children). The review did not identify any new studies that would result in changes to the toxicological end points selected for either public or occupational health assessments. The end points used in the NRA (2000) review were considered to be valid. No toxicological effects were observed at doses lower than those that resulted in inhibition of plasma cholinesterase activity in a human volunteer study. On the basis of this effect in humans at a dose of 0.1 mg/kg/day, with no effects seen at 0.03 mg/kg/day, the ADI at 0.003 mg/kg/day was established, with a 10-fold safety factor used to account for inter-individual variability.
International		
WHO (2011) and JMPR (1983, 2000)	ADI = 0.01 mg/kg/day	ADI adopted in derivation of the current WHO DWG and JMPR (1983, 2000) is based on a NOAEL of 0.1 mg/kg/day based on effects of chlorpyrifos on brain acetylcholinesterase activity in animal studies, and erythrocyte acetylcholinesterase inhibition in human subjects, and an uncertainty factor of 10. Review of this data by APVMA (2009) noted that both of these measures of toxicity are less sensitive than the inhibition of plasma cholinesterase activity, and hence the JMPR ADI is higher (i.e. less conservative) than that set by the OCS.
ATSDR (1997)	Oral MRL = 0.001 mg/kg/day	Chronic oral MRL based on a NOAEL for acetylcholinesterase inhibition in rats exposed to 0.1 mg/kg/day of chlorpyrifos in feed for 2 years, and an uncertainty factor of 100.
US EPA (IRIS 2012)	Not available	The previous evaluation (oral RfD of 0.003 mg/kg/day) was withdrawn by the US EPA in 2011. No new evaluation is available.

The ADI of 0.003 mg/kg/day identified and considered current in the most recent review by APVMA (2009) and NRA (2000) is consistent with that considered in the derivation of the ADWG (NHMRC 2011) and listed in the ADI List (OCS 2012). The value is considered relevant for the derivation of a soil HIL in Australia.

No dermal or inhalation-specific studies or data are available. For the presence of chlorpyrifos in soil, it is considered appropriate to consider use of the available ADI for all pathways of exposures.

6.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for chlorpyrifos in the derivation of HILs:

Recommendation for Chlorpyrifos

Oral TRV (TRV_O) = 0.003 mg/kg/day (OCS 2012; NRAAVC 2000; APVMA 2009) for all pathways of exposure

Dermal absorption factor (DAF) = 0.03 (or 3%) (APVMA 2009)

Background intakes from other sources (as % of TRV):

BI_O = 50% for oral and dermal intakes

BIi = 50% for inhalation

Note that background intakes in areas where chlorpyrifos products used need to be considered on a site-specific basis.

6.5 Calculated HILs

On the basis of the above, the following HILs have been derived for chlorpyrifos (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	160	72	--	28	<1
Residential B	340	38	--	62	<1
Recreational C	250	55	--	45	<1
Commercial D	2000	31	--	69	<1

-- Pathway not included in derivation of HIL

6.6 References

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WHO 2004, Chlorpyrifos in Drinking Water, Background Document for Development of WHO Drinking Water Quality, World Health Organization, Geneva.

WHO 2011, Guidelines for drinking-water quality, 4th edn, World Health Organization, Geneva, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html.

7 Bifenthrin

7.1 General

Several comprehensive reviews of bifenthrin in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 2003; US EPA 1999; Fecko 1999; Taylor & Di Marco 2003). The following provides a summary of the key aspects of bifenthrin that are relevant to the derivation of a soil HIL.

Bifenthrin is the common name for the compound (2-methyl-1, 1-biphenyl-3-y1)-methyl-3-(2-chloro-3,3,3-trifluoro-1-propenyl)-2,2-dimethylcyclopropanecarboxylate. It is referred to as a 'third generation' synthetic pyrethroid insecticide and is known to be more stable and persistent in the environment and have a greater insecticidal activity than previously synthesized pyrethroid compounds (Taylor & Di Marco 2003). Pure bifenthrin is a crystalline or waxy solid which is off-white to pale tan in colour.

Bifenthrin is used in the agricultural industry to control insects in a number of crops and to protect stored grains. It is also used in domestic and commercial settings as a barrier to repel or kill insects such as termites (Taylor & Di Marco 2003).

7.2 Previous HIL

No previous HIL is available for bifenthrin (NEPC 1999). It is noted, however, that review of bifenthrin by Taylor & Di Marco (2003) derived a soil investigation level (residential) of 300 mg/kg on the basis of a threshold toxicity reference value of 0.01 mg/kg/day (noted to be derived from the Therapeutic Goods Administration), 100% oral bioavailability, soil ingestion only, and an assumption that exposures from soil contribute (by default) 20% of the reference value.

7.3 Significance of Exposure Pathways

7.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of bifenthrin, hence a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

7.3.2 Dermal absorption

Insufficient data is available on the dermal absorption of bifenthrin from soil. Hence the default value of 0.1 (10%) suggested by US EPA (1995) for pesticides has been adopted in the derivation of HILs.

It is noted that review by ATSDR (2003) considered the limited human and animal data associated with dermal application of pyrethroids. Dermal absorption values in the range of 0.5% to 1.8% were identified. Hence the adoption of 10% is considered conservative.

7.3.3 Inhalation of Dust

Bifenthrin is not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

7.3.4 Plant Uptake

Limited information is available on the potential for plant uptake of bifenthrin. ATSDR (2003) notes that in soils, pyrethrins adsorb strongly and do not leach appreciably into groundwater. These

compounds are not considerably taken up by the roots of vascular plants; however, they are deposited upon the leafy region of vegetation following spraying.

Where the application of the product is not of concern, there is limited potential for bifenthrin to be present in soil solution, and available for plant uptake, due to its strong adsorption to soil and its limited solubility.

On this basis, the potential for plant uptake into home-grown fruit and vegetable crops is not considered to be significant and has not been considered in the derivation of a soil HIL.

7.3.5 Intakes from Other Sources – Background

Background intakes were evaluated by Taylor & Di Marco (2003). No Australian data was identified and intakes from water, food, air, consumer products and soil were assumed to comprise 20% of the adopted ADI, resulting in background intakes from sources other than soil as 80%.

Synthetic pyrethroid pesticides were included in *The 23rd Australian Total Diet Survey* (FSANZ 2011). Intake associated with the detected residues of bifenthrin for children aged 2–5 years was 0.072 µg/kg/day, and for children aged 6–12 years was 0.085 µg/kg/day, similar to the intake estimated for adults.

Limited other data is available in Australia, where a study on bifenthrin in air within a home after termite treatment did not detect bifenthrin concentrations (Richards 2003). Pyrethrins and pyrethroids are used in both indoor and outdoor settings to control insects; therefore, these compounds are frequently detected in the air of homes and buildings after their use. Data from the USA (ATSDR 2003) reported concentrations of pyrethrins in the order of 0.1–0.3 µg/m^3 sometime after application (up to 84 days after application). Intakes by toddlers associated with these concentrations are in the range of 0.06–0.2 µg/kg/day, significantly higher than estimated from dietary intakes. It is noted that if these insecticide sprays are regularly used, indoor air concentrations may be higher.

On the basis of the above, intakes associated with bifenthrin (assuming it comprises 100% of the pyrethrins reported in indoor air in the US) may comprise up to 0.28 µg/kg/day for toddlers, approximately 3% of the recommended oral TRV. For the purpose of establishing an HIL, intakes from other sources has been taken to be 10% of the adopted TRV.

7.4 Identification of Toxicity Reference Values

7.4.1 Classification

The International Agency for Research on Cancer (IARC) and US EPA have not classified bifenthrin as to carcinogenicity. It is noted that the Joint Meeting on Pesticide Residues (JMPR 1993) has reviewed bifenthrin, which was evaluated as unlikely to pose a carcinogenic hazard to humans.

7.4.2 Review of Available Values/Information

A summary of health effects and information is presented by Taylor & Di Marco (2003). Limited data is available for the assessment of carcinogenicity, though the available data suggests that bifenthrin was not likely to pose a carcinogenic risk to humans.

On the basis of the available information it is considered appropriate that a threshold dose–response approach be adopted for bifenthrin. The following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG	No evaluation available	

Source	Value	Basis/Comments
OCS (2012)	ADI = 0.01 mg/kg/day	The ADI of 0.01 mg/kg/day based on maternal tremors in a developmental rat study. The value was set in 1992. The ADI is also used by FSANZ (2003).
International		
JMPR (1993)	ADI of 0–0.02 mg/kg/day	ADI established on the basis of a NOAEL of 1.5 mg/kg/day in a 1-year study in dogs, and a 100-fold uncertainty factor. The study was supported by the same NOAEL in the rat teratology study. ADI presented has been rounded by JMPR.
WHO	No evaluation available	
RIVM (2001)	No evaluation available	
ATSDR	No evaluation available	
US EPA (IRIS 2012)	RfD = 0.015 mg/kg/day	US EPA has established an oral RfD of 0.015 mg/kg/day based on a NOEL of 1.5 mg/kg/day associated with tremors in a 1-year dog study, and 100-fold uncertainty factor.

Based on the available data, the current Australian ADI of 0.01 mg/kg/day is considered current and relevant.

No dermal or inhalation-specific studies or data are available. For the presence of bifenthrin in soil (not during use), it is considered appropriate to consider use of the available threshold reference value for all pathways of exposures.

7.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for bifenthrin in the derivation of HILs:

Recommendation for Bifenthrin

Oral TRV (TRV_O) = 0.01 mg/kg/day (OCS 2012) for all pathways of exposure

Dermal absorption factor (DAF) = 0.1 (or 10%) (US EPA 1995)

Intakes allowable from soil (as % of TRV) = 80%

Background intakes from other sources (as % of TRV):

BI_O = 10% for oral and dermal intakes

BIi = 10% for inhalation

Note background intakes in areas where insecticides are regularly used may need to be considered on a site-specific basis.

7.5 Calculated HILs

On the basis of the above, the following HILs have been derived for bifenthrin (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)

Residential A	600	43	--	57	<1
Residential B	840	16	--	84	<1
Recreational C	730	27	--	73	<1
Commercial D	4500	12	--	88	<1

-- Pathway not included in derivation of HIL

7.6 References

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US EPA 1999, *Environmental Fate Assessment for the Synthetic Pyrethroids*. Environmental Fate Effects Division, United States Environment Protection Authority, www.epa.gov:80/opp00001/SAP/1999/february/pyreth.pdf.

US EPA (IRIS 2012), data and information available from the *Integrated Risk Information System*, an online database, available from http://www.epa.gov/iris/.

8 Shortened forms

ADI	acceptable daily intake
ADWG	Australian Drinking Water Guidelines
AI	adequate intake
ANZECC	Australia and New Zealand Environment and Conservation Council
APVMA	Australian Pesticides and Veterinary Medicines Authority
ATDS	Australian Total Diet Survey
ATSDR	Agency for Toxic Substances and Disease Registry
BA	bioavailability
BI	background intake
BMD	benchmark dose
BMDL	Benchmark dose lower confidence limit
CCME	Canadian Council of Ministers of the Environment
CICAD	Concise International Chemicals Assessment Document
CNS	central nervous system
DAF	dermal absorption factor
DW	dry weight
EA	Environment Agency (England and Wales)
EHC	Environmental Health Criteria
EPA	Environment Protection Authority
FSANZ	Food Standards Australia and New Zealand
GAF	gastrointestinal absorption factor
HCB	hexachlorobenzene
HEC	human equivalent concentration
HED	human equivalent dose
HIARC	Hazard Identification Assessment Review Committee
HIL	health investigation level
HSDB	Hazardous Substances Data Bank
HSL	health screening level
IARC	International Agency for Research on Cancer
IEUBK	Integrated exposure uptake biokinetic model

IRIS	Integrated Risk Information System
JECFA	Joint FAO/WHO Expert Committee on Food Additives
JMPR	WHO/FAO Joint Meeting on Pesticide Residues
LOAEL	lowest observed adverse effect level
LOEL	lowest observed effect level
MF	modifying factor
MOA	mode (or mechanism) of action
MRL	maximum residue limit
MRL	minimal risk level
NDI	negligible daily intake
NEPC	National Environment Protection Council
NEPM	National Environment Protection Measure
NHMRC	National Health and Medical Research Council
NOAEL	no observable adverse effect level
NOEL	no observable effect level
NSW DECC	New South Wales Department of Environment and Climate Change
OCS	Office of Chemical Safety
POP	persistent organic pollutant
PTDI	provisional tolerable daily intake
PTMI	provisional tolerable monthly intake
PTWI	provisional tolerable weekly intake
RAIS	Risk Assessment Information System
RDI	recommended daily intake
REL	reference exposure level
RfC	reference concentration
RfD	reference dose
RME	reasonable maximum exposure
SF	slope factor
TC	tolerable concentration
TD	tumorigenic dose
TDI	tolerable daily intake

TRV	toxicity reference value
UF	uncertainty factor
UL	upper limit
UR	unit risk
US EPA	United States Environmental Protection Agency
WHO	World Health Organization
WHO DWG	World Health Organization Drinking Water Guidelines

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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Prepared by the Office of Parliamentary Counsel, Canberra

Volume 10: Schedule B7 - Appendix 1
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About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

Schedule B7

APPENDIX A5

The Derivation of HILs for PCBs and PBDEs

Contents
The Derivation of HILs for PCBs and PBDEs

1 PCBs

1.1 General

Polychlorinated biphenyls (PCBs) are a group of synthetic organic compounds comprising two benzene rings joined together, with between one and ten chlorine atoms attached. There are 209 possible PCB variants (congeners) though PCBs are typically found as a complex mixture in commercial products and in the environment (WHO 1993). Of the 209 possible congeners, 12 are able to assume the same flat shape as dioxins and can cause impacts via the same mechanism. Consequently, it is normal to consider the PCB contribution to dioxin toxicity by measuring those congeners specifically. Some or all of these 12 congeners are always going to be present in any PCB contamination. There is evidence that using the dioxin-like PCBs as the basis for assessing risk from PCBs is also protective for the risks from the non-dioxin-like PCBs, i.e. the non-dioxin-like PCBs are less toxic than the dioxin-like PCBs.

The following relates to the assessment of non-dioxin-like PCBs only. The assessment of dioxins and dioxin-like PCBs needs to be conducted on a site-specific basis where there is the potential for a PCB source (such as PCB oil contamination) to be present at a site.

Several comprehensive reviews of PCBs in the environment and their toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 2000; WHO 1993; WHO 2003; EPHC 2003). The following provides a summary of the key aspects of PCBs that are relevant to the derivation of a soil HIL.

PCBs are typically in the form of an oily liquid or solid and are colourless to light yellow. Some PCB congeners may also exist as a vapour in air. They are odourless and tasteless. PCBs do not burn easily and have good insulating properties. They are both chemically and thermally stable. PCBs are relatively insoluble in water with the solubility decreasing with increasing chlorine content (ATSDR 2000).

Commercial PCB mixtures are also known by their trade names, such as Aroclor (USA), Phenochlor (France), Clophen (Germany), Kanechlor (Japan), Fechlor (Italy) and Sovol (USSR). Information on the toxicity and behaviour of a number of commercial PCB mixtures, Aroclors, is available, with Aroclor 1254 most commonly used as an indicator for the assessment of PCB mixtures. WHO (2003) provides a review of the most common commercial Aroclor mixtures with respect to the composition and toxicity of congeners present, and the various mixtures of indicator congeners (that differ from that of Aroclor 1254) may need to be considered on a site-specific basis.

Due to the thermal and chemical stability of PCBs, they are widely used as coolants and lubricants in transformers, capacitors and other electrical equipment (ATSDR 2000). In Australia, PCBs were also used in the manufacture of plastics, adhesives, paints and varnishes and were found in consumer products such as pesticides, fluorescent lighting and carbonless copy paper. PCBs were used in Australia between the 1930s and 1970s, when the importation of PCBs was banned.

1.2 Previous HIL

The derivation of the previous HIL (HIL A = 10 mg/kg) for PCBs is presented by Di Marco & Buckett (1993) and NEPC (1999). In summary, the HIL was derived on the basis of the following:

- Background intakes from air, water and food were estimated to be 5.4 ng/kg/day for a child and 4.4 ng/kg/day for an adult, estimated to be approximately 5% of the adopted PTDI (derived PTDI of 0.0001 mg/kg/day for Aroclor 1016).
- Due to the lack of published data for PCBs, the lowest threshold value derived for Aroclors 1016 and 1248 were considered. A PTDI of 0.0001 mg/kg/day was derived for Aroclor 1016 based on a NOAEL of 0.0125 mg/kg/day, and a safety factor of 100.

- Intakes derived from ingestion (assuming 30% bioavailability), inhalation of dust (assuming 50% bioavailability) and dermal absorption (10% absorption) were considered in the derivation of the soil HIL of 10 mg/kg.

1.3 Significance of Exposure Pathways

1.3.1 Oral Bioavailability

Bioavailability of PCBs in soil appears to be important due to their high affinity for soil particles and organic matter. Bioavailability was considered in the derivation of the current HIL (Di Marco & Buckett 1993) with 30% assumed for oral intakes and 50% assumed for inhalation. The basis for this assumption is not available and no more detailed reviewed of PCB bioavailability (oral or inhalation) in soil is available.

Insufficient data is available to adequately define the bioavailability of PCBs in the range of contaminated sites that may need to be considered in Australia. On this basis, a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

1.3.2 Dermal absorption

US EPA (2004) recommends a dermal absorption value of 0.14 (14%) for PCB Aroclors 1254/1242 and other PCBs, based on a study by Wester et al. (1993). A range of dermal absorption values is presented by ATSDR (2000). Review of these studies suggests that, while the data is limited, the value recommended by US EPA (2004) is adequately representative.

1.3.3 Inhalation of Dust

PCBs are not considered sufficiently volatile to be of significance and inhalation exposures associated with particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

1.3.4 Plant Uptake

PCBs accumulate in terrestrial vegetation by the following possible mechanisms: uptake from soil through the roots; dry deposition on aerial parts (particle-bound or gaseous); and wet deposition on aerial parts (particle-bound or solute).Where PCBs are sorbed to soil and organic matter, the potential for plant uptake is reduced; however, it remains of potential significance (CCME 1999). The uptake of PCBs (in soil) into edible fruit and vegetable crops has been the subject of a number of studies with a range of bioaccumulation factors derived for different crops (ATSDR 2000), with adsorption onto root surfaces most significant compared with translocation within the root or upper portions of the plant (CCME 1999). On this basis, the potential for the uptake of PCBs into home-grown produce has been considered in the derivation of an HIL A. This has been undertaken on the basis of the equations presented in Appendix B, with the following parameters and plant uptake factors estimated:

Parameter	Value	Reference/Comment
Parameters		
K_{oc}	131 000 (cm^3/g)	RAIS (2010) for Aroclor 1254
log K_{ow}	6.79	RAIS (2010) for Aroclor 1254
Diffusivity in water	$6.75x10^{-6}$ (cm^2/s)	RAIS (2010) for Aroclor 1221
Calculated Plant Uptake Factors (mg/kg produce fresh weight per mg/kg soil)		
Green vegetables	0.00026	calculated
Root vegetables	0.0038	calculated
Tuber vegetables	0.079	calculated
Tree fruit	0.00096	calculated

1.3.5 Intakes from Other Sources – Background

Background intakes (5.4 ng/kg/day for a child) were estimated by Di Marco & Buckett (1993) in the derivation of the previous HIL. Review of information available from FSANZ (2003) indicates that PCBs remain undetected in Australian and New Zealand food supplies, information consistent with that identified by Di Marco & Buckett (1993). Hence, intakes from food are considered negligible.

Intakes estimated by WHO (2003) are 0.3–3 ng/kg/day from air (including data derived from close-to-stack emissions from industrial/hazardous waste sources) and less than 0.2 ng/kg/day, from water. These values are similar to those noted above. Air concentrations reported by WHO (2003) from areas away from significant sources ranged from 0.002–0.95 ng/m^3 with PCBs in air noted to be slowly declining since the early 1980s. Based on these concentrations, intake of PCBs in air away from significant sources is approximately 0.3 ng/kg/day (the lower end of the range reported by WHO). Intakes estimated by RIVM (2001) are dominated by food (particularly where seafood dominates the diet), where the total intake is estimated to be 10 ng/kg/day. More recent review of intakes of PCBs from food by RIVM (2003) suggests that median lifelong intakes are estimated to be 5.6 ng/kg/day, similar to those estimated by Di Marco & Buckett (1993).

If the intakes estimated by WHO (2003) for air (away from significant sources) and water are considered relevant to current background intakes in Australia (where intakes from food are negligible), these comprise approximately 0.5 ng/kg/day, approximately 2.5% of the recommended oral TRV. These intakes are considered negligible.

1.4 Identification of Toxicity Reference Values

1.4.1 Classification

The International Agency for Research on Cancer (IARC 1987) has classified PCBs as Group 2A—probably carcinogenic to humans. This evaluation is based on limited evidence in humans (occupational studies) and sufficient evidence in experimental animals, where some PCBs (particularly those with greater than 50% chlorination) produced liver neoplasms in mice and rats after oral administration.

It is noted that US EPA has classified PCBs as Group B2—probable human carcinogen.

1.4.2 Review of Available Values/Information

PCBs have been associated with carcinogenic effects (in particular, hepatocarcinogenic effects have been seen in animals for PCBs with higher levels of chlorination) but the mode of action is of prime importance for determining the most appropriate dose–response approach to adopt for establishing an HIL. Review by WHO (2003) notes that the results of in vitro and in vivo genotoxicity studies on PCB mixtures are generally negative and suggest that PCB mixtures do not pose a direct genotoxic threat to humans. Although the mechanistic basis of the hepatocarcinogenicity of PCB mixtures in rodents is not clearly understood, it apparently is not due to genotoxicity. This is consistent with information provided by ATSDR (2000) and RIVM (2001).

On the basis of the available information, it is considered appropriate that a threshold dose–response approach be adopted for PCBs. The following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG	No evaluation available	
OCS (2012)	No evaluation	

Source	Value	Basis/Comments
	available	
International		
WHO (2003)	TDI = 0.00002 mg/kg/day	Derived on the basis of a LOAEL of 0.005 mg/kg/day for Aroclor 1254 associated with immunological effects in a 23-month study in monkeys, and an uncertainty factor of 300. WHO considers this TDI relevant to mixtures of PCBs.
WHO (2011)	No evaluation available	
RIVM (2001)	TDI = 0.00001 mg/kg/day TC = 0.0005 mg/m^3	TDI based on a LOAEL of 0.005 mg/kg/day for Aroclor 1254 associated with immunological effects in a 23-month study in monkeys, and an uncertainty factor of 270 (approx. 300). An additional factor of 2 has been applied that relates the TDI derived from Aroclor 1254 to that relevant to PCB mixtures, where the seven indicator PCBs are present in Aroclor 1254 between 40 and 50%. Hence the assessment of mixtures has been undertaking by assuming 50% of the TDI for Aroclor 1254. TC is based on a LOAEC (adjusted) of 0.3 mg/m^3 for Aroclor 1254 associated with marginal effects in experimental animals, and an uncertainty factor of 300. The additional 50% factor noted above is also applied to the Aroclor TC.
ATSDR (2000)	Oral MRL = 0.00002 mg/kg/day	Chronic oral MRL based on the same study as considered by RIVM and WHO (2003), with no additional adjustment for PCB mixtures. No inhalation MRL has been derived.
US EPA (IRIS 2012)	RfD = 0.00002 mg/kg/day	US EPA RfD (last reviewed in 1994) derived on the same basis as that presented by ATSDR and WHO (2003). US EPA also presents a non-threshold oral slope factor for PCBs which is not considered relevant in this assessment.

All the currently available oral threshold values for PCBs, based on Aroclor 1254, are derived from the same study with the only difference being the application of an additional factor by RIVM (2001) to address PCB mixtures. WHO (2003) considers that the available TDI for Aroclor 1254 is adequate to address PCB mixtures with no further adjustment. Hence the value derived by WHO (2003), also adopted by ATSDR and US EPA, is recommended for use in the derivation of a soil HIL.

Few inhalation-specific studies are available, with RIVM deriving an inhalation-specific value based on limited data. No dermal or inhalation-specific studies or data are available. As the data is limited and does not suggest the toxicity of PCBs is significantly different via inhalation, the oral TDI is recommended for the assessment of all pathways of exposure.

1.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for PCBs in the derivation of HILs

1.5 Calculated HILs

On the basis of the above, the following HILs have been derived for PCBs (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of	Ingestion of	Dermal	Inhalation

		Soil/Dust	Home-grown Produce	Absorption of Soil/Dust	(dust)
Residential A	1	19	46	35	<1
Residential B	1	12	--	88	<1
Recreational C	1	21	--	79	<1
Commercial D	7	9	--	91	<1

-- Pathway not included in derivation of HIL

1.6 References

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CCME 1999, *Polychlorinated Biphenyls (total), Canadian Soil Quality Guidelines for the Protection of Environmental and Human Health,* Canadian Council of Ministers of the Environment, 1999.

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WHO 2003, *Concise International Chemical Assessment Document 55, Polychlorinated Biphenyls*: Human Health Aspects, World Health Organization, Geneva.

2 Polybrominated Diphenyl Ethers (Br1 to Br9)

2.1 General

Polybrominated diphenyl ethers (PBDE) are a group of compounds manufactured for their flame retardant properties. They consist of two phenyl groups bound to a single oxygen atom with the hydrogen atoms on the phenyl groups substituted with between one and ten bromine atoms. The group consists of 209 congeners, which differ in the number and location of substituted bromine atoms. The internationally accepted numbering system for PBDE congeners is the acronym 'BDE' followed by a number from 1 to 209 (NICNAS 2007).

Several comprehensive reviews of PBDEs in the environment and their toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 2004; NICNAS 2007; UNEP 2009). The following provides a summary of the key aspects of these compounds that are relevant to the derivation of a soil HIL.

The literature to date indicates that the toxicity and environmental fate of PBDEs with a lower number of substituted bromine atoms (penta-BDE to hexa-BDE) is different from higher brominated BDEs (deca-BDE to BDE-209). Lower brominated BDEs have been demonstrated to be more toxic in animal studies, have a higher bioavailability and are more readily transported in the environment. As a result, ATSDR has recommended separating deca-BDE from lower brominated BDEs (ATSDR 2004). For the purpose of this assessment, lower brominated BDEs are considered to be BDEs containing between one and nine substituted bromines and it is these lower brominated BDEs for which HILs have been derived.

It is noted that the toxicity of higher BDEs is less certain, hence if significant levels of PBDE that include higher BDEs are present, a site-specific assessment should be conducted.

Further studies regarding the toxicity and environmental fate of lower brominated BDEs may result in this grouping being revised to a smaller proportion of significant congeners in future reviews.

PBDE are manufactured compounds, which have been widely used in industrial and consumer applications. A review of the compounds conducted by scientific and regulatory bodies has culminated in tetra- and penta-BDEs (components of technical penta-BDE) and hexa- and hepta-BDEs (components in technical octa-BDE) being listed as a Persistent Organic Pollutants (POPs) under the Stockholm Convention in May 2009 (UNEP 2009). All production and use of these compounds has subsequently been banned, with the exception of recycling activities (UNEP 2009). PBDEs are not manufactured in Australia but were historically imported and used until 2005 (NICNAS 2007). Importation of products pre-treated with PBDEs is expected to decrease following the recent ban. Technical penta-BDE was mainly used in polyurethane foams (such as in furnishings) whereas technical octa-BDE and deca-BDE were mainly used in hard plastics (such as for electrical equipment) (NICNAS 2007). The articles treated with PBDEs usually have long lives and, as such, articles containing PBDEs are still expected to be in use (NICNAS 2007). Deca-BDE was declared a priority existing chemical in Australia and is currently being assessed as to its environment and human health risks (NICNAS 2007).

2.2 Previous HIL

No previous HIL is available for lower BDEs (NEPC 1999).

2.3 Significance of Exposure Pathways

2.3.1 Oral Bioavailability

Insufficient data is available to adequately define the bioavailability of lower BDEs, hence a default approach of assuming 100% oral bioavailability has been adopted in the derivation of an HIL. It is noted that a site-specific assessment of bioavailability can be undertaken where required.

2.3.2 Dermal absorption

Insufficient data is available on the dermal absorption of lower BDEs from soil. Hence the default values of 0.1 (10%) suggested by US EPA (2004) for semi-volatile organic compounds has been adopted in the derivation of HILs.

It is noted that EU (2004) estimated a dermal absorption value of 1% as a maximum for deca-BDE, based on assumptions associated with the lipophillic nature of the compound and analogies to PCB. However, it is also noted in this review that dermal absorption may also be associated with accumulation in the stratum corneum, which may behave as a storage site, resulting in a low systemic release over time.

2.3.3 Inhalation of Dust

Lower BDEs are not considered sufficiently volatile to be of significance and inhalation exposures associated with dust particulates outdoors and indoors are expected to be of less significance than ingestion of soil. While likely to be negligible, potential inhalation exposures associated with dust have been considered in the HIL derived.

2.3.4 Plant Uptake

Limited data is available on the potential for lower BDEs to be taken up by plants from soil into edible fruit and vegetable crops. ATSDR notes that PBDEs will be strongly adsorbed to soil, hence PBDEs present in soil-pore water will bind to soil organic matter. Because PBDEs adsorb strongly to soil, they will have very low mobility, and leaching of PBDEs from soil to groundwater will be insignificant.

Review of plant uptake of deca-PBDE (BDE-209) into plants from soil by Huang et al. (2010) suggests that deca-BDE is taken up and translocated within the plants assessed (ryegrass, alfalfa, pumpkin, squash, maize and radish). Nineteen lower brominated (di- to nona-) PBDEs were detected in the soil and plant samples and five hydroxylated congeners were detected in the plant samples, indicating debromination and hydroxylation of BDE-209 in the soil−plant system. Evidence of a relatively higher proportion of penta- through to di-BDE congeners in plant tissues than in the soil indicates that there is further debromination of PBDEs within plants or lower brominated PBDEs are more readily taken up by plants.

On the basis of the available information, the potential for the uptake of lower BDEs into home-grown produce has been considered in the derivation of an HIL A. This has been undertaken on the basis of the equations presented in Appendix B with the following parameters and plant uptake factors estimated:

Parameter	Value	Reference/Comment
Parameters		
K_{oc}	1 698 000 (cm^3/g)	Refer to note below*
log K_{ow}	6.84	RAIS (2010) for penta-BDE (BDE-99)
Diffusivity in water	$5.32x10^{-6}$ (cm^2/s)	Estimated as per Guan et al. (2009)
Calculated Plant Uptake Factors (mg/kg produce fresh weight per mg/kg soil)		
Green vegetables	0.00026	calculated
Root vegetables	0.0038	calculated
Tuber vegetables	0.079	calculated

Parameter	Value	Reference/Comment
Tree fruit	0.00096	calculated

* The estimation of potential plant uptake of BDE is sensitive to the value of K_{oc} adopted. The data would normally be derived from RAIS (2010) for consistency; however, the data provided is only for penta-BDE with data from no other lower BDEs presented for comparison. Data presented in ATSDR (2001) suggests log K_{oc} ranges from 2.89–5.1 for penta-BDE and from 5.92–6.22 for octa-BDE. Review by Guan et al. (2009) provides log K_{oc} values for the lower BDEs (BDE-28 to BDE-208) that range from 5.73–6.49. Due to the range of values provided for the lower BDEs, the average of values presented by Guan et al. (2009), log K_{oc} = 6.23, has been adopted.

2.3.5 Intakes from Other Sources – Background

Background intakes were evaluated by NICNAS (2007) on the basis of PBDE levels in blood rather than as an intake. The presence of PBDEs in blood lipids indicates exposure by the general population; however, the data does not determine the major source of exposure. Data available from FSANZ (2007) suggests that dietary sources are likely to be low, therefore house dust may be the major source, but there is little correlation between exposure levels and house construction/contents. FSANZ notes a review by USA where dietary exposures did not explain the current body burden and exposures to hose dust were estimated to account for 82% of the total intake. Based on information presented in the available reviews, the following can be noted with respect to background intakes of PBDEs:

- A range of dietary intakes has been determined by FSANZ (2007) for all age groups. Estimated 95th percentile dietary intakes from FSANZ (2007) for a child aged 2–5 years ranged from 7 ng/kg/day (lower-bound) to 389 ng/kg/day (upper-bound). These intakes are consistent with data reported from other countries, including Canada and USA, and corresponded with a margin of exposure (MoE) of 300 or greater where a threshold of 0.1 mg/kg/day was considered. The MoE was greater for all other age groups considered in the study.
- PBDE in dust reported in indoor air in Australian buildings (Toms et al. 2006) ranged from 0.5–179 pg/m^3 for homes and 15–487 pg/m^3 for offices. Dust concentrations ranged from 87 ng/g–3070 ng/g. PBCEs were detected in 9 out of 10 surface wipe samples. No estimation of intake associated with measured levels in air and dust was presented. The study size was limited and showed dust levels similar to or lower than those conducted overseas in Canada and USA.
- Upper-bound total intakes of PBDEs from all sources (ambient and indoor air, dietary and dust) in Canada (Health Canada 2006) have been estimated to be approximately 0.95 µg/kg/day for children aged 0.5–4 years. Higher intakes (2.6 µg/kg/day) are noted for breastfed infants. Recent review of total intakes from food, dust and air of PBDEs in USA (Schecter et al. 2008) range from 1.2 ng/kg/day for adults to 307 ng/kg/day for infants.
- Based on the Australian data noted above, intakes by young children may range from 0.007–0.5 µg/kg/day. The higher value is half that estimated by Health Canada (2006), both of which exceed the recommended oral TRV.
- On the basis of the above, total intakes (and those reported from Australia) vary and may comprise a significant proportion of the recommended threshold value. Hence, consideration of 80% of the recommended TRV as background intakes is considered appropriate.

2.4 Identification of Toxicity Reference Values

2.4.1 Classification

The International Agency for Research on Cancer (IARC 1999) has classified technical deca-BDE as Group 3—not classifiable. No classification is available for other BDEs.

It is noted that US EPA has a classification for deca-BDE where it is classified as Group C—possible human carcinogen. US EPA has classified technical penta-BDE and technical octa-BDE as Group D—not classifiable.

2.4.2 Review of Available Values/Information

Review of PBDEs, in particular, penta-BDE and octa-BDE by NICNAS (2007), indicated there is insufficient information on the carcinogenic potential of these PBDEs, and that the overall conclusion relating to penta-BDE is that it is not genotoxic. Further review of octa-BDE, PBDE mixtures and penta-BDE (JECFA 2006) suggests that PBDE mixtures and individual congeners are not genotoxic. On the basis of the available information, it is considered appropriate that a threshold dose–response approach be adopted for PBDEs.

The following are available for the lower BDEs from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2004)	No evaluation available	
OCS (2012)	No evaluation available	
NICNAS (2007)	No ADI/TDI established	Based on review of PBDEs and available studies, the highest toxicity was associated with penta-BDE associated with neurodevelopmental effects in pups and dams where the LOAELs were 0.8 mg/kg/day in pups and 0.06 mg/kg/day in dams.
FSANZ (2007)	No ADI/TDI established	Review of dietary intakes considered a margin of exposure (MoE) approach where a threshold value of 0.1 mg/kg/day was considered, based on a review by JECFA.
International		
JECFA (2006)	No ADI/TDI established	Due to the complexity of PBDEs and the lack of adequate data, a provisional maximum tolerable daily intake or provisional tolerable weekly intake has not been derived for PBDEs. Limited data suggests that, for more toxic PBDE congeners, adverse effects would be unlikely to occur in rodents at doses less than approximately 0.1 mg/kg/day.
WHO (2011)	No evaluation available	
Health Canada (2006)	No ADI/TDI established	A threshold value of 0.8 mg/kg/day was identified for penta-BDE, based on neurobehavioural effects in neonatal mice, considered the critical effects and appropriate for undertaking a MoE approach to the assessment of risk.
ATSDR (2004)	No chronic duration MRLs derived	No chronic duration MRLs have been derived for lower brominated BDEs, due to insufficient data. An intermediate duration oral MRL of 0.007 mg/kg/day has been derived on the basis of a LOAEL of 2 mg/kg/day associated with liver effects in rats exposed to penta-BDE. An intermediate duration inhalation MRL of 0.006 mg/m^3 has been derived based on a NOAEL of 1.1 mg/m^3 for thyroid effects in rats exposed to commercial octa-BDE mixture.
US EPA	RfD = 0.0001	RfD established (in 2008) for BDE-99 (penta-BDE) on the

Source	Value	Basis/Comments
(IRIS 2012)	mg/kg/day for penta-BDE (BDE-99) RfD = 0.0002 mg/kg/day for hexa-BDE (BDE-153) RfD = 0.0001 mg/kg/day for tetra-BDE (BDE-47) RfD = 0.003 mg/kg/day for octa-BDE	basis of a benchmark dose approach and a $BMDL_{1SD}$ of 0.29 mg/kg/day associated with neurobehavioral effects in mice, and an uncertainty factor of 3000. Hexa-BDE RfD established (in 2008) for BDE-153 on the basis of a NOAEL of 0.45 mg/kg/day associated with neurobehavioral effects in mice, and an uncertainty factor of 3000. Tetra-BDE RfD established (in 2008) for BDE-47 on the basis of a benchmark dose approach and a $BMDL_{1SD}$ of 0.35 mg/kg/day associated with neurobehavioral effects in mice, and an uncertainty factor of 3000. Octa-BDE RfD (established in 1986) for octa-BDE based on a NOAEL of 2.51 mg/kg/day associated with liver effects in rats, and an uncertainty factor of 1000. Note the US EPA (2008) review established an RfD = 0.007 mg/kg/day for deca-BDE (BDE-209), based on a NOAEL of 2.22 mg/kg/day associated with neurobehavioral effects in mice, and application of a 300-fold uncertainty factor. While not part of the lower-BDEs evaluated for the derivation of the soil HIL, this evaluation indicates that deca-BDE is less toxic than the lower BDEs.

Limited quantitative data is available for the characterisation of chronic exposures to lower BDEs. The more recent evaluations by US EPA (IRIS 2012) for individual congeners BDE-99, BDE-153 and BDE-47 have considered threshold values (BMDLs or NOAELs) that are consistent with those identified in reviews by NICNAS (2007), JECFA (2006) and Health Canada (2006), that are associated with the more sensitive end point of neurobehavioral/developmental effects. These end points are more sensitive than those considered by ATSDR in the derivation of intermediate duration MRLs and considered in older reviews by US EPA for penta-BDE and octa-BDE. The uncertainty factor applied by US EPA to the individual congeners considered, 3000, includes an additional 10-fold factor to address database deficiencies.

There is no evaluation of a chronic threshold value that would be applicable to all lower BDEs as a group, hence application of the US EPA values requires an assumption that the congeners studied are an appropriate indicator for total lower BDEs. This is likely to be conservative but no more detailed evaluations are available. The individual congener studies by US EPA are noted by NICNAS (2007) to be those within commercial penta-BDE that are of most importance in biomonitoring and environmental sampling.

The lower RfD of 0.0001 mg/kg/day derived by US EPA for BDE-99 and BDE-47, similar to that derived for BDE-153, is recommended for use in the derivation of a soil HIL for lower BDEs. As noted in most other reviews, the available database is poor and limited with respect to identification of a threshold associated with chronic exposures to the group of congeners. Hence, the use of this threshold TRV requires further review and update in the future when further studies are undertaken.

No dermal or inhalation-specific chronic studies or data are available. For the presence of lower BDEs in soil, it is considered appropriate to consider use of the available threshold value for all pathways of exposures.

2.4.3 Recommendation

On the basis of the discussion above, the following toxicity reference values (TRVs) have been adopted for lower BDEs in the derivation of HILs:

Recommendation for Lower BDEs

Oral TRV (TRV_O) = 0.0001 mg/kg/day (US EPA (IRIS 2012)) for BDE-99 and BDE-47) for all pathways of exposure

Dermal absorption factor (DAF) = 0.1 (or 10%) (US EPA 2004)

Intakes allowable from soil (as % of TRV) = 20%

Background intakes from other sources (as % of TRV):

BI_O = 80% for oral and dermal intakes

BIi = 80% for inhalation

Uptake in home-grown produce considered in derivation of HIL A.

2.5 Calculated HILs

On the basis of the above, the following HILs have been derived for lower BDEs (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	HIL (mg/kg)	Percentage Contribution from Exposure Pathways			
		Ingestion of Soil/Dust	Ingestion of Home-grown Produce	Dermal Absorption of Soil/Dust	Inhalation (dust)
Residential A	1	39	8	53	<1
Residential B	2	16	--	84	<1
Recreational C	2	27	--	73	<1
Commercial D	10	12	--	88	<1

-- Pathway not included in derivation of HIL

2.6 References

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US EPA, 2004, Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment), Final, EPA/540/R-99/005, OSWER 9285.7-02EP, July 2004.

3 Shortened forms

ADI	acceptable daily intake
ADWG	Australian Drinking Water Guidelines
AI	adequate intake
ANZECC	Australia and New Zealand Environment and Conservation Council
APVMA	Australian Pesticides and Veterinary Medicines Authority
ATDS	Australian Total Diet Survey
ATSDR	Agency for Toxic Substances and Disease Registry
BA	bioavailability
BI	background intake
BMD	benchmark dose
BMDL	Benchmark dose lower confidence limit
CCME	Canadian Council of Ministers of the Environment
CICAD	Concise International Chemicals Assessment Document
CNS	central nervous system
DAF	dermal absorption factor
DW	dry weight
EA	Environment Agency (England and Wales)
EHC	Environmental Health Criteria
EPA	Environment Protection Authority
FSANZ	Food Standards Australia and New Zealand
GAF	gastrointestinal absorption factor
HCB	hexachlorobenzene
HEC	human equivalent concentration
HED	human equivalent dose
HIARC	Hazard Identification Assessment Review Committee
HIL	health investigation level
HSDB	Hazardous Substances Data Bank
HSL	health screening level
IARC	International Agency for Research on Cancer
IEUBK	Integrated exposure uptake biokinetic model

IRIS	Integrated Risk Information System
JECFA	Joint FAO/WHO Expert Committee on Food Additives
JMPR	WHO/FAO Joint Meeting on Pesticide Residues
LOAEL	lowest observed adverse effect level
LOEL	lowest observed effect level
MF	modifying factor
MoA	mode (or mechanism) of action
MoE	margin of exposure
MRL	maximum residue limit
MRL	minimal risk level
NDI	negligible daily intake
NEPC	National Environment Protection Council
NEPM	National Environment Protection Measure
NHMRC	National Health and Medical Research Council
NOAEL	no observable adverse effect level
NOEL	no observable effect level
NSW DECC	New South Wales Department of Environment and Climate Change
OCS	Office of Chemical Safety
PBDE	polybrominated diphenyl ether
POP	persistent organic pollutant
PTDI	provisional tolerable daily intake
PTMI	provisional tolerable monthly intake
PTWI	provisional tolerable weekly intake
RAIS	Risk Assessment Information System
RDI	recommended daily intake
REL	reference exposure level
RfC	reference concentration
RfD	reference dose
RME	reasonable maximum exposure
SF	slope factor
TC	tolerable concentration

TD	tumorigenic dose
TDI	tolerable daily intake
TRV	toxicity reference value
UF	uncertainty factor
UL	upper limit
UR	unit risk
US EPA	United States Environmental Protection Agency
WHO	World Health Organization
WHO DWG	World Health Organization Drinking Water Guidelines

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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This compilation has been split into 22 volumes

Volume 1: sections 1–6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

Schedule B7

APPENDIX A6

The Derivation of Interim HILs for Volatile Organic Chlorinated Compounds

Contents
The derivation of interim HILs for volatile organic chlorinated compounds

1 Trichloroethene (TCE)

1.1 General

Several comprehensive reviews of trichloroethene (TCE) in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 1997; WHO 1985; EU 2004; CCME 2007; NICNAS 2009; US EPA 2011). The following provides a summary of the key aspects of TCE that are relevant to the derivation of interim HILs.

TCE is a colourless, non-flammable, volatile liquid, with a characteristic slightly sweet odour. Most people can begin to smell TCE in air at a concentration of 100 ppm (ATSDR 1997).

TCE was not thought to occur naturally in the environment until the recent discovery in 1995 that several species of marine macro-algae and at least one species of micro-algae produce the compound. The importance of this release and potential exposure route is not currently known. TCE is mainly used as an industrial solvent in a variety of industries, primarily metal degreasing and cleaning operations. TCE can also be found in some household products, including correction fluid, paint removers, adhesives, and spot removers. TCE has also been used as a carrier solvent for the active ingredients of insecticides and fungicides; as a solvent for waxes, fats, resins, and oils; and as an anaesthetic for medical and dental use. It has also been used to extract spice oleoresins and caffeine from coffee (ATSDR 1997; WHO 1985).

TCE was manufactured in Australia for approximately 30 years from the early 1950s to the early 1980s. At present, the Australian market demand for TCE is entirely met by imports of the chemical. TCE is used widely in both large and small industries, mainly as a degreasing agent (NICNAS 2009).

If released into the environment, the following can be noted with respect to TCE (WHO 1985):

- Air – TCE is expected to remain in the vapour phase. Removal is primarily through reaction with hydroxyl radicals to produce low levels of phosgene, dichloroacetyl chloride, formyl chloride and other degradation products. The half-life of TCE varies from 1 day to months.
- Soil and Water – TCE is expected to volatilise from surface soils and water. TCE may leach through soil into groundwater where it may persist for years, depending on conditions.
- Water – Depending on conditions, reductive dehalogenation to vinyl chloride may occur. Under anaerobic conditions TCE can be intrinsically biodegraded to form dichloroethene (1,1-DCE and isomers of 1,2-DCE) and vinyl chloride.

1.2 Previous HIL

No previous HIL is available for TCE (NEPC 1999).

1.3 Proposed Interim HIL

Review of available information in relation to the presence of TCE in soil indicates that the vapour inhalation pathway is the most significant/important. This pathway should be assessed on the basis of measured vapour data, in particular, soil vapour data. There are significant limitations in the derivation of a soil HIL, in particular, the modelling of phase partitioning from soil to soil vapour and the field measurement of volatiles in soil, hence an interim HIL has been derived for soil vapour only.

The following presents the values adopted for the calculation of a soil vapour interim HIL. In addition, other information that is relevant to the assessment of TCE in soil (relevant to other pathways of exposure) is presented.

1.4 Significance of Exposure Pathways

1.4.1 Inhalation

TCE is a volatile compound and, as such, the derivation of the HIL has considered the vapour inhalation pathway as the most significant. The approach adopted for the quantification of potential vapour migration to outdoor air and intrusion indoors is outlined in the main text of Schedule B7. Due to limitations with the vapour modelling approach adopted, the HILs derived are considered interim.

The inhalation of particulates outdoors and indoors is considered essentially insignificant, compared with vapour inhalation.

1.4.2 Dermal absorption

Insufficient data is available on the dermal absorption of TCE from soil. Given the volatility of the compound, dermal absorption is expected to be low, however, as there is insufficient data available to further assess dermal absorption from soil, a default value of 0.03 (3%) has been adopted for the volatile organic compounds (US EPA 1995).

1.4.3 Plant Uptake

Limited data is available on the potential for TCE to be taken up by home-grown produce. According to ATSDR (1997), TCE has been detected in small amounts in fruits and vegetables, suggesting a potential for limited phytoaccumulation. Laboratory studies with carrot and radish plants and radioactively labelled TCE (Schroll et al. 1994) showed some uptake, though it is noted that the experiment indicated that uptake occurred mainly through the foliage (from the air) as opposed to the roots in these plants (with subsequent translocation throughout the plant tissues). Schnabel et al. (1997) looked at the uptake of TCE in edible garden plants (carrots, spinach and tomatoes) and identified that TCE, when taken up, was transformed and bound to plant tissues in a form that was less toxic than the parent compound.

On the basis of the above, the use of the more commonly adopted equations for quantifying plant uptake (as presented in the text of Schedule B7) that do not address uptake of volatiles (from air) rather than the root, or transformations within the plant, are not considered appropriate and relevant for the assessment of TCE.

It is expected that the potential for plant uptake will be of less significance in the derivation of a soil HIL, when compared with the assessment of vapour inhalation, and given the limitations involved in providing a meaningful evaluation of plant uptake, it has not been considered in the derivation of HILs.

1.4.4 Intakes from Other Sources – Background

As TCE is highly volatile, background intakes will be dominated by inhalation exposures. Concentrations of TCE in industrial, urban and regional areas are available in Australia. Data collected in NSW (DEC 2003) from urban and regional areas in NSW report average concentrations of TCE of approximately 0.1 ppbv (0.0005 mg/m^3), close to the analytical limit of reporting with most samples noted to be not detected, with a maximum concentration in the Sydney CBD of 3.6 ppbv (0.019 mg/m3). Concentrations in an industrial area in Brisbane (Hawas et. al. 2001) have been reported with average and maximum concentrations of 0.0002 mg/m3 (also close to the limit of reporting) and 0.0005 mg/m3 respectively. Background air concentrations in Canada (CCME 2007) are considered to be approximately 0.0014 mg/m3, consistent with the range reported by DEC (2003). Background intakes (dominated by inhalation) were estimated by WHO (2011) to be approximately 0.04 µg/kg/day for children and 0.01 µg/kg/day for adults. Based on average concentrations reported in NSW and in Brisbane, intakes by young children are estimated to be approximately 0.3 µg/kg/day. These intakes comprise approximately 10% of the recommended inhalation TRVs for non-carcinogenic effects. It is noted that other sources found indoors (from a wide range of common products) are likely to be

present and may contribute more significantly to background exposures. These sources may need to be addressed on a site-specific basis.

1.5 Identification of Toxicity Reference Values

1.5.1 Classification

The International Agency for Research on Cancer (IARC 1995) has classified TCE as Group 2A—probably carcinogenic to humans.

Review by US EPA (2011) characterised TCE as carcinogenic in humans by all routes of exposure. This conclusion is based on convincing evidence of a causal association between TCE exposure in humans and kidney cancer. The human evidence of carcinogenicity from epidemiologic studies of TCE exposure is strong for non-Hodgkin Lymphoma but less convincing than for kidney cancer, and more limited for liver and biliary tract cancer. Less human evidence is found for an association between TCE exposure and other types of cancer, including bladder, oesophageal, prostate, cervical, breast, and childhood leukaemia. Further support is derived from positive results in multiple rodent bioassays, similar toxicokinetics between rodents and humans, mechanistic data supporting a mutagenic mode of action for kidney tumours.

1.5.2 Review of Available Values/Information

Some epidemiological studies indicate a possible association between exposure to TCE and an increased cancer risk, with IARC (1995) noting elevated risk for cancer of the liver and biliary tract and a modestly elevated risk for non-Hodgkin's lymphoma in three cohort studies. In animals, TCE induces tumours at several sites and in different species. Tumours have been seen in mouse liver and lung and rat kidney and testis. On the basis of the available information, most current reviews by IARC (1995), WHO (2011), CCME (2007) and US EPA (2011) consider TCE to be carcinogenic (with responses tending to increase with dose), via all routes of exposure.

The potential mode of action (MoA) for TCE is reviewed and discussed in the current WHO DWG (2011) and US EPA (2011) review.

The WHO DWG (2011) review concluded that the MoA for tumour induction by TCE may be attributed to non-genotoxic processes (related to cytotoxicity, peroxisome proliferation and altered cell signalling); genotoxic processes, (such as the production of genotoxic metabolites (e.g., chloral and DCVC[1])); or the production of reactive oxygen species related to peroxisomal induction in the liver. The potential role of several mutagenic or carcinogenic metabolites of TCE cannot be ignored. Hence TCE appears to be at least weakly genotoxic and evaluation of carcinogenicity on the basis of a non-threshold approach is considered appropriate (as is undertaken in the current WHO DWG (2011) and WHO Air Quality Guidelines (2000)).

The most recent US EPA review (2011) provides a detailed assessment of genotoxicity (of TCE and metabolites) and mutagenicity. With respect to genotoxicity, although it appears unlikely that TCE, as a pure compound, causes point mutations, there is evidence for TCE genotoxicity with respect to other genetic end points, such as micronucleus formation. In addition, several TCE metabolites have tested positive in genotoxicity assays. It is noted that uncertainties with regard to the characterisation of TCE genotoxicity remain, particularly because not all TCE metabolites have been sufficiently tested in the standard genotoxicity screening battery to derive a comprehensive conclusion. However, the metabolites that have been tested, particularly DCVC, have predominantly resulted in positive data, supporting the conclusion that these compounds are genotoxic.

[1] DCVC is the abbreviation for the metabolite S-(1,2-dichlorovinyl)-L-cysteine.

The MoA relevant to specific target organs in laboratory animals has been reviewed by US EPA. Only in the case of the kidney is it concluded that the data is sufficient to support a particular MoA being operative. For the kidney, the predominance of positive genotoxicity data in the database of available studies of TCE metabolites, together with toxicokinetic data, supports the conclusion that a mutagenic MoA is operative in TCE-induced kidney tumours. Hence a linear (non-threshold) approach is recommended for the quantification of carcinogenic effects.

There is some evidence that certain populations may be more susceptible to exposure to TCE. Because the weight of evidence supports a mutagenic MoA being operative for TCE carcinogenicity in the kidney, and there is an absence of chemical-specific data to evaluate differences in carcinogenic susceptibility, early-life susceptibility is recommended by US EPA to be assumed and the age-dependent adjustment factors (ADAFs) should be applied.

On the basis of the above, it is reasonable to consider a non-threshold approach for the assessment of carcinogenicity in relation to TCE. It is noted that a number of guidelines (such as WHO 2011) have been derived on the basis of both carcinogenic and non-carcinogenic end points, with non-carcinogenic end points noted to be more sensitive for at least oral intakes. Hence both non-threshold and threshold reference values available have been noted in the following.

The following quantitative values are available for TCE from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	No health-based value derived	Not derived due to insufficient data.
International		
WHO (2011)	SF = 0.00078 $(mg/kg/day)^{-1}$ TDI = 0.00146 mg/kg/day	The WHO guideline of 0.02 mg/L is based on the lower value derived from carcinogenic and non-carcinogenic end points. It is noted that the guideline derived on the basis of reproductive/developmental (threshold) effects was most conservative. The oral slope factor adopted is from Health Canada (range of values derived) and based on combined tubular cell adenomas and adenocarcinomas of the kidneys in rats following oral exposure to TCE for 103 weeks and a linear multistage model. The oral TDI derived from a $BMDL_{10}$ of 0.146 mg/kg/day associated with reproductive/developmental effects in rats, and an uncertainty factor of 100.
WHO (2000 and 2010)	UR = $4.3x10^{-7}$ $(\mu g/m^3)^{-1}$	Inhalation unit risk derived on the basis of Leydig-cell tumours in the testes of rats and a linear multistage model. Inhalation unit risk from rat study is supported by a similar unit risk of 9 $x10^{-7}$ $(\mu g/m^3)^{-1}$ derived from increased incidence of hepatic tumours in a cohort study of occupationally exposed adults. The non-threshold approach was adopted by the WHO as TCE was considered genotoxic and carcinogenic.
EU (2004)	SF = 0.0019 $(mg/kg/day)^{-1}$	TCE gives rise to concern for humans owing to possible mutagenic and carcinogenic effects and because it is not possible to identify a threshold exposure level below which these effects would not be expressed. For non-carcinogenic effects, the most sensitive threshold effect evaluated was associated with CNS disturbance following repeated dose where a NOAEL of 38

Source	Value	Basis/Comments
		mg/kg/day was considered. The EU has presented a calculation of lifetime cancer risk based on the T25 method in relation to non-Hodgkin lymphoma. From an inhalation study in female mice a HT25 dose descriptor for humans was derived as 130 mg/kg/day. Following the approach presented, the EU calculated increased cancer risk for TCE for all groups using an equivalent slope factor of 0.0019 $(mg/kg/day)^{-1}$. This value was used in the quantification of risk associated with exposure from oral, dermal and inhalation pathways for workers, consumers and environmental exposures.
Health Canada (2005)	SF = 0.000811 $(mg/kg/day)^{-1}$ UR = $1.2x10^{-7}$ $(\mu g/m^3)^{-1}$ TDI = 0.00146 mg/kg/day	Oral slope factor derived on the basis of the same study noted in WHO (2011), however a slightly different value is quoted. Inhalation unit risk based on renal adenocarcinomas in rats following inhalation exposures for 104 weeks in males (a lower, less conservative value was derived for females). Note that the derivation of drinking water guidelines has also considered the oral TDI noted in the WHO DWG which results in a lower guideline than is derived on the basis of the oral slope factor.
CCME (2007)	SF = 0.000811 $(mg/kg/day)^{-1}$ UR = $6.4x10^{-7}$ $(\mu g/m^3)^{-1}$ TDI = 0.00146 mg/kg/day TC = 0.04 mg/m^3	Slope factor based on same study noted by Health Canada (2005). Inhalation unit risk based on older evaluation from Health Canada where a TC_{05} (concentration expected to cause a 5% incidence in cancer) of 0.082 mg/m^3 and extrapolation based on an excess lifetime cancer risk of 10^{-6}. TDI and TC values also presented for non-carcinogenic end points. TDI as noted by WHO DWG TC adopted from the former US EPA RfC (currently withdrawn pending finalisation of the 2009 draft) associated with effects on the CNS, kidney, liver and endocrine system in inhalation studies where a point of departure (POD) of 38 mg/m^3 was identified, and an uncertainty factor of 1000 adopted.
RIVM (2001)	PTDI = 0.05 mg/kg/day PTC = 0.2 mg/m^3	Provision threshold values derived for TCE due to limited data and an assumption that the genotoxic mechanism for TCE (numerical chromosome aberration *in vivo*) exhibits a threshold. The basis for these values is not listed here as the evaluation is considered dated.
ATSDR (1997)	No chronic MRLs derived	No chronic oral or inhalation MRL has been established.
New York State (NYS DH 2006)	GV = 0.005 mg/m^3	An air guideline value (GV) of 0.01 mg/m^3 was derived for non-carcinogenic effects (CNS effects in humans) is based on review of all available studies and associated end points. The lowest guideline value has been adopted and is noted to be protective of the general population including sensitive life stages of infants, children, the infirm and elderly. The GV resulted in carcinogenic risk estimates at the lower end of the risk range ($1x10^{-6}$ to $1x10^{-4}$). The guideline value was then reduced by a factor of 2 based on the consideration of additional factors (data gaps, concern regarding methods for

Source	Value	Basis/Comments
		evaluating risks to children and concerns regarding human carcinogenicity) in addition to background levels and analytical capabilities. The resulting air guideline derived was 0.005 mg/m^3.
US EPA (2011)	SF = 0.05 $(mg/kg/day)^{-1}$ UR = $4x10^{-6}$ $(\mu g/m^3)^{-1}$ RfD = 0.0005 mg/kg/day RfC = 0.002 mg/m^3	Oral slope factor based on PBPK model-based route-to-route extrapolation from the inhalation value based on human kidney cancer risks. The value is also supported by data from oral bioassays. Inhalation unit risk derived on the basis of non-Hodgkin's lymphoma , renal cell carcinoma and liver tumours from a human inhalation (epidemiology) studies, adjusted (by a factor of 4) to address potential risk of tumours at multiple sites. The value is derived from linear extrapolation from the point of departure (LEC01). It is noted that even with the consideration of the 4-fold factor, the inhalation unit risk value derived is within the range of values derived from a wide range of studies. Application of the ADAF for kidney cancer risks due to evidence supporting a mutagenic MoA is recommended. RfD based on critical effects of heart malformations (rats), adult immunological effects (mice) and developmental immunotoxicity (mice), which is further supported by an oral study for the toxic nephropathy (rats) and route extrapolation from an inhalation study. RfC based on route-extrapolation from and oral studies for the critical effects of heart malformations in rats and immunotoxicity in mice, and incorporation of uncertainty factors ranging from 10 to 100.

For TCE the health end points associated with carcinogenic (non-threshold) and non-carcinogenic (threshold) effects are similar in sensitivity. Hence it is appropriate that the derivation of a guideline consider all relevant end points to ensure that the value derived is adequately protective of all effects.

Many of the reviews conducted by WHO (2011), CCME (2007), RIVM (2001) and ATSDR (1997) have considered limited and dated databases of information (as noted). The most recent comprehensive review of TCE toxicity has been conducted by US EPA (2011), where the most recent studies and health end points have been addressed. The more recent review by WHO (2010), in relation to inhalation toxicity, considered some of the more recent studies, though the review has not considered non-carcinogenic end points, and the key studies considered by US EPA (2011) for the derivation of the inhalation unit risk were not considered in the WHO (2010) review. On this basis it is considered appropriate that the more recent evaluation conducted by US EPA (2011) be used for the purpose of establishing soil vapour Interim HILs.

The US EPA review has concluded that there is sufficient weight of evidence that TCE operates through a mutagenic mode of action (MoA) for kidney tumours and there is a lack of TCE-specific quantitative data in relation to early lifetime susceptibility. Hence it is appropriate to consider increased susceptibility associated with early lifetime exposures through the adjustment of exposure factors. This adjustment, however is noted to be relevant to the kidney cancer component of the total risk (note the inhalation unit risk includes a factor of 4-fold to address the risk of tumours at multiple sites). The effect of considering theses age-adjusted exposure factors to only the kidney cancer portion of the unit risk has been evaluated by US EPA and determined to be of minimal impacts to the total cancer risk, except when exposure only occurs during early life (if these effects occur). In addition to this evaluation, a number of uncertainties have been identified in relation to applying the age

adjustment factors for a more complex carcinogenic MoA, as identified for TCE. Hence, for the purpose of deriving HILs where long-term exposures are considered, no further adjustments to account for potential early lifetime susceptibility have been incorporated.

1.5.3 Recommendation

In relation to TCE, only soil vapour Interim HILs have been derived. Hence only the inhalation pathway has been quantified in the development of these HILs. On the basis of the discussion above, the following inhalation toxicity reference values (TRVs) have been adopted for TCE:

Recommendation for TCE (quantitative inhalation toxicity values)

Carcinogenic end points evaluated on the basis of:

Inhalation TRV (TRV_I) = 0.004 $(mg/m^3)^{-1}$ (US EPA 2011)

Non-Carcinogenic end points evaluated on the basis of:

Inhalation TRV (TRV_I) = 0.002 mg/m^3 (US EPA 2011)

Background intakes from other sources (as % of TRV):

BIi = 10% for inhalation

1.6 Calculated Interim HILs

Based on the evaluation presented above, a range of approaches has been identified for the quantification of exposure and toxicity. The following comments relate to the derivation of the interim soil vapour HIL A (also note the methodology and assumptions adopted, as outlined in the text of Schedule B7):

- The calculated interim soil vapour HIL for TCE on the basis of the adopted threshold TRVs noted above is 0.02 mg/m^3.
- The calculated interim soil vapour HIL for TCE on the basis of the adopted non-threshold TRVs noted above is 0.06 mg/m^3.

The most sensitive end point for the derivation of the interim soil vapour HIL is the assessment of threshold (non-carcinogenic) effects.

On the basis of the above, the following interim soil vapour HILs have been derived for TCE (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	Interim Soil Vapour HIL# (mg/m^3)
Residential A	0.02
Residential B	0.02
Recreational C	0.4
Commercial D	0.08

Interim soil gas HILs are conservative soil gas concentrations that can be adopted for the purpose of screening sites where further investigation is required on a site-specific basis. They are based on the potential for vapour intrusion indoors using an indoor air-to-soil gas attenuation factor of 0.1 for HILs A, B and D and an outdoor attenuation factor of 0.05 for HIL C.

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2 1,1,1-Trichloroethane

2.1 General

Several comprehensive reviews of 1,1,1-trichloroethane (1,1,1-TCA) in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 1997; ATSDR 2006; WHO 1990). The following provides a summary of the key aspects of 1,1,1-TCA that is relevant to the derivation of interim HILs.

1,1,1-TCA is a synthetic chemical that does not occur naturally in the environment. It is a colourless, volatile liquid, with a characteristic sharp, sweet odour, and a vapour that is denser than air. It is slightly soluble in water, and is found in a number of solvents in a variety of domestic and industrial uses. 1,1,1-TCA is typically non-flammable under normal conditions however, at higher vapour concentrations (10 %), it can burn when it contacts a spark (ATSDR 1997).

No natural sources of 1,1,1-TCA have been identified. 1,1,1-TCA is a chlorinated hydrocarbon which is manufactured from vinyl chloride by chlorination. 1,1,1-TCA had many industrial and household uses, however its production has been limited to essential industrial use and is to be phased out due to its effects on the ozone layer (ATSDR 1997). It is widely used as a cleaning solvent, and is used to clean electrical equipment, motors, electronic components, printed circuit boards, photographic film and various metal and plastic parts. It is also used as a lubricant in metal-cutting oils and as a component in inks, correction fluid and drain cleaners (NHMRC 2011).

2.2 Previous HIL

No previous HIL is available for 1,1,1-TCA (NEPC 1999).

2.3 Proposed Interim HIL

Review of available information in relation to the presence of 1,1,1-TCA in soil indicates that the vapour inhalation pathway is the most significant/important. This pathway should be assessed on the basis of measured vapour data, in particular, soil vapour data. There are significant limitations in the derivation of a soil HIL, in particular the modelling of phase partitioning from soil to soil vapour and the field measurement of volatiles in soil. Hence an interim HIL has been derived for soil vapour only.

The following presents the values adopted for the calculation of a soil vapour interim HIL. In addition other information that is relevant to the assessment of 1,1,1-TCA in soil (relevant to other pathways of exposure) is presented.

2.4 Significance of Exposure Pathways

2.4.1 Inhalation

1,1,1-TCA is a volatile compound and, as such, the derivation of the HIL has considered the vapour inhalation pathway. The approach adopted for the quantification of potential vapour migration to outdoor air and intrusion indoors is outlined in Schedule B7. It is noted that the derived HIL is dominated by the assessment of these pathways of exposure. Due to limitations with the vapour modelling approach adopted, the HILs derived are considered interim.

The inhalation of particulates outdoors and indoors is considered essentially insignificant, compared with vapour inhalation.

2.4.2 Dermal absorption

Insufficient data is available on the dermal absorption of 1,1,1-TCA from soil. Given the volatility of the compound, dermal absorption is expected to be low though, as there is insufficient data available

to further assess dermal absorption from soil, a default value of 0.03 (3%) has been adopted for the volatile organic compounds (US EPA 1995).

2.4.3 Plant Uptake

No data is available on the potential for 1,1,1-TCA to be taken up by home-grown produce. Given the volatility of this compound, the potential for plant uptake is expected to be similar to that of TCE, which was considered to be limited. As with the assessment presented for TCE, the use of the more commonly adopted equations for quantifying plant uptake (as presented in the text of Schedule B7) that do not address uptake of volatiles (from air) rather than the root or transformations within the plant, are not considered appropriate and relevant for the assessment of 1,1,1-TCA.

It is expected that the potential for plant uptake will be of less significance in the derivation of an HIL, when compared with the assessment of vapour inhalation, and given the limitations involved in providing a meaningful evaluation of plant uptake, it has not been considered in the derivation of HILs.

2.4.4 Intakes from Other Sources – Background

As 1,1,1-TCA is highly volatile and not persistent, background intakes will be dominated by inhalation exposures. TCA has been reported in sampling undertaken in urban, suburban and industrial areas in NSW (DEC 2003) where the average concentration reported was 0.1 ppbv (0.5 $\mu g/m^3$) and the maximum reported in Beresfield was 0.3 ppbv (1.6 $\mu g/m^3$). Concentrations of 1,1,1-TCA in industrial air in Brisbane (Hawas et al. 2001) were similar (mean of 0.15 ppbv and maximum of 0.4 ppbv). These concentrations are lower than the average urban concentration assumed by ATSDR (2006) of 1 ppbv. Indoor air sources may also be significant; however, there are no estimates of exposure or intake from these sources.

Based on the recommended inhalation TRV for 1,1,1-TCA, these concentrations are essentially negligible.

It is noted that other sources found indoors (from a wide range of common products) are likely to be present and may contribute more significantly to background exposures. These sources need to be addressed on a site-specific basis.

2.5 Identification of Toxicity Reference Values

2.5.1 Classification

The International Agency for Research on Cancer (IARC 1999) has classified 1,1,1-TCA as Group 3—not classifiable.

Review by US EPA (2007) noted that for 1,1,1-TCA there is ‘inadequate information to assess carcinogenic potential’.

2.5.2 Review of Available Values/Information

There is insufficient data available to determine carcinogenicity of 1,1,1-TCA (WHO 2011, ATSDR 2006 and US EPA 2007). Review by US EPA (2007) has noted that 1,1,1-TCA has been tested extensively for genotoxic potential. The chemical has shown little capacity to produce genotoxic effects in bacteria or fungi. Results in mammalian test systems in vitro and in vivo were more mixed, but still predominantly negative for assays other than cell transformation. The chemical has been shown to interact weakly with DNA. The overall weight of evidence suggests that 1,1,1-TCA is not considered genotoxic.

On the basis of the available information, it is considered appropriate that a threshold dose–response approach be adopted for 1,1,1-TCA. Few quantitative toxicity values are available but the following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	No guideline established	No guideline established in current ADWG (NHMRC 2011) due to insufficient data.
International		
WHO (2011)	TDI = 0.6 mg/kg/day	No guideline is established as 1,1,1-TCA concentrations in drinking water are well below those of health concern. The review notes that a health-based guideline of 2 mg/L can be derived based on a TDI of 0.6 mg/kg/day based on a NOAEL of 600 mg/kg associated with liver and kidney effects from a short-duration oral study in rats, and an uncertainty factor of 1000.
RIVM (1993)	MPC = 4.8 mg/m^3	Maximum permissible concentration (MPC) in air derived on the basis of a duration corrected NOAEL of 482 mg/m^3 associated with liver effects in a 2-year rat inhalation study, and an uncertainty factor of 100.
ATSDR (2006)	No chronic MRLs derived	
US EPA (IRIS 2012)	RfD = 2 mg/kg/day RfC = 5 mg/m^3	Oral reference dose (RfD, last reviewed in 2007) of 2 mg/kg/day derived on the basis of a benchmark approach with a BMDL10 of 2155 mg/kg/day associated with reduced body weight in a 90-day mouse study, and an uncertainty factor of 1000 (including 3 for database deficiencies). RfC (last reviewed in 2007) derived on the basis of a NOAEL (HEC) of 1553 mg/m^3 associated with liver effects in mice and rats, and an uncertainty factor of 100.

In relation to inhalation exposures (the only pathway considered in development of soil vapour Interim HILs) the most recent review conducted by US EPA (which is consistent with the older review from RIVM) has been adopted.

2.5.3 Recommendation

In relation to 1,1,1-TCA, only soil vapour Interim HILs have been derived. Hence only the inhalation pathway has been quantified in the development of these HILs. On the basis of the discussion above, the following inhalation toxicity reference values (TRVs) have been adopted for 1,1,1-TCA:

Recommendation for 1,1,1-TCA

Inhalation TRV (TRV_I) = 5 mg/m^3 (US EPA)

Background intakes from other sources (as % of TRV):

BIi = 0% for inhalation

2.6 Calculated Interim HILs

On the basis of the above, the following interim soil vapour HILs have been derived for 1,1,1-TCA (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	Interim Soil Vapour HIL# (mg/m^3)
Residential A	60
Residential B	60
Recreational C	1200
Commercial D	230

Interim soil gas HILs are conservative soil gas concentrations that can be adopted for the purpose of screening sites where further investigation is required on a site-specific basis. They are based on the potential for vapour intrusion indoors using an indoor air-to-soil gas attenuation factor of 0.1 for HILs A, B and D and an outdoor attenuation factor of 0.05 for HIL C.

2.7 References

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3 Tetrachloroethene (PCE)

3.1 General

Several comprehensive reviews of tetrachloroethene in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 1997; WHO 2006; NICNAS 2001; US EPA 2012). The following provides a summary of the key aspects of PCE that is relevant to the derivation of interim HILs.

Tetrachloroethene, also known as perchloroethylene (PCE) and tetrachloroethylene, is a synthetic, colourless, volatile, non-flammable liquid, with a characteristic sharp, sweet odour. It has a relatively low solubility in water and is commonly used as a dry-cleaning and metal degreasing solvent (ATSDR 1997). PCE manufacture in Australia ceased in 1991. Use in Australia has declined from 1995, consistent with declining use worldwide. PCE is primarily imported in its 'pure' form with approximately 80 % used in the dry cleaning industry in Australia (NICNAS 2001)

PCE is widespread in the environment and is found in trace amounts in water, aquatic organisms, air, foodstuffs, and human tissue. The highest environmental levels of PCE are found in the commercial dry-cleaning and metal degreasing industries. PCE may degrade in the environment to more toxic compounds, including vinyl chloride (WHO 2006).

3.2 Previous HIL

No previous HIL is available for PCE (NEPC 1999).

3.3 Proposed Interim HIL

Review of available information in relation to the presence of PCE in soil indicates that the vapour inhalation pathway is the most significant/important. This pathway should be assessed based on measured vapour data, in particular, soil vapour data. There are significant limitations in the derivation of a soil HIL, in particular, the modelling of phase partitioning from soil to soil vapour and the field measurement of volatiles in soil; hence, an interim HIL has been derived for soil vapour only.

The following presents the values adopted for the calculation of a soil vapour interim HIL. In addition other information that is relevant to the assessment of PCE in soil (relevant to other pathways of exposure) is presented.

3.4 Significance of Exposure Pathways

3.4.1 Inhalation

PCE is a volatile compound and, as such, the derivation of the HIL has considered the vapour inhalation pathway. The approach adopted for the quantification of potential vapour migration to outdoor air and intrusion indoors is outlined in Schedule B7. It is noted that the derived HIL is dominated by the assessment of these pathways of exposure. Due to limitations with the vapour modelling approach adopted, the HILs derived are considered interim.

The inhalation of particulates outdoors and indoors is considered essentially insignificant, compared with vapour inhalation.

3.4.2 Dermal absorption

Insufficient data is available on the dermal absorption of PCE from soil. Given the volatility of the compound, dermal absorption is expected to be low though, as there is insufficient data available to further assess dermal absorption from soil, a default value of 0.03 (3%) has been adopted for the volatile organic compounds (US EPA 1995).

3.4.3 Plant Uptake

Limited data is available on the potential for PCE to be taken up by home-grown produce. Some data is available on the effects of PCE vapours on plant growth with a predicted no effect concentration (PNEC) of 8.2 µg/m^3 identified. ATSDR (1997) also notes that food products can absorb PCE from the atmosphere over time; hence, some studies on the level of PCE in food products are expected to reflect this process, rather than plant uptake from the roots. Given the volatility of this compound, the potential for plant uptake is expected to be limited. As with the assessment presented for TCE, the use of the more commonly adopted equations for quantifying plant uptake (as presented in the text of Schedule B7) that do not address uptake of volatiles (from air) rather than the root, or transformations within the plant, are not considered appropriate and relevant for the assessment of PCE.

It is expected that the potential for plant uptake will be of less significance in the derivation of an HIL, when compared with the assessment of vapour inhalation and, given the limitations involved in providing a meaningful evaluation of plant uptake, it has not been considered in the derivation of HILs.

3.4.4 Intakes from Other Sources – Background

As PCE is highly volatile and not persistent, background intakes will be dominated by inhalation exposures. Concentrations of PCE in industrial, urban and regional areas are available in Australia. Data collected in NSW (DEC 2003) from urban and regional areas in NSW report average concentrations of PCE of approximately 0.1 ppbv, or 0.0007 mg/m^3 (<5% of inhalation TRV) with a maximum concentration in the Sydney CBD of 1.6 ppbv, or 0.01 mg/m^3 (5% of inhalation TRV) A study of concentrations in an industrial area in Brisbane (Hawas et. al. 2001) has reported average and maximum concentrations of 0.015 mg/m^3 (7.5% of inhalation TRV) and 0.085 mg/m^3 (42% of inhalation TRV) respectively. These concentrations are consistent with those reported in other cities in Australia (NICNAS 2001).

Other significant exposures of the general public are likely to occur through the use of dry-cleaning. Variable concentrations of PCE in homes and where dry-cleaned clothes are stored and worn are reported by NICNAS (2001) and WHO (2000). A study on the effect of wearing dry-cleaned clothes reported median personal air concentrations ranging from 0.032 mg/m^3 to 0.22 mg/m^3, depending on the garment. These exposures, together with exposures to paint solvents and cleaning material containing PCE were considered potentially significant. No estimate of intake by the general public is provided in the NICNAS review. Median indoor air concentration reported by WHO (2006) for homes not located in the same building as dry-cleaners was 0.004 mg/m^3 (note that concentrations indoors were much higher in buildings with a dry-cleaner with indoor air levels ranging from 0.05 to 6.1 mg/m^3). This value is also essentially negligible compared with the recommended inhalation TRV. While there is the potential for increased background intakes depending on consumer use of products and frequency of dry-cleaning, average intakes are considered low, with a conservative average intake of 10% assumed in the derivation of HILs.

It is noted that other sources found indoors (from a wide range of common products) are likely to be present and may contribute more significantly to background exposures. These sources need to be addressed on a site-specific basis.

3.5 Identification of Toxicity Reference Values

3.5.1 Classification

The International Agency for Research on Cancer (IARC 1995) has classified PCE as Group 2A—probably carcinogenic to humans, based on limited evidence in humans and sufficient evidence in experimental animals.

Review of PCE by US EPA (2012) classified it as *'Likely to be Carcinogenic to Humans'* by all routes of exposure, based on suggestive evidence of carcinogenicity in epidemiologic studies and conclusive evidence that the administration of PCE, either by ingestion or by inhalation to sexually mature rats and mice, increases tumour incidence.

3.5.2 Review of Available Values/Information

Some epidemiological studies indicate a possible association between chronic exposure to PCE and an increased cancer risk. Review of these studies has indicated that the evidence provided is inconclusive (US EPA 2012). This is mainly due to concurrent exposure to other petroleum solvents as well as PCE, confounding factors (smoking, alcohol, socio-economic status) and small numbers of cancers in the studies.

An association between exposure to PCE (inhalation and ingestion) and an increased risk of cancer (mononuclear cell leukaemia and hepatic tumours) in animals has been suggested. Review of PCE by WHO (2000) indicates that PCE is a non-genotoxic animal carcinogen. Review of the possible mechanisms of tumour formation by PCE in animals suggests that the tumours observed may have little relevance for humans. This is subject to some debate, though recent reviews by WHO (2006) and US EPA (2012) have noted that, in the absence of suitable supporting evidence to the contrary, it must be concluded that the cancers produced by PCE in rodents are of potential relevance to humans.

From the weight of evidence, PCE does not appear to have significant genotoxic potential, however some of the possible metabolites are recognised Ames bacterial mutagens (WHO 2000; WHO 2006, RIVM 2001). Review of the available studies by WHO (2006) suggests that non-genotoxic mechanisms have been recognised for the formation of kidney tumours in male rats and liver tumours in mice for some chemicals. The available data on MoA for PCE are limited, and the dose–response data related to these recognised mechanisms are not consistent with the dose–response relationships for cancer induction by PCE. WHO (2006) has derived a threshold inhalation value for PCE that is considered protective of key end points including carcinogenicity. Hence it may be considered appropriate that a threshold dose-response approach be adopted for PCE.

Review of PCE by US EPA (2012) suggests that PCE has been shown to induce some genotoxic effects. There are a number of limitations noted in the assessment presented by US EPA, in particular, the fact that the MoA for PCE that induces carcinogenesis is not yet fully characterised or understood and that the role of genotoxicity in hepatocarcinogenicity is uncertain. Where US EPA lacks certainty, the default position is to be conservative and, as such, it has suggested considering PCE having a mutagenic MoA, where a non-threshold approach is recommended for the assessment of carcinogenicity and mutagenicity. This is not consistent with the approach adopted in this assessment (consistent with NHMRC 1999 guidance). The assessment of PCE should be updated should additional data become available that supports the US EPA review.

The following quantitative values are available for PCE from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	TDI = 0.014 mg/kg/day	The current ADWG (NHMRC 2011) have derived a guideline of 0.05 mg/L for PCE based on a NOEL of 14 mg/kg/day from a 90-day drinking water study in rats and mice, and an uncertainty factor of 1000. The uncertainty factor includes an additional 10-fold factor to address possible carcinogenicity.
International		
WHO (2011)	TDI = 0.014 mg/kg/day	WHO DWG TDI based on the same study and uncertainty factor as noted in the ADWG (NHMRC 2011).
WHO (2006 and 2010)	TC = 0.2 mg/m^3 TC = 0.25 mg/m^3 TDI = 0.05 mg/kg/day	TC in air derived on the basis of the most sensitive end point, namely neurotoxicological effects, based on a mean LOAEC (adjusted) of 20 mg/m^3 from an occupational inhalation study (mean exposure of 10 years) (Seeber 1989), and an uncertainty factor of 100. The TC derived is lower than that from other key end points such as kidney and liver effects and reproductive/developmental effects. Potential carcinogenic effects have been assessed on the basis of a benchmark dose approach with a $BMCL_{10}$ of 20 mg/m^3 and if a multistage model were considered the TC of 0.2 mg/m^3 would be associated with a risk of 1 x10^{-3}. The assessment presented by WHO (2006) is an update of the earlier assessment presented in the WHO Air Quality Guidelines (2000) where a TC of 0.25 mg/m^3 was derived based on a LOAEL of 102 mg/m^3 in dry-cleaning workers, with adjustment for exposure duration (to LOAEL of 24.3 mg/m^3) (Mutti et al. 1993), and an uncertainty factor of 100. Further review of PCE by WHO (2010) re-confirmed the guideline of 0.25 mg/m^3.
RIVM (2001)	TDI = 0.016 mg/kg/day TC = 0.25 mg/m^3	TDI derived on the basis of a NOAEL of 16 mg/kg/day associated with liver effects in a 4-week oral study in rats, and an uncertainty factor of 1000. TC adopted based on older WHO (2000) evaluation derived from a LOAEL (adjusted) of 23 mg/m^3 from an occupational inhalation study, and an uncertainty factor of 100.
Health Canada (1993)	TDI = 0.014 mg/kg/day TC = 0.36 mg/m^3	TDI derived on the same basis as noted for the WHO DWG and ADWG. TC derived from a LOAEL of 363 (adjusted) mg/m^3 associated with multiple effects in mice, and an uncertainty factor of 1000.
ATSDR (1997)	No chronic oral MRL Inhalation MRL =0.24 mg/m^3	Nor chronic oral MRL has been established. The chronic inhalation MRL has been derived on the basis of a LOAEL (adjusted) of 24 mg/m^3 associated with neurobehavioural effects in an occupational

Source	Value	Basis/Comments
		inhalation study, and an uncertainty factor of 100.
US EPA (2012)	RfD = 0.006 mg/kg/day RfC = 0.04 mg/m^3	RfD derived based on route extrapolation from the inhalation studies. RfC derived on the basis of the midpoint of RfCs derived from 2 studies. An RfC of 0.015 mg/m^3 was derived from a LOAEL of 15 mg/m3 associated with colour confusion in an adult occupational study (Cavalleri et al. 1994), and application of a 100-fold uncertainty factor. An RfC of 0.056 mg/m^3 was derived from a LOAEL of 56 mg/m^3 associated with cognitive and reaction time effects in an adult occupational study (Echeverria et al. 1995), and application of a 100-fold uncertainty factor. The derived value is consistent with that derived for liver effects from the study by Mutti et al. (1993), and 1000-fold uncertainty factor. The 100-fold uncertainty factor applied to these key studies includes a 10-fold factor to address database deficiencies in relation to characterising the hazard and dose response in the human population. The US EPA review also identified non-threshold values not considered relevant in this evaluation.

In relation to the identification of an appropriate inhalation TRV for use in the derivation of a soil vapour interim HIL, the following is noted from the above studies:

- The point of departure (LOAELs in this case) from key studies by WHO (2006; 2010) and US EPA (2012) are similar, ranging from 0.02 to 0.056 mg/m^3;
- The key studies identified in the US EPA (2012) review were also considered in the WHO (2006 and 2010) reviews, with the WHO (2006) review determining that the study conducted by Cavalleri et al. (1994) (used by US EPA as the lower end of the range of two principal RfCs derived) provided results that were difficult to interpret and hence not suitable for the determination of a threshold criterion. The other principal study considered by US EPA was not used as a key study by WHO. Similarly, the key study adopted by WHO (2006), while initially identified by US EPA as an appropriate key study, was not considered due to concerns regarding discrepant results;
- The key difference between the WHO and US EPA reviews and derived inhalation TRVs is the application of uncertainty factors. The WHO reviews have consistently applied an uncertainty factor of 100 to address intra-species variability and the use of a LOAEL. US EPA (2012) has applied an additional factor of 10-fold to address database deficiencies in relation to characterising the hazard and dose–response in the human population (i.e. residents rather than workers). The WHO (2006) review considered the use of occupational studies to be conservative for the general population, as worker exposures are likely to include short duration peaks of higher concentrations. This approach (by WHO) is consistent with that adopted in the assessment of exposures by the general public, based on occupational studies, in Australia.
- Based on the above, both the WHO and US EPA reviews have considered the same key studies and database of information. However, the interpretation of uncertainty in relation to the use of occupational studies for establishing criteria for the general public differs. Where the range of potential RfCs (from suitable available studies) was considered by US EPA (including consideration of uncertainty factors), the inhalation

value of 0.2 mg/m^3 derived by WHO (2006) lies at the lower end of the range of criteria derived. Hence adopting the WHO (2006) inhalation TRV of 0.2 mg/m^3 is considered appropriate for the derivation of soil vapour Interim HILs.

3.5.3 Recommendation

In relation to PCE, only soil vapour Interim HILs have been derived. Hence, only the inhalation pathway has been quantified in the development of these HILs. On the basis of the discussion above, the following inhalation toxicity reference values (TRVs) have been adopted for PCE:

Recommendation for PCE

Inhalation TRV (TRV_I) = 0.2 mg/m^3 (WHO 2006)

Background intakes from other sources (as % of TRV):

BIi = 10% for inhalation

3.6 Calculated Interim HILs

On the basis of the above, the following interim soil vapour HILs have been derived for PCE (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	Interim Soil Vapour HIL# (mg/m^3)
Residential A	2
Residential B	2
Recreational C	40
Commercial D	8

Interim soil gas HILs are conservative soil gas concentrations that can be adopted for the purpose of screening sites where further investigation is required on a site-specific basis. They are based on the potential for vapour intrusion indoors using an indoor air-to-soil gas attenuation factor of 0.01 for scenarios A, B and D and an outdoor attenuation factor of 0.005 for scenario C.

3.7 References

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WHO 2010. WHO Guidelines for Indoor Air Quality, Selected Pollutants. WHO Regional Office for Europe.

WHO 2011, *Guidelines for drinking-water quality, 4th edn*, World Health Organization, Geneva, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html.

4 Cis-1,2-Dichloroethene (DCE)

4.1 General

Several comprehensive reviews of *cis*-1,2-dichloroethene (DCE) in the environment and its toxicity to humans are available and should be consulted for more detailed information (ATSDR 1996; WHO 2011). The following provides a summary of the key aspects of DCE that is relevant to the derivation of interim HILs.

DCE is a colourless, volatile and flammable liquid with a characteristic sharp, harsh odour. It is one of two isomers of 1,2-DCE, the second being *trans*-1,2-DCE. *cis*-1,2-DCE is considered to be more toxic than *trans*-1,2-DCE and hence the HILs derived for the *cis*-isomer are adequately protective of exposures associated with the *trans*-isomer.

DCE is not known to occur naturally. It is most commonly used as a chemical intermediate to produce chlorinated solvents and chemical compounds. It is also used in rubber extraction, pharmaceutical manufacturing, as a refrigerant and in the extraction of oils from meats and fish. DCE has also historically been used as a solvent for a variety of waxes, resins, perfumes, dyes, lacquers, acetyl cellulose, thermoplastics and phenols (ATSDR 1996).

4.2 Previous HIL

No previous HIL is available for DCE (NEPC 1999).

4.3 Proposed Interim HIL

Review of available information in relation to the presence of DCE in soil indicates that the vapour inhalation pathway is the most significant/important. This pathway should be assessed on the basis of measured vapour data, in particular, soil vapour data. There are significant limitations in the derivation of a soil HIL, in particular, the modelling of phase partitioning from soil to soil vapour and the field measurement of volatiles in soil. Hence, an interim HIL has been derived for soil vapour only.

The following presents the values adopted for the calculation of a soil vapour interim HIL. In addition, other information that is relevant to the assessment of DCE in soil (relevant to other pathways of exposure) is presented.

4.4 Significance of Exposure Pathways

4.4.1 Inhalation

DCE is a volatile compound and, as such, the derivation of the HIL has considered the vapour inhalation pathway. The approach adopted for the quantification of potential vapour migration to outdoor air and intrusion indoors is outlined in Schedule B7. It is noted that the derived HIL is dominated by the assessment of these pathways of exposure. Due to limitations with the vapour modelling approach, adopted the HILs derived are considered interim.

The inhalation of particulates outdoors and indoors is considered essentially insignificant, compared with vapour inhalation.

4.4.2 Dermal absorption

Insufficient data is available on the dermal absorption of DCE from soil. Given the volatility of the compound, dermal absorption is expected to be low though, as there is insufficient data available to further assess dermal absorption from soil, a default value of 0.03 (3%) has been adopted for the volatile organic compounds (US EPA 1995).

4.4.3 Plant Uptake

No data is available on the potential for DCE to be taken up by home-grown produce. Given the volatility of this compound, the potential for plant uptake is expected to be limited. As with the assessment presented for TCE, the use of the more commonly adopted equations for quantifying plant uptake (as presented in the text of Schedule B7) that do not address uptake of volatiles (from air) rather than the root, or transformations within the plant, are not considered appropriate and relevant for the assessment of DCE.

It is expected that the potential for plant uptake will be of less significance in the derivation of an HIL, when compared with the assessment of vapour inhalation, and given the limitations involved in providing a meaningful evaluation of plant uptake, it has not been considered in the derivation of HILs.

4.4.4 Intakes from Other Sources – Background

As DCE is highly volatile and not persistent, background intakes will be dominated by inhalation exposures. DCE is not considered to be a typical urban air contaminant and little data is available for Australian cities. *Cis*-1,2-DCE has been detected in VOC sampling from Perth (WA DEP 2000), with average concentrations of 0.2 ppb (0.8 $\mu g/m^3$) and a maximum reported concentration of 2.1 ppb (8.3 $\mu g/m^3$). These values were comparable to average concentrations reported in air in the USA and used by RIVM (2001) to estimate background intake of 1,2-DCE (both isomers) of approximately 0.13 µg/kg/day. Based on the recommended TRV for DCE, this intake is less than 5% and considered negligible (0%).

It is noted that other sources found indoors (from a wide range of common products) are likely to be present and may contribute more significantly to background exposures. These sources need to be addressed on a site-specific basis.

4.5 Identification of Toxicity Reference Values

4.5.1 Classification

The International Agency for Research on Cancer (IARC) has not classified DCE.

US EPA (2010) has classified 1,2-DCE as 'inadequate information to assess the carcinogenic potential'.

4.5.2 Review of Available Values/Information

There is no adequate data available to assess the carcinogenicity of DCE. Review of available genotoxicity studies by WHO (2011) provided equivocal results. Review by RIVM (2001) suggested that cis-1,2-DCE could be considered genotoxic in vivo, producing gene mutations and chromosome aberrations. However, no carcinogenic toxicity values have been derived for the *cis*- isomer. A more recent review of genotoxicity provided by US EPA (2010) suggested that, overall, data for 1,2-DCE (both isomers) is not positive for genotoxicity and mutagenicity. The positive results (considered by RIVM) are considered inconsistent by US EPA and need further confirmation. On the basis of the available information, it is considered appropriate that a threshold dose–response approach be adopted for DCE. Few quantitative toxicity values are available; however, the following are available from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	TDI = 0.017 mg/kg/day for *trans*-isomer	The Australian Drinking Water Guidelines (NHMRC 2011) have derived a drinking water guideline of 0.06 mg/L for 1,2-DCE (both isomers) following guidance from WHO (refer below).
International		
WHO	TDI = 0.017	WHO (2011) has derived a guideline of 0.05 mg/L

Source	Value	Basis/Comments
(2011)	mg/kg/day for *trans*-isomer	based on a TDI of 0.017 mg/kg/day associated with a NOAEL of 17 mg/kg from a 90-day study in mice administered *trans*-1,2-DCE in drinking water, and an uncertainty factor of 1000. This guideline is relevant to the sum of both *cis*- and *trans*- isomers, however this is due to WHO adopting a conservative approach where there is no data available for the derivation of a *cis*- isomer value.
RIVM (2001)	TDI = 0.006 mg/kg/day TC = 0.03 mg/m^3	A TDI of 0.006 mg/kg/day has been established for *cis*-1,2-DCE based on a NOAEL of 32 mg/kg/day from a 90-day oral rat study (using the *cis*- isomer), and an uncertainty factor of 5000. Inhalation tolerable concentrations (TC) were derived for *cis*-1,2-DCE using route extrapolation from the oral study, resulting in a TC of 0.03 mg/m^3
ATSDR (1996)	No chronic MRLs derived	
US EPA (2010)	RfD = 0.002 mg/kg/day for *cis*-isomer	RfD derived on the basis of a $BMDL_{10}$ of 5.1 mg/kg/day associated with increased kidney weight in male rats and a 3000-fold uncertainty factor (includes 3-fold factor for database deficiencies). No inhalation RfC was derived for the cis-isomer. For the trans-isomer an oral RfD of 0.02 mg/kg/day was derived and no inhalation RfC was derived.

For the assessment of inhalation exposures (relevant to the derivation of soil vapour Interim HILs), there are no specific TRVs derived from inhalation studies associated with cis-1,2-DCE. An inhalation value can be derived from route extrapolation from an oral value (as undertaken by RIVM). In relation to the available oral TRVs, the most recent evaluation conducted by US EPA is considered the most appropriate. From this oral TRV, an inhalation TRV of 0.007 mg/m^3 can be derived (for a 70 kg adult where 20 m^3 of air is inhaled each day).

4.5.3 Recommendation

In relation to cis-1,2-DCE, only soil vapour Interim HILs have been derived. Hence only the inhalation pathway has been quantified in the development of these HILs. On the basis of the discussion above, the following inhalation toxicity reference values (TRVs) have been adopted for cis-1,2-DCE:

Recommendation for cis-1,2-DCE

Inhalation TRV (TRV_I) = 0.007 mg/m^3 (US EPA 2010)

Background intakes from other sources (as % of TRV):

BIi = 0% for inhalation

4.6 Calculated Interim HILs

On the basis of the above, the following interim soil vapour HILs have been derived for DCE (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	Interim Soil Vapour HIL# (mg/m^3)
Residential A	0.08
Residential B	0.08
Recreational C	2
Commercial D	0.3

Interim soil gas HILs are conservative soil gas concentrations that can be adopted for the purpose of screening sites where further investigation is required on a site-specific basis. They are based on the potential for vapour intrusion indoors using an indoor air-to-soil gas attenuation factor of 0.1 for HILs A, B and D and an outdoor attenuation factor of 0.05 for HIL C.

4.7 References

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5 Vinyl Chloride

5.1 General

Several comprehensive reviews of vinyl chloride in the environment and its toxicity to humans are available and should be consulted for more detailed information not presented in this summary (ATSDR 2006; WHO 1999; IARC 2008). The following provides a summary of the key aspects of vinyl chloride that is relevant to the derivation of interim HILs.

Vinyl chloride is a colourless, flammable gas, with a characteristic slightly sweet odour. It has a high vapour pressure, a high value for Henry's Law constant, a relatively low solubility in water, and is heavier than air. It is also soluble in most organic solvents. Under pressure, vinyl chloride is easily liquefied, and is commonly stored and transported as a liquid and made into polyvinyl chloride (PVC) (ATSDR 2006).

Vinyl chloride is not known to occur naturally. Vinyl chloride is predominantly used in the plastics industry, in the production of polyvinyl chloride (PVC). PVC is used in numerous industries including packaging, building, electrical appliances, medical care, agriculture, automobiles and toys. Vinyl chloride is also used in limited quantities as a refrigerant and an intermediate in the production of chlorinated compounds (WHO 1999).

Vinyl chloride is a degradation product of PCE/TCE/1,2-DCE and 1,1-DCE and its presence in the environment may not be due to a primary source, but rather it may be due to degradation of other chlorinated sources.

5.2 Previous HIL

No previous HIL is available for vinyl chloride (NEPC 1999).

5.3 Proposed Interim HIL

Review of available information in relation to the presence of vinyl chloride in soil indicates that the vapour inhalation pathway is the most significant/important. This pathway should be assessed on the basis of measured vapour data, in particular, soil vapour data. There are significant limitations in the derivation of a soil HIL, in particular, the modelling of phase partitioning from soil to soil vapour and the field measurement of volatiles in soil. Hence, an interim HIL has been derived for soil vapour only.

The following presents the values adopted for the calculation of a soil vapour interim HIL. In addition, other information that is relevant to the assessment of vinyl chloride in soil (relevant to other pathways of exposure) is presented.

5.4 Significance of Exposure Pathways

5.4.1 Inhalation

Vinyl chloride is a volatile compound and, as such, the derivation of the HIL has considered the vapour inhalation pathway. The approach adopted for the quantification of potential vapour migration to outdoor air and intrusion indoors is outlined in Schedule B7. It is noted that the derived HIL is dominated by the assessment of these pathways of exposure. Due to limitations with the vapour modelling approach adopted, the HILs derived are considered interim.

It is noted that there is the potential for vinyl chloride to undergo biodegradation within the soil profile. Available data (Scheutz 2002) suggests that the degradation of vinyl chloride is complex, involving both anaerobic and aerobic processes. Vinyl chloride is rapidly degraded in the presence of oxygen and is considered one of the least stable chlorinated chemicals in soil gas. NJ DEP (2005)

notes that, due to these processes, vinyl chloride is seldom found in soil gas above a contaminated source. Hence, while the potential for vapour migration to be significant has been modelled and considered in the HILs, due to the potential for degradation, this approach is expected to be conservative for vinyl chloride.

The inhalation of particulates outdoors and indoors is considered essentially insignificant, compared with vapour inhalation.

5.4.2 Dermal absorption

Insufficient data is available on the dermal absorption of vinyl chloride from soil. Given the volatility of the compound, dermal absorption is expected to be low though, as there is insufficient data available to further assess dermal absorption from soil, a default value of 0.03 (3%) has been adopted for the volatile organic compounds (US EPA 1995).

5.4.3 Plant Uptake

No data is available on the potential for vinyl chloride to be taken up by home-grown produce. It is noted that vinyl chloride can be absorbed by produce packaged in PVC plastic. Concentrations reported in these products are not associated with plant uptake from soil. Given the volatility of this compound, the potential for plant uptake is expected to be limited. As with the assessment presented for TCE, the use of the more commonly adopted equations for quantifying plant uptake (as presented in the text of Schedule B7) that do not address uptake of volatiles (from air) rather than the root, or transformations within the plant, are not considered appropriate and relevant for the assessment of vinyl chloride.

It is expected that the potential for plant uptake will be of less significance in the derivation of an HIL, when compared with the assessment of vapour inhalation and, given the limitations involved in providing a meaningful evaluation of plant uptake, it has not been considered in the derivation of HILs.

5.4.4 Intakes from Other Sources – Background

As vinyl chloride is highly volatile and not persistent, background intakes will be dominated by inhalation exposures. Concentrations of vinyl chloride in industrial, urban and regional areas are available in Australia. Data collected in NSW (DEC 2003) from urban and regional areas in NSW note that vinyl chloride was rarely detected (<1% of samples) with the maximum reported from the Sydney CBD of 0.3 ppbv (0.0008 mg/m^3). Vinyl chloride was not detected in ambient air sampling undertaken in Perth (WA DEP 2000). In addition, vinyl chloride has not been detected in drinking water and low levels are expected in food (NHMRC 2011). Low levels have been historically reported in some consumer products. Background intakes expected from vinyl chloride are expected to be low, with conservative intakes estimated by Health Canada (1992) of approximately 0.005 mg/kg/day and RIVM (2001) of approximately 0.00006 mg/kg/day (predominantly from inhalation). It is noted that, as the most sensitive end point is carcinogenicity, which is assessed on the basis of a non-threshold approach, background intakes are not used in the derivation of the HIL.

5.5 Identification of Toxicity Reference Values

5.5.1 Classification

The International Agency for Research on Cancer (IARC 2008) has classified vinyl chloride as Group 1—carcinogenic to humans.

Vinyl chloride is also classified as a known human carcinogen (Category A) by US EPA for the inhalation route of exposure, and by analogy for the oral route of exposure. It is also considered highly likely to be carcinogenic by the dermal route.

5.5.2 Review of Available Values/Information

Exposure to vinyl chloride via inhalation has been associated with increases in liver cancer, including a rare form of angiosarcoma and biliary tract cancer. Other studies have indicated increase incidence of CNS and brain cancer. While most data is associated with inhalation exposures, ingestion studies suggest evidence of carcinogenicity via oral exposure (WHO 1999 and ATSDR 2006).

Vinyl chloride has been identified as genotoxic and mutagenic (WHO 1999, ATSDR 2006 and US EPA 2000). The US EPA (2000) review notes that vinyl chloride toxicity occurs via a genotoxic pathway (identified from a number of lines of evidence) that is understood in some detail. On this basis, the assessment of carcinogenicity on the basis of a non-threshold (linear) approach is appropriate.

The US EPA (2000) review also noted that chemically induced human liver carcinogenicity is associated with mutational alteration of multiple genes, consistent with a mutagenic mode of action. In addition, several studies of partial lifetime exposure suggest that the lifetime cancer risk depends on age at exposure, with higher lifetime risks attributable to exposures at younger ages. This is also noted by WHO (2000; 2011). Consistent with US EPA guidance, the derivation of non-threshold values for vinyl chloride has incorporated factors that address early life susceptibility and hence, if the US EPA non-threshold values are adopted, (also considered in the WHO values) no additional adjustment is required in the quantification of exposure. It is noted, however, that the application of the US EPA values for exposures by adults only (such as workers) needs to adopt the most correct values that do not include early-life susceptibility.

The most sensitive end point for vinyl chloride (particularly inhalation, which will dominate the derivation of an HIL) is carcinogenicity (noting that in the derivation of the ADWG both carcinogenic and non-carcinogenic effects were considered as sensitive for the oral pathway). Hence, the selection of appropriate non-threshold values for the assessment of vinyl chloride exposure is relevant.

The following quantitative non-threshold values are available for vinyl chloride from Level 1 Australian and International sources:

Source	Value	Basis/Comments
Australian		
ADWG (NHMRC 2011)	Adopted WHO non-threshold approach.	Current guideline derived on the basis of the WHO non-threshold value and additional consideration of non-carcinogenic effects with a TDI of 0.00013 mg/kg/day associated with a no-effect level of 0.13 mg/kg/day from lifetime studies in rats, and 1000-fold uncertainty factor.
OCS (2012)	No evaluation available	

International		
WHO DWG (2011)	SF = 1.15 $(mg/kg/day)^{-1}$ (for exposures from birth) SF = 0.7 $(mg/kg/day)^{-1}$ (for exposures as adults)	WHO (2011, last review in 2004) derived on the basis of linear extrapolation from dose response data for all liver tumours from an oral exposure study in rats and assuming a doubling of the risk of exposure from birth (incorporating the 2-fold uncertainty identified by the US EPA (2000) review to address early life sensitivity. Exposures by workers (only adults) can be calculated on the basis of a slope factor that is 2 times lower.
WHO (2000)	UR = $1x10^{-6}$ $(\mu g/m^3)^{-1}$	Inhalation unit risk derived on the basis of occupational exposures studies associated with haemangiosarcoma and a linear multistage model. The value derived is noted to be limited as it does not address early life sensitivity identified in newborn animals (relevant to exposures by children to 10 years).
Health Canada (1992)	SF = 0.26 $(mg/kg/day)^{-1}$	Slope factor based on the upper value from a free extrapolation method associated with hepatocellular angiosarcomas in female rats. The evaluation is older than that considered by WHO and US EPA and does not include any consideration of early life sensitivity.
RIVM (2001)	SF = 0.17 $(mg/kg/day)^{-1}$ UR = $2.8x10^{-5}$ $(\mu g/m^3)^{-1}$	Slope factor derived on the basis of hepatocellular carcinomas, angiosarcomas and neoplastic nodules in female rats as markers for carcinogenic response, and a linear extrapolation model. Inhalation unit risk derived on the basis liver effects in an inhalation study on female rats and mice and an extrapolation model. No consideration of early-life sensitivity was considered by RIVM. Threshold values were also derived for non-carcinogenic effects with a TDI = 0.0013 mg/kg/day which is based on the same study as considered in the ADWG, but with a less conservative uncertainty factor of 100. An inhalation TC = 0.056 mg/m^3 was derived based on an inhalation study. RIVM notes that the carcinogenic end points are most sensitive.
ATSDR (2006)	No quantitative assessment of carcinogenic effects	ATSDR does not provide quantitative estimates of carcinogenic effects. However for non-carcinogenic effects a chronic oral MRL = 0.003 associated with non-neoplastic effects in livers from a chronic oral rat study was derived.

US EPA (IRIS 2012)	SF = 1.5 $(mg/kg/day)^{-1}$ (for exposures over lifetime) SF = 0.75 $(mg/kg/day)^{-1}$ (for exposures as adult) UR = $8.8x10^{-6}$ $(\mu g/m^3)^{-1}$ for exposures over lifetime) UR = $4.4x10^{-6}$ $(\mu g/m^3)^{-1}$ for exposures as adult)	Slope factor (last reviewed in 2000) derived on the basis of hepatocellular carcinomas, angiosarcomas and neoplastic nodules in female rats as markers for carcinogenic response, a PBPK model to estimate human equivalent dose and linearised multistage model. Based on animal evidence of age-dependent sensitivity an additional 2-fold uncertainty has been included to address early-life sensitivity in exposures from birth. Inhalation unit risk derived on the basis liver angiosarcomas, angiomas, hepatomas and neoplastic nodules in an inhalation study on female rats and mice and an extrapolation model. Based on animal evidence of age-dependent sensitivity an additional 2-fold uncertainty has been included to address early-life sensitivity in exposures from birth. The US EPA review also identified threshold values for the assessment of non-carcinogenic effects with an oral RfD = 0.003 mg/kg/day (same as derived by ATSDR) and an RfC = 0.1 mg/m^3 based on route-extrapolation from the oral value.

Both WHO and US EPA recognise age-sensitivity is important with respect to the assessment of exposure to vinyl chloride and hence it is appropriate to adopt toxicity values that take these issues into consideration. On this basis, of the non-threshold reference values available, the inhalation values presented by US EPA are the most relevant and current (and adequately address early lifetime exposures) and suitable for the derivation of soil vapour Interim HILs.

5.5.3 Recommendation

In relation to vinyl chloride, only soil vapour Interim HILs have been derived. Hence only the inhalation pathway has been quantified in the development of these HILs. On the basis of the discussion above, the following inhalation toxicity reference values (TRVs) have been adopted for vinyl chloride:

Recommendation for Vinyl Chloride (quantitative inhalation toxicity values)

Carcinogenic end points most sensitive and evaluated on the basis of:

Inhalation TRV = 0.0088 $(mg/m^3)^{-1}$ (US EPA (IRIS 2012)) for inhalation exposures from birth (HIL A, B and C)

Inhalation TRV = 0.0044 $(mg/m^3)^{-1}$ (US EPA (IRIS 2012)) for inhalation exposures as adults (HIL D)

5.6 Calculated Interim HILs

On the basis of the above, the following interim soil vapour HILs have been derived for vinyl chloride (refer to Appendix B for equations used to calculate the HILs and Appendix C for calculations):

HIL Scenario	Interim Soil Vapour HIL# (mg/m^3)
Residential A	0.03
Residential B	0.03

Recreational C	0.5
Commercial D	0.1

Interim soil gas HILs are conservative soil gas concentrations that can be adopted for the purpose of screening sites where further investigation is required on a site-specific basis. They are based on the potential for vapour intrusion indoors using an indoor air-to-soil gas attenuation factor of 0.1 for HILs A, B and D and an outdoor attenuation factor of 0.005 for HIL C.

5.7 References

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WHO 2004, *Vinyl Chloride in Drinking-water, Background document for development of WHO Guidelines for Drinking Water Quality*, World Health Organization, available from http://www.who.int/water_sanitation_health/dwq/chemicals/en/index.html

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6 Shortened forms

ADAF	age-dependent adjustment factor
ADI	acceptable daily intake
ADWG	Australian Drinking Water Guidelines
AI	adequate intake
ANZECC	Australia and New Zealand Environment and Conservation Council
APVMA	Australian Pesticides and Veterinary Medicines Authority
ATDS	Australian Total Diet Survey
ATSDR	Agency for Toxic Substances and Disease Registry
BA	bioavailability
BI	background intake
BMD	benchmark dose
BMDL	Benchmark dose lower confidence limit
CCME	Canadian Council of Ministers of the Environment
CICAD	Concise International Chemicals Assessment Document
CNS	central nervous system
DAF	dermal absorption factor
DCE	dichloroethene
DW	dry weight
EA	Environment Agency (England and Wales)
EHC	Environmental Health Criteria
EPA	Environment Protection Authority
FSANZ	Food Standards Australia and New Zealand
GAF	gastrointestinal absorption factor
GV	guideline value
HCB	hexachlorobenzene
HEC	human equivalent concentration
HED	human equivalent dose
HIARC	Hazard Identification Assessment Review Committee
HIL	health investigation level
HSDB	Hazardous Substances Data Bank

HSL	health screening level
IARC	International Agency for Research on Cancer
IEUBK	Integrated exposure uptake biokinetic model
IRIS	Integrated Risk Information System
JECFA	Joint FAO/WHO Expert Committee on Food Additives
JMPR	WHO/FAO Joint Meeting on Pesticide Residues
LOAEL	lowest observed adverse effect level
LOEL	lowest observed effect level
MF	modifying factor
MoA	mode (or mechanism) of action
MoE	margin of exposure
MPC	maximum permissible concentration
MRL	maximum residue limit
MRL	minimal risk level
NDI	negligible daily intake
NEPC	National Environment Protection Council
NEPM	National Environment Protection Measure
NHMRC	National Health and Medical Research Council
NOAEL	no observable adverse effect level
NOEL	no observable effect level
NSW DECC	New South Wales Department of Environment and Climate Change
OCS	Office of Chemical Safety
PBDE	polybrominated diphenyl ether
PCB	polychlorinated biphenyl
PCE	perchloroethene (tetrachloroethene)
PNEC	predicted no-effect concentration
POP	persistent organic pollutant
PTDI	provisional tolerable daily intake
PTMI	provisional tolerable monthly intake
PTWI	provisional tolerable weekly intake
PVC	polyvinyl chloride

RAIS	Risk Assessment Information System
RDI	recommended daily intake
REL	reference exposure level
RfC	reference concentration
RfD	reference dose
RME	reasonable maximum exposure
SF	slope factor
TC	tolerable concentration
TCA	trichlorethane
TCE	trichlorethene
TD	tumorigenic dose
TDI	tolerable daily intake
TRV	toxicity reference value
UF	uncertainty factor
UL	upper limit
UR	unit risk
US EPA	United States Environmental Protection Agency
VOC	volatile organic compound
WHO	World Health Organization
WHO DWG	World Health Organization Drinking Water Guidelines

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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Volume 1: sections 1–6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

Schedule B7

APPENDIX B

Equations for Derivation of HILs and Interim HILs

Contents
Equations for derivation of HILs and interim HILs

1 Equations for derivation of HILs and interim HILs

1.1 Introduction

This appendix presents the equations used in the derivation of soil health investigation levels (HILs) and interim soil vapour HILs. The appendix does not present all equations and methodologies that may be considered in conducting a site-specific assessment, rather it presents those equations used in deriving the HILs presented in Schedule B7. The derivation of HILs requires the consideration of a number of exposure pathways. With respect to the soil HILs, the following pathways are considered (as relevant for the exposure scenarios and compounds considered):

- Ingestion of soil and/or dust (indoors). The ingestion rate adopted for the characterisation of this pathway is a combined value reflecting both sources; hence, the calculation undertaken is a combined calculation.
- Dermal absorption during contact with soil and/or dust (indoors that may be derived from outdoor soil). As with the calculation of ingestion, the calculation of dermal absorption is based on absorption from both sources combined.
- Inhalation of dust generated from outdoor soil (where surface cover is poor) both outdoors and indoors (including resuspension of dust indoors).

Inhalation of volatile chemicals in soil indoors and outdoors has been considered in the derivation of interim soil vapour HILs.

Worked examples of the HIL A calculations using the equations presented in this Appendix for cadmium and benzo(a)pyrene are included in Attachments A and B respectively.

1.2 General equations

The approach adopted in the derivation of soil HILs is consistent with the approach adopted in the derivation of previous HILs (NEPC 1999) and in other jurisdictions including the USA (in the derivation of preliminary remediation goals (US EPA 1992; US EPA 2002) and regional screening levels (US EPA 2012)) and the UK and New Zealand (in the derivation of soil guideline values (MfE 2011; EA 2009])).

Very generally, a soil health investigation level (HIL) for an exposure pathway (x), where a threshold approach is adopted, can be back-calculated by setting the estimated intake for a chemical (i) to the acceptable intake allowable from soil for that chemical (i), then rearranging the equation as follows:

$$HIL_{x,i}\ (mg/kg) = \frac{\text{Acceptable Intake}}{\text{Intake from Contamination}} = \frac{(\text{acceptable intake}_i \text{ from soil}) \times (\text{body weight}) \times (\text{averaging time})}{(\text{contact rate}_i) \times (\text{exposure frequency}) \times (\text{exposure duration})}$$

Equation 1

Similarly, HILs can be derived for other pathways of exposure and for non-threshold carcinogenic effects as relevant. The final HIL is calculated by combining the pathway-specific HILs as noted below:

$$HIL\,(mg/kg) = \frac{1}{\left[\frac{1}{HIL_{ingestion}}\right] + \left[\frac{1}{HIL_{dermal}}\right] + \left[\frac{1}{HIL_{plant\ uptake}}\right] + \left[\frac{1}{HIL_{dust}}\right]}$$

Equation 2

where:

$HIL_{ingestion}$ = derived soil guideline associated with the ingestion of soil and dust by young child and/or adult, refer to **Equations 3, 4 and 5**

HIL_{dermal} = derived soil guideline associated with dermal absorption of contaminant in soil/dust by young child and/or adult, refer to **Equations 6, 7 and 8**

$HIL_{plant\ uptake}$ = derived soil guideline associated with ingestion of contaminant in home-grown fruit and vegetable produce by young child and/or adult (where relevant), refer to **Equations 15 to 18**

HIL_{dust} = derived soil guideline associated with inhalation of contaminants in dust by young child and/or adult, refer to **Equations 9, 10 and 11**

This approach assumes that the pathways of exposure are all complete and are additive, and that the toxicological end point considered for all pathways of exposure are the same or additive.
The contribution of each individual pathway ($HIL_{pathway}$) to the total HIL has been calculated (and presented in Appendix A) as follows:

$$\%\text{pathway contribution} = (1/HIL_{pathway})/(1/HIL) \times 100\ (\%)$$

For volatile compounds, only interim soil vapour HILs have been derived. This has been conducted on the basis of calculations relevant to inhalation of volatile contaminants in air by a young child and/or adult, refer to **Equations 12, 13 and 14.**

1.3 Pathway-specific equations

1.3.1 Ingestion of soil/dust

Threshold contaminants (2–3-year-old child for HILs A, B and C and adult for HIL D)

$$HIL_{ingestion}\ (mg/kg) = \frac{(TRV_o(100\% - BI_o)) \times BW_C \times AT_T}{IR_{SC} \times BA_o \times CF \times EF \times ED}_C$$ **Equation 3**

where:

TRV_o = toxicity reference value relevant for the quantification of oral intakes, (as mg/kg/day for threshold contaminants)

BI_o = background intakes relevant to oral/dermal exposures (from sources other than soil, which include food, water, air and consumer products where relevant) (as % of the TRVo)

IR_{SC} = ingestion rate of soil/dust by young child (for HILs A, B and C) and adult (HIL D) (mg/day)

BA_o = oral bioavailability (unitless, expressed as a fraction of 1)

CF = conversion factor of $1x10^{-6}$ to convert mg to kg

EF = exposure frequency (days/year)

ED_C = exposure duration for young child (for HILs A, B and C) and adult (HIL D) (years)

BW_C = body weight of young child (for HILs A, B and C) and adult (HIL D) (kg)

AT_T = averaging time for threshold contaminants (days, = ED x 365 days)

Non-threshold contaminants (lifetime exposures)

$$HIL_{ingestion}\ (mg/kg) = \frac{TR}{\text{Intake Factor}_o \times TRV_o}$$ **Equation 4**

$$\text{Intake Factor}_o\ (\text{kg / kg / day}) = \sum\left(\frac{\text{IRs}_x \times \text{BAo} \times \text{CF} \times \text{EF}_x \times \text{ED}_x}{\text{BW}_x \times \text{AT}_{NT}}\right)$$ **Equation 5**

where:

TRV_o = toxicity reference value relevant for the quantification of oral intakes, (as $(mg/kg/day)^{-1}$ for non-threshold contaminants)

TR = target risk for non-threshold contaminants (unitless)

$\sum$ = signifies the sum over all receptor groups *x* considered (in the HILs derived these groups include a child (C) and adult (A))

IRs_X = ingestion rate of soil/dust by each receptor group *x* (mg/day)

BAo = oral bioavailability (unitless)

CF = conversion factor of $1x10^{-6}$ to convert mg to kg

EFx = exposure frequency relevant to exposures by each receptor group *x* (days/year)

EDx = exposure duration relevant to exposures by each receptor group *x* (years)

BWx = body weight relevant to each receptor group *x* (kg)

AT_{NT} = averaging time for non-threshold contaminants (days, = 70 years x 365 days)

1.3.2 Dermal contact with soil/dust

Threshold contaminants (2–3-year-old child for HILs A, B and C and adult for HIL D)

$$HIL_{dermal}\ (mg/kg) = \frac{(TRV_D(100\% - BI_o)) \times BW_C \times AT_T}{SA_C \times AF \times DAF \times CF \times EF \times ED_C}$$ **Equation 6**

where:

TRV_D = toxicity reference value relevant for the quantification of dermal intakes, (as mg/kg/day for threshold contaminants)

BI_O = background intakes relevant to oral/dermal exposures (from sources other than soil, which include food, water, air and consumer products where relevant) (fraction relevant to the % allocated to background intakes)

SA_C = exposed skin surface area for young child (for HILs A, B and C) and adult (HIL D) (cm^2)

AF = soil-to-skin adherence factor ($mg/cm^2/day$)

DAF = dermal absorption factor, (chemical-specific) (unitless)

CF = conversion factor of $1x10^{-6}$ to convert mg to kg

EF = exposure frequency (days/year)

ED_C = exposure duration for young child (for HILs A, B and C) and adult (HIL D) (years)

BW_C = body weight of young child (for HILs A, B and C) and adult (HIL D) (kg)

AT_T = averaging time for threshold contaminants (days, = ED x 365 days)

Non-threshold contaminants (lifetime exposures)

$$HIL_{dermal}\ (mg/kg) = \frac{TR}{Intake\ Factor_D \times TRV_D}$$ **Equation 7**

$$Intake\ Factor_D\ (kg/kg/day) = \sum\left(\frac{SA_x \times AF \times DAF \times CF \times EF_x \times ED_x}{BW_x \times AT_{NT}}\right)$$ **Equation 8**

where:

TRV_D = toxicity reference value relevant for the quantification of dermal intakes, (as $(mg/kg/day)^{-1}$ for non-threshold contaminants)

TR = target risk for non-threshold contaminants (unitless)

$\sum$ = signifies the sum over all receptor groups *x* considered (in the HILs derived these groups include a child (C) and adult (A))

SA_x = exposed skin surface area for all receptor groups *x* (cm^2)

AF = soil-to-skin adherence factor ($mg/cm^2/day$)

DAF = dermal absorption factor, (chemical-specific) (unitless)

CF = conversion factor of $1x10^{-6}$ to convert mg to kg

EF_x = exposure frequency relevant to exposures by all receptor groups *x* (days/year)

ED_x = exposure duration relevant to exposures by all receptor groups *x* (years)

BW_x = body weight relevant to each receptor group *x* (kg)

AT_{NT} = averaging time for non-threshold contaminants (days, = 70 years x 365 days)

1.3.3 Inhalation of dust

Threshold contaminants (2–3-year-old child for HILs A, B and C and adult for HIL D)

$$HIL_{dust}\ (mg/kg) = \frac{(TRV_i(100\% - BI_i)) \times AT_T}{\left[\left[\frac{1}{PEF_o} \times ET_{co}\right] + \left[\frac{1}{PEF_i} \times TF \times ET_{ci}\right]\right] \times RF \times EF \times ED_C} \qquad \textbf{Equation 9}$$

where:

TRV_i = toxicity reference value relevant for the quantification of inhalation intakes, (as $mg/m^{3)}$

BI_i = background intakes relevant to inhalation exposures (from sources other than soil, which include food, water, air and consumer products where relevant) (fraction relevant to the % allocated to background intakes)

$PEF_{i,o}$ = particulate emission factor (or dust loading) for outdoor (O) or indoor (I) air (m^3/kg)

$ET_{ci,co}$ = exposure time outdoors (O) or indoors (I) for young child (for HILs A, B and C) and adult (HIL D) (hours/day)

TF = indoor dust transport factor (unitless)

RF = lung retention factor relevant for the inhalation of dust from site (unitless)

EF = exposure frequency (days/year)

ED_C = exposure duration for young child (for HILs A, B and C) and adult (HIL D) (years)

AT_T = averaging time for threshold contaminants (hours, = ED x 365 days x 24 hours)

Non-threshold contaminants (lifetime exposures)

$$HIL_{dust}\ (mg/kg) = \frac{TR}{\text{Intake Factor}_{dust} \times TRV_i} \qquad \textbf{Equation 10}$$

$$\text{Intake Factor}_{dust}\ (kg/m^3) = \sum \left(\frac{\left[\left[\frac{1}{PEF_o} \times ET_o\right] + \left[\frac{1}{PEF_i} \times TF \times ET_i\right]\right] \times RF \times EF_x \times ED_x}{AT_{NT}} \right) \qquad \textbf{Equation 11}$$

where:

TRV_i = toxicity reference value relevant for the quantification of inhalation intakes, (as $(mg/m^{3)-1}$ for non-threshold contaminants)

TR = target risk for non-threshold contaminants (unitless)

$\sum$ = signifies the sum over all receptor groups *x* considered (in the HILs derived, these groups include a child (C) and adult (A))

$PEF_{i,o}$ = particulate emission factor (or dust loading) for outdoor (O) or indoor (I) air (m^3/kg)

$ET_{i,o}$ = exposure time indoors (I) and outdoors (O) for adults and children (as relevant) (hours/day)

TF = indoor dust transport factor (unitless)

RF = lung retention factor relevant for the inhalation of dust from site (unitless)

EF_x = exposure frequency for all receptor groups *x* (days/year)

ED_x = exposure duration for all receptor groups *x* (years)

AT_{NT} = averaging time for non-threshold contaminants (hours, = 70 years x 365 days x 24 hours)

1.3.4 Inhalation of volatiles

No soil HILs have been derived for volatile compounds, hence this section only presents the approach adopted in the derivation of interim soil vapour HILs.

For the derivation of soil vapour HILs, an attenuation factor has been adopted that relates the indoor air concentration to the soil vapour concentration.

The interim soil vapour HIL (based on indoor air exposures) has then been derived on the basis of the following equations:

Threshold contaminants (2–3-year-old child for HILs A, B and C and adult for HIL D)

$$\text{Interim soil vapour HIL}\,(\text{mg}/\text{m}^3) = \frac{(\text{TRV}_i(100\% - \text{BI}_i)) \times \text{AT}_T}{\alpha \times \text{ET}_{Ci} \times \text{EF} \times \text{ED}_C} \qquad \textbf{Equation 12}$$

where:

TRV_i	= toxicity reference value relevant for the quantification of inhalation intakes, (as $mg/m^{3)}$
BI_i	= background intakes relevant to inhalation exposures (from sources other than soil, which include food, water, air and consumer products where relevant) (fraction relevant to the % allocated to background intakes)
α	= soil vapour to indoor air attenuation factor (unitless)
ET_{ci}	= exposure time indoors (I) for young child (for HILs A, B and C) and adult (HIL D) (hours/day)
EF	= exposure frequency (days/year)
ED_C	= exposure duration for young child (for HILs A, B and C) and adult (HIL D) (years)
AT_T	= averaging time for threshold contaminants (hours, = ED x 365 days x 24 hours)

Non-threshold contaminants (lifetime exposures)

$$\text{Interim soil vapour HIL}\,(\text{mg}/\text{m}^3) = \frac{\text{TR}}{\text{Intake Factor}_{\text{volatile}} \times \text{TRV}_i} \qquad \textbf{Equation 13}$$

$$\text{Intake Factor}_{\text{volatile}}\,(\text{unitless}) = \sum\left(\frac{\alpha \times \text{ET}_{ix} \times \text{EF}_x \times \text{ED}_x}{\text{AT}_{NT}}\right) \qquad \textbf{Equation 14}$$

where:

TRV_i	= toxicity reference value relevant for the quantification of inhalation intakes, (as $(mg/m^{3)-1}$ for non-threshold contaminants)
TR	= target risk for non-threshold contaminants (unitless)
$\sum$	= signifies the sum over all receptor groups *x* considered (in the HILs derived, these groups include a child (C) and adult (A))
α	= soil vapour to indoor air attenuation factor (unitless)
ET_i	= exposure time indoors (I) (hours/day)
EF_x	= exposure frequency for all receptor groups (days/year)
ED_x	= exposure duration for all receptor groups (years)
AT_{NT}	= averaging time for non-threshold contaminants (hours, = 70 years x 365 days x 24 hours)

1.3.5 Ingestion of produce

Intake factors relevant to the estimation of exposures associated with the ingestion of contaminants following uptake into home-grown fruit and vegetable crops (considered as below-ground tuber vegetables (tuber) and root vegetables (root) and above-ground green vegetables (green) and tree fruit (fruit)) are as follows:

Threshold contaminants (2–3-year-old child for HIL A only)

$$HIL_{plant\ uptake}\ (mg/kg) = \frac{(TRV_o(100\% - BI_O)) \times BW_C \times AT_T}{UF_V \times EF \times ED_C} \quad \textbf{Equation 15}$$

$$UF_{VC}\ (kg/day) = F_{HG} \times \left(\left[CF_{tuber} \times C_{tuber}\right] + \left[CF_{root} \times C_{root}\right] + \left[CF_{green} \times C_{green}\right] + \left[CF_{fruit} \times C_{fruit}\right]\right) \quad \textbf{Equation 16}$$

where:

TRV_o	= toxicity reference value relevant for the quantification of oral intakes, (as mg/kg/day for threshold contaminants);
BI_O	= background intakes relevant to oral/dermal exposures (from sources other than soil, which include food, water, air and consumer products where relevant) (fraction relevant to the % allocated to background intakes)
UF_{VC}	= plant uptake factor calculated for the consumption of home-grown produce by young children (kg/day)
CF_y	= plant concentration factors relevant for produce type (y), (chemical-specific) (mg/kg fresh weight produce to mg/kg dry weight soil)
C_y	= consumption rate of each produce type (y) (kg/day)
F_{HG}	= fraction of all fruit and vegetable produce consumed that is home-grown (unitless)
EF	= exposure frequency (days/year)
ED_C	= exposure duration for young children (years)
BW_C	= body weight of young child (kg)
AT_T	= averaging time for threshold contaminants (days, = ED x 365 days)

Non-threshold contaminants (lifetime exposures)

$$HIL_{plant\,uptake}\ (mg/kg) = \frac{TR}{Intake\,Factor_{plant} \times TRVo} \qquad \textbf{Equation 17}$$

$$Intake\,Factor_{plant}\ (kg/kg/day) = \sum\left(\frac{UF_{Vx} \times EF_x \times ED_x}{BW_x \times AT_{NT}}\right) \qquad \textbf{Equation 18}$$

where:

TRV_o = toxicity reference value relevant for the quantification of oral intakes, (as $(mg/kg/day)^{-1}$ for non-threshold contaminants)

TR = target risk for non-threshold contaminants (unitless)

$\sum$ = signifies the sum over all receptor groups *x* considered (in the HILs derived, these groups include a child (C) and adult (A))

UF_{yx} = plant uptake factors calculated using Equation 16 for both adults and children (kg/day)

EF_x = exposure frequency for all receptor groups *x* (days/year)

ED_x = exposure duration for all receptor groups *x* (years)

BW_x = body weight for all receptor groups *x* (kg)

AT_{NT} = averaging time for non-threshold contaminants (days, = 70 years x 365 days)

Note that the calculation of intakes derived from home-grown produce has been included in the derivation of HIL A where relevant. However, it is noted that, for some compounds such as metals, the assessment of intakes derived from the consumption of home-grown produce as well as intakes derived from the diet (as estimated from total diet surveys) results in double counting of intakes that may be derived from produce.

To address the potential for double counting of these intakes it is assumed that 50% of the intake derived from home-grown produce (10% of total intake) is already accounted for in the data available on intakes derived from all dietary sources. Hence, the derivation of the HIL for plant uptake for metals has been adjusted to address this issue (refer to Appendix A for compound-specific data).

1.4 Calculation of particulate emission factor

Soil-derived dust concentrations in outdoor air have been estimated using a particulate emission factor (PEF) using the approach outlined by US EPA (1996; 2002) and EA (2009). The PEF represents an estimate of the relationship between the concentration of a contaminant in soil and its concentration in air as a consequence of dust resuspension. Dust particles considered in the PEF are assumed to be less than 10 μm is diameter. This has been calculated using the following equation:

$$PEF_o\ (m^3/kg) = \frac{Q/C\ x\ 3600}{0.036\ x\ (1-V)\ x\ (\frac{U_m}{U_t})^3\ x\ F_x}$$ **Equation 19**

where:

PEF_O = particulate emission factor outdoors (mg/kg soil per mg/m^3 air)

Q/C = air dispersion factor which describes the dispersion of soil particles in the atmosphere of a theoretical outdoor box. A value of 90.8 (g/m^2/s per kg/m^3) has been used in the derivation of HILs. The value is a default value recommended by US EPA (2002) for small sites (0.5 acres).

V = the fraction of outdoor surface cover (0= bare soil), dimensionless (0.75 for HIL A, 0.9 for HIL B and 0.8 for HIL D)

U_m = mean annual wind speed at a height of 10m (m/s), assumed to be 8.75 km/hr (or 2.4 m/s) based on the average 9 am and 3 pm winds from Canberra

U_t = threshold value of wind speed at a height of 10m (m/s), which is how much wind is required to generate dust at a given site from an erodible surface. A default value of 7.2 m/s has been used in the derivation of HILs (EA 2009)

F_x = empirical function calculated based on the ratio of mean and threshold wind speeds as noted by EA (2009). For the derivation of HILs the following was used:

$$F_x = 0.18 \times (8x^3 + 12x)\exp(-x^2), \quad \text{where} \quad x = 0.886\frac{U_t}{U_m}$$ **Equation 20**

The PEF calculated for indoor air (and outdoors for HIL C) is based on a dust loading factor. The PEF is calculated as follows:

$$PEF_i\ (m^3/kg) = \frac{1}{DL \times 10^{-6}}$$ **Equation 21**

where:

DL = dust loading factor (mg dust/m^3 air)

10^{-6} = conversion factor for mg to kg

1.5 Calculation of plant concentration factors

The concentration of contaminants in edible portions of fruit and vegetables is estimated from the relationship between soil and plant and described using a soil-to-plant concentration factor (CF_x).
For inorganic contaminants, the CF_x values are derived from available literature (relevant to below- or above-ground crops).
For organic contaminants, there is a range of equations available that is based on experimental data. Where relevant, plant uptake of organic compounds has been estimated in the derivation of HILs using the equations presented by EA (2009), which are detailed as follows (refer to EA (2009) for further explanation of the basis for these equations):

Root Crops

$$CF_{root}\ (mg/kg\ fw\ plant\ per\ mg/kg\ dw\ soil) = \frac{\left(\frac{Q}{K_{oc} \times F_{oc}}\right)}{\frac{Q}{\left[\frac{W}{\rho_p} + \frac{L}{\rho_p} \times 1.22 K_{ow}^{0.77}\right]} + (k_g + K_m)\,\rho_p\ RV}$$

Equation 22

where:

Q	= transpiration stream flow rate, (cm^3/day) (assumed equal to the default of 1000)
K_{oc}	=organic carbon–water partition coefficient for the contaminant, (cm^3/g) (compound-specific)
F_{oc}	= fraction of organic carbon in the soil, (unitless)
K_{ow}	= octanol–water partition coefficient, (unitless) (compound-specific)
W	= root water content, (g/g) (assumed equal to the default of 0.89)
L	= root lipid content on a mass basis, (g/g) (assumed equal to the default of 0.025)
ρ_p	= plant root density, ($g/cm^{3)}$ (assumed equal to the default of 1)
k_g	= first order growth rate constant, per day (assumed equal to the default of 0.1)
K_m	= first order metabolism rate constant, (per day) (assumed equal to the default of 0)
RV	= root volume, ($cm^{3)}$ (assumed equal to the default of 1000)

Tuber Crops

Calculations presented for tuber crops are based on potatoes as representative crops for this group.

$$CF_{tuber}\ (mg/kg\ fw\ plant\ per\ mg/kg\ dw\ soil) = \frac{k_1}{k_2 + k_g}$$

Equation 23

where:

$$k_1 = k_2\left(\frac{K_{pw}}{K_{oc} \times F_{oc}}\right)$$

Equation 24

$$K_{pw} = \left(\frac{W}{\rho_p}\right) + (f_{ch}K_{ch}) + \left(\frac{L}{\rho_p}\right)1.22K_{ow}^{0.77}$$

Equation 25

$$k_2 = \frac{23\left(\frac{3600D_{water}(W^{7/3}/\rho_p)}{K_{pw}}\right)}{R^2}$$

Equation 26

where:

k_1 = rate of chemical flux into the potato, (per hour) (Equation 24)

k_2 = rate of chemical flux out of the potato, (per hour) (Equation 26)

k_g = exponential rate of growth of the potato, (per hour) (assumed equal to the default of 0.0014)

F_{oc} = fraction of organic carbon in the soil, (unitless)

K_{oc} =organic carbon–water partition coefficient for the contaminant, (cm^3/g) (compound-specific)

D_{water} = chemical diffusion coefficient in water, (m^2/s) (compound-specific)

ρ_p = potato tissue density, ($g/cm^{3)}$ (assumed equal to the default of 1)

R = radius of the potato, (m) (assumed equal to the default of 0.04)

W = water content of potato, (g/g) (assumed equal to the default of 0.79)

K_{pw} = equilibrium partition coefficient between potato and water, (cm^3/g) (Equation 25)

f_{ch} = fraction of carbohydrates in the potato, (unitless) (assumed equal to the default of 0.209)

L = lipid content of potato on a mass basis, (g/g) (assumed equal to the default of 0.001)

K_{ow} = octanol–water partition coefficient, (unitless) (compound-specific)

K_{ch} = carbohydrate–water partition coefficient, (cm^3/g) (calculated from chemical lipophilicity according to the following table)

Chemical log K_{ow}	Chemical K_{ch} (cm^3/g)
<0	0.1
≥0 but <1	0.2
≥1 but <2	0.5
≥2 but <3	1
≥3 but <4	2
≥4	3

Green Vegetables

$$CF_{green} = (10^{0.95 \log K_{OW} - 2.05} + 0.82) \times (0.784 \times 10^{-0.434(\log K_{OW} - 1.78)^2 / 2.44}) \times \left(\frac{\rho_S}{\theta_{WS} + (\rho_S . K_{OC} . f_{OC})} \right)$$

(mg/kg fresh weight [fw] plant per mg/kg dry weight [dw] soil)

Equation 27

where:

K_{oc} =organic carbon–water partition coefficient for the contaminant, (cm^3/g) (compound-specific)

f_{oc} = fraction of organic carbon in the soil, (unitless)

K_{ow} = octanol-water partition coefficient, (unitless) (compound-specific)

ρ_s = dry soil bulk density, ($g/cm^{3)}$

θ_{WS} = soil-water content by volume, ($cm^3/cm^{3)}$

Tree Fruit

$$CF_{fruit}\ (mg/kg\ \ fw\ plant\ \ per\ mg/kg\ dw\ soil) = \frac{0.001 \times (M_f Q_{fruit} DM_{fruit}) \left(\frac{C_{stem}}{K_{wood}} \right) / M_f}{C_{soil}}$$

Equation 28

where:

$$C_{stem}\ (mg/g) = \frac{\left[\left(\frac{C_{soil}}{K_{oc} F_{oc}} \right) 0.756 e^{\frac{-(\log K_{ow} - 2.5)^2}{2.58}} \right] \left[\frac{Q}{M} \right]}{\frac{Q}{K_{wood} M} + k_e + k_g}$$

Equation 29

$$\log K_{wood} = -0.27 + 0.632 \log K_{ow}$$

Equation 30

where:

M_f = mass of fruit, (g fw) (assumed equal to the default of 1)

Q_{fruit} = water flow rate per unit mass of fruit, (cm^3/g fw) (assumed equal to the default of 20)

DM_{fruit} = dry matter content of fruit, (g/g) (assumed equal to the default of 0.16)

C_{stem} = chemical concentration in the woody stem (mg/g) (Equation 29)

K_{wood} = wood–water partition coefficient, (mg/g dw wood per mg/cm^3 water) (Equation 30)

C_{soil} = total chemical concentration in soil, (mg/kg dw) (assumed to be 1 for establishing ratio)

K_{oc} = organic carbon–water partition coefficient for the contaminant, (cm^3/g) (compound-specific)

f_{oc} = fraction of organic carbon in the soil, (unitless)

K_{ow} = octanol–water partition coefficient, (unitless) (compound-specific)

Q = transpiration stream flow rate, (cm^3/year) (assumed equal to the default of 25,000,000)

M = mass of the woody stem, (g dw) (assumed equal to the default of 50,000)

k_e = rate of chemical metabolism, (per year) (assumed equal to the default of 0)

k_g = rate of dilution due to wood growth, (per year) (assumed equal to the default of 0.01)

1.6 Bibliography

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2 Attachment A

2.1 Worked Example: Calculation of HIL A for cadmium

This attachment provides further detail on the calculation of the low-density residential (HIL A) calculation for cadmium based on the equations presented in this appendix, exposure assumptions presented in Table 5 of the main schedule and the information presented in Appendix A for cadmium. The calculations presented are also summarised in Appendix C.
Based on the information presented in Appendix A, the HIL for cadmium has been undertaken on the basis that it is a threshold contaminant, where the most sensitive receptor is a child aged 2–3 years. Hence only threshold calculations have been undertaken for this chemical, where the following assumptions have been used from Appendix A:

Oral TRV (TRV_O) = 0.0008 mg/kg/day (WHO 2010)
Dermal absorption (DAF) = negligible (0%)
Inhalation TRV (TRV_I) = 0.000005 mg/m^3 (WHO 2000)
Background intakes from other sources:
BI_O = 60% for oral intakes
BI_i = 20% for inhalation

Calculation for Ingestion of Soil/dust

Based on Equation 3, the $HIL_{ingestion}$ is calculated for cadmium as follows:

$$HIL_{ingestion}\ (mg/kg) = \frac{(TRV_o(100\% - BI_o)) \times BW_C \times AT_T}{IR_{SC} \times BA_o \times CF \times EF \times ED} = \frac{0.0008\ x\ 40\%\ x15\ x\ 2190}{100\ x\ 100\%\ x\ 0.000001 x\ 365\ x\ 6} = 48\ (mg/kg)$$

Equation 3

where:

TRV_o = toxicity reference value relevant for the quantification of oral intakes, (as mg/kg/day for threshold contaminants) = 0.0008 mg/kg/day
BI_o = background intakes relevant to oral/dermal exposures (from sources other than soil, which include food, water, air and consumer products where relevant) (% of the TRVo) = 60% for oral intakes
IR_{SC} = ingestion rate of soil/dust by young child (mg/day) = 100 mg/day
BA_o = oral bioavailability (unitless, expressed as a fraction of 1) = 100% or 1 for cadmium
CF = conversion factor of $1x10^{-6}$ to convert mg to kg
EF = exposure frequency (days/year) = 365 days per year
ED_C = exposure duration for young child (years) = 6 years
BW_C = body weight of young child (kg) = 15 kg
AT_T = averaging time for threshold contaminants (days, = ED x 365 days) = 6 x 365 = 2190 days

Calculation for Dermal Absorption from Soil/dust

Based on information presented in Appendix A, dermal absorption of cadmium in soil in considered negligible and hence no calculation is required for this pathway.

Calculation for Inhalation of Dust

Based on Equation 9, the HIL_{dust} is calculated for cadmium as follows:

$$HIL_{dust}\ (mg/kg) = \frac{(TRV_i(100\% - BI_i)) \times AT_T}{\left[\left[\frac{1}{PEF_o} \times ET_{co}\right] + \left[\frac{1}{PEF_i} \times TF \times ET_{ci}\right]\right] \times RF \times EF \times ED_C} = \frac{0.000005 \text{ x } 80\% \text{ x } 52560}{\left[\left[\frac{4}{3x10^{10}}\right] + \left[\frac{0.5 \text{ x } 20}{2.6x10^{7}}\right]\right] \text{x } 0.375 \text{ x } 365 \text{ x } 6}$$

$$= 665\ (mg/kg)$$

Equation 9

where:

TRV_i = toxicity reference value relevant for the quantification of inhalation intakes, (as mg/m^3) = 0.000005 mg/m^3

BI_i = background intakes relevant to inhalation exposures (from sources other than soil, which include food, water, air and consumer products where relevant) (fraction relevant to the % allocated to background intakes) = 20% for inhalation intakes

$PEF_{i,o}$ = particulate emission factor (or dust loading) for outdoor (O) or indoor (I) air (m^3/kg) = calculated as below using Equations 19 to 21, PEFo = $3x10^{10}$ and PEFi = $2.6x10^7$ (m^3/kg)

$ET_{ci,co}$ = exposure time outdoors (O) or indoors (I) for young child (hours/day) = 4 hours/day outdoors and 20 hours per day indoors

TF = indoor dust transport factor (unitless) = 0.5

RF = lung retention factor relevant for the inhalation of dust from site (unitless) = 0.375

EF = exposure frequency (days/year) = 365 days per year

ED_C = exposure duration for young child (years) = 6 years

AT_T = averaging time for threshold contaminants (hours, = ED x 365 days x 24 hours) = 6 x 365 x 24 = 52 560 hours

$$PEF_o\ (m^3/kg) = \frac{Q/C \text{ x } 3600}{0.036 \text{ x } (1-V) \text{ x } (\frac{U_m}{U_t})^3 \text{ x } F_x} = \frac{90.8 \text{ x } 3600}{0.036 \text{ x } 0.25 \text{ x } (\frac{2.4}{7.2})^3 \text{ x } 0.032} = 3x10^{10}$$

Equation 19

where:

PEF_O = particulate emission factor outdoors (mg/kg soil per mg/m^3 air)

Q/C = air dispersion factor which describes the dispersion of soil particles in the atmosphere of a theoretical outdoor box. A value of 90.8 ($g/m^2/s$ per kg/m^3) has been used in the derivation of HILs. The value is a default value recommended by US EPA (2002) for small sites (0.5 acres).

V = the fraction of outdoor surface cover (0= bare soil), (unitless) = 0.75

U_m = mean annual wind speed at a height of 10m (m/s), assumed to be 8.75 km/hr (or 2.4 m/s) based on the average 9 am and 3 pm winds from Canberra

U_t = threshold value of wind speed at a height of 10m (m/s), which is how much wind is required to generate dust at a given site from an erodible surface. A default value of 7.2 m/s has been used in the derivation of HILs (EA 2009a)

F_x = empirical function calculated based on the ratio of mean and threshold wind speeds as noted by EA (2009a) = 0.032 based on the following:

$$F_x = 0.18 \times (8x^3 + 12x)\exp(-x^2), \quad \text{where} \quad x = 0.886\frac{U_t}{U_m} = 0.886\frac{7.2}{2.4} = 2.6$$

Equation 20

$$PEF_i\,(m^3/kg) = \frac{1}{DL \times 10^{-6}} = 2.6x10^7$$

Equation 21

where:

DL = dust loading factor (mg dust/m^3 air) = 39 µg/m^3 = 0.039 mg/m^3 (as per Section 5.3.3.2 of Schedule B7)

10^{-6} = conversion factor for mg to kg

Calculation for Ingestion of Cadmium via Home-grown Produce

Based on Equations 15 and 16, the $HIL_{plant\ uptake}$ is calculated for cadmium as follows:

$$HIL_{plant\,uptake}\,(mg/kg) = \frac{(TRV_o(100\% - BI_O)) \times BW_C \times AT_T}{UF_V \times EF \times ED_C} = \frac{0.0008\ x\ 40\%15\ x\ 2190}{0.00044\ x\ 365\ x\ 6} = 11(mg/kg)$$

Equation 15

$$UF_{VC}\,(kg/day) = F_{HG} \times \left(\left[CF_{tuber} \times C_{tuber}\right] + \left[CF_{root} \times C_{root}\right] + \left[CF_{green} \times C_{green}\right] + \left[CF_{fruit} \times C_{fruit}\right]\right)$$ **Equation 16**

where:

TRV_o = toxicity reference value relevant for the quantification of oral intakes, (as mg/kg/day for threshold contaminants) = 0.0008 mg/kg/day;

BI_O = background intakes relevant to oral/dermal exposures (from sources other than soil, which include food, water, air and consumer products where relevant) (fraction relevant to the % allocated to background intakes) = 60%

UF_{VC} = plant uptake factor calculated for the consumption of home-grown produce by young children (kg/day) = $4.4x10^{-4}$ kg/day based on Equation 16

CF_y = plant concentration factors relevant for produce type (y), (chemical-specific) (mg/kg fresh weight produce to mg/kg dry weight soil), see table below

C_y = consumption rate of each produce type (y) (kg/day), see table below

F_{HG} = fraction of all fruit and vegetable produce consumed that is home-grown (unitless) = 10% or 0.1 as per Schedule B7

EF = exposure frequency (days/year) = 365 days per year

ED_C = exposure duration for young children (years) = 6 years

BW_C = body weight of young child (kg) = 15 kg

AT_T = averaging time for threshold contaminants (days, = ED x 365 days) = 2190 days

For cadmium the plant uptake factors, or concentration factors, (CF_y) for the different produce types are presented in Appendix A. The consumption rate of each produce type, by young children, is presented in Table 7 in Schedule B7. These are both summarised for cadmium in the following table. These have been used in Equation 16 to calculate the plant uptake factor for young children.

Produce Group	Plant Uptake Factors or Concentration Factors CF_y (mg/kg produce fresh weight per mg/kg soil) (EA 2009c) – from Appendix A	Child consumption rate for each produce group (kg/day) – from Table 7 in Schedule B7
Green vegetables	0.052	0.055
Root vegetables	0.029	0.017
Tuber vegetables	0.031	0.028
Tree fruit	0.0014	0.18

As noted in Appendix A, and the calculation sheets in Appendix C, as background intakes (via ingestion) are dominated by intakes from food sources, the inclusion of uptakes from home-grown produce as well as all other food sources results in some double counting of cadmium intakes via food sources. As discussed in Section 1.3.5 to correct for this double counting, the calculated HIL from plant uptake has been adjusted by a factor of 2-fold (which has the effect of reducing the contribution from this pathway by 50%).
Hence the calculated HIL plant uptake = 21 mg/kg (after rounding)

Calculation of the Residential HIL from all Exposure Pathways

The final HIL is calculated by combining the pathway-specific HILs calculated above using Equation 2 (for the complete pathways of exposure) (as rounded):

$$HIL\,(mg/kg) = \frac{1}{\left[\frac{1}{HIL_{ingestion}}\right] + \left[\frac{1}{HIL_{plant\ uptake}}\right] + \left[\frac{1}{HIL_{dust}}\right]} = \frac{1}{\frac{1}{48} + \frac{1}{21} + \frac{1}{665}} = 15\,mg/kg$$

Equation 2

As noted in Appendix A, for cadmium an HIL A of 15 mg/kg has been calculated using the above equations. The value of 15 mg/kg is considered to be essentially the same (with consideration of uncertainties and accuracy of HIL calculations) as the existing HIL of 20 mg/kg. There is no new data available that suggests that the existing HIL is not adequately protective and that, given the level of uncertainty in the calculation of any HIL, the existing HIL A of 20 mg/kg has been retained in the NEPM.

2.2 Bibliography

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3 Attachment B

3.1 Worked Example: Calculation of HIL A for benzo(a)pyrene

This attachment provides further detail on the calculation of the low-density residential (HIL A) calculation for benzo(a)pyrene (BaP) based on the equations presented in this appendix, exposure assumptions presented in Table 5 of the main schedule and the information presented in Appendix A for BaP. The calculations presented are also summarised in Appendix C.
Based on the information presented in Appendix A, BaP has been considered to be a genotoxic carcinogen where the HIL has been calculated on the basis of a non-threshold approach, considering exposures over a lifetime (i.e. as a child and adult). The assessment of BaP is complex (as outlined in Appendix A), where the following have been considered in the derivation of the HIL:

Recommendation for BaP and carcinogenic PAHs as BaP TEF

Oral TRV (TRV_O) = 0.208 $(mg/kg/day)^{-1}$ (MfE 2011) for all routes of exposure

Value has been compared with TRV_O = 0.5 $(mg/kg/day)^{-1}$ (WHO 2011) for all routes of exposure

Dermal absorption factor (DAF) = 0.06 (or 6%) (MfE 2011)

Note: early lifetime exposures to BaP may need to be addressed in the quantification of exposure as per US EPA (2005).

As discussed in Appendix A, when determining the HIL A value, calculations have been undertaken for BaP where the TRVo from MfE (2011) and WHO (2011) have been considered, where early lifetime exposures have been considered and where the dermal-specific toxicity reference value has also been considered. For the purpose of this worked example, calculations have been presented that support the HIL A value adopted, which is based on the TRVo available from WHO (2011), consideration of early lifetime exposures and no additional consideration of the dermal-specific toxicity reference value.

Calculation for Ingestion of Soil/dust

Based on Equations 3 and 4, as well as the age-adjustment factors outlined by US EPA (2005), the $HIL_{ingestion}$ is calculated for BaP as follows:

$$HIL_{ingestion}\ (mg/kg) = \frac{TR}{Intake\,Factor_o \times TRV_o} \qquad \textbf{Equation 4}$$

$$Intake\,Factor_o\ (kg/kg/day) = \sum\left(\frac{IRs_x \times BAo \times CF \times EF_x \times ED_x}{BW_x \times AT_{NT}}\right) \qquad \textbf{Equation 5}$$

where:

TRV_o = toxicity reference value relevant for the quantification of oral intakes, (as $(mg/kg/day)^{-1}$ for non-threshold contaminants) = 0.5 $(mg/kg/day)^{-1}$

TR = target risk for non-threshold contaminants (unitless) = $1x10^{-5}$

$\sum$ = signifies the sum over all receptor groups *x* considered (in the HILs derived, these groups include a child (C) and adult (A))

IRs_X = ingestion rate of soil/dust by each receptor group *x* (mg/day) = 50 mg/day for adults and 100 mg/day for young children

BAo = oral bioavailability (unitless) = 100% or 1 for BaP

CF = conversion factor of $1x10^{-6}$ to convert mg to kg

EFx = exposure frequency relevant to exposures by each receptor group *x* (days/year) = 365 days/year for both adults and children

EDx = exposure duration relevant to exposures by each receptor group *x* (years) = 6 years for young children and 29 years for adults

BWx = body weight relevant to each receptor group *x* (kg) = 15kg for young children and 70 kg for adults

AT_{NT} = averaging time for non-threshold contaminants (days, = 70 years x 365 days) = 25 550 days

The calculated intake factor has taken into account age-adjustment factors that relate to the potential for exposures during childhood to be more sensitive than those later in life. This has been undertaken using the age adjustment factors (ADAF) outlined by US EPA (2005). The adjustment factors are as follows:

- ADAF = 10 during the first 2 years of life
- ADAF = 3 for ages 2 through to less than 16 years
- ADAF = 1 for ages 16 through to 70 years.

The lifetime risk calculations undertaken for non-threshold compounds (based on the equations in this appendix) are based on exposures that occur as a young child aged 0–5 years, and then as an adult from ages 6 and older. The ADAFs have been applied within these calculations as follows:

$$\text{Intake Factor}_{o}\ (\text{kg/kg/day}) = 10\ \text{x}\ \frac{\text{IR}_{c}(100\ \text{mg/day})\ \text{x}\ \text{BA}_{O}(1)\ \text{x}\ \text{CF}\ (1\text{x}10^{-6}\ \text{mg/kg})\ \text{x}\ \text{EF}_{C}(365\ \text{days/year})\ \text{x}\ 2\ (\text{years for ADAF})}{\text{BW}_{C}(15\text{kg})\ \text{x}\ \text{AT}\ (25550\text{days})}$$

$$+\ 3\ \text{x}\ \frac{\text{IR}_{c}(100\ \text{mg/day})\ \text{x}\ \text{BA}_{O}(1)\ \text{x}\ \text{CF}\ (1\text{x}10^{-6}\ \text{mg/kg})\ \text{x}\ \text{EF}_{C}(365\ \text{days/year})\ \text{x}\ 4\ (\text{years for ADAF from ages}\ 2-5)}{\text{BW}_{C}(15\text{kg})\ \text{x}\ \text{AT}\ (25550\text{days})}$$

$$+\ 3\ \text{x}\ \frac{\text{IR}_{A}(50\ \text{mg/day})\ \text{x}\ \text{BA}_{O}(1)\ \text{x}\ \text{CF}\ (1\text{x}10^{-6}\ \text{mg/kg})\ \text{x}\ \text{EF}_{A}(365\ \text{days/year})\ \text{x}\ 10\ (\text{years for ADAF from ages}\ 6-15)}{\text{BW}_{A}(70\text{kg})\ \text{x}\ \text{AT}\ (25550\text{days})}$$

$$+\ 1\text{x}\ \frac{\text{IR}_{A}(50\ \text{mg/day})\ \text{x}\ \text{BA}_{O}(1)\ \text{x}\ \text{CF}\ (1\text{x}10^{-6}\ \text{mg/kg})\ \text{x}\ \text{EF}_{A}(365\ \text{days/year})\ \text{x}\ 19\ (\text{years for ADAF from ages}\ 16-35)}{\text{BW}_{A}(70\text{kg})\ \text{x}\ \text{AT}\ (25550\text{days})}$$

$$= 3.5\text{x}10^{-6}\ (\text{kg/kg/day})$$

Based on the above the following is then calculated:

$$\text{HIL}_{\text{ingestion}}\ (\text{mg/kg}) = \frac{\text{TR}\ (1\text{x}10^{-5})}{\text{Intake Factor}_{o}\ (3.5\text{x}10^{-6}\ \text{kg/kg/day}) \times \text{TRV}_{o}(0.5\ (\text{mg/kg/day})^{-1})} = 5.6\ (\text{mg/kg})$$

Calculation for Dermal Absorption from Soil/dust

Based on Equations 7 and 8, as well as the age-adjustment factors outlined by US EPA (2005), the HI_{dermal} is calculated for BaP as follows:

Non-threshold contaminants (lifetime exposures)

$$HIL_{dermal}\ (mg/kg) = \frac{TR}{Intake\ Factor_D \times TRV_D} \qquad \textbf{Equation 7}$$

$$Intake\ Factor_D\ (kg/kg/day) = \sum\left(\frac{SA_x \times AF \times DAF \times CF \times EF_x \times ED_x}{BW_x \times AT_{NT}}\right) \qquad \textbf{Equation 8}$$

where:

TRV_D = toxicity reference value relevant for the quantification of dermal intakes, (as $(mg/kg/day)^{-1}$ for non-threshold contaminants) = 0.5 $(mg/kg/day)^{-1}$

TR = target risk for non-threshold contaminants (unitless) = $1x10^{-5}$

$\sum$ = signifies the sum over all receptor groups *x* considered (in the HILs derived, these groups include a child (C) and adult (A))

SA_x = exposed skin surface area for all receptor groups *x* (cm^2) = 2700 cm^2 for young children and 6300 cm^2 for adults

AF = soil to skin adherence factor ($mg/cm^2/day$) = 0.5 $mg/cm^2/day$

DAF = dermal absorption factor, (chemical-specific) (unitless) = 6% or 0.06 for BaP

CF = conversion factor of $1x10^{-6}$ to convert mg to kg

EF_x = exposure frequency relevant to exposures by all receptor groups *x* (days/year) = 365 days per year for adults and children

ED_x = exposure duration relevant to exposures by all receptor groups *x* (years) = 6 years as child from 0–5 years and 29 years as adult aged 6 and older

BW_x = body weight relevant to each receptor group *x* (kg) = 15 kg for young children and 70 kg for adults

AT_{NT} = averaging time for non-threshold contaminants (days, = 70 years x 365 days) = 25 550 days

As noted above for the calculation of the soil ingestion HIL, age-adjustment factors have been incorporated into the calculation of the intake factor, with the calculations considered as follows:

$$\text{Intake Factor}_D\ (\text{kg/kg/day}) =$$

$$10\,x\,\frac{SA_C(2700\,cm^2)\,x\,AF(0.5\,mg/cm^2/day)\,x\,DAF(0.06)\,x\,CF(1x10^{-6})\,x\,EF_C(365\,d/yr)\,x\,2\,(\text{years for ADAF})}{BW_C(15\,kg)\,x\,AT(25550\,\text{days})}$$

$$+3\,x\,\frac{SA_C(2700\,cm^2)\,x\,AF(0.5\,mg/cm^2/day)\,x\,DAF(0.06)\,x\,CF(1x10^{-6})\,x\,EF_C(365\,d/yr)\,x\,4\,(\text{years for ADAF from ages } 2-5)}{BW_C(15\,kg)\,x\,AT(25550\,\text{days})}$$

$$+3\,x\,\frac{SA_A(6300\,cm^2)\,x\,AF(0.5\,mg/cm^2/day)\,x\,DAF(0.06)\,x\,CF(1x10^{-6})\,x\,EF_A(365\,d/yr)\,x\,10\,(\text{years for ADAF from ages } 6-15)}{BW_A(70\,kg)\,x\,AT(25550\,\text{days})}$$

$$+1x\,\frac{SA_A(6300\,cm^2)\,x\,AF(0.5\,mg/cm^2/day)\,x\,DAF(0.06)\,x\,CF(1x10^{-6})\,x\,EF_A(365\,d/yr)\,x\,19\,(\text{years for ADAF from ages } 16-35)}{BW_A(70\,kg)\,x\,AT(25550\,\text{days})}$$

$$= 4.3x10^{-6}\ (\text{kg/kg/day})$$

Based on the above, the following is then calculated:

$$HIL_{dermal}\ (mg/kg) = \frac{TR\,(1x10^{-5})}{\text{Intake Factor}_D\,(4.3x10^{-6}\ \text{kg/kg/day}) \times TRV_D(0.5\,(mg/kg/day)^{-1})} = 4.6\ (mg/kg)$$

Calculation for Inhalation of Dust

Based on Equations 10 and 11, the HIL_{dust} is calculated for BaP as follows:

$$HIL_{dust}\ (mg/kg) = \frac{TR}{\text{Intake Factor}_{dust} \times TRV_i} \qquad \textbf{Equation 10}$$

$$\text{Intake Factor}_{dust}\ (kg/m^3) = \sum \left(\frac{\left[\left[\frac{1}{PEF_o} \times ET_o\right] + \left[\frac{1}{PEF_i} \times TF \times ET_i\right]\right] \times RF \times EF_x \times ED_x}{AT_{NT}} \right) \qquad \textbf{Equation 11}$$

where:

TRV_i = toxicity reference value relevant for the quantification of inhalation intakes, (as $(mg/m^{3)-1}$ for non-threshold contaminants) = 0.14 $(mg/m^{3)-1})$ (based on the TRVo and conversion based on inhalation of 20 m^3/day and a body weight of 70 kg)

TR = target risk for non-threshold contaminants (unitless) = $1x10^{-5}$

$\sum$ = signifies the sum over all receptor groups *x* considered (in the HILs derived, these groups include a child (C) and adult (A))

$PEF_{i,o}$ = particulate emission factor (or dust loading) for outdoor (O) or indoor (I) air (m^3/kg), calculated as outlined below

$ET_{i,o}$ = exposure time indoors (I) and outdoors (O) for adults and children (as relevant) (hours/day) = 20 hours indoors and 4 hours outdoors for both young children and adults

TF = indoor dust transport factor (unitless) = 0.5

RF = lung retention factor relevant for the inhalation of dust from site (unitless) = 0.375

EF_x = exposure frequency for all receptor groups *x* (days/year) = 365 days per year for both young children and adults

ED_x = exposure duration for all receptor groups *x* (years) = 6 years as child from 0–5 years and 29 years as adult aged 6 and older

AT_{NT} = averaging time for non-threshold contaminants (hours, = 70 years x 365 days x 24 hours) = 613 200 hours

$$PEF_o\,(m^3/kg) = \frac{Q/C \times 3600}{0.036 \times (1-V) \times (\frac{U_m}{U_t})^3 \times F_x} = \frac{90.8 \times 3600}{0.036 \times 0.25 \times (\frac{2.4}{7.2})^3 \times 0.032} = 3x10^{10}$$

Equation 19

where:

PEF_O = particulate emission factor outdoors (mg/kg soil per mg/m^3 air)

Q/C = air dispersion factor which describes the dispersion of soil particles in the atmosphere of a theoretical outdoor box. A value of 90.8 $(g/m^2/s$ per $kg/m^3)$ has been used in the derivation of HILs. The value is a default value recommended by US EPA (2002) for small sites (0.5 acres).

V = the fraction of outdoor surface cover (0= bare soil), (unitless) = 0.75

U_m = mean annual wind speed at a height of 10m (m/s), assumed to be 8.75 km/hr (or 2.4 m/s) based on the average 9 am and 3 pm winds from Canberra

U_t = threshold value of wind speed at a height of 10m (m/s), which is how much wind is required to generate dust at a given site from an erodible surface. A default value of 7.2 m/s has been used in the derivation of HILs (EA 2009)

F_x = empirical function calculated based on the ratio of mean and threshold wind speeds as noted by EA (2009) = 0.032 based on the following:

$$F_x = 0.18 \times (8x^3 + 12x)\exp(-x^2), \quad \text{where} \quad x = 0.886\frac{U_t}{U_m} = 0.886\frac{7.2}{2.4} = 2.6$$

Equation 20

$$PEF_i\,(m^3/kg) = \frac{1}{DL \times 10^{-6}} = 2.6x10^7$$

Equation 21

where:

DL = dust loading factor (mg dust/m^3 air) = 39 µg/m^3 = 0.039 mg/m^3 (as per Section 5.3.3.2 of Schedule B7)

10^{-6} = conversion factor for mg to kg

As noted above for the calculation of the soil ingestion HIL, age-adjustment factors have been incorporated into the calculation of the intake factor, with the calculations considered as follows:

$$\text{Intake Factor}_{\text{dust}}\ (\text{kg}/\text{m}^3) = 10\ \text{x}\ \frac{\left[\left[\frac{1}{\text{PEF}_o(3\text{x}10^{10})}\times \text{ET}_o(4\text{hrs})\right]+\left[\frac{1}{\text{PEF}_i(2.6\text{x}10^{7})}\times \text{TF}(0.5)\times \text{ET}_i(20\text{hrs})\right]\right]\times \text{RF}(0.375)\times \text{EF}_C(365\text{d}/\text{yr})\times \text{ED}_C(2\ \text{yrs for ADAF})}{\text{AT}_{NT}(613200\ \text{hrs})}$$

$$+\ 3\ \text{x}\ \frac{\left[\left[\frac{1}{\text{PEF}_o(3\text{x}10^{10})}\times \text{ET}_o(4\text{hrs})\right]+\left[\frac{1}{\text{PEF}_i(2.6\text{x}10^{7})}\times \text{TF}(0.5)\times \text{ET}_i(20\text{hrs})\right]\right]\times \text{RF}(0.375)\times \text{EF}_C(365\text{d}/\text{yr})\times \text{ED}_C(4\ \text{yrs for ADAF for ages}\ 2-5)}{\text{AT}_{NT}(613200\ \text{hrs})}$$

$$+\ 3\ \text{x}\ \frac{\left[\left[\frac{1}{\text{PEF}_o(3\text{x}10^{10})}\times \text{ET}_o(4\text{hrs})\right]+\left[\frac{1}{\text{PEF}_i(2.6\text{x}10^{7})}\times \text{TF}(0.5)\times \text{ET}_i(20\text{hrs})\right]\right]\times \text{RF}(0.375)\times \text{EF}_A(365\text{d}/\text{yr})\times \text{ED}_A(10\ \text{yrs for ADAF for ages}\ 6-15)}{\text{AT}_{NT}(613200\ \text{hrs})}$$

$$+\ 1\text{x}\ \frac{\left[\left[\frac{1}{\text{PEF}_o(3\text{x}10^{10})}\times \text{ET}_o(4\text{hrs})\right]+\left[\frac{1}{\text{PEF}_i(2.6\text{x}10^{7})}\times \text{TF}(0.5)\times \text{ET}_i(20\text{hrs})\right]\right]\times \text{RF}(0.375)\times \text{EF}_A(365\text{d}/\text{yr})\times \text{ED}_A(19\ \text{yrs for ADAF for ages}\ 16-35)}{\text{AT}_{NT}(613200\ \text{hrs})}$$

$$= 7\text{x}10^{-9}\ (\text{kg}/\text{m}^3)$$

Based on the above, the following is then calculated:

$$\text{HIL}_{\text{dust}}\ (\text{mg}/\text{kg}) = \frac{\text{TR}\ (1\text{x}10^{-5})}{\text{Intake Factor}_{\text{dust}}\ (7\text{x}10^{-9})\times \text{TRV}_i\ (0.143\ (\text{mg}/\text{m}^3)^{-1})} = 10000\ (\text{mg}/\text{kg})$$

Calculation for Ingestion of BaP via Home-grown Produce

As discussed in Appendix A, the potential for the uptake of BaP into plants is considered to be limited and hence this pathway has not been considered in the calculation of the HIL A.

Calculation of the Residential HIL from all Exposure Pathways

The final HIL is calculated by combining the pathway-specific HILs calculated above using Equation 2 (for the complete pathways of exposure) (as rounded):

$$\mathrm{HIL}\,(\mathrm{mg/kg}) = \frac{1}{\left[\frac{1}{\mathrm{HIL}_{\mathrm{ingestion}}}\right] + \left[\frac{1}{\mathrm{HIL}_{\mathrm{dermal}}}\right] + \left[\frac{1}{\mathrm{HIL}_{\mathrm{dust}}}\right]} = \frac{1}{\frac{1}{5.6} + \frac{1}{4.6} + \frac{1}{10000}} = 2.5\,\mathrm{mg/kg} = 3\,\mathrm{mg/kg}\ \text{(rounded to 1 s.f.)}$$

Equation 2

Based on these calculations, the HIL A for BaP = 3 mg/kg for the scenario presented.

3.2 Bibliography

EA 2009, *Updated technical background to the CLEA model,* Science report SC050021/SR3, Environment Agency, Bristol, UK.

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National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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This compilation has been split into 22 volumes

Volume 1: sections 1–6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of

Schedule B7

APPENDIX C

Derivation of Investigation Levels for Generic Land Uses

Appendix C Derivation of Investigation Levels for Generic Land Uses
HIL A - Low Density Residential

Summary of Exposure Parameters		Abbreviation	units	Parameter	References/Notes
Soil and Dust Ingestion Rate	- Young children (0-5 years)	IR_{SC}	mg/day	100	Schedule B7, Table 5
	- Adults	IR_{SA}	mg/day	50	Schedule B7, Table 5
Surface Area of Skin	- Young children (0-5 years)	SA_C	cm^2/day	2700	Schedule B7, Table 5
	- Adults	SA_A	cm^2/day	6300	Schedule B7, Table 5
Soil-to-Skin Adherence Factor		AF	mg/cm^2/day	0.5	Schedule B7, Table 5
Time Spent Outdoors		ETo	hours	4	Schedule B7, Table 5
Time Spent Indoors		ETi	hours	20	Schedule B7, Table 5
Lung Retention Factor		RF	-	0.375	Schedule B7, Table 5
Particulate Emission Factor		PEFo	(m^3/kg)	2.9E+10	Calculated for scenario, refer to Equations 19 and 20 and assumptions in Schedule B7
Indoor Air Dust Factor		PEFi	(m^3/kg)	2.6E+07	As per Equation 21 based assumptions presented in Schedule B7
Fraction of indoor dust comprised of outdoor soil		TF	-	0.5	Assume 50% soil concentration present in dust as noted in Schedule B7
Indoor Air-to-Soil Gas Attenuation Factor		α	-	0.1	Value adopted as discussed in Section 5.5 of Schedule B7
Body weight	- Young children (0-5 years)	BW_C	kg	15	Schedule B7, Table 5
	- Adults	BW_A	kg	70	Schedule B7, Table 5
Exposure Frequency		EF	days/year	365	Schedule B7, Table 5
Exposure Duration	- Young children (0-5 years)	ED_C	years	6	Schedule B7, Table 5
	- Adults	ED_A	years	29	Schedule B7, Table 5
Averaging Time (non-carcinogenic)		AT_T	days	ED*365	Calculated based on ED for each relevant age group, multiplied by 24 hours for the assessment of inhalation exposures
Averaging Time (carcinogenic)		AT_{NT}	days	25550	Based on lifetime of 70 years, multiplied by 24 hours for the assessment of inhalation exposures

Threshold Calculations - Young Child aged 2-3 years

Compound	Toxicity Reference Value Oral (TRV_O) (mg/kg/day)	GI Absorption (GAF) (unitless)	Toxicity Reference Value Dermal (TRV_D) (mg/kg/day)	Oral Bioavailability BA_O (%)	Dermal Absorption Factor (DAF) (unitless)	Background Intake Oral/Dermal (BI_O) (% of TDI)	Toxicity Reference Value Inhalation (TRV_I) (mg/m^3)	Background Intake Inhalation (BIi) (% of TC)		Plant Uptake Factor (incl % intake) Adults (kg/day) (eqn 16)	Plant Uptake Factor (incl % intake) Children (kg/day) (eqn 16)	Pathway Specific HILs (mg/kg)				Soil Vapour HIL (mg/m^3) (eqn 12)	Derived Interim Soil Gas HIL - Threshold (to 1 or 2 s.f.) (mg/m3)	Derived Soil HIL (not rounded) (mg/kg) (eqn 2 for relevant pathways)	Derived Soil HIL (to 1 or 2 s.f.) (mg/kg)	Notes
												Soil Ingestion (eqn 3)	Home-grown produce (eqn 15)	Dermal (eqn 6)	Dust (eqn 9)					
arsenic	0.002	1	0.002	100%	0.005	50%	0.001	0%			2.3E-05	1.5E+02	1.3E+03	2.2E+03	1.6E+05			126	**100**	
beryllium	0.002	0.007	0.000014	100%	0.001	30%	0.000020	0%			1.1E-05	2.1E+02	3.7E+03	1.1E+02	3.3E+03			69	**70**	
boron	0.2			100%		85%	0.7	85%				4.5E+03	NA	NA	1.7E+07			4499	**4500**	
cadmium	0.0008			100%		60%	0.000005	20%			4.5E-04	4.8E+01	2.1E+01	NA	6.6E+02			15	**15**	1
chromium (VI)	0.001			100%		10%	0.0001	0%			1.5E-04	1.4E+02	1.9E+02	NA	1.6E+04			78	**80**	1
cobalt	0.0014	1	0.0014	100%	0.001	20%	0.0001	0%			1.0E-04	1.7E+02	3.2E+02	1.2E+04	1.6E+04			109	**100**	
copper	0.14			100%		70%	0.49	70%				6.3E+03	NA	NA	2.4E+07			6298	**6000**	
manganese	0.16			100%		50%	0.00015	20%			3.1E-04	1.2E+04	7.8E+03	NA	2.0E+04			3822	**3800**	
methyl mercury	0.00023	1	0.00023	100%	0.001	80%	0.000805	80%				6.9E+00	NA	5.1E+02	2.6E+04			7	**7**	1
mercury (inorganic)	0.0006	0.07	0.000042	100%	0.001	40%	0.0002	10%			6.2E-05	5.4E+01	1.7E+02	2.8E+02	3.0E+04			36	**40**	
nickel	0.012	1	0.012	100%	0.005	60%	0.00002	20%			9.5E-05	7.2E+02	1.5E+03	1.1E+04	2.6E+03			397	**400**	
selenium	0.006			100%		60%	0.021	60%			1.2E-04	3.6E+02	5.9E+02	NA	1.4E+06			224	**200**	
zinc	0.5	1	0.5	100%	0.001	90%	1.75	90%				7.5E+03	NA	5.6E+05	2.9E+07			7398	**7400**	
cyanide (free) (no VI)	0.006	1	0.006	100%	0.1	50%	0.0008	0%				4.5E+02	NA	3.3E+02	1.3E+05			191	**200**	1
TCE							0.002	10%				NA	NA	NA	NA	2.2E-02	**0.02**			
1,1,1-TCA							5	0%				NA	NA	NA	NA	6.0E+01	**60**			
PCE							0.2	10%				NA	NA	NA	NA	2.2E+00	**2**			
cis-1,2-dichloroethene							0.007	0%				NA	NA	NA	NA	8.4E-02	**0.08**			
phenol	0.7	1	0.7	100%	0.1	30%	0.035	30%			2.3E-03	7.4E+04	3.1E+03	5.4E+04	4.0E+06			2848	**3000**	
pentachlorophenol	0.003	1	0.003	100%	0.24	0%	0.0105	0%				4.5E+02	NA	1.4E+02	1.7E+06			106	**100**	
cresols	0.1	1	0.1	100%	0.1	50%	0.35	50%			1.9E-03	7.5E+03	4.0E+02	5.6E+03	2.9E+07			357	**400**	
DDX	0.002	1	0.002	100%	0.018	0%	0.007	0%				3.0E+02	NA	1.2E+03	1.1E+06			241	**240**	
aldrin and dieldrin	0.0001	1	0.0001	100%	0.1	10%	0.00035	10%				1.4E+01	NA	1.0E+01	5.2E+04			5.7	**6**	
chlordane	0.0005	1	0.0005	100%	0.04	0%	0.00175	0%				7.5E+01	NA	1.4E+02	2.9E+05			49	**50**	
endosulfan	0.006	1	0.006	100%	0.1	30%	0.021	30%				6.3E+02	NA	4.7E+02	2.4E+06			268	**270**	
endrin	0.0002	1	0.0002	100%	0.1	0%	0.0007	0%				3.0E+01	NA	2.2E+01	1.1E+05			13	**10**	
heptachlor	0.0001	1	0.0001	100%	0.1	0%	0.00035	0%				1.5E+01	NA	1.1E+01	5.7E+04			6.4	**6**	
HCB	0.00016	1	0.00016	100%	0.1	0%	0.00056	0%				2.4E+01	NA	1.8E+01	9.2E+04			10	**10**	
methoxychlor	0.005	1	0.005	100%	0.1	0%	0.0175	0%				7.5E+02	NA	5.6E+02	2.9E+06			319	**300**	
mirex	0.0002	1	0.0002	100%	0.1	0%	0.0007	0%				3.0E+01	NA	2.2E+01	1.1E+05			13	**10**	
toxaphene	0.00035	1	0.00035	100%	0.1	10%	0.001225	10%				4.7E+01	NA	3.5E+01	1.8E+05			20	**20**	
2,4,5-T	0.01	1	0.01	100%	0.1	0%	0.035	0%				1.5E+03	NA	1.1E+03	5.7E+06			638	**600**	
2,4-D	0.01	1	0.01	100%	0.05	0%	0.035	0%				1.5E+03	NA	2.2E+03	5.7E+06			895	**900**	
MCPA	0.01	1	0.01	100%	0.1	0%	0.035	0%				1.5E+03	NA	1.1E+03	5.7E+06			638	**600**	
MCPB	0.01	1	0.01	100%	0.1	0%	0.035	0%				1.5E+03	NA	1.1E+03	5.7E+06			638	**600**	
mecoprop	0.01	1	0.01	100%	0.1	0%	0.035	0%				1.5E+03	NA	1.1E+03	5.7E+06			638	**600**	
picloram	0.07	1	0.07	100%	0.1	0%	0.245	0%				1.1E+04	NA	7.8E+03	4.0E+07			4468	**4500**	
atrazine	0.005	1	0.005	100%	0.1	0%	0.0175	0%				7.5E+02	NA	5.6E+02	2.9E+06			319	**320**	
chlorpyrifos	0.003	1	0.003	100%	0.03	50%	0.0105	50%				2.3E+02	NA	5.6E+02	8.6E+05			160	**160**	
bifenthrin	0.01	1	0.01	100%	0.1	10%	0.035	10%				1.4E+03	NA	1.0E+03	5.2E+06			574	**600**	
PCBs	0.00002	1	0.00002	100%	0.14	0%	0.00007	0%			2.5E-04	3.0E+00	1.2E+00	1.6E+00	1.1E+04			0.6	**1**	
PBDE Flame Retardants (Br1-Br9)	0.0001	1	0.0001	100%	0.1	80%	0.00035	80%			2.1E-05	3.0E+00	1.4E+01	2.2E+00	1.1E+04			1.2	**1**	

Includes factor of 2 to adjust for inclusion of metals in background food and plant uptake - see Appendix A

NA Pathway not of significance for chemical assessed (refer to Appendix A for chemical-specific details)

1 Calculated value differs from final HIL adopted in Schedule B7. For these compounds the calculated value, and basis, were not considered sufficiently different from the former HIL and hence the former HIL was retained - refer to Appendix A for details

Non-Threshold Effects - Lifetime Exposures [young child and adult]																					
Compound	**Toxicity Reference Value Oral (TRV_O)** (mg/kg/day)$^{-1}$	**GI Absorption (GAF)** (unitless)	**Non-Threshold Slope Factor Dermal (SFd)** (mg/kg/day)$^{-1}$	**Oral Bioavailability BA_O (%)**	**Dermal Absorption Factor (DAF)** (unitless)		**Toxicity Reference Value Inhalation (TRV_I)** (mg/m^3)$^{-1}$		**Target Risk (TR)**	**Plant Uptake Factor (incl % intake) Adults** (kg/day) (eqn 16)	**Plant Uptake Factor (incl % intake) Children** (kg/day) (eqn 16)	**Pathway Specific HILs (mg/kg)**				**Soil Vapour HIL (mg/m^3)** (eqns 13 and 14)	**Derived Interim Soil Gas IL - Threshold (to 1 or 2 s.f.)** (mg/m3)	**Derived Soil HIL (not rounded)** (mg/kg) (eqn 2 for relevant pathways)	**Derived Soil HIL (to 1 or 2 s.f.)** (mg/kg)	**Notes**
												Soil Ingestion (eqns 4 and 5)	**Home-grown produce** (eqns 17 and 18)	**Dermal** (eqns 7 and 8)	**Dust** (eqns 10 and 11)					
TCE							0.004		1E-05			NA	NA	NA	NA	6.0E-02	**0.06**			
vinyl chloride							0.00880		1E-05			NA	NA	NA	NA	2.7E-02	**0.03**			
benzo(a)pyrene	0.5	1	0.5	100%	0.06		1.43E-01		1E-05			2.3E+01		1.3E+01	2.3E+04			8.2	**8**	2
benzo(a)pyrene (Early-Life)	0.5	1	0.5	100%	0.06		1.43E-01		1E-05			5.6E+00		4.6E+00	9.8E+03			2.53	**3**	2

NA Pathway not of significance for chemical assessed (refer to Appendix A for chemical-specific details)
2 Refer to Appendix A for discussion on different calculations conducted for benzo(a)pyrene and basis for HIL adopted

Appendix C Derivation of Investigation Levels for Generic Land Uses
HIL B - High Density Residential

Summary of Exposure Parameters		Abbreviation	units	Parameter	References / Notes
Soil and Dust Ingestion Rate	- Young children (0-5 years)	IR_{SC}	mg/day	25	25% of HIL A assumption, Schedule B7, Table 5
	- Adults	IR_{SA}	mg/day	12.5	25% of HIL A assumption, Schedule B7, Table 5
Surface Area of Skin	- Young children (0-5 years)	SA_C	cm^2/day	2700	Schedule B7, Table 5
	- Adults	SA_A	cm^2/day	6300	Schedule B7, Table 5
Soil-to-Skin Adherence Factor		AF	mg/cm^2/day	0.5	Schedule B7, Table 5
Time Spent Outdoors		ETo	hours	1	Schedule B7, Table 5
Time Spent Indoors		ETi	hours	20	Schedule B7, Table 5
Lung Retention Factor		RF	-	0.375	Schedule B7, Table 5
Particulate Emission Factor		PEFo	(m^3/kg)	7.3E+10	Calculated for scenario, refer to Equations 19 and 20 and assumptions in Schedule B7
Indoor Air Dust Factor		PEFi	(m^3/kg)	2.6E+07	As per Equation 21 based assumptions presented in Schedule B7
Fraction of indoor dust comprised of outdoor soil		TF	-	0.5	Assume 50% soil concentration present in dust as noted in Schedule B7
Indoor Air-to-Soil Gas Attenuation Factor		α	-	0.1	Value adopted as discussed in Section 5.5 of Schedule B7
Body weight	- Young children (0-5 years)	BW_C	kg	15	Schedule B7, Table 5
	- Adults	BW_A	kg	70	Schedule B7, Table 5
Exposure Frequency		EF	days/year	365	Schedule B7, Table 5
Exposure Duration	- Young children (0-5 years)	ED_C	years	6	Schedule B7, Table 5
	- Adults	ED_A	years	29	Schedule B7, Table 5
Averaging Time (non-carcinogenic)		AT_T	days	ED*365	Calculated based on ED for each relevant age group, multiplied by 24 hours for the assessment of inhalation exposures
Averaging Time (carcinogenic)		AT_{NT}	days	25550	Based on lifetime of 70 years, multiplied by 24 hours for the assessment of inhalation exposures

Threshold Calculations - Young Child Aged 2-3 years

Compound	Toxicity Reference Value Oral (TRV_O) (mg/kg/day)	GI Absorption (GAF) (unitless)	Toxicity Reference Value Dermal (TRV_D) (mg/kg/day)	Oral Bioavailability BA_O (%)	Dermal Absorption Factor (DAF) (unitless)	Background Intake Oral/Dermal (BI_O) (% of TDI)	Toxicity Reference Value Inhalation (TRV_I) (mg/m^3)	Background Intake Inhalation (BIi) (% of TC)		Pathway Specific HILs (mg/kg)			Soil Vapour HIL (mg/m^3) (eqn 12)	Derived Interim Soil Gas HIL - Threshold (to 1 or 2 s.f.) (mg/m3)	Derived Soil HIL (not rounded) (mg/kg) (eqn 2 for relevant pathways)	Derived Soil HIL (to 1 or 2 s.f.) (mg/kg)	Notes
										Soil Ingestion (eqn 3)	Dermal (eqn 6)	Dust (eqn 9)					
arsenic	0.002	1	0.002	100%	0.005	50%	0.001	0%		6.0E+02	2.2E+03	1.6E+05			471	**500**	
beryllium	0.002	0.007	0.000014	100%	0.001	30%	0.000020	0%		8.4E+02	1.1E+02	3.3E+03			94	**90**	
boron	0.2			100%		65%	0.7	65%		4.2E+04	NA	4.0E+07			41956	**40000**	
cadmium	0.0008			100%		60%	0.000005	20%		1.9E+02	NA	6.6E+02			149	**150**	
chromium (VI)	0.001			100%		10%	0.0001	0%		5.4E+02	NA	1.6E+04			523	**500**	
cobalt	0.001	1	0.0014	100%	0.001	20%	0.0001	0%		6.7E+02	1.2E+04	1.6E+04			614	**600**	
copper	0.14			100%		60%	0.49	60%		3.4E+04	NA	3.2E+07			33565	**30000**	
manganese	0.16			100%		50%	0.00015	20%		4.8E+04	NA	2.0E+04			13963	**14000**	
methyl mercury	0.00023	1	0.00023	100%	0.001	80%	0.000805	80%		2.8E+01	5.1E+02	2.6E+04			26	**30**	
mercury (inorganic)	0.0006	0.07	0.000042	100%	0.001	40%	0.0002	10%		2.2E+02	2.8E+02	3.0E+04			121	**120**	
nickel	0.012	1	0.012	100%	0.005	60%	0.00002	20%		2.9E+03	1.1E+04	2.6E+03			1217	**1200**	
selenium	0.006			100%		60%	0.021	60%		1.4E+03	NA	1.4E+06			1438	**1400**	
zinc	0.5	1	0.5	100%	0.001	80%	1.75	80%		6.0E+04	1.1E+06	5.7E+07			56870	**60000**	
cyanide (free) (no VI)	0.006	1	0.006	100%	0.1	50%	0.0008	0%		1.8E+03	3.3E+02	1.3E+05			281	**300**	
TCE							0.002	10%		NA	NA	NA	2.2E-02	**0.02**			
1,1,1-TCA							5	0%		NA	NA	NA	6.0E+01	**60**			
PCE							0.2	10%		NA	NA	NA	2.2E+00	**2**			
cis-1,2-dichloroethene							0.007	0%		NA	NA	NA	8.4E-02	**0.08**			
phenol	0.7	1	0.7	100%	0.1	30%	0.035	30%		2.9E+05	5.4E+04	4.0E+06			45419	**45000**	
pentachlorophenol	0.003	1	0.003	100%	0.24	0%	0.0105	0%		1.8E+03	1.4E+02	1.7E+06			129	**130**	
cresols	0.1	1	0.1	100%	0.1	50%	0.35	50%		3.0E+04	5.6E+03	2.9E+07			4687	**4700**	
DDX	0.002	1	0.002	100%	0.018	0%	0.007	0%		1.2E+03	1.2E+03	1.1E+06			608	**600**	
aldrin and dieldrin	0.0001	1	0.0001	100%	0.1	10%	0.00035	10%		5.4E+01	1.0E+01	5.2E+04			8.4	**10**	
chlordane	0.0005	1	0.0005	100%	0.04	0%	0.00175	0%		3.0E+02	1.4E+02	2.9E+05			95	**90**	
endosulfan	0.006	1	0.006	100%	0.1	30%	0.021	30%		2.5E+03	4.7E+02	2.4E+06			394	**400**	
endrin	0.0002	1	0.0002	100%	0.1	0%	0.0007	0%		1.2E+02	2.2E+01	1.1E+05			19	**20**	
heptachlor	0.0001	1	0.0001	100%	0.1	0%	0.00035	0%		6.0E+01	1.1E+01	5.7E+04			9.4	**10**	
HCB	0.00016	1	0.00016	100%	0.1	0%	0.00056	0%		9.6E+01	1.8E+01	9.2E+04			15	**15**	
methoxychlor	0.005	1	0.005	100%	0.1	0%	0.0175	0%		3.0E+03	5.6E+02	2.9E+06			469	**500**	
mirex	0.0002	1	0.0002	100%	0.1	0%	0.0007	0%		1.2E+02	2.2E+01	1.1E+05			19	**20**	
toxaphene	0.00035	1	0.00035	100%	0.1	10%	0.001225	10%		1.9E+02	3.5E+01	1.8E+05			30	**30**	
2,4,5-T	0.01	1	0.01	100%	0.1	0%	0.035	0%		6.0E+03	1.1E+03	5.7E+06			937	**900**	
2,4-D	0.01	1	0.01	100%	0.05	0%	0.035	0%		6.0E+03	2.2E+03	5.7E+06			1621	**1600**	
MCPA	0.01	1	0.01	100%	0.1	0%	0.035	0%		6.0E+03	1.1E+03	5.7E+06			937	**900**	
MCPB	0.01	1	0.01	100%	0.1	0%	0.035	0%		6.0E+03	1.1E+03	5.7E+06			937	**900**	
mecoprop	0.01	1	0.01	100%	0.1	0%	0.035	0%		6.0E+03	1.1E+03	5.7E+06			937	**900**	
picloram	0.07	1	0.07	100%	0.1	0%	0.245	0%		4.2E+04	7.8E+03	4.0E+07			6561	**6600**	
atrazine	0.005	1	0.005	100%	0.1	0%	0.0175	0%		3.0E+03	5.6E+02	2.9E+06			469	**470**	
chlorpyrifos	0.003	1	0.003	100%	0.03	50%	0.0105	50%		9.0E+02	5.6E+02	8.6E+05			343	**340**	
bifenthrin	0.01	1	0.01	100%	0.1	10%	0.035	10%		5.4E+03	1.0E+03	5.2E+06			844	**840**	
PCBs	0.00002	1	0.00002	100%	0.14	0%	0.00007	0%		1.2E+01	1.6E+00	1.1E+04			1.4	**1**	
PBDE Flame Retardants (Br1-Br9)	0.0001	1	0.0001	100%	0.1	80%	0.00035	80%		1.2E+01	2.2E+00	1.1E+04			1.9	**2**	

NA Pathway not of significance for chemical assessed (refer to Appendix A for chemical-specific details)

Non-Threshold Effects - Lifetime Exposures [young child and adult]

Compound	Toxicity Reference Value Oral (TRV_O) $(mg/kg/day)^{-1}$	GI Absorption (GAF) (unitless)	Non-Threshold Slope Factor Dermal (SFd) $(mg/kg/day)^{-1}$	Oral Bioavailability BA_O (%)	Dermal Absorption Factor (DAF) (unitless)		Toxicity Reference Value Inhalation (TRV_I) $(mg/m^3)^{-1}$		Target Risk (TR)	Pathway Specific HILs (mg/kg)			Soil Vapour HIL (mg/m^3) (eqns 13 and 14)	Derived Interim Soil Gas IL - Threshold (to 1 or 2 s.f.) (mg/m3)	Derived Soil HIL (not rounded) (mg/kg) (eqn 2 for relevant pathways)	Derived Soil HIL (to 1 or 2 s.f.) (mg/kg)	Notes
										Soil Ingestion (eqns 4 and 5)	Dermal (eqns 7 and 8)	Dust (eqns 10 and 11)					
TCE							0.004		1E-05	NA	NA	NA	6.0E-02	**0.06**			
vinyl chloride							0.0088		1E-05	NA	NA	NA	2.7E-02	**0.03**			
benzo(a)pyrene	0.5	1	0.5	100%	0.06		1.43E-01		1E-05	9.2E+01	1.3E+01	2.3E+04			11	**10**	1
benzo(a)pyrene (Early-Life)	0.5	1	0.5	100%	0.06		1.43E-01		1E-05	2.3E+01	4.6E+00	8.5E+03			3.8	**4**	1

NA Pathway not of significance for chemical assessed (refer to Appendix A for chemical-specific details)

1 Refer to Appendix A for discussion on different calculations conducted for benzo(a)pyrene and basis for HIL adopted

Appendix C Derivation of Investigation Levels for Generic Land Uses
HIL C - Recreational

Summary of Exposure Parameters		Abbreviation	units	Parameter	References/Notes
Soil and Dust Ingestion Rate	- Young children (0-5 years)	IR_{SC}	mg/day	50	50% of HIL A assumption, Schedule B7, Table 5
	- Adults	IR_{SA}	mg/day	25	50% of HIL A assumption, Schedule B7, Table 5
Surface Area of Skin	- Young children (0-5 years)	SA_C	cm^2/day	2700	As per enHealth (2012)
	- Adults	SA_A	cm^2/day	6300	As per enHealth (2012) for male and female combined
Soil-to-Skin Adherence Factor		AF	mg/cm^2/day	0.5	Schedule B7, Table 5
Time Spent Outdoors		ETo	hours	2	Schedule B7, Table 5
Time Spent Indoors		ETi	hours	0	Schedule B7, Table 5
Lung Retention Factor		RF	-	0.375	Schedule B7, Table 5
Particulate Emission Factor		PEFo	(m^3/kg)	2.6E+07	As per Equation 21 based assumptions presented in Schedule B7
Outdoor Air-to-Soil Gas Attenuation Factor		α	-	0.05	Value adopted as discussed in Section 5.5 of Schedule B7
Body weight	- Young children (0-5 years)	BW_C	kg	15	Schedule B7, Table 5
	- Adults	BW_A	kg	70	Schedule B7, Table 5
Exposure Frequency		EF	days/year	365	Schedule B7, Table 5
Exposure Duration	- Young children (0-5 years)	ED_C	years	6	Schedule B7, Table 5
	- Adults	ED_A	years	29	Schedule B7, Table 5
Averaging Time (non-carcinogenic)		AT_T	days	ED*365	Calculated based on ED for each relevant age group, multiplied by 24 hours for the assessment of inhalation exposures
Averaging Time (carcinogenic)		AT_{NT}	days	25550	Based on lifetime of 70 years, multiplied by 24 hours for the assessment of inhalation exposures

Threshold Calculations - Young Child Aged 2-3 years

Compound	Toxicity Reference Value Oral (TRV_O) (mg/kg/day)	GI Absorption (GAF) (unitless)	Toxicity Reference Value Dermal (TRV_D) (mg/kg/day)	Oral Bioavailability BA_O (%)	Dermal Absorption Factor (DAF) (unitless)	Background Intake Oral/Dermal (BI_O) (% of TDI)	Toxicity Reference Value Inhalation (TRV_I) (mg/m^3)	Background Intake Inhalation (BIi) (% of TC)		Pathway Specific HILs (mg/kg)			Soil Vapour HIL (mg/m^3) (eqn 12)	Derived Interim Soil Gas HIL - Threshold (to 1 or 2 s.f.) (mg/m3)	Derived Soil HIL (not rounded) (mg/kg) (eqn 2 for relevant pathways)	Derived Soil HIL (to 1 or 2 s.f.) (mg/kg)	Notes
										Soil Ingestion (eqn 3)	Dermal (eqn 6)	Dust (eqn 9)					
arsenic	0.002	1	0.002	100%	0.005	50%	0.001	0%		3.0E+02	2.2E+03	8.2E+05			264	**300**	
beryllium	0.002	0.007	0.000014	100%	0.001	30%	0.000020	0%		4.2E+02	1.1E+02	1.6E+04			86	**90**	
boron	0.2			100%		65%	0.7	65%		2.1E+04	NA	2.0E+08			20998	**20000**	
cadmium	0.0008			100%		60%	0.000005	20%		9.6E+01	NA	3.3E+03			93	**90**	
chromium (VI)	0.001			100%		10%	0.0001	0%		2.7E+02	NA	8.2E+04			269	**300**	
cobalt	0.001	1	0.0014	100%	0.001	20%	0.0001	0%		3.4E+02	1.2E+04	8.2E+04			326	**300**	
copper	0.14			100%		60%	0.49	60%		1.7E+04	NA	1.6E+08			16798	**17000**	
manganese	0.16			100%		50%	0.00015	20%		2.4E+04	NA	9.8E+04			19296	**19000**	
methyl mercury	0.00023	1	0.00023	100%	0.001	80%	0.000805	80%		1.4E+01	5.1E+02	1.3E+05			13	**13**	
mercury (inorganic)	0.0006	0.07	0.000042	100%	0.001	40%	0.0002	10%		1.1E+02	2.8E+02	1.5E+05			78	**80**	
nickel	0.012	1	0.012	100%	0.005	60%	0.00002	20%		1.4E+03	1.1E+04	1.3E+04			1157	**1200**	
selenium	0.006			100%		60%	0.021	60%		7.2E+02	NA	6.9E+06			720	**700**	
zinc	0.5	1	0.5	100%	0.001	80%	1.75	80%		3.0E+04	1.1E+06	2.9E+08			29208	**30000**	
cyanide (free) (no VI)	0.006	1	0.006	100%	0.1	50%	0.0008	0%		9.0E+02	3.3E+02	6.6E+05			243	**240**	
TCE							0.002	10%		NA	NA	NA	4.3E-01	**0.4**			
1,1,1-TCA							5	0%		NA	NA	NA	1.2E+03	**1200**			
PCE							0.2	10%		NA	NA	NA	4.3E+01	**40**			
cis-1,2-dichloroethene							0.007	0%		NA	NA	NA	1.7E+00	**2**			
phenol	0.7	1	0.7	100%	0.1	30%	0.035	30%		1.5E+05	5.4E+04	2.0E+07			39651	**40000**	
pentachlorophenol	0.003	1	0.003	100%	0.24	0%	0.0105	0%		9.0E+02	1.4E+02	8.6E+06			120	**120**	
cresols	0.1	1	0.1	100%	0.1	50%	0.35	50%		1.5E+04	5.6E+03	1.4E+08			4054	**4000**	
DDX	0.002	1	0.002	100%	0.018	0%	0.007	0%		6.0E+02	1.2E+03	5.7E+06			404	**400**	
aldrin and dieldrin	0.0001	1	0.0001	100%	0.1	10%	0.00035	10%		2.7E+01	1.0E+01	2.6E+05			7.3	**10**	
chlordane	0.0005	1	0.0005	100%	0.04	0%	0.00175	0%		1.5E+02	1.4E+02	1.4E+06			72	**70**	
endosulfan	0.006	1	0.006	100%	0.1	30%	0.021	30%		1.3E+03	4.7E+02	1.2E+07			341	**340**	
endrin	0.0002	1	0.0002	100%	0.1	0%	0.0007	0%		6.0E+01	2.2E+01	5.7E+05			16	**20**	
heptachlor	0.0001	1	0.0001	100%	0.1	0%	0.00035	0%		3.0E+01	1.1E+01	2.9E+05			8.1	**10**	
HCB	0.00016	1	0.00016	100%	0.1	0%	0.00056	0%		4.8E+01	1.8E+01	4.6E+05			13	**10**	
methoxychlor	0.005	1	0.005	100%	0.1	0%	0.0175	0%		1.5E+03	5.6E+02	1.4E+07			405	**400**	
mirex	0.0002	1	0.0002	100%	0.1	0%	0.0007	0%		6.0E+01	2.2E+01	5.7E+05			16	**20**	
toxaphene	0.00035	1	0.00035	100%	0.1	10%	0.001225	10%		9.5E+01	3.5E+01	9.0E+05			26	**30**	
2,4,5-T	0.01	1	0.01	100%	0.1	0%	0.035	0%		3.0E+03	1.1E+03	2.9E+07			811	**800**	
2,4-D	0.01	1	0.01	100%	0.05	0%	0.035	0%		3.0E+03	2.2E+03	2.9E+07			1277	**1300**	
MCPA	0.01	1	0.01	100%	0.1	0%	0.035	0%		3.0E+03	1.1E+03	2.9E+07			811	**800**	
MCPB	0.01	1	0.01	100%	0.1	0%	0.035	0%		3.0E+03	1.1E+03	2.9E+07			811	**800**	
mecoprop	0.01	1	0.01	100%	0.1	0%	0.035	0%		3.0E+03	1.1E+03	2.9E+07			811	**800**	
picloram	0.07	1	0.07	100%	0.1	0%	0.245	0%		2.1E+04	7.8E+03	2.0E+08			5676	**5700**	
atrazine	0.005	1	0.005	100%	0.1	0%	0.0175	0%		1.5E+03	5.6E+02	1.4E+07			405	**400**	
chlorpyrifos	0.003	1	0.003	100%	0.03	50%	0.0105	50%		4.5E+02	5.6E+02	4.3E+06			249	**250**	
bifenthrin	0.01	1	0.01	100%	0.1	10%	0.035	10%		2.7E+03	1.0E+03	2.6E+07			730	**730**	
PCBs	0.00002	1	0.00002	100%	0.14	0%	0.00007	0%		6.0E+00	1.6E+00	5.7E+04			1.3	**1**	
PBDE Flame Retardants (Br1-Br9)	0.0001	1	0.0001	100%	0.1	80%	0.00035	80%		6.0E+00	2.2E+00	5.7E+04			1.6	**2**	

NA Pathway not of significance for chemical assessed (refer to Appendix A for chemical-specific details)

Non-Threshold Effects - Lifetime Exposures [young child and adult]																	
Compound	**Toxicity Reference Value Oral (TRV_O)** ($(mg/kg/day)^{-1}$)	**GI Absorption (GAF)** (unitless)	**Non-Threshold Slope Factor Dermal (SFd)** ($(mg/kg/day)^{-1}$)	**Oral Bioavailability BA_O (%)**	**Dermal Absorption Factor (DAF)** (unitless)		**Toxicity Reference Value Inhalation (TRV_I)** ($(mg/m^3)^{-1}$)		**Target Risk (TR)**	**Pathway Specific HILs (mg/kg)**			**Soil Vapour HIL (mg/m^3)** (eqns 13 and 14)	**Derived Interim Soil Gas IL - Threshold (to 1 or 2 s.f.)** (mg/m3)	**Derived Soil HIL (not rounded)** (mg/kg) (eqn 2 for relevant pathways)	**Derived Soil HIL (to 1 or 2 s.f.)** (mg/kg)	**Notes**
										Soil Ingestion (eqns 4 and 5)	**Dermal** (eqns 7 and 8)	**Dust** (eqns 10 and 11)					
TCE							0.004		1E-05	NA	NA	NA	1.2E+00	**1**			
vinyl chloride							0.0088		1E-05	NA	NA	NA	5.5E-01	**0.5**			
benzo(a)pyrene	0.5	1	0.5	100%	0.06		1.43E-01		1E-05	4.6E+01	1.3E+01	1.1E+05			9.9	**10**	1
benzo(a)pyrene (Early-Life)	0.5	1	0.5	100%	0.06		1.43E-01		1E-05	1.1E+01	4.6E+00	4.3E+04			3.3	**3**	1

NA Pathway not of significance for chemical assessed (refer to Appendix A for chemical-specific details)
1 Refer to Appendix A for discussion on different calculations conducted for benzo(a)pyrene and basis for HIL adopted

Appendix C Derivation of Investigation Levels for Generic Land Uses
HIL D - Commercial / Industrial

Summary of Exposure Parameters	Abbreviation	units	Parameter	References / Notes
Soil and Dust Ingestion Rate - Adults	IR_{SA}	mg/day	25	50% of HIL A assumption, Schedule B7, Table 5
Surface Area of Skin - Adults	SA_A	cm^2/day	3800	Based on 19% total skin area of 20000 cm^2 exposed (Schedule B7, Table 5)
Soil-to-Skin Adherence Factor	AF	mg/cm^2/day	0.5	Schedule B7, Table 5
Time Spent Outdoors	ETo	hours	1	Schedule B7, Table 5
Time Spent Indoors	ETi	hours	8	Schedule B7, Table 5
Lung Retention Factor	RF	-	0.375	Schedule B7, Table 5
Particulate Emission Factor	PEFo	(m^3/kg)	3.7E+10	Calculated for scenario, refer to Equations 19 and 20 and assumptions in Schedule B7
Indoor Air Dust Factor	PEFi	(m^3/kg)	2.6E+07	As per Equation 21 based assumptions presented in Schedule B7
Fraction of indoor dust comprised of outdoor soil	TF	-	0.5	Assume 50% soil concentration present in dust as noted in Schedule B7
Indoor Air-to-Soil Gas Attenuation Factor	α	-	0.1	Value adopted as discussed in Section 5.5 of Schedule B7
Body weight - Adults	BW_C	kg	70	Schedule B7, Table 5
Exposure Frequency	EF	days/year	240	Schedule B7, Table 5
Exposure Duration - Adults	ED_C	years	30	Schedule B7, Table 5
Averaging Time (non-carcinogenic)	AT_T	days	ED*365	Calculated based on ED for each relevant age group, multiplied by 24 hours for the assessment of inhalation exposures
Averaging Time (carcinogenic)	AT_{NT}	days	25550	Based on lifetime of 70 years, multiplied by 24 hours for the assessment of inhalation exposures

Threshold Calculations - Adult Worker

Compound	Toxicity Reference Value Oral (TRV_O) (mg/kg/day)	GI Absorption (GAF) (unitless)	Toxicity Reference Value Dermal (TRV_D) (mg/kg/day)	Oral Bioavailability BA_O (%)	Dermal Absorption Factor (DAF) (unitless)	Background Intake Oral/Dermal (BI_O) (% of TDI)	Toxicity Reference Value Inhalation (TRV_I) (mg/m^3)	Background Intake Inhalation (BIi) (% of TC)		Pathway Specific HILs (mg/kg)			Soil Vapour HIL (mg/m^3) (eqn 12)	Derived Interim Soil Gas HIL - Threshold (to 1 or 2 s.f.) (mg/m3)	Derived Soil HIL (not rounded) (mg/kg) (eqn 2 for relevant pathways)	Derived Soil HIL (to 1 or 2 s.f.) (mg/kg)
										Soil Ingestion (eqn 3)	Dermal (eqn 6)	Dust (eqn 9)				
arsenic	0.002	1	0.002	100%	0.005	50%	0.001	0%		4.3E+03	1.1E+04	6.2E+05			3071	**3000**
beryllium	0.002	0.007	0.000014	100%	0.001	30%	0.000020	0%		6.0E+03	5.5E+02	1.2E+04			483	**500**
boron	0.2			100%		65%	0.7	65%		3.0E+05	NA	1.5E+08			297503	**300000**
cadmium	0.0008			100%		60%	0.000005	20%		1.4E+03	NA	2.5E+03			881	**900**
chromium (VI)	0.001			100%		10%	0.0001	0%		3.8E+03	NA	6.2E+04			3611	**3600**
cobalt	0.001	1	0.0014	100%	0.001	20%	0.0001	0%		4.8E+03	6.3E+04	6.2E+04			4138	**4000**
copper	0.14			100%		60%	0.49	60%		2.4E+05	NA	1.2E+08			238002	**240000**
manganese	0.16			100%		50%	0.00015	20%		3.4E+05	NA	7.5E+04			61373	**60000**
methyl mercury	0.00023	1	0.00023	100%	0.001	80%	0.000805	80%		2.0E+02	2.6E+03	1.0E+05			182	**180**
mercury (inorganic)	0.0006	0.07	0.000042	100%	0.001	40%	0.0002	10%		1.5E+03	1.4E+03	1.1E+05			730	**730**
nickel	0.012	1	0.012	100%	0.005	60%	0.00002	20%		2.0E+04	5.4E+04	1.0E+04			5963	**6000**
selenium	0.006			100%		60%	0.021	60%		1.0E+04	NA	5.2E+06			10200	**10000**
zinc	0.5	1	0.5	100%	0.001	80%	1.75	80%		4.3E+05	5.6E+06	2.2E+08			395040	**400000**
cyanide (free) (no VI)	0.006	1	0.006	100%	0.1	50%	0.0008	0%		1.3E+04	1.7E+03	5.0E+05			1481	**1500**
TCE							0.002	10%		NA	NA	NA	8.2E-02	**0.08**		
1,1,1-TCA							5	0%		NA	NA	NA	2.3E+02	**230**		
PCE							0.2	10%		NA	NA	NA	8.2E+00	**8**		
cis-1,2-dichloroethene							0.007	0%		NA	NA	NA	3.2E-01	**0.3**		
phenol	0.7	1	0.7	100%	0.1	30%	0.035	30%		2.1E+06	2.7E+05	1.5E+07			238835	**240000**
pentachlorophenol	0.003	1	0.003	100%	0.24	0%	0.0105	0%		1.3E+04	7.0E+02	6.6E+06			664	**660**
cresols	0.1	1	0.1	100%	0.1	50%	0.35	50%		2.1E+05	2.8E+04	1.1E+08			24752	**25000**
DDX	0.002	1	0.002	100%	0.018	0%	0.007	0%		8.5E+03	6.2E+03	4.4E+06			3594	**3600**
aldrin and dieldrin	0.0001	1	0.0001	100%	0.1	10%	0.00035	10%		3.8E+02	5.0E+01	2.0E+05			44.6	**45**
chlordane	0.0005	1	0.0005	100%	0.04	0%	0.00175	0%		2.1E+03	7.0E+02	1.1E+06			527	**530**
endosulfan	0.006	1	0.006	100%	0.1	30%	0.021	30%		1.8E+04	2.4E+03	9.2E+06			2079	**2000**
endrin	0.0002	1	0.0002	100%	0.1	0%	0.0007	0%		8.5E+02	1.1E+02	4.4E+05			99	**100**
heptachlor	0.0001	1	0.0001	100%	0.1	0%	0.00035	0%		4.3E+02	5.6E+01	2.2E+05			49.5	**50**
HCB	0.00016	1	0.00016	100%	0.1	0%	0.00056	0%		6.8E+02	9.0E+01	3.5E+05			79	**80**
methoxychlor	0.005	1	0.005	100%	0.1	0%	0.0175	0%		2.1E+04	2.8E+03	1.1E+07			2475	**2500**
mirex	0.0002	1	0.0002	100%	0.1	0%	0.0007	0%		8.5E+02	1.1E+02	4.4E+05			99	**100**
toxaphene	0.00035	1	0.00035	100%	0.1	10%	0.001225	10%		1.3E+03	1.8E+02	6.9E+05			156	**160**
2,4,5-T	0.01	1	0.01	100%	0.1	0%	0.035	0%		4.3E+04	5.6E+03	2.2E+07			4950	**5000**
2,4-D	0.01	1	0.01	100%	0.05	0%	0.035	0%		4.3E+04	1.1E+04	2.2E+07			8868	**9000**
MCPA	0.01	1	0.01	100%	0.1	0%	0.035	0%		4.3E+04	5.6E+03	2.2E+07			4950	**5000**
MCPB	0.01	1	0.01	100%	0.1	0%	0.035	0%		4.3E+04	5.6E+03	2.2E+07			4950	**5000**
mecoprop	0.01	1	0.01	100%	0.1	0%	0.035	0%		4.3E+04	5.6E+03	2.2E+07			4950	**5000**
picloram	0.07	1	0.07	100%	0.1	0%	0.245	0%		3.0E+05	3.9E+04	1.5E+08			34653	**35000**
atrazine	0.005	1	0.005	100%	0.1	0%	0.0175	0%		2.1E+04	2.8E+03	1.1E+07			2475	**2500**
chlorpyrifos	0.003	1	0.003	100%	0.03	50%	0.0105	50%		6.4E+03	2.8E+03	3.3E+06			1946	**2000**
bifenthrin	0.01	1	0.01	100%	0.1	10%	0.035	10%		3.8E+04	5.0E+03	2.0E+07			4455	**4500**
PCBs	0.00002	1	0.00002	100%	0.14	0%	0.00007	0%		8.5E+01	8.0E+00	4.4E+04			7.3	**7**
PBDE Flame Retardants (Br1-Br9)	0.0001	1	0.0001	100%	0.1	80%	0.00035	80%		8.5E+01	1.1E+01	4.4E+04			9.9	**10**

NA Pathway not of significance for chemical assessed (refer to Appendix A for chemical-specific details)

Non-Threshold Effects - Lifetime Exposures [adult]

Compound	Toxicity Reference Value Oral (TRV_O) $(mg/kg/day)^{-1}$	GI Absorption (GAF) (unitless)	Non-Threshold Slope Factor Dermal (SFd) $(mg/kg/day)^{-1}$	Oral Bioavailability BA_O (%)	Dermal Absorption Factor (DAF) (unitless)		Toxicity Reference Value Inhalation (TRV_I) $(mg/m^3)^{-1}$		Target Risk (TR)	Pathway Specific HILs (mg/kg)			Soil Vapour HIL (mg/m^3) (eqns 13 and 14)	Derived Interim Soil Gas IL - Threshold (to 1 or 2 s.f.) (mg/m3)	Derived Soil HIL (not rounded) (mg/kg) (eqn 2 for relevant pathways)	Derived Soil HIL (to 1 or 2 s.f.) (mg/kg)
										Soil Ingestion (eqns 4 and 5)	Dermal (eqns 7 and 8)	Dust (eqns 10 and 11)				
TCE							0.004		1E-05	NA	NA	NA	2.7E-01	**0.3**		
vinyl chloride							0.00880		1E-05	NA	NA	NA	1.2E-01	**0.1**		
benzo(a)pyrene	0.5	1	0.5	100%	0.06		1.43E-01		1E-05	2.0E+02	4.4E+01	1.0E+05			35.7	**40**

NA Pathway not of significance for chemical assessed (refer to Appendix A for chemical-specific details)

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

Compilation start date:	16 May 2013
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This compilation has been split into 22 volumes

Volume 1: sections 1–6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of

Schedule B7

APPENDIX D

Blood lead model assumptions

IEUBK Modelling Input Parameters - Child Receptors

Parameter	Unit	Child Resident (0-1)	Child Resident (1-2)	Child Resident (2-3)	Child Resident (3-4)	Child Resident (4-5)	Child Resident (5-6)	Child Resident (6-7)	Source
Background Exposure Parameters									
Air									
Ratio of indoor dust lead concentration to corresponding outdoor concentration	%	30	30	30	30	30	30	30	IEUBK default value (US EPA 1989a)
Outdoor air dust lead concentration (constant value)	ug/m^3	0.1	0.1	0.1	0.1	0.1	0.1	0.1	IEUBK default value (US EPA 1989a)
Daily time spent outdoors on-site (HIL A)	hr/day	1	2	3	4	4	4	4	IEUBK default value (US EPA 1989a) also consistent with data from Brinkman et al. (1999)
Daily time spent outdoors on-site (HIL B)	hr/day	1	1	1	1	1	1	1	HIL B Exposure Scenario
Daily time spent outdoors on-site (HIL C)	hr/day	1	2	2	2	2	2	2	HIL C Exposure Scenario, also considering data from Brinkman et al. (1999) for infants.
Lung absorption	%	32	32	32	32	32	32	32	IEUBK default value (US EPA 1989a)
Ventilation rate (HIL A, HIL B)	m^3/day	5.7	8.77	9.76	10.64	11.4	12.07	12.25	Mean inhalation rates as per US EPA (2008), as per Table 6-16.
Ventilation rate (HIL C)	m^3/day	18.7	18.7	18.7	23	23	23	23	Mean inhalation rates as per US EPA (2008) for short-duration exposures, moderate activity.
Diet									
Lead dietary intake (HIL A, HIL B, HIL C)	ug/day	5.1	5.8	6.7	3.2	3.6	4.1	4.7	Food Standards (2003), *The 20th Australian Total Diet Survey* , with conversion to ug/day using mean body weights from US EPA (2008).
Bioavailability of lead in food	unitless	0.5	0.5	0.5	0.5	0.5	0.5	0.5	IARC (2006)
Soil/Dust									
Outdoor soil lead concentration	ug/g	100	100	100	100	100	100	100	Arbitrary value
Indoor dust lead concentration (multiple source analysis) (HIL A & HIL B)	ug/g	70	70	70	70	70	70	70	Calculated by the IEUBK model using multiple source analysis to calculate lead concentration of indoor dust using a 70% contribution of soil to indoor dust.
Indoor dust lead concentration (multiple source analysis) (HIL C)	ug/g	0	0	0	0	0	0	0	HIL C Exposure Scenario has no building
Contribution of soil lead to indoor building dust lead	%	70	70	70	70	70	70	70	IEUBK default value (US EPA 1994)
Percent of total soil and dust ingestion that is soil	%	50	50	50	50	50	50	50	enHealth (2004)
Bioavailability of lead in soil/dust	%	50	50	50	50	50	50	50	IARC (2006)
Ingestion rate of soil and dust (HIL A)	g/day	0.032	0.1	0.1	0.1	0.1	0.1	0.1	As per exposure factors adopted for HIL C, NEPM B7
Ingestion rate of dust (HIL B)	g/day	0.008	0.025	0.025	0.025	0.025	0.025	0.025	As per exposure factors adopted for HIL C, NEPM B7
Ingestion rate of soil and dust (HIL C)	g/day	0.016	0.05	0.05	0.05	0.05	0.05	0.05	As per exposure factors adopted for HIL C, NEPM B7
Other									
Fraction passive/total accessible	unitless	0.2	0.2	0.2	0.2	0.2	0.2	0.2	IEUBK default value
Half saturation level	ug/day	100	100	100	100	100	100	100	IEUBK default value
Drinking Water									
Lead concentration in drinking water	ug/L	0.7	0.7	0.7	0.7	0.7	0.7	0.7	Average concentrations in SA drinking water, considered representative.
Bioavailability of lead in water	unitless	0.5	0.5	0.5	0.5	0.5	0.5	0.5	IARC (2006)
Water consumption	L/day	0.49	0.308	0.356	0.417	0.417	0.417	0.48	US EPA (2008) mean values
Background Lead Allocation									
Maternal blood lead concentration	ug/dL	1	1	1	1	1	1	1	IEUBK default value

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National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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This compilation has been split into 22 volumes

Volume 1: sections 1–6, Schedules A and B
Volume 2: Schedule B1
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Volume 4: Schedule B3
Volume 5: Schedule B4
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Volume 9: Schedule B6

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Volume 10: Schedule B7 - Appendix 1
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About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

Schedule B8

GUIDELINE ON

Community Engagement and Risk Communication

Explanatory note

The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

The original Schedule B8 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document.

The National Environment Protection Council (NEPC) acknowledges the contribution of the Commonwealth Department of Health and Ageing, enHealth, South Australian Department of Health and the Western Australian Department of Environment and Conservation to the development of this Schedule.

Contents
Community engagement and risk communication

1 Purpose and application

This Schedule provides a systematic approach to effective community engagement and risk communication in relation to the assessment of site contamination. It is not intended to be prescriptive but is intended to be used as a tool for effective engagement by consultants and regulators and should also provide a useful reference for all stakeholders including industry, government, landholders and the wider community. It should be noted that, in addition to this Schedule, each state or territory has its own regulatory requirements regarding notification of contamination/pollution to the appropriate regulatory agency.

There are three principles to the approach taken in the preparation of this Schedule:

- that an evaluation regarding the probable need, nature and extent of community engagement for a project should be carried out by site managers with expertise in risk communication at an early stage in the preliminary assessment of site contamination, and should detailed investigations identify contamination that has the potential (or the perceived potential) to have an impact on any stakeholder
- that interaction with the community cannot simply be a technical process; it requires skills in listening and communicating and should be a two-way process
- that for sites with contentious issues, engagement with the community is considered to be essential. This is particularly the case when the contamination at the site has the potential (or the perceived potential) to have an impact on any stakeholder and where impacts are known to extend outside the boundaries of the site.

As an indication, engagement with the community is likely to be particularly beneficial in the following situations:

amenity/nuisance – when the assessment or decisions on and implementation of remediation strategies informed by the assessment of the site may affect the amenity of the locality, for example, by way of temporary noise, odour, emissions or dust

significant contamination – where a high level of contamination has the potential to affect the adjacent community, or where the contaminant types are controversial

site proximity – where the site is near to residential areas or particularly sensitive receptors and/or vulnerable sub-populations, such as childcare centres, schools or nursing homes, and sensitive ecological receptors

controversial sites – where the site or locality has a controversial history that may be related to the site contamination, or the development of the site is controversial for political, economic or social reasons, or where the characteristics or toxicity of the contamination may be controversial, or where contamination has moved outside the site boundaries, or a remediation method may be proposed that is perceived as controversial or that is likely to affect the amenity of the locality or give rise to nuisance conditions.

2 Benefits of community engagement and risk communication

When managed well, community engagement and risk communication can benefit the assessment and management of site contamination by helping site managers to:

- understand public perceptions and concerns, and more accurately anticipate community response to actions and decisions
- increase the effectiveness of risk management decisions and empower the community by involving them
- improve communication and trust and reduce unwarranted tension between the wider community and decision-makers
- explain risk more effectively, to ensure that the community gains a more accurate understanding of the risks.

Simply distributing information without regard for the complexities and uncertainties of the issues does not ensure effective engagement and risk communication. A well-developed community engagement plan will help ensure that messages and actions are constructively formulated, communicated and received.

Two-way engagement, which effectively conveys information and enables community participation in the decision-making process, can provide significant cost savings and improve credibility for organisations involved in site assessment. The community also benefits by contributing to: improved risk assessment inputs, increased ownership of negotiated decision processes, and more acceptable site-management options.

3 Key principles of community engagement and risk communication

The United States Environmental Protection Agency (US EPA) has identified seven overarching principles which should guide risk communication as part of community engagement (US EPA 2007). Corvello et al. (1989) have adapted these seven principles, as follows:

Accept and involve the community as a legitimate partner

- Involve the community early.
- Involve all groups that have an interest in or are potentially affected by the issue.
- Focus on informing the public to enable their participation.
- Never underestimate the level of technical knowledge of community members.
- Invite the public to become involved in the design and evaluation of the public engagement process.

Plan carefully

- Clearly define the objectives of the communication strategy.
- Identify and address the particular concerns of specific groups and stakeholders.
- Educate staff in risk communication.
- Develop a timeline that allows sufficient time for the engagement process,
- Include allowance for new developments or changes — be flexible and responsive.

Listen to the community's specific concerns

- Do not make assumptions about what people know, think or feel — take time to find out.
- Allow all interested parties the opportunity to be heard.
- Be empathetic; put yourself in the place of the community and try to understand their concerns.
- Trust, credibility, competence, fairness and empathy can be of as equal or greater importance to the community as facts and figures.
- Develop a community engagement plan that has the involvement and support of the community.

Be honest, frank and open at all times

- Do not expect to be trusted, and remember that once trust is lost, it is very difficult to regain.
- Acknowledge when you do not have all the answers, and commit to getting back to people with the answers in a given timeframe.
- Disclose information, including 'bad news', as soon as it comes to hand.
- Do not exaggerate or minimise the level of risk; be honest.
- Share more, not less, information.

Coordinate and collaborate with other credible sources

- Build bridges with other organisations and groups that can provide reliable, credible information and advice.
- Try to issue communications jointly with other credible sources – conflict and disagreement between organisations makes communication difficult and results in loss of credibility.

Meet the needs of the community

- Consider opportunities to assist the community in participating in the engagement process, e.g. by providing assistance with travel to meetings, access to office facilities, free methods to respond to published material (e.g. free phone numbers, return envelopes), information in other languages if appropriate.
- Be aware of and sensitive to different cultural behaviours and preferred methods of communication.
- Ensure that information is readable, credible and publicly accessible, and written in a style and format (including site maps and diagrams) that encourages the community to comment about general and specific issues, especially where technical detail is involved.

Meet the needs of the media

- Be accessible to the media, be open with information and respect deadlines.
- Provide information tailored to the needs of each type of media.
- Prepare in advance and provide background information to issues.
- Provide feedback (praise or criticism) to the media when appropriate.
- Where possible, establish a good working relationship with media personnel.
- Nominate one person within the organisation to liaise with the media and provide the main point of contact; this helps to avoid conflicting or confused messages.
- Remember that the media will want to report danger rather than safety, simplicity rather than complexity, and politics rather than risk.

Speak clearly and with compassion, kindness and respect

- Always use clear, plain language.
- Simplify language, not content.
- Acknowledge and respond to emotions expressed by the community including anger, fear, outrage and helplessness.
- Do not be patronising or condescending; show respect for the community's intelligence.
- Respectfully restate a person's questions or statements in your own words to make sure you understand their question before answering it.
- Discuss what you can do and what you will do.
- It is essential to do what you promise.
- Remember to tell people what you can't do, and why.
- People can understand risk information, but they may not agree with you; some people will not be satisfied.

Evaluate effectiveness

- Monitor and evaluate the effectiveness of the risk communication and community engagement program during and at the end of each stage of the process.
- Record accurately and comprehensively the nature and detail of community contributions and responses made throughout the engagement program.
- Establish feedback processes and monitor and review the effectiveness of the engagement.
- Learn from your mistakes.

4 A step-by-step guide to community engagement and risk communication

4.1 Planning and preparation

A community engagement and risk communication plan is an integral part of the wider goal of successful assessment and management of site contamination. Effective communication relies on a commitment to planning, focusing the response to address community concerns and ongoing evaluation with the aim of continuous improvement. Engagement and communication goals should be quite specific, must be well understood by the consultant and should be communicated to the wider community at the beginning of, and during, any engagement plan.

A good plan should help you to:

- integrate the engagement and communication efforts with the risk assessment and management process
- increase the effectiveness of the engagement and communication
- allocate appropriate resources to engagement and communication efforts
- increase dialogue and mutual understanding, and reduce unwarranted tension with the wider community.

Engagement should start as early as possible and continue throughout the site assessment. The community should be informed of possible risks as soon as an issue is identified that may pose a risk to health or the environment or raise public concern. This can mean starting the engagement process before all the information is known and before all options for managing the risk have been identified and considered.

The early initiation of the engagement process is often difficult for those responsible for the site, as they may be unused and unwilling to publicise possible risks associated with the site until they are sure what those risks may be and how they will be managed. However, by consulting early, the community is allowed to actively participate in the decision-making process and members will feel that they have some control over and involvement in the risk assessment and management process. When the community participates in a risk management decision, it is more likely to accept it.

For more complex or contentious sites, a better outcome is often achieved if the engagement and communication role is undertaken by a third party such as a consultant or professional facilitator. This can help to ensure a more open exchange of information and reduce tension if the community is already mistrustful of those responsible for the site assessment.

Open and honest information exchange between organisations (including government agencies) and the community is vital in the management of site contamination. Community members have a right to information about environmental factors that affect their lives and they can contribute valuable local knowledge to the decision-making processes. However, when engaging with the community, there are some legislative issues to consider that may limit or modify the information provided.

For example:

- Commercial-in-confidence materials should not be disclosed.
- Privacy legislation restrains the giving out of personal information to any other person without the permission of the person named.
- Freedom of information (FOI) legislation means that written material can be requested and viewed by any citizen with an interest in it. FOI covers all forms of 'writing', including emails and sticky notes.

- Coroners' courts will investigate incidents where there has been a fire or a death. The court will review information that has been provided.

In planning communication, the first contact should be with the assessor's organisational communication or liaison officer. Planning should also involve government agencies and emergency services (if necessary) to ensure that procedures are understood and that everyone involved agrees on roles and procedures.

A communication plan starts by answering the following questions:

- **Why** do you need to communicate? (purpose of communication)
- **Who** do you need to communicate with? (target audience/s)
- **What** is your message? (what you need to say or what information you need to gather)
- **How** will you communicate? (communication methods and tools)
- **How** will you use the information you gather? (evaluate and review).

4.1.1 Identify the purpose of communication

It is essential to have a clear understanding of the purpose of communicating. Is it:

- to simply inform (the decision has already been made)?
- to consult with the community (obtain their input for consideration)?
- to involve the community in the final decision-making process?

In order to manage expectations, the purpose of the communication activity should be made clear to the community, including the elements that have already been decided upon and are non-negotiable, and what aspects are open for discussion and decision.

4.1.2 Identify your target audience and undertake audience analysis

Once the purpose of the communication has been identified, it is important to identify and analyse the target audience including for cultural and religious sensitivities. If communication efforts are aimed too broadly, the message may not reach key persons. The more tailored messages are to specific audiences, the more effective they will be. Audience analysis will also provide an insight as to what communication methods and tools will best reach each target audience. The communication plan should identify all of the stakeholders—including those beyond the affected community. This includes local and state officials and politicians, other agencies and organisations and, if relevant, emergency and health services.

Establish the project's area of impact. Determining how far interest in the project extends, and determining the location of geographic boundaries and communities of interest will help identify who should be engaged in the engagement process.

Contact key community leaders. Crises tend to push forward local community leaders and groups who become active in voicing community concerns. Identify those people and groups and involve them early on in communication and decision-making activities. Also include council staff and local politicians to brief them about the impending project if appropriate. The longer a delay in involving community representatives and groups, the harder it can be to gain their support and trust. It might also be useful to obtain expert advice about the local community and any outstanding issues that may have an impact on the plan.

It is vital to consider community languages when planning communication activities. Where required, provide printed information in languages other than English. Translators may also be required for verbal communication activities.

There are also a number of protocols for effectively engaging Aboriginal and Torres Strait Islander people. These should be considered prior to initiating communication activities. It is, for example, essential to have an appreciation of cultural difference, to use accurate and non-offensive language, and to show respect when communicating with Aboriginal and Torres Strait Islander people and organisations. Most jurisdictions have guidelines or principles for building good communication skills and channels with Aboriginal and Torres Strait Islander people, communities and organisations. For further information, contact the relevant state or territory health and indigenous affairs departments.

In planning particular sessions or modes of communication, it is important to consider matters of wheelchair accessibility and the possible need for services for people with vision or hearing impairment.

4.1.2.1 Audience analysis

There are a number of resources and sources of information available which are useful in audience analysis. The Australian Bureau of Statistics (ABS) website (www.abs.gov.au) has tools that enable the extracting and viewing of census data for specific geographical areas. This data can be used to build a demographic profile of the local community, including information about male-to-female ratios, number of children and elderly people, socio-economic status, level of educational attainment, minority groups and languages spoken at home. These factors should be carefully considered when planning any communication activity, and may also influence the audience's perception of risk.

Other sources of information that may be helpful in building a profile of a community include:

- internet research — many communities and community or interest groups have websites, usually written in the language and style preferred in the area
- local newspapers — articles and letters to the editor in local newspapers and/or magazines may give you an indication of what issues are of most concern to the community and which groups are most vocal
- local political groups
- local media advertising profiles — local newspapers, magazines and television and radio stations may be willing to share this information (they may charge a fee to do this)
- environmental impact statements (EISs) — many EISs contain information about the local communities and economy, and can often be viewed online.

4.1.3 Identify stakeholders

The area of relevance to assessment of site contamination typically contains a variety of stakeholders, all of whom should be taken into consideration when planning communication activities. A general outline of the various stakeholders that may typically be involved in risk communication and engagement in relation to site contamination and assessment is discussed below. However, it should be remembered that even within these groups there may be a diverse range of perspectives, expectations and concerns, and each group may also be comprised of people of different cultural and socio-economic backgrounds.

Stakeholders include:

- industry — industry's aim is to improve community confidence in its operations. Some companies are successful in achieving this and are good environmental citizens, adopting an 'open door' approach to the scrutiny of their operations, such as holding open days and inviting complainants to visit the site to attempt to pinpoint particular problems. Conversely, some companies may view the community as 'the enemy' and will avoid interaction with the community at all costs, commonly holding the view that, as their activities have not impacted on the community, they have no need to consult. It should also be noted that companies can be constrained by commercial confidentiality in terms of undertaking engagement and risk communication, or may not be able to fund or meet all the expectations of the community. In general, industry is moving towards a more open stance in regard to communicating with the wider community and it is likely that this trend will continue.
- government agencies and departments — the actions of government agencies and departments are dictated primarily by their statutory responsibilities, with different agencies having different roles and functions. For example, some will have responsibility for overall management of an assessment and remediation program, while others will have responsibility for a specific aspect of assessment such as public health or occupational health and safety. However, most are also involved in balancing a range of expectations from the wider community.
- local government — conscious of the increasing environmental awareness of communities, local government has been instrumental in responding to the need for more community participation, greater accountability and better communication between all stakeholders. Both local and state government organisations are coming under increased pressure from reduced budgets and may find it difficult to fully resource the range of expertise and involvement required to manage a wide range of site-assessment responsibilities.
- residents — no residential community of any size is a homogeneous entity. It is not possible to generalise about the role or attitude of the residential community. For example, not all the residents will be involved, even though they may be concerned, or want to be involved in community engagement; others will have an intense interest and some residents who are not involved initially may change their minds later. Moreover, some act and think autonomously, while others represent the views of an organisation or group. For this reason, audience analysis is an important aspect of planning engagement and communication activities
- non-government organisations — non-government organisations include environment groups, special interest groups, and committees and associations that comprise various representatives from industry, council, non-government agencies and departments, and residents. To those managing the site contamination assessment, the 'activists' (who may either support or oppose the situation) within the non-government organisations are often seen as a threat because of the scientific skills couched within the agenda of a pseudo-political organisation. However, to local residents, the advice and assistance from such organisations can be instrumental in understanding the issues and learning how to frame their concerns.
- employees, unions and associations — employees, unions and associations are generally concerned that, in undertaking a site assessment or site remediation, adequate health protection measures are in place. Accordingly, health risks associated with site contamination should be communicated to employees and all other persons working on the site. Briefing on risk management and safety precautions is essential and should form part of the engagement plan.

- media — media coverage can focus either on the negative or positive aspects of the issues involved, which can then determine whether the community feels threatened and defensive or confident and cooperative. Accordingly, it is important to ensure that the material available to the media is framed in a rational, consistent and non-inflammatory manner. A good working relationship with media personnel can provide the opportunity for information dissemination outlets to the community. For consultants who deal with the media, it is sensible to nominate one person within the organisation to liaise with the media and act as the main point of contact (this helps to avoid conflicting or confused messages being disseminated).

4.1.4 Risk perception

The term 'risk perception' generally refers to the perceptions of that part of the community that is outside the regulatory, scientific research and risk assessment spheres. In engaging with the community, it is important to remember that perception of risk can be influenced by numerous factors beyond just the scientific data. It is for this reason that what may scientifically constitute a 'negligible risk' can still give rise to anger and resentment within the community. People see risk as multidimensional and not as being represented by a numerical value alone, judging risk according to its characteristics and context. For example, trauma and death as the result of an involuntary catastrophic reaction is likely to be dreaded more than as the result of a situation where the risk is assumed voluntarily and the person feels some degree of control over it (for example, motor vehicle crashes).

A study by the Centre for Population Studies in Epidemiology, (**Starr & Taylor** 2000), investigated health risk perception in a national sample population. Major findings indicated that risk perception is largely influenced by age, gender and education, and that certain kinds of risks tend to arouse heightened levels of concern.

Concerns about risk tend to be heightened where risks are:

- involuntary or imposed on the community
- man-made rather than natural
- inescapable
- controlled by parties outside the community
- likely to have little or no benefit to the community
- subject to media attention
- unfairly distributed
- related to a distrusted source
- exotic or unfamiliar
- likely to affect children or pregnant women
- likely to affect identifiable rather than anonymous people
- the cause of insidious and irreversible damage
- the cause of dreaded health effects such as cancer
- poorly understood by science
- subject to contradictory statements from responsible sources (or, even worse, from the same source)
- related to situations where the risk makers are not the risk takers.

While medical doctors were viewed with greater trust, nearly 40% of study participants identified the media (including newspapers, magazines, television and radio) as their primary source of information.

4.1.5 Develop the message

It is often helpful to develop key messages as part of the risk communication planning process. This can help to focus communication activities on the most important information and, by helping to ensure that messages are consistent, can also assist in building trust with communities.

It is important to remember that message development is not 'spin' and is not manipulative, and nor is it a substitute for two-way communication. The key to good message development is to avoid bombarding the audience with too much information or with information that does not address their needs. This can be achieved by understanding community concerns and focusing messages on answering those concerns in a clear and concise manner.

In developing key messages, it is helpful to collate maps, diagrams and reports relevant to the project and identify data which may be useful in providing information, explaining decisions, and so on.

The most important part of message development is focusing on what information the community wants. In general, people are interested in receiving information on the following subjects:

- description of the risk – people want more than just technical descriptions of risk. Risk should also be conveyed in ways that are accessible and relatable for people with non-technical backgrounds. It may be helpful for risk communicators to provide familiar analogies that assist an understanding of the risk.
- risk consequences – this includes effects and the level of danger associated with the risk.
- level of control about the risk and its consequences – people want to know the answers to questions such as "what should I do?" and "what are agencies doing?"
- exposure information – this includes risk intensity, duration, acceptable risk levels and how they are measured, how long the exposing agent is dangerous, how long it persists, and how it accumulates in the body.

As part of an engagement process, the following kinds of questions may be asked, relating to numerous types of concerns.

Note: these are generalisations and these questions are NOT provided as a substitute for identifying the community concerns through two-way communication.

Health and lifestyle concerns

- What is the danger to my health and that of my family?
- Can I drink the water, eat vegetables from my garden, etc.?
- What can I do to find out if my health has already been affected?
- What can I do to reduce the damage already done?
- What can I do to prevent further damage?
- What about my children?
- We are already at risk because of X. Will Y increase our risk?
- How will this affect our quality of life/property values?
- How will we be affected by the stigma of X being attached to our community?
- How will we be protected in an accident?

- How will we be compensated for the loss of value of our homes?

Data and information concerns

- How sure are you?
- What is the worst-case scenario?
- What do these numbers mean and how did you get them?
- How do we know your studies are correct?
- What about other opinions on this issue?
- How do our exposures compare to the standards?
- You say X can't happen, why not?

Process concerns

- How will we be involved in the decision-making?
- How will you communicate with us?
- Why should we trust you?
- How and when can we reach you?
- Who else are you talking to?
- When will we hear from you?

Risk management concerns

- When will the problem be corrected?
- Why did you let this happen and what are you going to do about it?
- What are the other opinions? Why do you favour option X?
- Why are you moving so slowly to correct the problem?
- What other agencies are involved and in what roles?
- What kind of oversight will we have?

In formulating key messages, it is often useful to convey information in more than one way, for example, to use visual representations of information in addition to just words. If you need to communicate numerical risk information it is also useful to consider the following techniques:

- highlight the most important information
- pre-test symbols and graphics
- align data with general thinking (e.g. in a choice of one to five, the highest number would be the best)
- if you state probabilities as '1 chance in X', keep 'X' as a constant
- give visual clues as to the importance of information (e.g. use larger fonts or bold items).

Consider expressing risks in terms of absolute risk (1 in 10) rather than relative risk (10%), and do not use decimals.

4.1.6 Determine requirements for engagement

Following audience analysis and identification of stakeholders, requirements should be determined for engagement and stakeholder involvement including:

- what stage(s) of the project will require engagement
- the role the community and its representatives will have in the engagement process
- appropriate notices about the project and the engagement process (include media and public involvement techniques and existing communication avenues such as council newsletters and local newspapers).

4.1.7 Incorporate an evaluation process

Plan to involve all parties in evaluation and feedback on the effectiveness of the engagement and communication throughout implementation of the community engagement plan, as well as after the conclusion of the process. This will allow for midcourse improvements to be made, where necessary. The effectiveness of a community engagement plan can be measured by evaluating the implementation of engagement techniques and actions, the quality and quantity of stakeholder interactions, and by reviewing stakeholder relationships.

4.1.8 Develop an engagement and communication protocol

This kind of public document should include the following information:

- a brief, clear statement of the issues and background information
- a clear statement of issues that are not negotiable within the engagement
- a broad description of who is affected
- a statement of what kind of information is being sought and how it will, or won't, be used
- a timeline for the engagement program that allows sufficient time for stakeholders to discuss and form opinions on the issues
- a list of engagement techniques to be used
- identity of author, accessible point of contact, phone number, email address and website link (if available)
- a list of staff and funding resources available for engagement.

4.1.9 Reporting on community engagement

Following the implementation of a community engagement plan, reporting and subsequent feedback to the community should be undertaken, which should address the following:

- the extent of community engagement undertaken should be documented and justified
- details of the engagement process including names of potential stakeholders (individuals and groups) who were identified and invited to participate, method or techniques of engagement used, names of community members who participated, details of how, when and where engagement was carried out
- information provided to the community
- input and comments received from the community
- how the community's input was considered and incorporated in the decision-making process
- availability of all documentation to the community.

4.2 Key messages for contaminated land practitioners

The ten key take-home messages (adapted from Heath et al. 2010) for contaminated land practitioners in regards to community engagement are:

4.2.1 Community perceptions

Risk, in the context of contaminated land, is an inherently predictive, multidimensional estimate that is useful in trying to prevent future harm from happening. Because predictions of risk inevitably rely on a mixture of evidence, assumptions and judgment, characterising any differing beliefs of the public about risk as being just 'perception' is guaranteed to undermine trust and mutual respect, if not create open conflict and further outrage.

4.2.2 Credibility is based on more than scientific and technical competence

Scientific competence is essential to establish credibility, but is by itself not sufficient to ensure trust. Openness, honesty and transparency are also necessary to demonstrate credibility and warrant trust. This includes a frank and honest approach to dealing with uncertainty, which is inevitable in any risk assessment. Denial of uncertainty (both knowledge uncertainty and uncertainty caused by variability) will eventually backfire and undermine credibility.

4.2.3 Effective communication is necessary but not sufficient

Scientific and technical evidence is often complex and difficult to understand. If an audience is presented with confusing information they can at best ignore it or at worst be angered by it. However, regardless of how carefully or compassionately it is presented, scientific or technical evidence is unlikely to have a constructive impact if the public is outraged.

4.2.4 Avoiding community engagement will guarantee trouble

There is no all-purpose, sure way to avoid problems simply by engaging communities. However, it is equally certain that failing to engage a community about an issue of concern will create problems that could be reduced, if not avoided, by effective community engagement.

4.2.5 Do not promise more than you can deliver

Overly zealous claims (even if they are sincere) about what or how quickly something can be achieved will, when not achieved, cause disappointment that may boil over into distrust. It is better to be realistic from the outset. With the public engaged from the beginning, they can make the journey through a project with some sense of ownership and reality that can lead to tolerance of missed targets.

4.2.6 An unfair process will generate outrage

People who believe they are being treated unfairly, in a condescending manner, or being ignored altogether, will become aggrieved, possibly to the point of active opposition. It is extremely difficult to engage an outraged public in a constructive manner.

4.2.7 Effective communication must be a two-way process

One-way communication is simply preaching or selling. Any risk communication process that lacks an effective means to listen to community concerns, a commitment to seriously seek to understand those concerns and respond to them will be dismissed by the community as merely public relations.

4.2.8 Resolving disputes requires a dedicated process

Because proponent objectives for dealing with contaminated land may not coincide with the objectives of other stakeholders, there is always potential for disputes that are unlikely to be resolved purely by communication. Because litigation is expensive and often ineffective, dedicated alternative dispute resolution methods, such as negotiation or meditation, should be pursued before disputes become unmanageable.

4.2.9 Validate your messages and behaviour

Everyone involved in a project will have associates, whether they are family members, friends or non-technical staff, who can offer perspectives on key issues that will not be based on, or limited to, narrow scientific and technical interpretation. Talk with them to remind yourself of the lay person's view.

4.2.10 Trust and credibility are both essential

Trust and credibility are closely related and interdependent. Credibility (being worthy of confidence) is usually necessary to establish trust, but credibility alone does not guarantee trust. Because we are all busy and we already have more things to think about than we have time for, we inevitably have to rely on the views of others for most of the things that we face in our lives. When we rely on the views of others rather than analysing a problem for ourselves firsthand, we are placing trust in others. In essence, trust often serves as a means for dealing with complexity that we have insufficient time to resolve for ourselves.

5 Community engagement techniques

An effective community engagement plan includes all affected stakeholders and uses techniques that ensure that those who wish to participate in the engagement are able to do so. Achieving effective engagement with stakeholders relies on selecting methods of communication that will reach the target groups.

Determining the extent of engagement depends upon the nature and impact of the contaminants, the proximity of the community, and the particular stage of the assessment process. As a general guide, the more significant the impact of the contamination on the community, the more community participation is expected. It is important to recognise that there is no single stakeholder and that different techniques need to be used to reach different stakeholders. It is also important to recognise that a combination of one or more techniques may need to be used to effectively engage with a particular stakeholder. Moreover, engagement is most likely to be effective if it builds on or creates an ongoing relationship between various stakeholders.

The choice of techniques will depend on a number of factors including:

- the purpose of involving the wider community
- the stage of the process
- the nature of the wider community and their willingness to participate
- the likely impact of the contaminants and the assessment process
- timelines
- the skills and resources that are available.

A description of a range of engagement techniques, and the advantages and disadvantages of each, is provided below.

5.1 Engagement techniques: summary of advantages and disadvantages

Group techniques			
Technique	Description and Guidelines	Advantages	Disadvantages
Public meetings	Usually more than 20 people, self-selection by advertised invitation, formalised proceedings aimed at presenting information to large audience, conducted at a time and location to suit most people, needs to be widely publicised.	Provides a forum for information dissemination and exchange with large numbers, may incorporate other techniques such as workshops, brings a wide range of people together.	Focused discussion on one issue is difficult, more articulate and better prepared members of the community may dominate, less vocal sections of the community may not express their views.
On-site meetings	Open-air community meetings held on-site or adjacent to the affected site to provide information, gauge interest and explain process and procedures.	Enables interested individuals to gain an understanding of the issues involved. Useful for site contamination as standing on the site can remove some aura of the unknown.	Accessibility to site not always possible (for example, for aged or disabled community members, or for safety concerns).

Group techniques			
Technique	Description and Guidelines	Advantages	Disadvantages
Search conference	Usually 20–30 participants selected to be heterogeneous but sharing an interest, staged discussion aimed at identifying broad cross-section of views on a variety of issues, lasting a day, weekend or longer.	Can assist in the early stages of the engagement process to identify community characteristics and relevant issues, program devised with participants, future orientated, allows lengthy discussion to develop and refine ideas.	Large time commitment, may appear to be an elite group, participants may not have necessary information, may tend to result in 'wish list' of unrealistic future requirements.
Design meeting	Community members meet to work on maps, scale representations and photographs to gain better idea of the effect on their community of proposals and options, expert presenters may be required.	Allows community members to better express their views and visualise the impact of changes, enables consultant to understand how a proposal appears to the community.	Numbers of participants limited, limited technique if complete socio-economic and environmental impact to be determined.
Workshops	Participants are usually homogeneous in terms of skills and concerns, structured sessions aimed at encouraging open discussion between participants and producing proposals for solutions.	Provides opportunity for all stakeholders to contribute, a flexible technique that can be used at all stages of the engagement process, can provide a forum for testing alternatives, training opportunities, information gathering and dissemination, receiving feedback and refining input.	If the participants are specifically selected then the nature of this technique can result in it appearing exclusive, the specific workshops may restrict discussion and debate.
Seminars	A meeting where a particular subject is explored in depth for some length of time under expert guidance.	Opportunity for learning and information sharing, detailed discussion and inquiry can take place, all participants can question or contribute.	The 'right' expert may not be available, participants may not be adequately prepared, experts may dominate and inhibit discussion.

Individual techniques			
Technique	**Description and Guidelines**	**Advantages**	**Disadvantages**
Individual discussion	Selected individuals consulted by telephone, meetings and doorknocking an area.	Provides a quick and efficient means of disseminating information and identifying a range of issues and views.	Provides limited opportunities for large numbers of community members to participate in the process, does not allow for broadscale exchange of ideas.

Individual techniques			
Technique	**Description and Guidelines**	**Advantages**	**Disadvantages**
Submission	Oral or written submissions to enable people to register their ideas and concerns, open to the general community and usually undertaken in the early or later stages of engagement.	Political and institutional demonstration of commitment to open engagement, provides focus for groups to organise a basis from which to lobby, provides consultant with some information on viewpoints of key stakeholders.	Limited role as submissions are unlikely to draw response from minority groups in the community, only 'organised' and articulate stakeholders are likely to respond, the formality of hearings may intimidate some.
Survey	Structured questioning of community sample that statistically represents the whole population or sector, used to gather information about objective characteristics or attitudes of a community.	Provides data for analysis of characteristics of a community, and to document probable effects of a proposal and for gauging likely public reaction to a proposal.	Minimal discussion and no interaction between members of the community, respondents may be indifferent to the subject matter and require persuasion.
Open houses	Informal arrangement where tables or booths are manned by knowledgeable government staff or consultants who are able to discuss what individuals in the community want.	Sets up a comfortable discussion situation for staff and members of the public. Especially useful early in the process to establish rapport and explain complex processes.	Attendances may be low if distrust of the consultants and government by the public is already high.
Display and exhibitions	Means of disseminating information to the community, mobile or permanent exhibition, may be staffed for seeking response and giving detailed explanation.	Opportunity to inform and meet with the wider community who can speak directly to the consultants, opportunity to demonstrate commitment to engagement.	May be costly and ineffective, particularly if the community does not perceive the issues as being of high importance.
Observations	Means of gathering information and establishing contacts in a community.	Provides a thorough understanding of the community in preparation for engagement.	This technique is generally only suitable in the early information collection stage of engagement.
Information bulletins and brochures	Regular information bulletins and brochures distributed to households and/or made available to the community at key public outlets.	Provides ongoing information on the project.	Generic flyers may be perceived as junk mail and may be ignored.
Site office	Temporary accommodation for consultants in the area, provides information for the wider community, needs to be suitably located and staffed.	Provides consultants with a convenient base from which to work and establish contact in the area, satisfies some community needs for individual attention to their issues and concerns.	Does not involve interaction between members of the community and may be costly, has limited value in the overall engagement process if used alone.

Individual techniques			
Technique	**Description and Guidelines**	**Advantages**	**Disadvantages**
Open door	Conducting periodic open days to invite interested people and complainants to visit the site.	Can shift community confidence in current and proposed operations, pinpoint particular problems and result in problems being address and resolved.	May not be possible for commercial confidentiality or occupational health, safety and welfare reasons.
Hotline	A telephone service to provide information and to record comments, concerns and suggestions.	Ensures that information is available; provides the opportunity for the wider community with mobility problems.	Would not reach all people from non-English speaking backgrounds unless hotline is available in different languages.
Websites	Information dissemination through an interactive web page, aimed at informing and generating interest.	Keeps the public and other interested parties informed. Can be updated quickly and easily. Allows people to access large amounts of information and provide feedback.	Can only be accessed by those with access to a computer with web connection. Tends not to be available to minority groups such as the elderly, poor, people with non-English speaking backgrounds. Can contribute to information overload if not managed effectively.
Use of media	Information dissemination through printed and electronic media, can be aimed at informing or generating interest and feedback.	Political and institutional advantages of ensuring that information is provided, keeps the community informed, provides opportunity for all of the community to contribute.	Will not reach all groups unless special attention is given to minority groups by the use of ethnic media, and other avenues to reach other target groups.

The above information was sourced and adapted from *The human services planning kit*, (SA Department of Housing and Urban Development 1994).

An extensive list of community engagement methods and techniques can also be found in *Effective engagement: building relationships with community and other stakeholders, Book 3: the engagement toolkit*, published by the Department of Sustainability and Environment Victoria (DSE VIC 2005) (www.dse.vic.gov.au/engage).

5.2 Engagement and communication DOs and DON'Ts

	DO	**DON'T**
Abstractions	DO use examples, anecdotes and analogies to establish a common understanding	DON'T generalise too much or use hypothetical situations
Attacks	DO attack the issue	DON'T attack the person or organisation
Blame	DO take responsibility for your share of the problem	DON'T try to shift blame or responsibility to others
Clarity	DO ask whether you have made yourself clear	DON'T assume you have been understood
Guarantees	DO emphasise ongoing efforts and	DON'T say there are no guarantees

	DO	DON'T
	achievements made and explain any limitations on the guarantee and why they exist	
Humour	DO use humour wisely — if used, direct it at yourself	DON'T use humour in relation to safety, health or environmental issues
Jargon	DO define all technical terms and acronyms (e.g. NATA)	DON'T use language that may not be understood by your audience
Length of presentation	DO limit presentation to 15 mins to allow for longer question & answer periods	DON'T ramble or fail to plan the time well
Money	DO refer to the importance you attach to health, safety and environmental issues; your moral obligation to protect public health and the environment outweighs financial considerations	DON'T refer to the amount of money spent as if it proved your concern
Negative allegations	DO refute allegations	DON'T repeat or refer to them
Negative words and phrases	DO use positive or neutral terms	DON'T minimise or trivialise the risk
Non-verbal messages	DO be sensitive to non-verbal messages you are communicating; make them consistent with what you are saying	DON'T allow your body language, your position in the room, or your dress to be inconsistent with your message
'Off the record'	DO assume everything you say and do is part of the public record	DON'T make side comments or 'confidential' remarks
Organisational identity	DO use personal pronouns (i.e. I, we)	DON'T take on the identity of a large organisation
Promises	DO promise only what you can deliver. Set and follow strict orders	DON'T make promises you can't keep or fail to follow up
Reliance on words	DO use visuals to emphasise key points	DON'T rely entirely on words
Risk comparisons	DO use comparisons, when asked, to help put risks in perspective	DON'T compare unrelated risks
Speculations	DO provide information on what is being done	DON'T speculate about worst-case scenarios
Technical details and debates	DO base your remarks on empathy, competence, honesty and dedication	DON'T provide too much detail or take part in protracted technical debates
Temper	DO remain calm. Use a question or allegation as a springboard to say something positive	DON'T let your feelings interfere with your ability to communicate positively

6 Case studies

Examples where effective community engagement practices were implemented early in the assessment of site contamination are provided below. Further case studies and examples of effective and ineffective engagement practices can be found in Heath et al. 2010.

6.1 Case study 1: Radioactive site in metropolitan area

Background
In 1997, a relatively undeveloped site in a metropolitan area was alleged to contain radioactive contamination. A site history and a radioactive survey were undertaken to assess the level of any immediate risks to public health. Following this, an engagement plan was developed prior to conducting a detailed site investigation.

Community engagement and risk communication plan
The following broad plan was formulated with the assistance of local government officers and elected members:

- a consultation process, initially to inform targeted key members of the wider community prior to the detailed site assessment
- following the site assessment, a wider engagement program with the local community to enable the community to contribute to decisions that could affect them.

Engagement and communication
The initial engagement involved informing and conducting meetings with:

- identified community representatives
- peak trade unions
- elected members of local government
- relevant government authorities and organisations.

Outcomes
The main outcomes of the initial engagement were that:

- key members of the wider community were well informed about the contamination and the engagement process to be undertaken
- these key stakeholders responded well and appeared satisfied that the issue was being managed in a logical and comprehensive manner
- a level of trust and confidence in the consultants was established in the minds of the key stakeholders at the outset, which assisted further engagement with the community during the site assessment and remediation phases.

6.2 Case study 2: Ardeer, Victoria

Background
In 1989, severe lead contamination was confirmed in soil of a residential area in the Melbourne suburb of Ardeer. The site was used previously for secondary lead smelting and lead-acid battery manufacture. Measures were put in place to relocate residents of the severely affected properties and to assess contamination in the surrounding area. Accordingly, 19 properties had their soil remediated and ceiling dust was removed from 65 properties. The site assessment and the clean-up process necessitated engagement and communication with the residents.

Community engagement and risk communication plan
Following the establishment of a broad snapshot of the local Ardeer community, the EPA developed an engagement plan. The engagement process extended over three and a half years, from initial assessment to completion of the remediation. The plan was based upon the following principles:

- identifying the affected community
- being clear about the purpose of conveying information
- accepting the rights of the residents and groups to contribute to decisions that could affect them.

Engagement techniques
The EPA used various engagement techniques including:

- doorknocking residents
- discussions with principals and teachers of education establishments in proximity to the site
- production and dissemination of ongoing multilingual information bulletins to the community in the area and the relevant action group
- intensive contact and personal visits undertaken with those with contaminated properties
- advising residents of sampling results
- periodically issuing media releases.

Outcomes
The main outcomes of the process were that:

- the community was well informed about the contamination and the remediation process
- the local community was able to contribute to decisions that affected them
- overall, the engagement plan was successful as the residents generally appeared satisfied that their safety was not compromised.

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8 Glossary

Community engagement is the process of communicating and deliberating with the community and other stakeholders. It can include a variety of project-specific approaches:

Inform	one-way communication or delivery of information
Consult	providing for ongoing public feedback
Involve	a two-way process to ensure community concerns are considered as part of the decision-making process
Collaborate	developing partnerships with the community to make recommendations
Empower	allowing the community to make decisions and to implement and manage change.

Community means those individuals and/or groups residing in the locality where a site assessment is to be conducted and who may be affected by the assessment and/or possible site contamination physically (for example, through risks to health or the environment, loss of amenity) or non-physically (for example, via concern about possible contamination).

Contamination means the condition of land or water where any chemical substance or waste has been added as a direct result or indirect result of human activity at above background level and represents, or potentially represents, an adverse health or environmental impact.

EPA means the relevant environment protection authority or equivalent agency responsible for the regulation and management of contaminated land.

Exposure occurs when a chemical, physical or biological agent makes contact with the human body through breathing, skin contact or ingestion; for example, contaminants in soil, water and air.

Hazard is the intrinsic capacity of a chemical, biological, physical or social agent to produce a particular type of adverse health or ecological effect.

Community engagement consultant means an appropriately skilled professional employed to develop and implement the community engagement and risk communication plan.

Remediation means the clean-up or mitigation of pollution or of contamination of soil or water by various methods.

Risk assessment means the process of estimating the potential impact of a chemical, physical, microbiological or social hazard on a specified human population or ecosystem under a specific set of conditions within a certain timeframe.

Risk communication means an interactive process involving the exchange among individuals, groups and institutions of information and expert opinion about the

nature, severity and acceptability of risks and the decisions to be taken to combat them. Risk communication is delivered most efficiently in the context of a well-structured community engagement process.

Risk management means the decision-making process to analyse and compare the range of options for site management and select the appropriate response to a potential health or environmental hazard. It may involve considerations of political, social, economic, environmental and engineering factors.

Risk means the probability in a certain timeframe that an adverse outcome will occur in a person, group, or ecological system that is exposed to a particular dose or concentration of a hazardous agent; that is, it depends on both the level of toxicity of hazardous agent and the level of exposure.

Risk perception is the subjective judgment that people make about the characteristics and severity of a risk.

Site managers are those responsible for environmental site assessment, risk assessment and risk management and may include landowners, contaminated land consultants, contractors or environmental auditors.

Site means the parcel of land being assessed for contamination.

Stakeholder means one who has an interest in a project or who may be affected by it.

Sustainable development means development that meets the needs of the present without compromising the ability of future generations to meet their own needs.

Wider community means individuals and/or groups, not necessarily residing in the locality of a site assessment, who may have an interest in the assessment.

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

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This compilation has been split into 22 volumes

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About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of Site Contamination) Measure 1999 National Environment Protection (Assessment of

Schedule B9

GUIDELINE ON

Competencies & Acceptance of Environmental Auditors and Related Professionals

Explanatory note

The following guideline provides general guidance in relation to investigation levels for soil, soil vapour and groundwater in the assessment of site contamination.

This Schedule forms part of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and should be read in conjunction with that document, which includes a policy framework and assessment of site contamination flowchart.

The original Schedule B10 to the National Environment Protection (Assessment of Site Contamination) Measure 1999 has been repealed and replaced by this document.

The National Environment Protection Council (NEPC) acknowledges the contribution of SA Environmental Protection Authority and Queensland Department of Environment and Heritage Protection to the development of this Schedule.

Contents
Competencies and acceptance of environmental auditors and related professionals

1 Introduction

The assessment of contaminated sites is a specialised professional area involving a number of disciplines. Practitioners must have a range of competencies and be able to recognise the need for supporting professional advice beyond their own expertise when assessing contamination and its effects on land use and the environment.

The extent to which these competencies are required varies with the level and nature of work being carried out by the professional. For example, the professional may be operating as an accredited auditor, a third-party reviewer, a specialist professional certifying work under statute or an environmental consultant involved in carrying out contaminated site assessments. The complexity of contamination issues will vary on individual sites from a single known contaminant with limited site distribution to sites with multiple contaminants of unknown vertical and lateral spread, off-site impacts and obvious human health and environmental risks.

Professional assessments of site contamination deal with health and environmental issues of concern to landowners, occupiers, regulators, local government, planning authorities and the public. These assessments are required by regulatory and planning authorities for the management of contaminated land and in development approval processes.

This Schedule should assist the development of arrangements to provide consistency in the recognition of competent professionals for contaminated site assessment across Australia.

2 Purpose

The purpose of this Schedule is to:

- describe the competencies and experience that are essential for professionals involved in contaminated site assessment including auditors, third-party reviewers and professionals who are certifying assessments of complex contaminated sites
- provide a general framework for the appointment or acceptance by regulatory authorities of contaminated land professionals who are required under statute to certify site assessments.

3 Use of these guidelines

This Schedule is primarily intended for use by regulatory authorities within the scope of their environmental and planning legislation. Its application in individual states and territories will assist in establishing a consistent minimum level of knowledge, experience and technical competencies for environmental professionals carrying out contaminated site assessment within Australia, and the mutual recognition of these professionals.

Individual states and territories may have specific legislative requirements relating to the appointment or acceptance of:

- auditors appointed or accredited for the independent third-party auditing of site contamination
- third-party reviewers accepted to conduct independent third-party reviews for the certification of assessment and remediation
- specialised professionals who are required under statute to demonstrate relevant qualifications and experience when presenting contamination assessment reports to regulatory authorities and to certify assessment work under statutory declarations.

To be recognised in these roles individuals must be professionals with significant technical expertise and experience in the assessment of site contamination. The application of a high level of technical competency assessment is to be applied to the appointment of accredited auditors and to third-party reviewers and the acceptance of professionals who are certifying assessments of contaminated sites.

While regulatory authorities in individual states and territories may require specific knowledge and understanding of legislation and guidelines relevant to their jurisdiction, it is intended the broad assessment process and minimum criteria described in this Schedule be used to establish the professional competencies required and to then determine the technical skills, experience and proficiency of these individuals.

Relevant aspects of this Schedule provide advice on appropriate qualifications, experience and competencies of environmental consultants involved in the assessment and/or remediation of contaminated sites. These considerations may also be applied to assess the abilities of environmental professionals and their companies not otherwise subject to specific legislative requirements for appointment or accreditation, in order to assess their capability to carry out specific assessments of site contamination.

To improve the quality of site contamination assessment work and encourage professional specialisation in this area, regulatory authorities may use this Schedule as the basis of advice for stakeholders, including professional associations, on the competence of practitioners.

This Schedule may also assist members of the community in decision-making regarding the employment of environmental professionals for contaminated site assessment work, by informing them of the broad range of competencies, knowledge and experience that should be held by environmental professionals in designing and carrying out contaminated site assessments.

4 Professional roles in the assessment of site contamination

Professionals involved in the assessment of site contamination need to demonstrate appropriate competence, knowledge and experience relative to their role and the complexity of site contamination.

4.1 Auditors and third-party reviewers

Auditors and third-party reviewers appointed under legislative requirements typically only act in the capacity of that role when they are carrying out an audit or a third-party review in accordance with those legislative requirements. In other situations, for example, when that person is involved in any other site assessment and/or remediation, that individual is acting as an environmental consultant.

The role of an auditor or third-party reviewer acting under statute is to carry out reviews of the assessment and/or remediation work carried out by environmental consultants and to provide independent expert opinion regarding any potential impacts to human health and/or the environment relating to site contamination, and the suitability of land for its intended use.

Auditors and third-party reviewers must be able to demonstrate that:

- they have exercised their own professional judgment
- they have taken appropriate specialised advice when the contamination issue is outside their expertise
- their opinions have been reached independently
- in forming those opinions, they have not been unduly influenced by the views or actions of others who may have an interest in the outcome of the review.

Legislative requirements may include provisions in relation to conflicts of interest and ethical codes of conduct and integrity. Individuals may be subject to penalties for any breaches of those requirements.

There is a clear distinction between the roles of an auditor or third-party reviewer acting under statute, and an environmental consultant. Jurisdictions typically have legislation regarding the provision of false and misleading information relating to statutory decisions.

Individuals applying to regulatory authorities for the purpose of appointment or acceptance of certification in these roles need to demonstrate significant knowledge and extensive experience in site contamination assessments. This should include the ability to meet all of the assessment criteria described in Section 6 of this Schedule.

The multidisciplinary nature of site contamination assessment requires that auditors and third-party reviewers are able to identify when there is an issue that is not within their own expertise and to obtain the additional professional advice required. In considering applications, it should be recognised by regulatory authorities that it is unlikely an individual could demonstrate all technical competencies relevant to site contamination.

4.2 Environmental consultants

An environmental consultant is usually a company that employs a range of professional and technical staff, or it can be an individual person. An environmental consultant can be engaged to carry out site contamination assessments for a variety of reasons. Engagement of a consultant is undertaken in accordance with the terms and conditions of that company or, in some cases, to complete an agreed scope of works. Their role is to design, prepare and carry out the assessment and/or remediation work in accordance with the scope of works.

Although not necessarily subject to specific legislative requirements, environmental consultants responsible for the assessment of contaminated sites and the preparation of assessment reports should

demonstrate relevant qualifications and experience to a level appropriate to the contamination issues relevant to the site under investigation.

While not having to demonstrate meeting all assessment criteria identified in Section 6.1 of this Schedule for auditors and third-party reviewers under statute, environmental consultants should be able to demonstrate:

- qualifications consistent with this Schedule
- competencies relevant to the work to be undertaken
- demonstrated relevant experience in site assessment
- comprehensive knowledge of relevant legislation and guidelines
- knowledge of relevant scientific literature for assessment of the impacts of site contamination on human health and the environment
- a demonstrated commitment to training and professional development
- relevant memberships and/or accreditation with professional societies.

Further information about qualifications and experience is provided in Section 6 of this Schedule.

Consultants should provide evidence that addresses these factors when it is requested. Individual jurisdictions may accredit consultants for certain activities or provide guidance on selection criteria and should be contacted for further advice as appropriate.

5 Application for acceptance

The application requirements described in this section relate to individuals applying to regulatory authorities for the appointment or acceptance of certification in the role of an environmental auditor or third-party reviewer under statute.
Subject to the specific legislative, policy and guideline frameworks applying in each state and territory, regulatory authorities reviewing applications from professionals for acceptance of their qualifications and experience should require the following information to be supplied for assessment.

1. A detailed current curriculum vitae that identifies relevant qualifications and the number of years' relevant experience held by the applicant in the assessment of contaminated sites.
2. A detailed statement of the applicant's knowledge, experience and expertise in relation to the assessment of contaminated sites and environmental issues, addressing the required technical competencies.
3. A statement demonstrating the applicant's understanding of the relevant provisions of environmental legislation in the particular state or territory and knowledge of policy, regulations and procedures.
4. A statement demonstrating the applicant's knowledge and understanding of the relevant provisions of guidelines issued or approved in the particular state or territory.
5. Nomination of people or companies who will provide support to the applicant in the competencies in which the applicant is not an expert.
6. Information that demonstrates the applicant's experience in forming and managing appropriate multidisciplinary teams for complex assessments.
7. A commitment that a professional liability insurance policy is, or will be, held by the applicant or on the applicant's behalf by the company employing the applicant, that demonstrates an appropriate level of coverage. Policies should cover the person for the activities to be undertaken and should not contain any exclusion that may have the effect of limiting cover for work carried out.
8. Examples of two or more relevant reports or studies on site contamination, which were authored or substantially prepared by the applicant and prepared no more than two years prior to the date of application. The reports should demonstrate the applicant's expertise in the assessment of contamination and their written communication skills. The report should clearly support the statements made by the applicant under items 2, 3 and 4 above. The role of the applicant in conducting the study (consultancy) and in preparing the report must be clearly indicated. Individual regulatory authorities may have specific requirements relating to requiring consent to be obtained from the client(s) for the reports to be submitted with the application, and may decide to return the reports to the applicant.
9. Summary information about additional reports and studies in which the applicant has made a major contribution may be presented, indicating the title of the project, the date of the report, the role of the applicant and the purpose of the project.
10. Nomination of referees. Referees should include people not directly associated with the applicant or the company employing the applicant, who have direct and recent knowledge of the applicant and can confirm the applicant's experience and expertise as stated under items 2, 3 and 4.

6 Assessment Criteria

This section details appropriate minimum criteria that should be considered by regulatory authorities in the assessment of individual applicants seeking acceptance for certification of contaminated site assessment work. Individual regulatory authorities may have further specific requirements for the criteria.

The ability of environmental consultants not otherwise subject to legislative requirements to demonstrate these criteria may also be used to assist in the decision-making process regarding the use of environmental consultants to carry out the assessment of site contamination.

6.1 Technical basis of application

The applicant should be able to demonstrate extensive experience and a high level of expertise in the core competencies required in each state and territory. In general, this will comprise such experience and expertise in all or a majority of the following:

- assessment of contaminant exposure pathways
- contaminated site assessment and management
- evaluation and interpretation of chemical and analytical data
- soil sampling design and methodology
- soil gas sampling design and methodology
- groundwater sampling design and methodology
- identification of potential human health and environmental risks
- quality control/quality assurance procedures
- risk communication.

The applicant should have basic proficiency in and be able to demonstrate experience and expertise relating to site contamination in the following areas, or otherwise have access to such expertise, to the level required by individual regulatory authorities:

- air quality (volatile emissions and dust) assessment relating to contamination
- assessment of impacts on groundwater from contaminated sites
- contaminant fate and transport
- environmental chemistry
- environmental sampling
- environmental toxicology
- geology
- human health and ecological risk assessment relating to contamination
- human toxicology
- hydrogeology
- identification of contaminants of concern from past industrial land uses
- work health and safety relating to contamination
- remediation technologies and geo-technology
- soil science
- statutory and environmental planning.

6.2 Legislative and guideline knowledge and understanding

The applicant should be able to demonstrate knowledge and an understanding of relevant legislation, regulations and policies relating to site contamination in each state or territory for which acceptance is sought.

The applicant should be able to demonstrate knowledge and an understanding of relevant guidelines issued or approved in each individual state and territory for which acceptance is sought.

The applicant should also be able to demonstrate consistency with relevant legislation and guidelines, in their carrying out and reporting of contaminated site assessments.

6.3 National framework

The applicant should be able to demonstrate an understanding of the National Environment Protection (Assessment of Site Contamination) Measure 1999 and other national guidance documents relevant to contamination.

The applicant should also be able to demonstrate consistency with the National Environment Protection (Assessment of Site Contamination) Measure 1999, in their carrying out and reporting of contaminated site assessments.

6.4 Experience and expertise

The applicant should demonstrate his/her expertise in the competencies identified in Section 6.1, to the level required by individual states and territories.

Where a competency, other than a core competency, is not able to be demonstrated by the applicant to the level required, the applicant should demonstrate access to relevant expertise in that competency.

All applicants should be required to nominate an expert support team of specialised professionals on whom they would rely for site issues beyond their areas of expertise.

Regulatory authorities in individual states and territories may have specific requirements relating to the qualifications, experience and expertise of expert support team members, and their use by applicants. However, an individual nominated as an expert in an auditor's support team should:

- be able to demonstrate a high level expertise or knowledge in the competencies where the applicant does not personally possess such expertise or knowledge to the level required
- hold qualifications relevant to and supporting the nominated competencies
- have at least eight years' relevant experience
- be actively working in the field of the nominated competencies
- be a current member of professional organisations/associations relevant to the field of the nominated competencies
- be able to demonstrate an ongoing commitment to professional training and development.

The applicant should demonstrate a sound ability and experience in forming and managing a multidisciplinary team for complex site assessment which contains the appropriate balance of expertise.

6.5 Qualifications

The applicant should hold qualifications as required by the regulatory authorities in individual states and territories (for example, a relevant bachelor's degree from a recognised institution).

6.6 Professional societies

The applicant should be required to demonstrate individual membership of and/or accreditation from one or more relevant professional societies, for example, Engineers Australia, the Royal Australian Chemical Institute, the Australian Institute of Geoscientists, the Environment Institute of Australia and New Zealand. In addition, applicants should also be able to demonstrate membership and/or accreditation of professional associations where relevant to nominated technical competencies (identified in section 6.1).

When considering professional societies that may be acceptable, regulatory authorities should consider the following criteria:

- discipline or area of expertise or interest relates directly to the assessment and management of contaminated sites
- membership is qualification-based
- membership requires adherence to an appropriate code of ethics.

Regulatory authorities should also take into consideration whether the maintenance of the membership and/or accreditation by the applicant is active and current. Maintenance of memberships should be in accordance to any code of ethics relevant to the particular society, and adherence to professional standards.

6.7 Professional experience

Regulatory authorities in individual states and territories may have specific requirements for the number of years of experience that applicants would be expected to have. Applicants should be expected to have had at least eight years' continuous relevant experience in the assessment and management of contaminated sites for appointment as accredited auditors or for acceptance as professionals involved in preparation and certification of assessments of complex contaminated sites. Individual regulatory authorities may also consider applicants with less than the required years' contaminated land experience but with significant years of relevant and related environmental experience, including assessment and management of major environmental issues involving complex sampling design and chemical or hydrogeological data collection and interpretation, where this experience is relevant.

It is generally desirable that an applicant's experience include at least two years of relevant work in Australia and two years in the role of project manager involving a multidisciplinary team approach to contaminated land or related environmental assessment and management.

It is preferable that the experience in contaminated sites work is broadly based in terms of the scale of work undertaken, the range of contaminants encountered and the scope of work performed, and includes contaminated site or environmental auditing experience, for example, as a member of an accredited auditor's expert support team or as an auditor's assistant.

6.8 Principles of audits

The applicant should be able to demonstrate a thorough understanding of the principles of, and methods for, conducting contaminated site assessments and environmental audits as required by the relevant state and territory, and be able to act independently using balanced professional judgement based on site-specific data and the advice of specialised support professionals.

6.9 Literature

The applicant should be able to demonstrate up-to-date knowledge of relevant scientific, technical developments and regulatory literature relating to new legislation and court proceedings and decisions relating to contaminated sites.

6.10 Professional development

The applicant should be able to demonstrate an active commitment to ongoing training and professional development relevant to the technical competencies (identified in Section 6.1) and the assessment and/or remediation of contaminated sites. Applicants should be able to provide evidence of continuous professional development and learning outcomes.

7 Acceptance processes and general conditions

This section provides general guidance on processes for the acceptance and ongoing review of applicants seeking certification of contaminated site assessment work.

7.1 General acceptance processes

Regulatory authorities may apply the following processes for the assessment, selection and review of auditors or third-party reviewers in accordance with legislative requirements and operational policies applying in each jurisdiction.

The regulatory authority may consider the establishment of a panel to assess applications. Typical panels would have not less than three professionals including a suitably qualified chairperson. Panels need to be able to adequately assess all of the competencies relevant to contaminated land assessment and management. The panel must consider the applicant's ability to meet all of the assessment criteria identified in Section 6, including the composition and relevance of their expert support team, their demonstrated ability to act independently on the basis of factual evidence, and their adherence to ethical and professional standards of conduct.

7.2 Ongoing practice

Once appointed or accepted for certification, regulatory authorities should ensure that professionals continue to update their training and experience in relation to the assessment of contamination, and comply with the relevant legislative requirements of the individual states and territories. This may be carried out through the implementation of a quality assurance program by the regulatory authority and review of a person's appointment, particularly at times of renewal. Applicants for renewal should also be able to demonstrate they are actively auditing.

The regulatory authority in individual states and territories may conduct independent audits and peer reviews of assessment work and adopt a system that involves the periodic review of the status of appointed professionals. Reviewers within the regulatory authorities should have appropriate qualifications and experience.

In the event of proven malpractice, such as a breach of legislative requirements by accepted persons, the regulatory authority may suspend or revoke the acceptance and may apply appropriate additional penalties in accordance with their legislative requirements.

National Environment Protection (Assessment of Site Contamination) Measure 1999

as amended

made under section 14(1) of the

National Environment Protection Council Act 1994 (Cwlth), the *National Environment Protection Council (New South Wales) Act 1995* (NSW), the *National Environment Protection Council (Victoria) Act 1995* (Vic), the *National Environment Protection Council (Queensland) Act 1994* (Qld), the *National Environment Protection Council (Western Australia) Act 1996* (WA), the *National Environment Protection Council (South Australia) Act 1995* (SA), the *National Environment Protection Council (Tasmania) Act 1995* (Tas), the *National Environment Protection Council Act 1994* (ACT) and the *National Environment Protection Council (Northern Territory) Act 1994* (NT)

Compilation start date:	16 May 2013
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This compilation has been split into 22 volumes

Volume 1: sections 1–6, Schedules A and B
Volume 2: Schedule B1
Volume 3: Schedule B2
Volume 4: Schedule B3
Volume 5: Schedule B4
Volume 6: Schedule B5a
Volume 7: Schedule B5b
Volume 8: Schedule B5c
Volume 9: Schedule B6

Prepared by the Office of Parliamentary Counsel, Canberra

Each volume has its own contents

About this compilation

The compiled instrument

This is a compilation of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* as amended and in force on 16 May 2013. It includes any amendment affecting the compiled instrument to that date.

This compilation was prepared on 22 May 2013.

The notes at the end of this compilation (the ***endnotes***) include information about amending Acts and instruments and the amendment history of each amended provision.

Uncommenced provisions and amendments

If a provision of the compiled instrument is affected by an uncommenced amendment, the text of the uncommenced amendment is set out in the endnotes.

Application, saving and transitional provisions for amendments

If the operation of an amendment is affected by an application, saving or transitional provision, the provision is identified in the endnotes.

Modifications

If a provision of the compiled instrument is affected by a textual modification that is in force, the text of the modifying provision is set out in the endnotes.

Provisions ceasing to have effect

If a provision of the compiled instrument has expired or otherwise ceased to have effect in accordance with a provision of the instrument, details of the provision are set out in the endnotes.

Contents

Endnotes

Endnote 1—Legislation history

This endnote sets out details of the legislation history of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* .

Title	Gazettal or FRLI registration date	Commencement date	Application, saving and transitional provisions
National Environment Protection (Assessment of Site Contamination) Measure (F2008B00713)	22 Dec 1999 (*see Gazette* 1999, No. GN51)	22 Dec 1999	
National Environment Protection (Assessment of Site Contamination) Amendment Measure 2013 (No. 1)	15 May 2013 (*see* F2013L00768)	16 May 2013	—

Endnote 2—Amendment history

This endnote sets out the amendment history of the *National Environment Protection (Assessment of Site Contamination) Measure 1999* .

ad. = added or inserted am. = amended rep. = repealed rs. = repealed and substituted exp. = expired or ceased to have effect

Provision affected	How affected
Introductory note	am. 2013 No. 1
s. 3	am. 2013 No. 1
s. 6	am. 2013 No. 1
s. 7	am. 2013 No. 1
s. 8	am. 2013 No. 1
s. 9	am. 2013 No. 1
s. 10	am. 2013 No. 1
Schedule A	
Schedule A	rs. 2013 No. 1
Schedule B	
Schedule B	rs. 2013 No. 1

Endnote 3—Uncommenced amendments [none]

There are no uncommenced amendments.

Endnote 4—Misdescribed amendments [none]

There are no misdescribed amendments.