Ministry for the Environment
Manatū Mō Te Taiao

Draft Methodology for Deriving Soil Guideline Values Protective of Human Health

New Zealand Government

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Status of this Document

Together with the *Draft Toxicological Intake Values for Priority Contaminants in Soil,* this document serves as a technical reference in support of the Ministry for the Environment's Discussion Document: *Proposed National Environmental Standard for Assessing and Managing Contaminants in Soil*.

Final versions of these documents will be published when work is completed on the National Environmental Standard.

Comments are invited on the content of this document, and in particular, the noting of any errors would be appreciated. Since this document is in draft form, changes may be made following the receipt of comments.

Please send comments by 19 April 2010, either electronically to cmlg@mfe.govt.nz, or by post to: SGVs for Contaminants in Soil, Ministry for the Environment, PO Box 10362, Wellington 6143.

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Executive Summary

This technical report sets out a risk-based methodology for deriving soil contaminant concentrations protective of human health. Together with the *Draft Toxicological Intake Values for Priority Contaminants in Soil (MfE, 2010b),* this document serves as a technical reference in support of the *Proposed National Environmental Standard for Assessing and Managing Contaminants in Soil: Discussion Document* (Ministry for the Environment, 2010a), and should be read in conjunction with it.

This report addresses a question that is fundamental to the science of contaminated land management: What contaminant concentration in soil can people be exposed to and yet not be subject to an appreciable risk of harm? The answer to this question varies internationally because each jurisdiction frames its response to fit within unique risk policy and legislative frameworks. So, although the technical approach to risk assessment of contaminated land is shared broadly by most countries, there are significant differences in detail such that a standard adopted by one country may not suit another.

As an alternative to adopting standards from another country, the Ministry for the Environment has examined the science of risk assessment and compiled a soil guideline derivation methodology for application in New Zealand under the Resource Management Act 1991. This initiative comes at a time when it is appropriate also to review the soil guideline values contained within the Ministry's existing suite of contaminated land guidelines. The soil guideline values for human health, $SGVs_{(health)}$ contained in this technical report, are intended to supersede the 'soil acceptance criteria' used in previous New Zealand guidelines; the new methodology also resolves technical differences between them. The Ministry intends to apply the same approach when reviewing the petroleum hydrocarbon contaminants.

The term $SGVs_{(health)}$ specifically refers to a soil contaminant concentration that, if exceeded, may result in health effects that are more than minor for some people, such an exceedance being unacceptable. Conversely, if actual soil concentrations are less than or equal to the $SGVs_{(health),}$ then this is judged to be acceptable because any adverse effects on human health for most people are likely to be no more than minor. $SGVs_{(health)}$ have been calculated for five generic land-use exposure scenarios to illustrate firstly their method of derivation and secondly, to show how they can be applied in practice. It is important to note that these values are intended to be protective of human health only, and not to other environmental receptors. When it is relevant to protect other valued elements of the environment, separate consideration of appropriate values to achieve this is required.

This report presents:

- a national risk-based methodology for deriving soil contaminant concentrations protective of human health
- a suite of draft numerical criteria for priority contaminants as examples of the national methodology
- background information on the risk assessment methodologies and exposure parameters.

As outlined in the accompanying proposed national environmental standard discussion document (MfE, 2010a), the $SGVs_{(health)}$ may be applied as Tier 1 or screening criteria; as conservative clean-up targets, to inform on-site management actions; or to trigger further investigation within a Tier 2 assessment.

Draft SGVs$_{(health)}$ are derived for the following priority contaminants: arsenic, boron, cadmium, chromium, copper, inorganic lead, inorganic mercury (but not elemental mercury), benzo(a)pyrene (representing the carcinogenic polycyclic aromatic hydrocarbons), DDT (as the sum of DDT, DDD and DDE), dieldrin, dioxin (as 2,3,7,8-tetrachlorodibenzo-*p*-dioxin) and dioxin-like polychlorinated biphenyls (PCBs), and pentachlorophenol. The SGV$_{(health)}$ calculations draw on toxicological intake values and background exposures set out in the companion document *Draft Toxicological Intake Values for Contaminants in Soil: Technical Report* (MfE, 2010b). Numerical values are calculated for a generic suite of land-use scenarios, and utilise standardised receptors and exposure parameters.

A summary of the SGVs$_{(health)}$ derived is presented in Tables ES1 and ES2. Contaminated-land practitioners are referred to the more detailed version of these tables set out in section 7 of this report, in which additional residential subscenarios have also been derived.

Table ES1: Summary of soil guideline values for inorganic substances (mg/kg)

	Arsenic	Boron	Cadmium (pH 5)[1]	Chromium		Copper	Inorganic lead	Inorganic mercury
				III[2]	VI			
Rural residential / lifestyle block 10% produce	20	34,000	5	280,000	560	32,000	730	380
Residential 10% produce	24	34,000	5	280,000	560	32,000	730	380
High-density residential	50	75,000	370	890,000	1,800	60,000	1,600	1,200
Recreation	100	220,000	1,100	NL	5,200	170,000	4,700	3,500
Commercial / industrial outdoor worker	70	400,000	1,600	NL	6,300	290,000	7,000	4,200

1 Default value is for pH 5.

2 The SGVs$_{(health)}$ for boron, chromium III and copper represent levels well in excess of concentrations that would affect the health of plants.

NL = No limit

Table ES2: Summary of soil guideline values for organic compounds (mg/kg[1])

Scenario	BaP	DDT	Dieldrin	PCP	Dioxin (µg/kg TEQ)	
					TCDD	Dioxin-like PCBs
Rural residential / lifestyle block 10% produce	85	90	3.1	70	0.19	0.15
Residential 10% produce	100	90	3.1	70	0.19	0.15
High-density residential	240	270	50	130	0.41	0.38
Recreation	440	750	110	230	1.1	0.9
Commercial / industrial outdoor worker	300	1,000	160	360	1.4	1.2

1 All values in mg/kg dry weight except dioxins which are in µg/kg.

1 Introduction

1.1 Background

The Ministry for the Environment (MfE) has determined the need for a better policy framework for managing contaminated land in New Zealand. After extensive public consultation, the Ministry published a position paper in September 2007 (MfE, 2007). The paper identified, among other things, an inconsistent and variable use by contaminated-land practitioners of numerical contaminated land guidelines used to assess the risk that contaminated soil might pose to human health. The Ministry then determined that, as a matter of priority, it would develop:

- a national risk-based methodology for deriving soil contaminant concentrations protective of human health
- a suite of numerical criteria for priority contaminants as examples of the national methodology
- site management options and actions that follow from applying the above criteria; ie, the numerical criteria may:
  - serve as Tier 1 or screening criteria to assess whether there is a potential risk to human health
  - when the criteria are exceeded, serve as conservative clean-up targets for many situations, ie, where further investigation or site-specific risk assessment is not warranted or economic
  - inform on-site management actions to reduce the potential for adverse effects
  - trigger further investigation to better assess the risk and/or determine site-specific criteria as a Tier 2 assessment.

The Ministry intends that the derivation methodology be incorporated by reference into a national environmental standard (NES), as proposed in the accompanying policy Discussion Document (*Proposed National Environmental Standard for Assessing and Managing Contaminants in Soil,* MfE, 2010a). Although the technical detail behind the newly derived soil guideline values would stay within this separate guideline document, a soil guideline value would be applied in accordance with the NES and within the context of the derivation methodology.

This technical report introduces and sets out the risk-based methodology the Ministry proposes be adopted as government policy for deriving $SGVs_{(health)}$: soil guideline values protective of human health. Draft $SGVs_{(health)}$ are derived for a limited group of priority contaminants to demonstrate the methodology; they are specifically: arsenic, boron, cadmium, chromium, copper, inorganic lead, inorganic mercury, benzo(a)pyrene, DDT, dieldrin, dioxin (as 2,3,7,8-tetrachlorodibenzo-*p*-dioxin) and dioxin-like polychlorinated biphenyls (PCBs), and pentachlorophenol. These calculations draw on toxicological values and background exposures set out in a companion document *Draft Toxicological Intake Values for Contaminants in Soil: Technical Report* (MfE, 2010b).

The numerical values are restricted to consideration of human-health risks and are based on conservative standard land-use scenarios, using standardised receptors and exposure parameters.

The consideration of exposure pathways includes (following MfE, 2007):

- estimating the levels of contaminants in the media (eg, air, water, food) that potentially convey the contaminants from soil to people (the 'receptor')
- identifying the typical physical characteristics of New Zealanders (such as area of skin, weight, air breathed, food and water ingested) that collectively determine a standard exposure model in association with the exposure pathways. This requires consideration of the:
  - groups of people (eg, children, adults, workers) who are potentially exposed, as well as the group considered the most sensitive to the toxic effect of each contaminant
  - time periods over which exposure occurs
  - age-related contact rates such as ingestion and inhalation rates
  - consumption of produce from their own home gardens.

The United Kingdom, Australia and elsewhere have recently reviewed the partitioning and vapour migration models used to derive guidelines for the inhalation of volatile contaminants. Therefore, this pathway is not considered in detail and no guidelines for volatile contaminants have been derived. It is beyond the scope of, and premature for, this document to review or derive more appropriate inhalation models. Further research is being carried out internationally on applying such models.

For ease of reference, $SGVs_{(health)}$[1] are often referred to in this document simply as SGVs, or soil guideline values. Where a health protection context is thought useful, the term $SGVs_{(health)}$ is retained.

1.1.1 The purpose of $SGVs_{(health)}$

$SGVs_{(health)}$ are soil contaminant concentrations appropriate to five generic land-use exposure scenarios at which the exposure to levels equal to or less than the SGV is judged be acceptable, because any adverse effects on human health for most people are likely to be no more than minor.

1.1.2 Purpose

This technical report sets out in a transparent manner a methodology to derive $SGVs_{(health)}$. In addition to the five standard scenarios, some additional residential subscenarios have been derived for illustrative purposes. The additional residential scenarios are for proportions of home-grown produce other than the standard 10 per cent. The additional derivations show the effect on the derivations for zero and 50 per cent home-grown produce.

$SGV_{(health)}$ derivations are given for:

- eight elements – arsenic, boron, cadmium, chromium (in trivalent and hexavalent forms), copper, inorganic lead and inorganic mercury
- six organic compounds, or groups of compounds – benzo(a)pyrene (to represent the carcinogenic polycyclic aromatic hydrocarbons); DDT (as the sum of DDT and its metabolites DDD and DDE); dieldrin (or aldrin or the sum of aldrin and dieldrin); tetrachlorodibenzo-*p*-dioxin (to represent dioxin mixtures); dioxin-like polychlorinated biphenyls; and pentachlorophenol.

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[1] In this document the term 'soil guideline values' replaces the term 'soil acceptance criteria' used in previous New Zealand guidelines.

1.1.3 Document organisation

This document is divided into the following parts. After this introduction (section 1), background information on the principles of the soil quality guidelines derivation is presented in section 2. Section 3 reviews and recommends generic exposure scenarios. Soil guideline derivation equations are provided in section 4. The proposed exposure pathways, and specific parameters for these pathways for each exposure scenario, are given in section 5, with toxicological values. The equations and parameters, together with contaminant-specific values from MfE (2010b) are then used to derive soil guideline values for selected priority contaminants in section 6. A summary of the derived values is provided in section 7.

The details pertaining to $SGV_{(health)}$ calculations for each individual contaminant in accordance with exposure assumptions are set out in appendix 1, with pH-dependant details provided for cadmium in appendix 2, and the details for dioxin with egg consumption in appendix 4. Appendix 3 sets out the produce consumption estimates made. Details of SGV derivation methodologies for selected overseas jurisdictions, and specific derivation methodologies for existing New Zealand guidelines, are given in appendix 5. This is followed by a glossary and a comprehensive references section.

2 General Approach to Generic Guideline Derivation

2.1 Concepts

Soil guideline values are generic quality standards adopted in many countries to regulate the management of contaminated land. They are usually in the form of concentration thresholds (mg/kg soil dry weight) of contaminants in soil, above which certain actions are recommended or enforced. The implications of exceeding the soil guidelines vary according to the regulatory framework of the particular national or regional jurisdiction. They range from the need for further investigations to the need for remedial actions. Given the different purposes in national regulatory frameworks, soil guidelines have been given various names (translated as appropriate): trigger values, reference values, target values, intervention values, clean-up values, cut-off values, and many others (Carlon, 2007).

Generic risk-based human-health soil guideline values are fundamental to the risk assessment process for contaminated land. Risk assessment is a process in which information is analysed to determine if an environmental hazard might cause harm to exposed persons and ecosystems (US EPA, 2004a). In this case the environmental hazard is land contamination. The risk assessment process is a multi-step process (Defra and EA, 2002a) consisting of:

1. defining the conceptual exposure model
2. collecting human exposure characteristics and contaminant fate and transport data
3. selecting contaminant concentration in soil from review of site investigation data
4. quantitative modelling of exposure
5. comparing predicted human exposure with health criteria values, based on contaminant toxicity
6. evaluating significance of risk to human health where exposure is close to or exceeds health-based reference values.

The above process is complex and time-consuming if carried out from first principles on every site. Multiple-exposure scenarios, receptors, exposure mechanisms and chemicals must be evaluated to determine their significance. Thousands of pages of guidance have been written by authorities such as the United States Environmental Protection Agency (US EPA) to ensure consistency of process. Most sites do not warrant the detail or expenses of a full risk assessment, which is why many jurisdictions have developed ‘screening level’ assessment processes using standardised exposure scenarios. The basis of screening assessments (otherwise known as Tier 1 assessments) is the use of SGVs derived for these standardised exposure scenarios.

The assessment process then becomes a much simpler process of taking soil samples (in accordance with, for example: MfE, 2004), analysing them for relevant contaminants and comparing the results with an appropriate soil guideline value. If the sample results are below the SGV, then the risk to human health is deemed acceptable; if the results exceed the SGV, a potential risk to human health is indicated and some form of action or management may be necessary. For most sites the assessment process stops there: the site has been dealt with as appropriate. However, for some sites it is worth making a more accurate assessment, using the more detailed site-specific assessment process (often called Tier 2 assessment).

The generic values are deliberately conservative to be protective of the great majority of potentially exposed people (often defined as the 95th percentile of the population). However, for some large sites this could lead to very expensive remediation or other forms of site management to limit exposure. In these situations the additional expense of a more detailed and less conservative assessment process may be far outweighed by the cost saving in remediation or management.

It is important that the $SGVs_{(health)}$ are not too conservative, as they may be used as clean-up guidelines where the cost of further assessment is not warranted. If screening guidelines are too conservative, then more clean-up than is warranted could occur. The US EPA uses the concept of reasonable maximum exposure (RME) in combining upper-bound and average exposure factors to arrive at exposure scenarios that are protective but reasonable (US EPA, 1989b, 2004a). Under this approach, some intake variables may not be at their individual maximum values, but in combination with other variables will result in estimates of the RME. However, the US EPA has been criticised for this approach on the basis that the RME approach combines too many upperbound parameter estimates, resulting in unrealistic over-conservative assessments (US EPA, 2004a).

Key components of the generic human health guidelines include standard human exposure scenarios relevant to a variety of land uses (eg, residential, commercial, parkland) and exposure through a variety of pathways (eg, inhalation, ingestion, skin absorption). The soil guidelines are then calculated to ensure that some pre-determined allowable daily intake of the contaminant is not exceeded. The allowable daily intakes (otherwise known as a reference health standard – RHS) are discussed in greater detail in MfE (2010b). These scenarios and exposure pathways are explored in subsequent sections.

The Canadian Council of Ministers of the Environment has developed a useful set of guiding principles for the development of SGVs (CCME, 2006). These are set out, in modified form, below:

1. There should be no appreciable risk to humans from a contaminated site. For each specified land use, there should be no restrictions as to the extent or nature of the interaction with the site. All activities normally associated with the intended land use should be free of any appreciable health risk.
2. Guidelines are based on defined, representative situations (exposure scenarios or land uses). Deriving numerical guidelines necessitates defining specific scenarios within which the exposure likely to arise on the site can be predicted with some degree of certainty.
3. Guidelines are derived by considering exposure through all relevant pathways.
4. A critical human receptor is identified for each land use. To ensure that the guidelines do not limit the application of a site within the intended land-use category, the defined exposure scenarios are usually based on the most sensitive receptor to the chemical, and the most critical health effect.
5. Guidelines should be reasonable, workable and usable. Guidelines are developed by applying scientifically derived information, backed by professional judgement where data gaps occur. Occasionally, defined exposure-based procedures produce numerical guidelines far below background levels of contamination occurring naturally in the soil. When this occurs, guidelines cannot be below background levels.

2.1.1 Generic equations

Soil guideline values for the protection of human health are typically based on generic assumptions about exposure incorporated into standard equations. These equations are essentially identical, regardless of jurisdiction, although the detail can vary considerably. The equations for a particular contaminant and exposure pathway i, follow the form:

$$\text{Intake}_i = \text{soil concentration} \times \text{contact rate}_i \times \text{exposure time} \qquad \text{Eqn 1}$$

The intake is usually normalised to an intake rate per unit of body weight (BW, in kilogram) and unit of time (day) by dividing by body weight and an averaging time. In addition, the exposure time is typically represented as exposure frequency in days per year multiplied by exposure duration in years, resulting in:

$$\text{Intake rate}_i = \frac{\text{soil concentration} \times \text{contact rate}_i \times \text{exposure frequency} \times \text{exposure duration}}{\text{body weight} \times \text{averaging time}} \qquad \text{Eqn 2}$$

The intake rate (in kg BW/day) is then compared with some acceptable intake rate for the substance (the reference health standard) and, for some substances, the particular pathway – with a human health risk indicated for exposure to that particular soil concentration and pathway if the intake rate exceeds the acceptable intake rate.

The acceptable intake is either the tolerable daily intake (TDI) for threshold compounds, or the dose that yields a specified increased cancer risk (the risk-specific dose). In New Zealand the specified acceptable cancer risk for non-threshold compounds is one additional cancer in 100,000 people (10^{-5}). This is discussed in greater detail in MfE (2010b).

Contact rate may be for soil ingestion, inhalation (particulates and vapours), dermal absorption, produce uptake, contaminated water and a variety of other things, depending on the policy of the particular jurisdiction. Contact rate may be modified by absorption, or by matrix factors to account for different modes of absorption into the body from contaminated soil, or by enrichment factors that account for the enriched concentration of contaminants on fine particles that are more likely to be inhaled or stick to the skin. The averaging time is the length of time over which exposure is averaged to give an average daily rate.

An allowable soil concentration (a soil guideline value) can be back-calculated by setting the intake rate to the acceptable intake rate and rearranging the equation so that the soil concentration is on the left hand side:

$$\text{Soil guideline value}_i = \frac{\text{accceptable intake}_i \times \text{body weight} \times \text{averaging time}}{\text{contact rate}_i \times \text{exposure frequency} \times \text{exposure duration}} \qquad \text{Eqn 3}$$

Similar values can be derived for each exposure pathway considered relevant. Some jurisdictions then choose the lowest value from the various pathway values, as the generic guideline for the substance in question. If there is more than one significant exposure pathway, the chosen soil guideline will result in an intake greater than the allowable intake, unless the allowable intake is factored down somehow to take the multiple pathways into account. As seen in appendix 5, Canada and some New Zealand guidelines adopt this approach.

A more usual approach is to back-calculate the guideline value combined over the relevant pathways. This is achieved by equating the sum of the hazard quotients (HQ) for each pathway with 1 (unity). The hazard quotient for a particular pathway is simply the ratio of the intake rate over the allowable intake.

$$\frac{\text{intake rate}_1}{\text{accceptable intake}_1} + \frac{\text{intake rate}_2}{\text{accceptable intake}_2} + \frac{\text{intake rate}_3}{\text{accceptable intake}_3} + \cdots = HQ_1 + HQ_2 + HQ_3 + \cdots = 1 \quad \text{Eqn 4}$$

The soil guideline value is then calculated from equation 4 by substituting equation 2 into equation 4 for each exposure pathway, and then redefining the soil concentration (assumed to be the same for each pathway) as the desired generic guideline value. After rearranging to bring the soil guideline value to the left hand side, the equation becomes:

$$\text{Soil guideline value} = \frac{1}{\left(\frac{1}{\text{soil guideline value}_1} + \frac{1}{\text{soil guideline value}_2} + \frac{1}{\text{soil guideline value}_3} + \cdots \right)} \quad \text{Eqn 5}$$

This is the basic algorithm used by the US EPA (1996a; 1996b) to calculated generic soil screening levels (SSLs) for multiple pathways and in the Ministry's 'Timber Treatment Guidelines' (MfE and MoH, 1997), and 'Sheep-dip Guide' (MfE, 2006a). The earlier equation 3 takes a slightly different form for each exposure pathway, being expanded with additional factors so the individual contact rates, which are generically expressed in mass of contaminant per day (mg/day), can be calculated from the exposure pathway-specific contact rates.

This additive approach is only valid if the acceptable intake has the same value for each pathway (typically defined by the oral pathway), or if different for one or more pathways (eg, the acceptable intake is different for the dermal pathway), so that all pathways have the same mode of toxic action. If the mode of action is different the pathway(s) with the different mode(s) of action must be considered separately, because the effects cannot be assumed to be additive.

Similarly, the additive approach is only valid if exposure to the soil in question can physically occur through multiple pathways. For example, it is not valid to combine exposure in a residential scenario to soil and produce ingestion, dermal absorption and inhalation of volatiles for deeper soil (eg, hydrocarbons on the water table at 2 metres) – because it is not physically possible for a person to be exposed to deeper soil through all these pathways simultaneously. In this instance, surface soil would need to be treated separately from deeper soil; and volatile inhalation is likely to be the only relevant pathway for deeper soil.

2.1.2 Generic exposure scenarios

Exposure scenarios are explored in greater detail in subsequent sections. However, as noted above, there is a considerable similarity to the basic approaches taken around the world to defining generic exposure scenarios, based around land use. Universally there is some form of residential exposure and generally some form of commercial or industrial exposure. Some jurisdictions consider an agricultural scenario – not for protection of produce, but for the protection of workers and/or farm residents. Some jurisdictions have also developed generic guidelines for various recreational scenarios.

Table 1 summarises exposure scenarios used in New Zealand's industry-based guidelines and several countries that New Zealand has traditionally looked to for guideline values, when a New Zealand guideline does not exist: specifically Australia, the United States, the United Kingdom, Canada and the Netherlands. All these countries have well-developed contaminated land assessment frameworks and guideline derivation methodologies.

The various scenarios presuppose a variety of exposure pathways and critical receptors. These are discussed in the next section.

Table 1: Scenarios for generic numeric values in various jurisdictions

Country	Scenario	Source documents
New Zealand 'Timber Treatment Guidelines'	Agricultural / horticultural[1] (includes home-grown prduce) Residential (high and typical home-grown produce intake)[2] Industrial – paved, unpaved Subsurface maintenance workers	MfE and MoH (1997)
New Zealand 'Gasworks Guidelines'	Agricultural / horticultural[1] (includes home-grown produce) Standard residential High-density residential (limited soil contact, no vegetable gardens) Commercial / industrial Parkland / recreational Maintenance workers	MfE (1997)
New Zealand 'Oil Industry Guidelines'	Agricultural / horticultural[1] (includes home-grown produce) Residential Commercial / industrial Maintenance Protection of groundwater for potable use	MfE (1999)
New Zealand 'Sheep-dip Guide'	Lifestyle block (includes home-grown produce) Standard residential (includes home-grown produce) High-density urban residential Parks / recreation Commercial / industrial (unpaved)	MfE (2006a)
Australia	Standard residential (includes children's daycare centres, kindergartens, preschools, and primary schools) Residential with minimal soil contact (includes dwellings with fully and permanently paved yard space, eg, high-rise apartments and flats) Parklands / recreational land use Commercial / industrial land use	NEPC (1999a)
US EPA Soil Screening Levels	Residential Industrial: indoor worker, outdoor worker Construction	US EPA (1996a and 2002a)
US EPA Regional PRGs[3]	Residential Industrial	US EPA (2008)
Canada	Agricultural Residential / parkland Commercial Industrial	CCME (1996, 2006)
UK	Allotments Residential with and without plant uptake Commercial / industrial	Defra and EA (2002a, 2002b)
The Netherlands	Residential with garden (standard scenario) Children's play areas Residential with kitchen garden Agriculture (equivalent to standard residential) Nature Green areas (parks and recreational areas) Other green areas, buildings and industry	VROM (2000), Brand et al (2007)

1 Agricultural and agricultural / horticultural is used interchangeably in these documents.
2 The high value for home-grown produce is described as typical of 'rural residential'.
3 Preliminary remediation goals.

2.1.3 Exposure pathways

There is sufficient commonality around the world with respect to exposure pathways that some can be considered generic. Similarly, there is, in general, a common approach internationally to quantitative risk assessment. Soil ingestion is universally considered, as is one or both of dermal exposure and inhalation of particulates and/or vapours. Beyond these exposure pathways, there is considerable variation between countries as to what particular pathway should be included. For example, the United States uses few pathways for generic screening values with relatively conservative parameters, leaving further consideration to site-specific assessment; by contrast, the UK and the Netherlands consider multiple pathways using spreadsheet-based models. Further detail is described for these countries in appendix 5.

2.1.4 Contaminant characteristics

The characteristics of a particular contaminant affect which exposure pathways may be significant, the form of the exposure equations, and (in some instances) the critical receptor. All jurisdictions considered as part of this study divide contaminants into:

- threshold and non-threshold in terms of their toxicological behaviour
- volatile or non-volatile with respect to the inhalation pathway.

New Zealand guideline documents differentiate between genotoxic and non-genotoxic carcinogens with the latter treated as threshold contaminants. Threshold and non-threshold contaminants are discussed in greater detail in MfE (2010b).

Threshold contaminants are considered to manifest toxic effects if exposure exceeds a threshold concentration, and conventionally (including in New Zealand), are considered to include both non-genotoxic carcinogens and non-carcinogens. For these contaminants, a reference health standard (commonly as a tolerable daily intake) is typically established, being the estimated daily amount that can be taken into the body without any detrimental health effects occurring.

Non-threshold contaminants are conventionally considered to include genotoxic carcinogens, and are considered to have effects at all levels of exposure. The potency of non-threshold contaminants is typically expressed as a slope factor or maximum likelihood estimate (both represent the increased risk per daily dose), or as a risk-specific dose or index dose (analogous to a tolerable daily intake for a minimal and acceptable human health risk). The risk-specific or index dose is obtained by dividing the acceptable increased risk level (ie, 10^{-5} in New Zealand) by the slope factor (MfE, 2010b).

The volatility of a contaminant affects whether the vapour pathway needs to be considered in addition to, or instead of, inhalation of contaminants attached to particulates (airborne dust). Inhalation of volatile organic compounds may be a significant part of overall exposure, whereas inhalation of particulates is typically not (and is ignored by some jurisdictions). Although of the same general form, the equations required to calculate exposure from particulates and volatiles have quite different subsidiary equations to derive the exposure rates. Subsidiary equations for particulates typically determine the amount of dust that is entrained into the air by the wind; equations for volatiles rely on the physico-chemical properties of the contaminant and soil, soil cover and building characteristics, to establish vapour concentrations that migrate to indoor or outdoor air spaces.

2.1.5 Critical receptor

Calculation of an $SGVs_{(health)}$ requires defining a critical receptor, with associated body weight, exposed skin areas, breathing rates and exposure rates for soil ingestion and dermal absorption, inhalation of particulates, and the like.

The critical receptor depends on land use / activity, and whether the contaminant is considered to be threshold or non-threshold. The critical receptor for the residential situation for threshold contaminants is almost universally a child (with associated low body weight and greater exposure rates to soil ingestion and dermal contact). The critical receptor in the industrial setting for a threshold substance is an adult worker.

For non-threshold substances, exposure over a lifetime is typically assumed: the critical receptor becomes a combination of childhood and adult exposure.

Further detail of this and other scenario-specific parameters is set out in section 5.

3 Exposure Scenarios

3.1 Introduction

Health risk assessment in contaminated-site practice is based on the assumption that individuals are exposed to contaminated soil while going about their normal activities. This exposure can occur in a number of ways: these are typically called exposure pathways and include soil ingestion, dermal absorption, and consumption of vegetables grown in contaminated soil. The exposure pathways used in the $SGV_{(health)}$ derivation are explained in greater detail in section 4.1.

An exposure scenario is a combination of exposure pathways typical of a particular activity in which exposure to soil contaminants is likely to occur: the intent is to estimate the intake of a contaminant for that scenario. For simple risk assessment, a number of generic scenarios are used, with a standard combination of exposure pathways for each scenario. The generic scenarios used for the SGV derivations are intended to be typical of the great majority of situations in which ordinary New Zealanders may be exposed to soil contaminants. They include three residential scenarios – standard, rural and high-density – to cover the range of residential situations in which most people live; parks / recreation to cover active play or sporting activities; and a commercial / industrial scenario to cover outdoor workers at work.

This section describes why the generic exposure scenarios have been chosen and details some others that have not been used as generic scenarios.

3.2 Rationale for the proposed scenarios

3.2.1 Residential scenarios – rural residential / lifestyle block, standard residential, high-density residential

The protection of humans in a residential setting is a major driving factor behind the development of soil guideline values. All jurisdictions have SGVs for some form of residential setting. Ingestion of contaminated soil and consumption of home-grown produce are typically the most significant pathways for inorganic compounds, while inhalation of volatile organic compounds and dermal absorption of some organics can add further significant pathways. The residential scenarios are characterised by a child being the critical receptor (because of their lower body weight and relatively higher exposure to soil than adults) and inclusion of home-grown produce consumption, for all but the high-density residential scenario. Not all overseas jurisdictions provide for produce consumption in generic guidelines but as growing vegetables at home is quite common in New Zealand, this pathway can make a significant difference to the derived value for a particular contaminant, depending on whether it is taken up into plants.

Standard residential

On average, separate houses form 73 per cent of private dwellings in main urban areas (eg, cities), 79–82 per cent in other urban areas, and 83–88 per cent in rural areas in New Zealand in 2001 (Statistics New Zealand, undated). Particularly in outer urban areas and rural areas, separate houses are likely to contain gardens, and therefore have the potential to grow vegetables. Given the high proportion of separate houses and the potential to consume home-grown vegetables from these dwellings, there needs to be a residential exposure scenario that considers consumption of home-grown produce.

This is supported by some local government research in Hamilton, Christchurch and Hastings. Work in Hamilton, reported in Cavanagh (2004a), indicates that about one-third of residential houses are likely to contain a garden in which home produce is grown. Similarly, in Christchurch, about 20 per cent of residential houses in recent subdivisions (less than 10 years old) contained a garden in which home produce is grown. A survey of residents' vegetable gardening habits and size of gardens carried out in Hastings and Havelock North found about two-thirds of the 121 households surveyed grew at least some of their vegetable consumption (Philip McKay, Hastings District Council, pers. comm).

High-density residential

The remainder of private dwellings in New Zealand are predominantly multiple-unit dwellings, multi-unit townhouses, blocks of flats and high-rise apartments. Almost 20 per cent of private dwellings in main urban areas were multi-unit dwellings in the 2001 Census; that proportion has probably risen since. Single-storey multi-unit dwellings are less likely to have gardens than separate houses and the gardens that do exist will tend to be small ornamental gardens, limiting the opportunity for soil contact. Significant growing of vegetables is not expected. The high-density residential scenario will therefore have lower soil ingestion rates than standard residential, and not include home-grown vegetable consumption.

A special case of high-density housing is inner city apartments, including high-rise developments. Residents would not be expected to have direct contact with soil, the only possible exposure being inhalation of volatiles for ground-floor residents where the floor is in direct contact with the ground. Ground-floor residences are akin to an industrial / commercial indoor worker scenario, except for the greater exposure frequency and being longer on site each day. Unless volatile contaminants are present there will be no soil ingestion, particulate inhalation or dermal contact – and therefore no risk. Risk from volatiles for ground-floor residents would be as calculated for the indoor worker, factored up to consider the greater exposure frequency and duration expected of a residential occupant. Inhalation exposure to volatile compounds is not considered in this document.

Rural residential / lifestyle blocks

Increasingly, land that was previously in agricultural or horticultural use is being subdivided into lifestyle blocks. This land may be contaminated from the historical widespread use of persistent agrichemicals and/or disused sheep dips. There is also a greater potential for a higher proportion of home-grown vegetables (eg, compared to urban residents) to be grown on a lifestyle block, so the significance of consuming contaminated produce may be greater than in the residential scenarios considered above.

However, there is no information on how much produce might be grown for own use within lifestyle blocks or rural areas. Depending on the circumstances, 10 per cent home-grown produce may be appropriate (ie, as for standard residential), whereas 50 per cent is expected to be towards the high end of a more self-sufficient lifestyle that some rural dwellers may adopt. The 100 per cent produce value used in the 'Timber Treatment Guidelines' (MfE and MoH, 1997) is considered unrealistic for most people. In the absence of information for rural-dwellers, the standard residential produce proportion of 10 per cent has been adopted as the default. For illustrative purposes, soil guideline values have also been derived for a higher proportion, 50 per cent, as an estimate of a possible extreme percentage. Ideally, the percentage of produce grown at home should be considered on a case-by-case basis for rural sites where a high proportion of produce is home-grown – particularly for those practising a ‘self-sufficiency’ lifestyle – and the SGV derived accordingly. The derived 50 per cent produce SGV may be suitable as a first approximation where the assessor deems it appropriate.

The rural residential scenario is not intended to cover the productive areas of agricultural land but is intended to be applicable to the immediate vicinity of the farmhouse or staff houses, an area of perhaps several hundred to a few thousand square metres. This will include kitchen garden areas, areas where children might routinely play and perhaps the ‘home paddock’, with the intention of protecting the farming family and any staff and their families. This is as agreed by the Technical Review Group for the Proposed National Environmental Standard for Assessing and Managing Contaminants in Soil, which suggested the scenario be called ‘rural’ so as to include residences on farms (MfE, 2005).

Poultry or grazing animals are often raised on a lifestyle block or farm, and consuming products from these animals (eggs, milk and meat) can constitute additional pathways of exposure to contaminants, particularly for lipophilic organic contaminants that bioaccumulate through the food chain. These pathways are highly variable in occurrence and highly influenced by site-specific considerations (contaminant, soil type, type of produce, degree of resident self-sufficiency), and should therefore be considered on a site-specific basis.

Locations for storing farm chemicals and fuel or for housing implements are generally similar to an unpaved commercial / industrial scenario with respect to human health effects, and should be considered accordingly.

3.2.2 Parks / recreation scenario

Some overseas jurisdictions including the Netherlands and Australia have park or recreational scenarios. Canada combines this scenario with residential use. The US EPA policy is to regard the use as site-specific. The scenario exists in two current New Zealand guidelines, the ‘Gasworks Guidelines’ (MfE, 1997) and the ‘Sheep-dip Guide’ (MfE, 2006a).

With New Zealanders’ relatively outdoor lifestyle, use of parks and urban green spaces is common. Further, inadvertent or deliberate use of contaminated land for park area or reserve areas is not uncommon in New Zealand: for example, many playing fields in Wellington were former landfills, and the soil under Victoria Park in Auckland contains gasworks wastes. Subdivision of former agricultural or horticultural land for new urban development typically includes setting some land aside for reserve areas and public rights-of-way. Some district councils are permitting disposal of contaminated ex-orchard soil into such reserve areas during the development process, provided the soil complies with a recreational soil guideline value.

Recreational activities are diverse. They range from walking through a park, where very little soil exposure occurs, to playing contact sports such as rugby, where players can end up being caked with mud during wet conditions. An intermediate scenario might be a child in a playground where the grass is worn (resulting in soil contact). A generic guideline has difficulty covering such a wide range of potential exposures, with selection of exposure parameters for the extremes described above resulting in quite different soil guideline values. Conservatively, a residential non-produce scenario could be used for parks / recreation, but typically the exposure frequency will be less than that compared to residential scenarios, and is not recommended.

An analysis of alternative scenarios is presented in section 5.3.3. The analysis shows that, if soil ingestion is the main consideration (true for most substances but not necessarily all organic contaminants), then the parks / recreation scenario can reasonably cover suburban reserves within residential areas and a sports field both for children and adults, and also a secondary school playing field. However, a primary school playing field should be the subject of site-specific assessment, given the lower body weight of the children. The scenario is too conservative for parks and reserves used for passive recreation (eg, walking in a park or public garden), but could be used as a first screening. Children's playgrounds are so variable in their layout and use that the scenario might be conservative or non-conservative, and therefore site-specific assessment is recommended.

The risk to park maintenance staff should be assessed separately using the commercial / industrial unpaved scenario (see below).

Park areas are often large and have multiples uses. It is necessary to firstly develop a good conceptual model of the site so that appropriate receptors and pathways are assigned to relevant sub-areas for screening assessment. It could be that a particular park is amenable to screening with SGVs over parts of the site, but site-specific assessment is necessary for other parts that potentially have much higher or lower soil contact than this scenario envisages. Table 2 gives a summary of parks / recreational subscenarios and the recommended approach.

Table 2: Parks / recreation scenarios – recommended approaches

Subscenario	Approach
Playing field	Included in scenario. Check occupational exposure for maintenance staff using commercial / industrial unpaved
Residential reserve where children play frequently	Included in scenario
Secondary school playing field	Included in scenario
Primary school playing field	Site-specific
Public green areas, reserves and gardens used for passive recreation	Scenario is very conservative but could be used for first screening, otherwise site-specific. Check exposure for park maintenance staff using commercial / industrial unpaved
Children's playground	Site-specific

3.2.3 Commercial / industrial scenario

Contamination of industrial land is common. This land may remain as industrial land or may be converted to commercial uses such as shopping centres, warehousing and office parks; and sometimes for residential use. Some industrial land may have considerable areas of exposed soil, while much commercial land is almost or completely paved or covered in buildings. This results in quite a wide range of potential exposures. Workers in factories or commercial buildings will have little, if any, direct exposure to soil, but may have exposure to volatiles.

The US EPA, in developing its soil screening levels, differentiated between an indoor worker, who spent most of his or her time indoors with little soil exposure, and an outdoor worker involved in outside maintenance activities (effectively the site caretaker). This approach has some attraction. Existing New Zealand guidelines have differentiated between paved and unpaved, but have not discussed how the paved scenario might apply to indoor workers. The industrial paved scenario implies virtually no exposure, even to volatiles, except where soil was exposed during excavation. However, the infrequency of subsurface maintenance suggests maintenance should not be a controlling factor in a commercial or industrial scenario.

The revisions to the Dutch CSOIL 2000 model (reported in Brand et al, 2007) put industry, infrastructure (which includes roads) and buildings all in the same category, with the same contact frequency and soil ingestion rates as the recreation scenario – in effect acknowledging the common low exposure of these land uses. The UK rejects soil exposure on fully paved sites as implausible, reserving the commercial / industrial scenario to indoor workers in single-storey buildings such as factories and warehouses.

Canadians separate the commercial and industrial scenarios (CCME, 2006), to differentiate between where activity is primarily commercial (eg, a shopping mall) and industrial, which is specifically for production or manufacturing of goods. It would appear the Canadian industrial scenario is for an unpaved site, which would make it equivalent to the existing commercial / industrial unpaved scenario. New Zealand commercial / industrial paved could then be considered akin to the Canadian commercial scenario, with low exposure.

Cavanagh (2004a) suggested that the paved commercial land-use scenario (eg, shopping malls, retail shops), for which there is no exposure to surface soil, and an unpaved industrial land use in which buildings (eg, factories) are located within an otherwise unpaved site, provide useful generic descriptions of common industrial land uses. However, Cavanagh rejected the paved commercial scenario as not providing useful values for metals and semi-volatile contaminants, and proposed the unpaved industrial scenario as covering all commercial / industrial scenarios. However, this is overly conservative for the commonly encountered indoor commercial and industrial situations. The lack of direct soil exposure with which to derive values for metals and semi-volatiles (which will have no concentration limit) does not invalidate such a scenario. It is therefore proposed that two commercial / industrial scenarios be continued with, but that they are renamed and redefined as follows:

Commercial / industrial (indoor worker) represents factory workers on commercial or industrial sites with little exposed soil, where workers spend the majority of their time indoors carrying out relatively low-intensity tasks. Direct exposure to soil is limited or zero but exposure to volatiles migrating to indoor air is possible. There would be no concentration limit for most metals and semi-volatile organics, given the protection by concrete floors.

Nevertheless potentially contaminated sites that are being redeveloped for commercial or industrial use should be investigated. While there may be little or no risk in the long term, there will be some risk to workers during construction (see next paragraph); and potential risk to the wider environment from disposal of surplus soil generated during redevelopment. Disposal must be guided by sampling results.

Commercial / industrial (outdoor worker or unpaved) represents the outdoor worker who carries out maintenance activities involving soil exposure to surface or near-surface soil through gardening and other landscaping activities, and occasional shallow excavation for routine underground service maintenance activities. Exposure to soil is less intensive and/or less frequent than would occur during construction or extensive excavation works, but occurs over a longer period.

A separate case is exposure to volatiles outdoors on an unpaved site, where workers not engaged in activities likely to incur direct soil exposure, may be exposed to vapours in going about their general work. Such exposure could be for most days at work, which is more frequent that that envisaged for direct soil exposure for an outdoor worker. The current 'Oil Industry Guidelines' (MfE, 1999) suggest that this scenario is unlikely to be critical, as the guideline values are generally high and concentrations exceeding the guideline values over wide areas (the assumption behind the derivation of the values) are not likely to occur in practice for most situations. The few situations where exposure could be significant, eg, major petroleum or chemical installations, are candidates for site-specific assessment. For completeness, this scenario should be considered on a chemical-specific case-by-case basis when guidelines for volatiles are reviewed.

3.2.4 Scenarios not adopted

Maintenance / excavation scenario

Existing New Zealand guidelines have a scenario for maintenance / excavation workers, to deal with the greater exposure to soil contaminants than for ordinary outdoor workers. Cavanagh (2004a) recommended this scenario as applicable to subsurface maintenance works and construction activities, being applicable for all land uses, including the assessment of commercial (completely paved) land. This scenario was also noted as the only scenario relevant for soil contamination at depth.

The exposure parameters for the current New Zealand maintenance / excavation scenario are unrealistic. The typical commercial / industrial site simply does not get dug up on 50 occasions each year, every year for 20 years, involving the same personnel. Even if excavations were carried out on a number of occasions on a site, such excavation would typically be by contractors using different personnel. Therefore it can be assumed that exposure of an individual would be no more than a few occasions per year, suggesting the current guidelines are conservative by a factor of perhaps 10 for threshold substances – and much more than that for non-threshold substances, for which the duration affects the final value.

The Technical Review Group agreed after lengthy discussion that such a scenario should not be part of the NES (MfE, 2005). Sites would not be cleaned up to this standard. The Review Group considered it was more appropriate that exposure be limited through the site-specific controls that are required under health and safety legislation. This is similar to the UK, where maintenance / excavation activities are considered to be covered by occupational health protection legislation (Defra and EA, 2002b).

Under New Zealand legislation, when a site is known to be contaminated there is an onus on the employer to be aware of the potential hazard. The Health and Safety in Employment (HSE) Act (1992) is intended to protect the safety of individual workers and requires the employer (and individuals) to take steps to identify and eliminate, isolate or minimise hazards. Carried out properly, this will reduce the exposure of excavation workers to acceptable limits. In any case, most excavations will be of short enough duration that exposure will not be great, and if the ground is particularly contaminated (eg, gasworks waste or leaking underground storage tanks) it is often sufficiently obvious that workers and supervisors could be expected to notice and take precautions.

Particular individuals might, on occasion, be exposed to contaminated soil on more than one site as part of their work, but it is not reasonable to base the assessment of soil on all commercial / industrial sites on this relatively rare, and person-specific, situation. Most sites are not significantly contaminated and most workers would not move from one contaminated site to another. Again, the HSE Act is intended to protect the safety of the individual workers and provides the appropriate approach here.

There is a small subset of workers involved in specialist maintenance and soil removal tasks involving site contamination. Given the limited duration and frequency of most excavation, exposure to volatiles or liquid contaminants is generally the most likely risk-creating scenario. A particular case is redevelopment and re-tanking of service station sites, where exposure to volatile compounds is common. This is already dealt with by employers under the HSE Act by the writing of safety plans and compliance with confined-space regulations which require the measurement of vapours. Such measures are routinely carried out during tank pit excavations. Generic soil guideline values are of little use in such situations, given the wide variety of soil conditions and site circumstances.

In summary, it is proposed to dispense with the commercial / industrial excavation worker / maintenance scenario, and leave this situation to health and safety legislation.

Agricultural scenario

The'Timber Treatment Guidelines' (MfE and MoH, 1997), the 'Gasworks Guidelines' (MfE, 1997) and 'Oil Industry Guidelines' (MfE, 1999) include an agricultural / horticultural land use scenario to protect the general public from concentrations of contaminants in produce that would pose a concern to public health. The scenario also protects the health of residents at any farm property from exposure via consumption of home grown livestock and produce, and through direct contact with contaminated soil. However, the more recent 'Sheep Dip Guidelines' (MfE, 2006a) dropped the agricultural scenario in favour of the lifestyle block scenario which removed consideration of protecting the productive capacity of land and exceeding the maximum residue levels in food.

The rural residential / lifestyle block scenario, proposed in this document, is intended to be applicable to the immediate vicinity of the farmhouse or staff houses with the intention of protecting the health of the farming family and any staff and their families.

The rural residential scenario is therefore not intended to cover the commercially productive areas of agricultural land. It is considered that the commercial uses of farm properties are either outside the scope of this proposal (ie, not directly related to effects on human health) or are dealt with by other legislation protecting public health:

- Plant growth and the health of soil micro organisms, while beneficial to maintaining productive capacity, are not directly related to human health effects.
- The safety of food produced for the general public is subject to the joint New Zealand Australian Food Standards. Testing under this jurisdiction is a more direct measure of determining whether this land is safe for human use.
- Farm worker's exposure is subject to the provisions of the Health Safety and Employment Act 1992. The requirements of an employer under this legislation are more fully discussed under the maintenance and excavation scenario described above.

In summary, it is proposed to dispense with the agricultural scenario, since the protection of farming families is addressed by the rural residential / lifestyle block scenario. The considerations relating to the productive parts of agricultural land being left to the food safety and health and safety legislation.

Other possible scenarios

From time to time, schools and/or childcare facilities raise another generic scenario that has been the subject of discussion (MfE, 2005). Few international protocols specifically mention this land use, and those that do typically include it with a residential scenario (eg, NEPC, 1999a). However, typical exposure frequencies and durations are less than for a residential scenario; produce consumption does not generally apply (or will not apply at the rates assumed for the residential scenario – daily produce consumption – even if the school or kindergarten has a vegetable garden for demonstration or other purposes).

Schools are typically required to be open for between 190 and 200 days, considerably less than the 350-day exposure frequency for the residential scenario in existing New Zealand guidelines. Also, a pupil is usually indoors for most of the day and there is typically a smaller proportion of exposed soil in a school, compared with a residential site, suggesting less opportunity for exposure by direct contact. While duration is not relevant for threshold substances, the duration for non-threshold substances of 30 years for residential exposure is greater than would be expected for the school scenario. The body weight typically assumed for childhood exposure (typically 13–15 kg, depending on jurisdiction) is clearly too small for the average school child, being increasingly conservative as the child grows.

The greatest exposure is likely for a childcare centre, where attendance may be every day that the parent goes to work (say up to 250 days a year) from as early as being a toddler (ie, the 13–15 kg body weight applies) and could involve more outdoor play where soil exposure might occur, than a typical school situation. On this basis a child at a childcare centre is probably at least twice more exposed (on a weight-normalised basis) than a child just starting primary school (assumed 20 kg body weight) and at least five times more exposed than a child just entering high school (assumed 50 kg body weight), without fully accounting for differences in the proportion of exposed soil and opportunity for contact in the two sorts of facilities.

Cavanagh (2004a) suggested that an alternative to providing a school or childcare facility scenario as a generic scenario would be to provide explicit information (eg, appropriate exposure frequencies and duration) to enable the ready derivation of a site-specific value. The Technical Review Group suggested that early childhood centres should be included under either the residential or high-density residential land uses, depending on how much paving the site had (while acknowledging that the residential land-use scenario includes produce consumption), but that calculations should be performed to determine the best fit.

Given the wide range of situations that a school or childcare scenario has to cover – early childhood, primary school, secondary school – with a wide range of body weights and potential exposure, a single generic scenario is not considered feasible. Instead, apart from the proposed use of the parks / recreation scenario for secondary school playing fields (section 3.2.2), a site-specific risk assessment is recommended as the appropriate approach for schools and childcare centres. However, providing there is no significant growing and consumption of vegetables (ie, a typical child is not receiving 10 per cent or more of its daily vegetable intake from site-grown vegetables), residential no-produce guidelines could be used as a conservative initial screening. Alternatively, the parks / recreational scenario could be used as an initial screening value for areas of secondary and primary schools other than playing fields, if justified on a case-by-case basis. Generally the exposure parameters for the parks / recreational scenario would be conservative for general areas of schools. In the small number of childcare centres where there is significant site-grown vegetable consumption, the standard residential guideline may be used as a conservative initial screening value.

Non-produce residential guideline values have been calculated (see detailed calculations in section 6) to facilitate site-specific assessment such as for schools, but it should be noted that non-produce residential scenarios are not part of the generic exposure scenarios and should not be used in other than site-specific assessment.

Groundwater

Groundwater is widely used in some areas of New Zealand for drinking-water, irrigation and stock water. Groundwater also discharges to the aquatic environment of surface water. Groundwater is included in the derivation of soil guideline values in some overseas jurisdictions, and soil guideline values protective of groundwater for human consumption are derived in the 'Oil Industry Guidelines' (MfE, 1999). These values are considered separately from values derived from soil ingestion, inhalation, produce ingestion and dermal exposure.

The development of soil guidelines for the protection of groundwater is currently beyond the scope of this document. It requires modelling the partitioning of the contaminant from soil to groundwater, and requires assumptions about soil type, area, depth and thickness of contamination, and hydrogeological properties of the underlying aquifer. Apart from the large variation in the way contaminated sites are contaminated, New Zealand is so varied geologically that selection of sensible generic parameters is a difficult task.

The consideration of groundwater contamination should be treated as a site-specific issue. If significant contamination is found on a site, and if groundwater is used there for consumption locally, then monitoring wells should be installed to measure groundwater concentrations directly.

4 Derivation Equations

4.1 General

All the jurisdictions reviewed for this report use essentially the same basic exposure equations, with variations in the:

- specific way they are applied
- subsidiary equations required to calculate indirect exposure from inhalation of particulates and volatiles, etc
- detail of the exposure scenarios and pathways that contribute to those scenarios.

Soil guideline values are based on assessing the intake of a particular contaminant and exposure pathway. Generic equations have been developed in section 2.1.1. In summary, for each pathway i the $SGV_{(health)}$ is:

$$\text{Soil guideline value}_i = \frac{\text{accceptable intake}_i \times \text{body weight} \times \text{averaging time}}{\text{contact rate}_i \times \text{exposure frequency} \times \text{exposure duration}}$$

and the combined soil guideline value is obtained from:

$$\text{Combined soil guideline value} = \frac{1}{\left(\frac{1}{\text{ingestion soil guideline}} + \frac{1}{\text{dermal soil guideline}} + \frac{1}{\text{produce soil guideline}} + \cdots\right)}$$

Existing New Zealand guidelines use the following exposure pathways:

- soil ingestion
- produce consumption (residential scenarios only)
- dermal exposure
- inhalation of particulates
- inhalation of volatiles.

Only the first three pathways are considered in this document. Inhalation of particulates is a minor pathway and can be dispensed with unless the toxicity of the contaminant of concern via the inhalation route is very much greater than the oral route. The particulate inhalation pathway typically contributes considerably less than one per cent to the total exposure, and is therefore well within the uncertainty of calculation of the other pathways; hence it can be safely ignored. It should be checked where the inhalation toxicity is much higher than the oral toxicity, or where site-specific conditions suggest dust is an unusually large contributor.

While inhalation of volatiles is important, it is beyond the scope of the current document. This issue is discussed further in section 4.7.2.

Some jurisdictions consider other exposure pathways in their generic derivations. However, in the New Zealand context it has been decided that for inorganic and semi-volatile contaminants, the three chosen pathways form the great majority of exposure for typical situations.

The two receptor groups adopted are the same as those used in existing New Zealand guidelines (eg, MfE and MoH, 1997). This follows on from US EPA practice. The age-groups are 1–6 years and 7–30 years. The 1–6-year-old child will be the critical receptor for non-threshold substances for residential and recreational scenarios. An 'adult', 7–30 years, will be the critical receptor for worker scenarios.

Following existing practice, exposure is combined across age groups for non-threshold substances, in the form of age-adjusted contact rates.

4.2 Background exposure

It is common in some jurisdictions to subtract background exposure from the reference health standard and using the residual to calculate the soil guideline value. This is only applied to threshold substances, because intakes of non-threshold contaminants are considered on the basis of an increase in risk, which is irrespective of background exposure. Not subtracting background exposure for threshold substances would theoretically permit exposure in excess of the reference health standard (RHS) at soil concentrations equal to or slightly less than the SGV

Some existing New Zealand SGV derivations follow the practice of subtracting the background whereas others do not. It has been determined (as reported in MfE, 2010b) that an allowance for background exposure should be made for deriving New Zealand soil guideline values for threshold substances. MfE (2010b) provides details on how background exposure should be determined. This report also provides recommended values for background exposure for the priority contaminants considered in this document.

It is possible for background exposure to exceed the RHS, in which case an SGV cannot be derived. The adopted approach around this problem is a variation of that adopted in the UK. In the SGV derivation protocol for the UK (EA, 2008a), when background exposure comprises greater than 50 per cent of the RHS then the background exposure is taken to be 50 per cent of the RHS.[2] Rather than using a fixed percentage when background is greater than 50 per cent of the RHS, the method adopted in this document is for the Toxicological Advisory Group of government toxicologists to consider the proportion allocated to exposure from soil on a case-by-case basis (MfE, 2010b). Further, in cases where background exposure is negligible (defined as less than 5 per cent of the RHS) or no data on background exposure exists, then a maximum of 95 per cent of the RHS should be allocated to exposure from soil. This is to provide a slight degree of precaution for substances for which determining the background exposure may be problematic (MfE, 2010b).

2 Until the recent updating of the UK protocol, the requirement was 20 per cent of the tolerable daily intake (TDI) (Defra and EA, 2002b) when the background intake exceeded 80 per cent of the TDI. This was changed as it was considered to result in excessively low SGVs.

4.3 SGV$_{(health)}$ derivation equations for each pathway

The main SGV$_{(health)}$ derivation equations for each exposure pathway are presented below. Exposure parameters values for each equation are provided in section 5. Common terms are listed below:

SGV_i	=	soil guideline value for pathway i (mg/day)
RHS	=	contaminant-specific reference health standard (mg/kg BW/day)
BI	=	background intake (mg/kg BW/day)
ED	=	exposure duration (years)
EF	=	exposure frequency (days/year)
AT	=	averaging time ED × 365 days for a threshold substance
	=	lifetime (75 years) × 365 = 27,375 days for non-threshold substance
BW	=	body weight (kg).

4.4 Soil ingestion

Threshold substance

$$SGV_{ing} = \frac{(RHS - BI) \times BW \times AT \times 10^6}{IR \times EF \times ED} \text{ mg/kg} \qquad \text{Eqn 6}$$

where: IR = soil ingestion rate (mg/day)

As AT = ED × 365, this reduces to:

$$SGV_{ing} = \frac{(RHS - BI) \times BW \times 365 \times 10^6}{IR \times EF} \text{ mg/kg} \qquad \text{Eqn 7}$$

Non-threshold substance

$$SGV_{ing} = \frac{RHS \times 27375 \times 10^6}{IR_{adj} \times EF} \text{ mg/kg} \qquad \text{Eqn 8}$$

with IR_{adj} being represented by:

$$IR_{adj} = \sum \frac{IR_i \times ED_i}{BW_i} \qquad \text{Eqn 9}$$

where:

IR_{adj}	=	the age-adjusted soil ingestion rate (mg year/kg day)
$\sum$		signifies summation over receptor groups i = 1 to n
IR_i	=	soil ingestion rate for receptor group i (mg/day)
BW_i	=	body weight for receptor group i (kg).

4.5 Dermal absorption

Threshold substance

$$SGV_d = \frac{(RHS - BI) \times BW \times AT \times 10^6}{AR \times AH \times AF \times EF \times ED \times EV} \text{ mg/kg} \qquad \text{Eqn 10}$$

where: AR = skin area of relevant exposed parts of the body (cm^2)
AH = soil adherence factor (mg/cm^2 – event)
AF = chemical specific dermal absorption factor
EV = events/day.

As AT = ED × 365 and for the default of one event per day, this reduces to:

$$SGV_d = \frac{(RHS - BI) \times BW \times 365 \times 10^6}{AR \times AH \times AF \times EF} \text{ mg/kg} \qquad \text{Eqn 11}$$

Non-threshold substance

$$SGV_d = \frac{RHS \times 27{,}375 \times 10^6}{AD_{adj} \times AF \times EF} \text{ mg/kg} \qquad \text{Eqn 12}$$

AD_{adj} being represented by: $AD_{adj} = \sum \frac{AR_i \times AH_i \times ED_i}{BW_i}$ Eqn 13

where: AD_{adj} = the age-adjusted dermal absorption factor (mg year/kg)
$\sum$ signifies summation over receptor groups i = 1 to n
AR_i = skin area of relevant exposed parts of the body for receptor group i (cm^2)
AH_i = soil adherence factor for receptor group i (mg/cm^2 event)
BW_i = body weight for receptor group i (kg).

The approach to dermal absorption is that of the US EPA (2001a; 2002a), which superseded US EPA's original approach used in the existing New Zealand guidelines. The approach is based on absorption per event, rather than being exposed for some proportion of the day, with the assumption that the soil adheres to the skin long enough for the contaminant to be absorbed into the body.[3] The soil adherence factor varies according to the type of exposure or activity, and varies according to the body part exposed. A single body-part-area-weighted adherence factor is used. The default is for there to be one dermal absorption event per day for residential or outdoor work situations. The equations are written for a single event.

[3] This assumption may not accord with the way the dermal absorption factor is derived. Some dermal absorption factors are derived on the basis of 24-hour exposure. Few people would not wash for 24 hours, therefore a 24-hour dermal exposure factor should be adjusted by a factor to take into account the period between washing. For the residential situation this is taken as 12 hours and for the occupational situation this is taken as 8 hours (from the *Sheep-dip Guide:* MfE, 2006a).

4.6 Produce ingestion

The equations for produce ingestion depend on the types of produce considered. As noted in section 5.4.2, only vegetable produce is considered and this has been divided into above-ground (leafy) and below-ground (roots and tubers), as contaminant uptake may be different for the different types of vegetable.

Threshold substance

$$SGV_P = \frac{(RHS - BI) \times BW \times 365}{IP \times Pg \times EF\ [(BCF_{root} + SL_{root}) \times p_{root} + (BCF_{tuber} + SL_{tuber}) \times p_{tuber} + (BCF_{leafy} + SL_{leafy}) \times p_{leafy}]} \qquad \text{Eqn 14}$$

where:
- IP = produce ingestion rate (kg dry weight/day)
- P_g = proportion of total daily produce consumption that is home-grown produce (dimensionless: no units)
- BCF = contaminant-specific bioconcentration factor (dry weight) (dimensionless)
- SL = produce-type-specific soil loading factor (dry weight) for soil attached to produce (no units)
- p = the proportion of total daily produce consumption for each produce type (dimensionless)

the subscripts leafy, root or tuber refer to above-ground edible vegetation and below ground roots (eg, carrots) and tubers (eg, potatoes), respectively.

As AT = ED × 365, this reduces to:

$$SGV_P = \frac{(RHS - BI) \times BW \times 365}{IP \times Pg \times EF\ [(BCF_{root} + SL_{root}) \times p_{root} + (BCF_{tuber} + SL_{tuber}) \times p_{tuber} + (BCF_{leafy} + SL_{leafy}) \times p_{leafy}]} \qquad \text{Eqn 15}$$

Where BCFs are determined from empirical data (eg, for metals) the soil loading factor is zero as the measured BCFs will include attached soil. Where BCFs are derived theoretically, SL is taken to be 0.001 for roots and tubers (dry weight) and 0.0002 for leafy vegetables, following EA (2008a).

Where no distinction is made between the type of vegetable, equation 10 reduces to:

$$SGV_P = \frac{(RHS - BI) \times BW \times 365}{IP \times Pg \times EF \times (BCF + SL)} \text{ mg/kg} \qquad \text{Eqn 16}$$

and SL = 0 or 0.0008, as appropriate, the latter calculated as a weighted average, assuming dry weight daily consumption for a child of 7.6 grams of root and tuber vegetable and 2.9 grams of leafy vegetables, respectively (see section A3). The adult ratio of root to leaf consumption is sufficiently similar that the same average SL can be used.

Non-threshold substance

$$SGV_P = \frac{RHS \times 27375}{IP_{adj} \times Pg \times EF\ [(BCF_{root} + SL_{root}) \times p_{root} + (BCF_{tuber} + SL_{tuber}) \times p_{tuber} + (BCF_{leafy} + SL_{leafy}) \times p_{leafy}]} \text{ mg/kg} \qquad \text{Eqn 17}$$

or where there is no vegetable type distinction:

$$SGV_P = \frac{RHS \times 27375}{IP_{adj} \times Pg \times EF \times (BCF + SL)} \text{ mg/kg} \qquad \text{Eqn 18}$$

with IP_{adj} being represented by: $IP_{adj} = \sum \frac{IP_i \times ED_i}{BW_i}$ Eqn 19

where: IP_{adj} = the age adjusted produce ingestion rate (kg year/kg day)
$\sum$ signifies summation over receptor groups i = 1 to n
IP_i = produce ingestion rate for receptor group i (kg dry weight/day)
BW_i = body weight for receptor group i (kg)

Specific exceptions in the use of the produce pathway equations exist for some contaminants, notably copper and boron. In the case of copper, at soil concentrations that do not constitute a health human risk, applying a BCF would theoretically result in plant tissue concentrations that would kill the plant, meaning it could not be harvested. This being the case, the use of the produce pathway equations is not appropriate. Instead, a maximum tissue concentration is assumed and an additional notional background intake subtracted from the reference health standard on the assumption that the produce is consumed at that concentration. A similar approach has been adopted for boron because reliable bioaccumulation factors could not be derived.

The details are provided in sections 6.2 (boron) and 6.5 (copper).

4.7 Inhalation

In this document, the inhalation pathway has not been used in calculating the $SGVs_{(health)}$. The equations are given here for completeness.

4.7.1 Contaminated particulates

Contaminated airborne particulates may be generated from bare soil on a contaminated site and be inhaled by residents or workers. As discussed previously, this pathway is minor and will not normally be a component of a generic soil guideline value. Given the minor role the particulate inhalation pathway plays, no differentiation between the contaminant concentration of indoor dust and outdoor dust is proposed (unlike the CLEA model for example: EA, 2008a).

Threshold substance

$$SGV_{ih} = \frac{(RHS\text{-}BI) \times BW \times PEF \times AT}{IH \times EF \times R \times ED} \text{ mg/kg}$$ Eqn 20

where: IH = inhalation rate (m^3/day)
R = proportion retained in lungs (dimensionless)
PEF = particle emission factor[4] (m^3/kg)

[4] The PEF is equivalent to the inverse of the proportion of the respirable (<10 micron) dust concentration coming from a contaminated source.

Non-threshold substance

$$SGV_{ih} = \frac{RHS \times PEF \times 27375}{IH_{adj} \times EF \times R} \quad \text{mg/kg} \qquad \text{Eqn 21}$$

with IR_{adj} being represented by:

$$IH_{adj} = \sum \frac{IH_i \times ED_i}{BW_i} \qquad \text{Eqn 22}$$

where:
- IH_{adj} = the age adjusted inhalation rate (m^3 year/kg day)
- $\sum$ signifies summation over receptor groups i = 1 to n
- IH_i = inhalation rate for receptor group i (m^3/day)
- BW_i = body weight for receptor group i (kg)

For some exposure scenarios (eg, occupational) a person will only be exposed for part of the day. This can be treated in two ways in the equations, either the inhalation rates are adjusted to reflect the shorter exposed period, or daily rates are used with the addition of a factor to reflect the proportion of the day spent exposed (eg, eight hours out of 24).

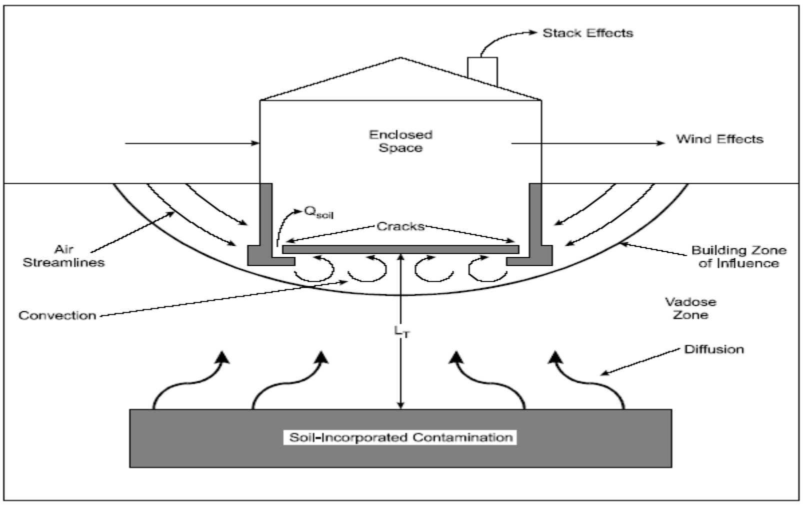
4.7.2 Volatiles inhalation

Exposure by inhalation of volatile organics presents a particular difficulty because the risk occurs indirectly from breathing vapours that have partitioned from contaminated soil at some depth below the surface. Figure 1 is a conceptual diagram of this process.

An equation similar to equation 11 can simply be written by replacing the concentration of contaminated particles being inhaled, with a vapour concentration. Two situations must be considered: volatilisation to outdoor air and volatilisation to indoor air.

Calculating soil guideline values for outdoor air inhalation is relatively straightforward. The vapour concentration to be put into the equation can be determined by applying a volatilisation factor to the soil concentrations. The volatilisation factor depends on generic soil properties, depth of contamination, contaminant-specific partition coefficients, diffusivity in air, and Henry's Law coefficients. Volatilisation factors can be calculated using the model by Jury et al (1983, 1984, 1990). Equations are presented in US EPA (2002a) with input factors for those equations. These factors, some of which are specific to US meteorological and geological conditions, can probably be applied to New Zealand conditions, if specific values are carefully selected.

Figure 1: Pathway for subsurface vapour intrusion into indoor air



Source: US EPA, 2004b.

Volatilisation to indoor air is a different proposition, however. There seems little point in calculating soil guideline values for outdoor inhalation without also addressing the more critical indoor air situation.

Volatilisation to indoor air requires modelling the migration of the vapour from the contaminant source, up through the soil and then through cracks or gaps in the building to the inside of the building. This has typically been carried out using a simple vapour migration model such as the Johnson and Ettinger (1991) model.

However, the vapour concentration is a non-linear function of the soil concentration, which means the equation must be solved for the soil concentration (the soil guideline value) in an iterative fashion. In other words, it is not just a matter of rearranging the equations to solve for the soil concentration, as it is with the other pathways. A further difficulty is that the vapour migration models have tended to be inaccurate, often by several orders of magnitude.

Davis et al (2004) provide a useful summary of some of the factors influencing vapour migration, each of which can contribute to inaccuracies if not model inadequacy:

- volume and location of the contamination relative to the surface and the groundwater table
- volatilisation and portioning from soil and groundwater
- diffusion
- sorption onto organic matter in the soil
- biodegradation
- soil properties such as soil moisture and permeability
- soil stratigraphy and layering
- temperature and barometric effects
- pressure effects due to wind
- density differences.

In addition, where migration into buildings is concerned, the design and configuration of the building has a large influence: for example whether it has a basement, is slab-on-grade, or is on piles with a crawl space. To date in New Zealand, only the slab-on-grade scenario has been considered. Yet houses with crawl spaces are just as common, if not more so. Attempting to model these two types of construction is beyond the scope of the present study and has proved difficult elsewhere.

As noted earlier, the UK Environment Agency (Evans et al, 2002) has reviewed a number of vapour migration models for use with both slab-on-grade and crawl space construction. They concluded that none of the models was sufficiently accurate to be recommended. They proposed further work on the issue.

Australia's Commonwealth Scientific and Industrial Research Organisation (CSIRO) has also reviewed vapour models with particular reference to Australian construction (Davis et al, 2004), which is not dissimilar to New Zealand's. CSIRO initially concluded that there were unacceptably large discrepancies between the models, with a lack of validation against field measurements. However, Davis et al (2008) have re-examined the Johnson and Ettinger (1991) model and recommended a modified form of it (not yet generally available) as the most suitable for Australian conditions when generating health screening levels for petroleum hydrocarbons, including for house with crawl space construction.

It is recommended that the Australian and UK work be examined as to its applicability for the New Zealand situation.

5 Exposure Parameters

5.1 General

The parameters used in the derivation of existing New Zealand soil guideline values are primarily based on US EPA data and some Australian data (eg, produce consumption). Some of these parameters have been updated since the guidelines were derived, for example, the US EPA has updated its approach to dermal exposure (US EPA, 2001a), further research has been carried out on soil ingestion (reviewed in US EPA, 2006a; Paustenbach et al, 2006; Van Holderbeke et al, 2007), and recent studies have become available on produce uptake (EA, 2006; Swartjes et al, 2007). In addition, information that is relevant to New Zealand is available and could be used instead of international data (eg, Russell et al, 1999). It is therefore appropriate to review the parameters currently in use.

In this report the parameters used to estimate exposure are divided into the following parameters:

- general exposure
- pathway-specific
- contaminant-specific (as a subset of pathway-specific).

General exposure parameters are dependent on the relevant receptors and the exposure scenario, but are independent of the pathway of exposure: they are common to all pathways. These include exposure frequencies, durations, averaging times and body weights.

Pathway-specific parameters are of two types. The first type is related to the type of receptor and activity (land use) in defining rates of exposure, eg, rates of soil ingestion, inhalation and produce consumption. The second type is dependent on the specific chemical and defines uptake rates, whether absorption rates through the skin or uptake into plants. These depend on the physico-chemical properties of the individual contaminants.

This section provides recommendations for the general exposure parameters, and then parameters used in individual exposure pathways.

5.2 Degree of conservatism

In selecting the appropriate values of individual parameters to derive guideline values, the degree of conservatism inherent in the individual parameters as well as in the derivation process, must be considered. This will ensure that the derived guideline values are not over- or under-conservative. Parameter values used internationally are influenced by different policy approaches.

As noted earlier, the US EPA (from which many of New Zealand's current parameter values are obtained) adopts a 'reasonable maximum exposure' approach. This is nominally aimed at providing a reasonable worst-case exposure scenario (that is, no more than 5 to 10 per cent of the population would be likely to exceed these exposures) and is based on a combination of average and upper-end exposures.

The US EPA has been criticised for being too conservative (US EPA, 2004a) because of combining too many upper-end estimates but has defended this approach as appropriate. Similar criticism has been levelled at the UK approach (Defra, 2006a), and the derivation process and associated input parameters were recently reviewed (Defra, 2008a; EA, 2008a).

Most countries use high-end estimates for exposure duration (this is only relevant for non-threshold substances because exposure duration drops out for threshold substances). High-end estimates are also used for exposure frequency, but average estimates are used for body weight, skin areas and inhalation rates. This is the existing New Zealand practice, based on US EPA practice and it is proposed to continue this.

Given reference health values also tend to be conservative (with some exceptions): the combination of high-end durations (for non-threshold substances) and frequencies should ensure adequate conservatism. Soil guideline values will frequently be used as clean-up values when the cost of site-specific assessment cannot be justified. Over-conservative generic SGVs could result in unnecessary remediation (with consequent cost: turning useful resource into a waste and unnecessarily using up landfill space if the remediation involved ‘dig and dump’) or in unnecessary abandonment of projects, when the human health risk was actually acceptable. It therefore proposed that the remaining parameters be central estimates to avoid over-conservatism. Many of these parameters are contaminant-specific, and because of varying properties in different soils, have a wide range of possible values. There is danger that choosing a central estimate will, for some particular site conditions, be non-conservative. Accordingly, use of central estimates for contaminant-specific parameters must be used with care.

5.3 General exposure parameters

The general exposure parameters include age ranges and body weights for those age ranges. These are dependent on the receptors considered important for a particular scenario. Averaging time, exposure frequency and exposure duration are scenario-dependent and in some cases also receptor-dependent.

General exposure parameters are required for children in all residential scenarios (lifestyle block, standard residential, high-density residential) and the parks and recreational scenario. General exposure parameters for adults are also required for all residential scenarios and the parks and recreation scenarios, and in addition for the commercial / industrial scenarios.

For children, the parameter values are primarily based on the age range over which soil and dust ingestion via inadvertent mouthing of non-food objects is important. Australian values are based on a two-year-old child, Canadian values are based on a toddler (six months to four years), whereas the US EPA (and New Zealand) values are nominally based on a child aged one to six years. In reality, the 15 kg body weight currently adopted in New Zealand and the United States, suggests a two to three-year-old child.

As noted earlier, for threshold contaminants the exposure duration is not important (other than being long enough for the exposure to be considered chronic) as it occurs in both numerator and denominator of the derivation equations, thus cancels. The most important general parameters are body weight, which is directly related to age in the case of children, and exposure frequency. However, for non-threshold contaminants, the defined childhood and adult exposure periods are used to calculate age-adjusted parameters in determining lifetime-average doses. In these cases, childhood parameters are used for the defined age range for a child, and adult parameters are used for the remainder of exposure duration.

Combining high-end estimates for exposure frequency and duration has the risk of greater conservatism for non-threshold substances than for an equivalent threshold calculation. However, given that the exposure duration is less than the 75-year[5] averaging time for a non-threshold substance, the potential over-conservatism is somewhat mitigated.

A summary of the age ranges and body weights of receptors used in existing national and international derivation protocols for deriving soil numeric values is given in Table 3.

Table 3: Age ranges and body weights of receptors considered in national and international derivation protocols

Country	New Zealand	Australia	US	Canada	Netherlands
Receptor age (years): child adult	 1–6 7–30	 2-year-old	 1–6 7–30	 6 months– 4 years	 0–6 7–70
Body weight (kg): child adult	 15 70	 13.2 na	 15 70	 13 70	 15 70

na = not applicable.

5.3.1 Body weight

The difference in the selected age range of concern for children is primarily reflected in the variable body weights used in different countries, which range from 13 to 15 kg in table 3. Fifteen kilograms is the approximate weight of a three-year-old child, or the average of a zero to six-year-old based on data provided in the *Exposure Factors Handbook* (US EPA, 1997), whereas a body weight of 13 kg is used for a two-year-old child. The difference in the body weight of a child applied internationally would give rise to a variation of about 5–15 per cent in a derived value for a threshold contaminant based on soil ingestion, if all other parameter values were identical. It is proposed to continue using a 15-kg body weight for a child on the basis that a younger child is likely to be less frequently exposed to outdoor activity that would result in soil ingestion or dermal exposure.

For adults, a body weight of 70 kg is typically used by all countries, with the exception of the UK (which bases the adult receptor on a female and uses a distribution with a mean weight of 68.5 kg). Other than the UK protocol, there seems to be little use of country-specific data for body weight. The rationale for selecting 70 kg as adult body weight appears to be that it provides a reasonable approximation of average adult body weight, as opposed to being based on more specific data. For example, the *Exposure Factors Handbook* recommends that a body weight of 71.8 kg be used in the US. This is derived as a mean of the average male body weight of 78.1 kg and the average female body weight of 65.4 kg based on data collected over 1976–1980 (US EPA, 1997).

The 1997 New Zealand National Nutrition Survey (Russell et al, 1999) established an average body weight for New Zealanders of 74.5 kg, based on the average of the average male and female body weights of 80.7 kg and 68.7 kg, respectively. These weights are for people aged 15 years and over, and would therefore over-estimate an average ‘adult’ weight from age seven. A body weight of 70 kg is used by the Ministry of Health for setting the *Drinking-water Standards* (MoH, 2008) and by the New Zealand Food Safety Authority for setting maximum residue limits in different foodstuffs.

5 Increased from the current 70 years, see section 5.3.2.

The Toxicological Advisory Group decided to continue using an adult body weight of 70 kg for consistency with other New Zealand health guideline setting authorities. While this is conservative (results in lower soil guideline values) for the average adult body weight, it is justified on the basis that it is close to the average female adult weight.

5.3.2 Averaging time

The averaging time selected depends on the type of toxic effect being assessed. When evaluating exposures to developmental toxicants, intakes are calculated by averaging over the exposure event (eg, a day or a single exposure incident). For acute toxicants, intakes are calculated by averaging over the shortest exposure period that could produce an effect, usually an exposure event or a day. When evaluating longer-term exposure to threshold toxicants, intakes are typically calculated by averaging intakes over the period of exposure (ie, subchronic or chronic daily intakes). For non-threshold toxicants, intakes are calculated by pro-rating the total cumulative dose over a lifetime (ie, chronic daily intakes, also called lifetime average daily intake). This approach for carcinogens is based on the assumption that a high dose received over a short period of time is equivalent to a corresponding low dose spread over a lifetime (US EPA, 1989a).

To this point the New Zealand guidelines have adopted the US EPA approach. For threshold contaminants the averaging time (in days) is typically the exposure duration (in years) for the critical receptor multiplied by 365, the number of days in a year. For the residential setting the critical receptor is typically a child. Given that exposure duration cancels out in the exposure equations for threshold substances, the time dependence of the exposure reduces to the proportion of the year exposed (ratio of exposure frequency and number of days in a year). However, averaging time is important for non-threshold contaminants if the US EPA approach to non-threshold contaminants is adopted. The convention is almost universally to use an averaging time of a 70-year lifetime, expressed as days, resulting in an estimate of exposure as an annual average daily rate. An exception is the Dutch protocol, which uses an averaging time of 70 years for all contaminants. This has the effect of reducing the emphasis on childhood exposure for threshold contaminants and therefore results in higher soil guideline values. The *New Zealand Drinking-water Standards* (MoH, 2008) use 70 years as a lifetime for non-threshold substances.

It has been decided to continue using the averaging time conventions as adopted in all the existing New Zealand guidelines; however, it is appropriate to consider whether the 70-year lifetime should continue – or whether the averaging time should be increased to reflect the increased life-expectancy enjoyed by New Zealanders. Seventy years appears to have been based on now-outdated statistics.

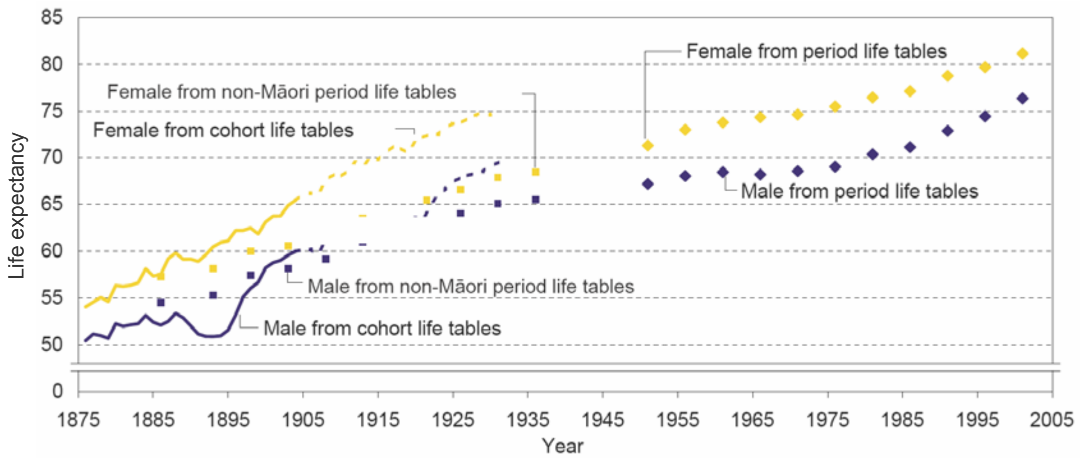
Despite the generic United States guidance still using 70 years, the *Exposure Factors Handbook* (US EPA, 1997) recommends using more recent statistics of life expectancy. The US life expectancy across males and females was 70.8 years in 1970 whereas it had increased to 75.5 years at the time of the 1993 Census. US EPA (1997) recommends 75 years as an appropriate value to reflect the average life expectancy of the general population in the US.

Life expectancy in New Zealand has also increased. Figure 2 shows the male and female life expectancy at birth in New Zealand for the years 1876 to 2002. A life expectancy at birth of 70 years for the general population was achieved in about 1955, with 75 years being achieved in about 1988. The latest Census has shown a further increase, to an average of 81.9 and 77.9 years, respectively, for a girl and boy born in 2007 (Statistics New Zealand, 2008).

Continuing to use an averaging time of 70 years will result in a guideline value for a non-threshold substance 12.5 per cent lower (more conservative) than if a life expectancy of 80 years was used, or 6.25 lower than if a life expectancy of 70 years was used.

Using an 80-year life expectancy is appropriate for people born now but not appropriate for the majority of the current population. Accordingly, based on these statistics it is recommended that the averaging time for non-threshold contaminants be increased to 75 years, to reflect the increased life expectancy of the current population.

Figure 2: Life expectancy at birth, 1876–2002



Source: Statistics New Zealand, 2006.

5.3.3 Exposure frequency

Exposure frequency is typically expressed as number of days per year, and relates to a given land-use scenario. The majority of countries use an exposure frequency of 365 days per year for residential (including parks / recreational exposure scenarios), and between 230 and 240 days per year for commercial / industrial scenarios. An exposure frequency of a little less than a year is used for residential scenarios in the existing New Zealand and US EPA protocols (350 days), based on US EPA data of an upper-end estimate of the time spent at home (95th percentile). The existing New Zealand protocols use 240 days per year for commercial / industrial scenarios. Note that the use of 365 days for residential was recently criticised in the UK on the basis that families routinely took holidays away from home every year; however, this was rejected on the basis of survey information (Defra, 2006a).

Residential

Limited data is available on time use by New Zealanders. The *New Zealand Time Use Survey* (Statistics New Zealand, 1999) provides the most extensive set of data on New Zealanders' activities. Based on the data in this report, New Zealanders, on average, spend about 70 per cent (males 66 per cent, females 75 per cent) of their time at home,[6] which equates to about 260 days

[6] Derived by Cavanagh (2004a) from data contained in Table 25 of the New Zealand Time Use Survey (Statistics New Zealand, 1999). Retrieved from http://www.stats.govt.nz

per year. However, this is not a very helpful average, as it says nothing about how many days are actually spent at home. Derivation of soil guideline values does not require the assumption of 24 hours at home for a soil ingestion or dermal absorption to occur. It just requires sufficient time for a dermal absorption or soil ingestion event to occur, which could be in the few hours after school, in the case of a child.

In the absence of better data, and given that a high-end estimate is desired, it is appropriate that the current residential exposure frequency of 350 days remains unchanged. This represents about two weeks out of the home, not necessarily as whole days but sufficiently long on the days when away from the home that soil exposure is unlikely to occur. This exposure frequency represents a 10 per cent ‘discount’ over the practice in Australia, many European countries and Canada, but is considered a high-end estimate.

Parks / recreation

No information on the frequency and duration of visiting parks and recreational facilities was found for New Zealand. A frequency of 350 days per year is used in the New Zealand ‘Gasworks Guidelines’ (MfE, 1997) and ‘Sheep-dip Guide’ (MfE, 2006a): this seems excessive, given the likelihood of bad weather preventing recreational activity for much greater than 15 days per year.

As noted in section 3.2.2, a wide range of activities can be fitted into the scenario, from simply walking in a park with little risk of soil exposure, through to frequent contact with potentially muddy ground while playing a sport such as rugby. The different possibilities will have different exposure frequencies and different exposure rates. Unfortunately it is not possible to determine an exposure frequency in isolation from the exposure-creating activity.

Possible scenarios include:

- An adult keen on both summer and winter sport who practises and plays on the same playing field perhaps two or three days a week in both summer and winter, for up to 50 weeks of the year (a maximum of 150 days per year). Soil ingestion rates are likely to be high-end estimates for both winter and summer sports, in winter being higher than summer.
- A small child plays junior sport on playing fields for limited summer and winter seasons, perhaps 26 weeks total. Winter soil ingestion rates would be greater than for residential but summer ingestion rates lower.
- A child plays on a grass-covered suburban reserve near home several days a week, more frequent in summer but less frequent in winter, say a total of 200 days a year as a matter of professional judgement (roughly equivalent to six days a week in summer, two days a week in winter and four days a week for the rest of the year). Opportunity for soil contact will be limited by the grass cover, but a child would likely get dirtier after rain and/or more active play than during dry conditions and/or more passive play. Again as a matter of judgement, soil ingestion rates would be half to a quarter of residential rates.

The question then arises as to what typical combination might be critical: frequent exposure at lower rates of more passive recreation; or less-frequent exposure at higher rates of more active recreation (eg, playing a common contact sport such as rugby in winter and sports such as cricket or softball, still involving some ground contact, in summer). For the purposes of calculating an initial comparative contaminant exposure, it can be assumed that most exposure will be through soil ingestion (although for some organic contaminants dermal absorption may also be important). Using the proposed child residential rate of 45 mg/day as a starting point, and 100 mg/day as a high-end estimate for both children and adults (see section 5.4.1) and factoring these values up or down for the various scenarios and activities as a matter of professional judgement, weight-normalised daily soil ingestion rates (mg/kg BW/day) can be calculated.

The comparison is shown in Table 4 for threshold and non-threshold substances. Age-adjusted exposure is necessary for threshold substance based on exposure durations. The proposed defaults for residential use have been assumed (see section 5.3.4), ie, six years exposure for a child and 14 years for an adult (ie, playing sport actively for 14 years on the same field). Soil intake for the equivalent standard residential scenario is shown for comparison.

Table 4: Comparison of daily soil intake for recreational scenarios

Receptor	Child (15 kg)	Child (15 kg)	Adult (70 kg)	Child (15 kg) / adult (70 kg)	
Scenario	Junior rugby and cricket: 26 weeks x 1 day/week	Suburban reserve most days. active half the time, passive half the time	Serious rugby and cricket player	Residential	
Days/year	26	200	150	350	
Soil ingestion mg/day	100 rugby, 50 cricket	20 active, 10 passive	100 rugby, 50 cricket	45	25
Rate mg/kg BW/day (threshold)	0.36	0.55	0.44	2.88	
Rate mg/kg BW/day (non-threshold)	0.03	0.04	0.08	0.29	

All three assumed recreational scenarios have a weight-normalised daily soil intake of a similar order of magnitude and significantly less than the residential scenario for both the threshold and non-threshold scenarios. In other words, the chosen recreational scenarios, despite being quite different, are not so different that it is unreasonable to take the worst value and also apply this to the other scenarios – the difference is not particularly large. However, the suburban child scenario is the critical scenario for threshold substances, while the adult sports scenario is critical for non-threshold substances; thus two scenarios have to be worked through depending on the type of substance. This will be further complicated if dermal absorption is a significant pathway.

It is therefore recommended that, for threshold substances, the default parks / recreation scenario be that of a child for 200 days per year with soil ingestion at a third of the rate of standard residential. The exposure frequency is considered a high-end estimate and will also cover the scenario of high-end sports field use by a child or an adult. For non-threshold substances it is proposed that the adult sports-field scenario is the default, with soil ingestion of 75 per cent of the high-end soil ingestion as an average for winter and summer sports. Appropriate dermal factors for sporting activity will also be required.

Commercial / industrial

Exposure frequency for commercial / industrial scenarios is typically based on a five-day working week, for 46 to 48 weeks (230–240 days per year). The exposure frequency for commercial / industrial scenarios in existing New Zealand industry-based guidelines is 240 days (48 weeks) per year. However, there are about two weeks of statutory holidays and, following the recent increase, an entitlement to four weeks annual leave. This equates to six weeks when most workers will not be at work, or 230 days when at work. It is therefore appropriate to reduce the current 240 days to 230 days. This is likely to represent a high-end estimate for those who work five days a week, as intended, but will be less conservative for those who frequently work a sixth day.

School scenario

While a school scenario is not proposed as a generic scenario, it is a scenario that concerns the community from time to time. For the purposes of site-specific assessments, an exposure frequency of 200 days per year is suggested. New Zealand primary and intermediate schools are required to be open for instruction for 394 half-days (197 days) and secondary and composite schools for 380 half-days (190 days). A frequency of 200 days per year should provide an upper-end estimate of exposure frequency for schools.

Early childhood education and childcare centres are more variable, ranging from a child attending perhaps two or three half days a week during the school term at a playcentre, through to the equivalent of a working year for a child in full-time care at a childcare centre while the parents work.

5.3.4 Exposure duration

Exposure duration is normally taken to be the length of time in years over which exposure occurs. However, the term is also used to describe the proportion of the day that any particular exposure event can occur.

Exposure duration – years

For threshold contaminants where children are the critical receptors (eg, residential and parks / recreation scenario), the duration of exposure is not important, as the averaging time in years is the same as the exposure duration. However, the critical age range is used to select other parameters appropriate to that age, which is taken as ages one to six.

Residential

The age ranges are important for the age-adjusted exposure calculations for non-threshold contaminants. As noted earlier, existing New Zealand and US protocols use an exposure duration of 30 years (6 years as a child and 24 years as an adult), which nominally represents a maximum time a resident will spend on one property, based on US data.

Census information on the length of time spent on a residential property is available for New Zealand. Statistics New Zealand has collected information on internal migration, from which some useful numbers can be derived. The 2006 Census provides information (available online at www.stats.govt.nz) on the number of continuous years at the same residence, broken down by age bracket. Within an age-bracket ‘x–y’ years, it is possible to determine the percentage of people who have inhabited the same residence for the full length of that age-bracket. Conservatively, this percentage is a high estimate of people in that age range who have inhabited the same residence all their lives. These statistics cannot help determine the number of people who have lived at a property as a child and perhaps teenager, leave home and then returned to live at the same house as an adult after a period away (possibly a relatively rare occurrence); nevertheless the data give some sense of what proportion of the population have lived continuously at one house both as a child and an adult.

For 20–24 years, 5.3 per cent of that age range had lived at a single residence for as long as their age in 2006. This translates to only 0.36 per cent of the total population; in other words, 99.6 per cent of the population had spent less than their first 20–24 years at a single residence. For 25–29 years, the closest age range for the currently assumed continuous exposure of

30 years, the same calculation produces 1.3 per cent having lived at the same residence almost all their lives. This group represents only 0.08 per cent of the total population. Only in the 15–19 age range does the percentage of the total population residing in the same house for their whole life rise to 1 per cent; still a small number.

Only 3.2 per cent of children aged four years have stayed all their lives in the same residence, so 96.8 per cent have not. Although data does not exist for up to six years of age (relevant to the childhood exposure scenario), the data on 0–4-year-olds suggests a similarly low percentage of children who have lived at the same house all their lives.

Considering the adult exposure scenario, the 2006 Census data shows that 6.2 per cent of the total population have stayed continuously in the same residence for 25 or more years; 9.7 per cent for 20 or more years; and 15.5 per cent for 15 or more years. Conversely, 84.5 per cent of the total population has lived in the same house fewer than 15 years, and 57.7 per cent fewer than 4 years.

It is not possible to determine from the 2006 Census the percentage of the population having resided in the same house both as a child and an adult. Clearly, continuous residence from birth to late teenage years is a small percentage, with residence as a child for up to six years being estimated at only 3 per cent of the population. In addition, the available information suggests only a very small percentage of adults are likely to reside in their childhood home for that length of time: most likely a small subset of rural dwellers who have been raised on a farm, and remain or return there as an adult. The current assumption of residing in the same house for 30 years, including as a child, therefore seems excessively conservative.

Given that New Zealand's population is mobile as demonstrated, it is appropriate for the standard residential scenario that the length of residence as an adult and child is reduced to 20 years: 6 years as a child and 14 years as an adult. This will still be a high-end estimate. The same duration is proposed for the high-density residential and parks / recreation scenarios.

A study by Sanson et al (2004) of several thousand lifestyle blocks in New Zealand, using census and land ownership records, found that 35.5 per cent of lifestyle blocks with a dwelling remained in the same ownership for more than 30 years. For the rural residential scenario, it is therefore proposed to retain the current 30-year duration (6 years as a child and 24 years as an adult).

It is worth examining whether some site-specific situations exist with extended adult residence being the critical case. This would only be for non-threshold contaminants where exposure durations affect the $SGV_{(health)}$ calculation. The 2006 Census found only 4.5 per cent of the general population stayed in the same house for more than 30 years. This suggests a duration of 30 years as an adult is a reasonable high-end estimate for the standard residential scenario. The Sanson et al (2004) study suggests that perhaps 40 years as an adult is a high-end estimate for the rural / lifestyle block scenario.

Check calculations using these extended adult durations show that the child / adult combination still remains critical, although the derived SGVs for extended adult durations are only marginally higher than the SGVs for the child / adult combination for some contaminants. If, in a site-specific study, the various default exposure parameters in this document were changed, then extended adult-only durations could become critical for some non-threshold substances; this would be particularly so if the ratio of adult to child rates of any of soil ingestion, produce ingestion, or dermal soil adherence was increased. A further check shows that increasing the adult-only duration to 35 years results in the derived SGVs for the standard residential scenario becoming the critical (lowest) value for some substances. This points to a need to ensure such scenarios are checked in site-specific analyses that might involve extended durations.

Commercial / industrial

For commercial / industrial scenarios, the exposure duration is based on the length of time in one job, and ranges from 20 years in New Zealand for current guidelines to 43 years in the UK. The 20-year exposure duration used in New Zealand is based on US EPA data for the 90th percentile for time spent in one job (MfE and MoH, 1997). US EPA protocols for deriving generic soil guideline values use an exposure duration of 25 years for commercial / industrial scenarios based on the 95th percentile for time spent in one job. The Australian protocol (NEPC, 1999a) also specifies a nominal exposure duration of 25 years for commercial / industrial scenarios, but because default exposure ratios (DER) are used to convert from residential to other scenarios, the nominal exposure duration value is not used.

There is limited data available on job tenure in New Zealand. The question has not been included in the five-yearly census. Statistics New Zealand's Linked Employer-Employee Data (LEED) matches tax data to statistics collected from employers; this provides limited job tenure information since the 1999 tax year. The most recent compilation, for the 2006 tax year (available online at www.stats.govt.nz), allows the percentage of employees who have worked seven years or greater with the same employer to be calculated. The data is limited in that tenure is not necessarily continuous and work is not necessarily full-time: the data only indicates whether work was performed in a particular month for an employer, not how much work. In addition, while the statistics are broken down to a regional level, they do not necessarily indicate work at a particular site. However, 12 per cent of workers have worked for the same employer for seven years or greater. Given the limitations, the percentage of workers working continuously at the same site is likely to be lower. This suggests that a 20-year job tenure is greater than the 90th percentile. In the absence of better information, it is proposed that the exposure duration for industrial / commercial land uses remain at 20 years, as a high-end estimate.

Exposure duration – hours per day

Exposure duration expressed as hours per day may also be used for some exposure scenarios or exposure pathways. Soil and dust ingestion is normally taken as a daily rate, based on a combination of indoor and outdoor exposure with no specified contact period during the day. While the assumption is that ingestion is spread out over the day, in reality it would consist of a number of individual hand-to-mouth contact events of only a few minutes duration. In addition, the available studies for estimating ingestion rates have been carried out over a number of days or weeks, and the rates are daily averages (see section 5.4.1); these render the duration of the activity which resulted in the soil ingestion irrelevant.

For dermal exposure, the UK protocol originally used a 12-hour exposure duration in a residential setting (Defra and EA, 2002a) but later changed that (EA, 2005a, 2008a) to the US EPA approach: this uses a 24-hour exposure duration, on the basis that soil is attached to the skin during an event and is then not washed off before it is absorbed (US EPA, 2001a). The default assumption is one exposure event per day. The Dutch CSOIL model (Brand et al, 2007) approach is to assign a period in hours for each of indoor and outdoor for both children and adults, because a dermal adsorption rate per hour is used in the model. Different durations are specified for different exposure scenarios. The approach of US EPA (2001a) and a later update, US EPA (2004c), are used in the current study. This may not be appropriate for all substances, depending on how the particular absorption rate is derived, and may need to be varied on a contaminant-specific basis.

For inhalation exposure, the durations of exposure are inextricably linked with inhalation rates, with an inhalation rate of m^3/day specified for the particular scenario and receptor, taking into account the duration. This is the approach adopted here, and is discussed in the next section.

In summary, the proportion of the day exposed is not important for soil ingestion and dermal exposure, if dermal absorption coefficients are based on event-based experiments. However, if the dermal absorption coefficient for a particular contaminant is based on 24-hour exposure, then the proportion of the day exposed has to be factored into the calculation. Following the existing 'Sheep-dip Guide' (MfE, 2006a), for the residential situation this is taken as 12 hours and for the occupational situation 8 hours. For the inhalation pathway, residential exposure is assumed to be 24 hours and occupational exposure 8 hours.

5.4 Pathway-specific parameters

The pathway-specific parameters are either exposure rates or combination of parameters that make up an exposure rate. These are the time-dependent soil ingestion, produce consumption and inhalation rates, and soil adherence values and skin surface areas necessary to calculate dermal soil loadings. For each pathway there are several scenario / receptor combinations, each of which must have a set of parameters – meaning a large number of parameters need to be decided on.

Soil ingestion and produce consumption are typically the pathways contributing most to residential guideline values, although dermal absorption can be significant for some contaminants. Soil ingestion and sometimes dermal absorption are the important pathways for high-density residential and the non-residential scenarios.

5.4.1 Soil ingestion

Soil ingestion can occur in both indoor and outdoor settings as a result of deliberate sucking and mouthing of objects by children, inadvertent hand-to-mouth transfer by children and adults, and ingestion of soil attached to produce. Early estimates of the amount of soil ingested were largely activity-based, whereby soil ingestion rates were estimated from factors such as time spent outside or doing certain activities, the number of hand-to-mouth events, and the degree of hand-soiling – whether measured or predicted. More recent studies have predominantly used tracer elements found in soil (typically aluminium, silicon and titanium, but also barium, manganese, vanadium, yttrium and zirconium) to provide a more direct measurement of soil ingestion. In these studies, soil ingestion rates are determined from the mass balance difference in the levels of tracers in materials ingested daily and the levels in urine and faeces. The better-designed studies took into account other sources of these tracers in food, medicines and consumer products.

Stanek et al (2001) have outlined the improvement in soil ingestion estimates from tracer studies as estimation techniques improved. Initial estimates of soil ingestion were based on individual trace elements (Binder et al, 1986; Calabrese et al, 1989; Davis et al, 1990). Since the estimated distributions from different trace elements often differed substantially, subsequent work focused on ways of identifying more reliable estimates (Stanek and Calabrese, 1991a; Calabrese and Stanek, 1991). This work led to the use of trace element food / soil ratios as a means of identifying potentially reliable trace elements, and the use of the median trace element estimate from among a subset of reliable elements (Stanek and Calabrese 1991b). The results of this work contributed to improvements in study designs for soil ingestion estimation via the inclusion of additional trace elements, longer study designs, and use of special diets.

While the relative understanding of soil ingestion has improved, average rates of ingestion in relation to specific activities and land-use remain uncertain. Limited data exists for child ingestion rates in a residential setting from a few reliable studies. Very little data exists for adult ingestion rates for a residential setting. Estimating ingestion rates for non-residential scenarios relies on professional judgement as no reliable tracer studies exist.

Residential soil ingestion – children

Children have been the primary focus for soil ingestion studies due to their inclination to mouth objects (hands, toys) and ingest dirt. Combined with low body weight, childhood soil ingestion is typically a major component of residential soil guideline values. Generally, it is considered that intensive mouthing diminishes after two to three years of age and negligible soil ingestion occurs after the age of six to seven (Paustenbach, 2000), making soil ingestion at later ages relatively less important.

A large range of soil ingestion rates is used by individual agencies in deriving guideline values (table 5), depending on the particular agency's philosophy (worst-case / high-end estimate versus best estimate) and which studies have been used to derive the estimates.

Table 5: Soil ingestion rates used in national and international protocols

Receptor	New Zealand	Australia	US	Canada	Netherlands	UK
Child	100	100	200	80	100[a]	100[b]
Adult – resident	25	na	100	20	50	na
Adult – worker	25	na	50 indoor 100 outdoor	20	na	50

a Reduced from 150 mg/day in 2001 (Brand et al, 2007).

b Originally a probability distribution with mean 100 mg/day and 95th percentile of 300 mg/day (Defra and EA, 2002a), subsequently modified to a single-point estimate of 100 mg/day (EA, 2008a).

na = Not applicable.

The value of 100 mg/day for soil ingestion by children used in current New Zealand protocols (eg, MfE and MoH, 1997) comes from ANZECC (1992) and is the same as that used in the derivation of the Australian health investigation levels as proposed by Langley and El Saadi (1991), based on a review by Taylor (1991). The value was considered to be a conservative value at the time but drew on work that pre-dated more recent tracer studies.

The *Exposure Factors Handbook* (US EPA, 1997) states a soil ingestion rate of 100 mg/day represents an average estimate for exposure assessments, but recommends as a high-end estimate a rate of 200 mg/day in risk assessment. Generic soil guidance in the US (eg, US EPA 1996a, 2002a) is based on 200 mg/day.

A report commissioned by the US EPA (Versar, 2001) considered the then available studies on childhood soil ingestion. The report noted that a number of studies had weaknesses and concluded that the best estimates were from several mass-balance studies using metal tracers conducted in the United States. Two Dutch studies (Clausing et al, 1987; Van Wijnen et al, 1990) were rejected because of study-design limitations. Versar (2001) also noted the possibility of different cultural practices between Dutch and US child rearing that might affect behaviour and soil ingestion.

The draft *Child Specific Exposure Factors Handbook* (US EPA, 2006a) reviews the science to that point and arrives at a mean childhood soil ingestion rate of 90 mg/day and median soil ingestion rate of 35 mg/day, with a 95th percentile value of 236 mg/day. Because of the skewed nature of the distribution, the mean is larger than the 75th percentile value. US EPA (2006a) arrives at these estimates by applying (unstated) weighting factors to the results of five 'key' mass-balance tracer studies to calculate a weighted average distribution: four original studies and one study re-analysing data from two of the other studies – in effect double counting some of the data in the weighted average. US EPA (2006a) acknowledges this double counting but does not explain why, nor discuss what effect this might have on the overall estimate. It is also not clear why some studies were considered 'key' whereas later studies re-analysing the same original data using improved statistical techniques were not considered 'key'. The later analyses generally resulted in lower soil ingestion estimates.

The four original tracer studies considered robust enough to be included in the weighted average of the US EPA (2006a) review were all conducted in the United States. These were:

- a study of 104 children from an semi-arid, three-city area in southeast Washington State (Davis et al, 1990)
- a further study a year later of a 19-child subset of the Washington study (Davis and Mirick, 2006)
- a study of 65 children from the Amherst area of Massachusetts (Stanek and Calabrese, 1995a, using data from Calabrese et al, 1989)
- a study of 254 children in Anaconda, Montana (Calabrese et al, 1997).

The fifth study included in the weighted average, Stanek and Calabrese (1995b), sought to provide better estimates by reanalysing Davis et al (1990) and Calabrese et al (1989) using the 'best tracer method' to correct for errors in the tracer input and output measurements, and error from ingestion of tracers from non-food and non-soil sources.

The review rejected some other original tracer studies because of study design limitations, particularly not accounting for tracers sources other than soil, for example in food and in medicines. Rejected studies included those of Binder et al (1986), Clausing et al (1987), and Van Wijnen et al (1990).

US EPA (2006a) goes on to recommend that the best estimate for mean child soil ingestion for ages one to seven years is 100 mg/day (rounding up of 90 mg/day) and further recommends a 95th percentile value for soil ingestion of 400 mg/day. It is notable that the latter recommendation is from soil and dust data, rather than soil alone, whereas the 95th percentile estimate for soil alone rounded to the nearest 100 would be 200 mg/kg. The recommendations in US EPA (2006a) essentially confirm earlier recommendations in the 2002 interim version of the same document (US EPA, 2002b) and the *Exposure Factors Handbook* (US EPA, 1997).

The double-counting of two of the earlier studies will have tended to bias the weighted averages calculated in US EPA (2006a). It is not possible to determine the precise bias because the weighting factors are not given (other than the incorporated sample size and 'other statistical factors'). However, if a simple average is used for the mean and median soil ingestions across the five studies, the values are 80 and 33 mg/day, compared with the weighted averages of 90 and 35 mg/day. If Davis et al (1990) and Stanek and Calabrese (1995a) are then not included in the average (as reanalysed in Stanek and Calabrese (1995b)), the simple averages of mean and median soil ingestion for the smaller number of key studies are 69 and 29 mg/day, respectively. This suggests the double-counting has biased the US EPA (2006a) result upwards by perhaps 10–20 mg/day for the mean and perhaps 5 mg/day for the median. The US EPA's rounded-up recommendation of 100 mg/kg for the mean soil ingestion therefore appears high by 20–30 mg/kg.[7]

It is notable that US EPA (2006a) has apparently not taken into account the view of one of the leading investigators in this field, Edward Calabrese: he was the principal author of the 1989 Amherst study and a collaborator with Edward Stanek in the later Anaconda study plus the many re-analyses of both studies. In a letter to the General Electric Company, Calabrese (2003) expressed the view that the 1989 Amherst study overestimated child soil ingestion. Calabrese instead favoured the reanalysed results of the Anaconda study (Stanek and Calabrese, 2000), in recommending a central tendency rate of 20 mg/day (based on a median) and an upperbound (95th percentile) rate of 100 mg/day.

Paustenbach et al (2006) reviewed the literature as part of determining input values for a probabilistic risk analysis for dioxins. They concluded that Stanek et al (2001) provided the most robust data set for determining a probability distribution for childhood soil ingestion. Stanek et al (2001) reanalysed earlier data from the Anaconda study (Calabrese et al, 1997), extrapolating the short-term measurements to long-term estimates, and concluded that earlier estimates (eg, the recommendations in US EPA, 1997) were too high. Their method was aimed at eliminating bias resulting from uncertainty in the daily estimate, or variability in soil ingestion from day to day – that bias overestimating soil ingestion for upper percentiles and underestimating it for lower percentiles. Paustenbach et al (2006) suggested the analysis of Stanek et al (2001) was a vast improvement over the original analysis of the studies considered to have an adequate design (Davis et al, 1990, Calabrese et al, 1989, Calabrese et al, 1997), with the improved method resulting in lower values.

Van Holderbeke et al (2007) provided a good summary of the various issues around attempting to estimate soil ingestion. They reviewed the literature to that time, including mass-balance tracer studies, behavioural hand-to-mouth and hand-soil loading-based studies, and various studies based on biomonitoring. Their aim was to arrive at soil ingestion estimates relevant to a project in the Kempen region of Belgium and the Netherlands. With respect to the tracer studies, they went over similar ground as US EPA (2006a) in determining ranges and arithmetic average across the various studies for the soil ingestion rate mean and percentiles. Like US EPA (2006a) they included in their calculations data from Davis et al (1990), Davis and Mirick (2006) and Calabrese et al (1997); but unlike US EPA (2006a) rejected the Amherst study (Calabrese et al, 1989) and subsequent re-analyses of that data. Data from the Amherst study was left out on the basis of the principal author subsequently stating that the results overestimated soil ingestion rates (Calabrese, 2003). Unlike US EPA (2006a) they also included results from Stanek and Calabrese (2000), a re-analysis of the Anaconda study – in effect double-counting that study – and also included data from Clausing et al (1987) and Van Wijnen et al (1990). The latter two studies were apparently included on the basis that they are European studies and therefore more relevant to Belgium, despite their design limitations.

Van Holderbeke et al (2007) concluded that the best central tendency estimates of child ingestion rates from tracer studies were, for the median value, 30 mg/day (average of medians ranging from 20–40) and for the mean value, 60 mg/day (average of means ranging from 40 to 80 mg/day).

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7 Since this section was written, the US EPA has released the final version of the *Child Specfic Exposure Factors Handbook* (EPA/600/R-06/096F, National Center for Environmental Assessment, Office of Research and Development, Washington, September 2008). Although this document arrives at a similar conclusion with respect to soil ingestion rates as the 2006 draft, and presents a similar range of scientific studies from the international literature, little detail is given to justify the recommended ingestion rates.

As noted earlier, Van Holderbeke et al (2007) also reviewed the literature for studies other than mass-balance tracer studies. Such other studies are generally considered less reliable. Without going into the details here, these authors determined that average child ingestion rates from hand loading studies range from 7 to 60 mg/day, from biomonitoring studies 50–100 mg/day, and from empirical relationships 20–70 mg/day. These ranges are generally consistent with the mass-tracer studies.

A summary of the child ingestion values proposed in the various references are set out in Table 6. They all draw on essentially the same information, with different interpretations as to which original studies or reanalyses of these studies should be considered.

Apart from Van Holderbeke et al (2007), the various reviews place little reliance on the European tracer studies, favouring the US studies because of their better study design. Van Holderbeke et al appear to have included the other studies only because they were European and hence might provide better values for European children, regardless of study design. This raises the issue of how relevant any of the studies might be for New Zealand conditions and children.

Table 6: Summary of child soil ingestion rate recommendations as reviewed

Reference	Age range (years)	Mean (mg/day)	Median (mg/day)	75th percentile (mg/day)	Upper bound (mg/day)	Comment
Versar (2001)	1–2	30	24	–	100 (90th percentile)	
	3–5	30	20	–	150 (90th percentile)	
	6–10	71	37	–	187 (90th percentile)	
Calabrese (2003)	1–4		20	–	100 (95th percentile)	From Stanek and Calabrese (2000)
US EPA (2006a)	1–7	100	35	78	400 (95th percentile)	
Paustenbach et al (2006)	1–4	31	24	42	91 (95th percentile)	From Stanek et al (2001)
Van Holderbeke et al (2007)	1–7	60	30	–	195	Tracer studies
		7–60	–	–	–	Behavioural studies
		50–100	–	–	–	Modelling / biomarkers
		20–70	–	–	–	Empirical relationships

Many physical and societal factors affect the opportunity for soil and dust exposure and a child's behaviour in potentially being exposed. The factors tend to be interrelated, and include:

- climate – opportunity for outdoor play, likelihood of soil sticking to shoes and being tracked inside, generation of dust
- style of residential development – house construction and dust-tightness, types of indoor floor coverings (and whether they gather dust), presence of gardens or other bare soil
- lifestyle – time spent outdoors, type of outdoor play, popularity of gardening activities, societal attitudes to how clean a house should be
- parental attitudes – attitude to allowing outside play and whether children are allowed to get dirty, insistence on hand washing.

Given these factors, and without information to quantify any differences, it cannot be determined how well the United States values translate to another location. At best it is a matter of judgement: in a general sense the New Zealand style of housing and lifestyle is possibly closer to those in the United States than Europe, suggesting the opportunity for outside play (and therefore exposure to soil) might be similar. For example, details provided in the US studies suggest a relatively high proportion of properties have gardens or unpaved driveways, providing the opportunity for soil contact. In addition, all the US tracer studies were carried out in summer or autumn with the particular locations' summer temperatures generally higher than is typical for New Zealand's temperate maritime climate. But in so far as a favourable temperature indicates a greater likelihood for outside play, the US studies perhaps represent a reasonable year-round estimate for New Zealand. This estimate is probably conservative because, even in New Zealand's temperate climate, outdoor activities will most likely be reduced in winter relative to summer and provide less opportunity for soil contact than the US summer studies would suggest.

If it is accepted that US tracer studies provide the best values for New Zealand and given that single-point estimates are to be used, a value must be chosen that avoids compounding conservatism. The necessary conservatism of SGVs is already obtained by using high-end estimates of other parameters (eg, exposure frequency and duration), and the conservatism of the substance-specific toxicity values. This means that the chosen soil ingestion values should be representing central tendency. The work of Van Holderbeke et al (2007) and others suggests the child ingestion rate of 100 mg/day used in current New Zealand guidelines is rather larger than a central estimate, lying perhaps between the 80th and 95th percentiles.

Central tendency is normally represented by the mean or median. In the case of soil ingestion, the distribution of values is quite skewed in the positive or right direction, with a long tail of a few high values. This results in the mean being considerably larger than the median, with consequent uncertainty as to which best represents central tendency. The median is often chosen as the best measure of central tendency with right-skewed distributions (eg, for house prices or income within a group of people) and is possibly appropriate here. The available studies suggest the median is in the range 20–30 mg/day, say 30 mg/day, with the mean being about twice that value (60 mg/day).

As a pragmatic compromise, a value halfway between the mean has been employed as the residential childhood rate, that is, 45 mg/day. This is close to the 75th percentile of the distribution calculated by Stanek et al (2001) of 42 mg/day.

Soil ingestion – pica children

Deliberate soil ingestion by so-called 'pica children[8]' is typically not taken into consideration in the estimated soil ingestion rates for children, because this eating disorder is considered to be rare rather than a chronic effect (Paustenbach, 2000). More recently, Paustenbach et al (2006) suggested that the upper bound estimates proposed by Stanek et al (2001) would account for the vast majority of typical children as well as mild or infrequent pica behaviours in 'a conservative risk assessment'.

It is proposed that pica behaviour is not taken into account, as behaviour modification is a more appropriate response. This was the view of the Technical Review Group for the NES (MfE, 2005). However, it may be appropriate to allow for pica behaviour in some site-specific

[8] Pica is a eating disorder characterized by an appetite for substances largely non-nutritive (eg, soil, clay, ash, metal, etc.)

situations, in which case choosing an upper-bound estimate for childhood soil ingestion (say 100 mg/day) will be appropriate.

Residential soil ingestion – adults and older children

There is little data for adult soil ingestion rates; development of appropriate soil ingestion rates is difficult (Paustenbach et al, 2006). Values ranging from 20 to 100 mg/day have been used by various regulatory agencies. US EPA (1997) recommends an adult ingestion rate of 50 mg/day as a reasonable central estimate, based on a review of three available studies. Considering the uncertainties in the central estimate, US EPA (1997) was unable to recommend an upper-bound estimate. This work noted that many past US EPA assessments had used 50 mg/kg for industrial settings and 100 mg/day for residential settings. Existing New Zealand guidelines use 25 mg/day for residential scenarios and 100 mg/day for industrial scenarios.

The few available studies indicate that adult soil ingestion is likely to be in the order of 5 to 25 mg/day. For example, Paustenbach (2000) concluded that average daily soil ingestion rates of 5 to 25 mg/day were reasonable, based on a review of the limited data. Similarly, Stanek et al (1997) estimated using tracer studies, that mean daily soil ingestion rates (over a four-week period) by adults was 6 mg/day. However, it should be noted that Otte et al (2001) suggested that this average was skewed by a low average in the fourth week of the four-week study, and that taking only the first three weeks' data was more appropriate. This would give rise to an average daily soil ingestion rate of 53 mg/day.

Calabrese's (2003) letter to the General Electric Company, referred to above, also commented on adult ingestion rates. He recommended an upper-bound adult soil ingestion rate of 50 mg/day and a central estimate of 10 mg/day, based on work by him and Stanek (Stanek et al, 1997). Calabrese noted that the US EPA had based their estimates on his earlier work reported in 1990, but that the more recent work was an improvement with more participants (20 instead of 6), a longer study period (28 days instead of 14), and better study design and analysis.

Paustenbach et al (2006) reviewed the available studies and found a log-normal distribution of adult soil ingestion appropriate for their probabilistic risk assessment, with a range of 10–100 mg/day and a mean of 30 mg/day. They viewed this distribution as appropriately conservative to address uncertainties with respect to exposure from adult activities with heavier soil contact (eg, gardening or construction).

Van Holderbeke et al (2007), also reviewing the available studies, suggested a median value of 25 mg/day and a mean value of 45 mg/day for the residential scenario with gardens.

Given the uncertainty, it has been decided to retain the adult soil ingestion rate for the residential scenario used in existing New Zealand guidelines, that is, 25 mg/day. This value falls towards the centre of the various estimates cited above.

Soil ingestion – high-density residential

Little or no data exists for soil ingestion for land uses other than the standard residential scenario. Professional judgement must be resorted to.

The high-density residential scenario has a reduced opportunity for soil contact relative to the standard residential scenario: it relates to a multi-unit or townhouse type of development with only small areas of land around the dwelling, and yards being largely paved or grassed, but perhaps with small ornamental gardens allowing some soil contact. The scenario does not include apartment-type developments for which no soil contact is expected. Few overseas jurisdictions provide for the high-density residential scenario. An exception is Australia (NEPC, 1999a), using default exposure ratio of four to factor up the standard residential guideline values. Australia does not specifically allow for home-grown produce consumption, soil ingestion for a two-year-old child being the major exposure pathway. Applying DER=4 is therefore an effective factoring down of the soil ingestion rate for a child to a quarter that of the standard residential rate.

Two existing New Zealand guidelines have high-density residential scenarios – the 'Gasworks Guidelines' (MfE, 1997) and the 'Sheep-dip Guide' (MfE, 2006a). The former reduced the childhood soil ingestion rate by a factor of four and the adult rate by a factor of five compared with standard residential, for unknown reasons. However, the 'Sheep-dip Guide' used the same rates as the standard residential scenario.

A reduction by a factor of four seems too high for the limited but definite potential exposure contemplated by the definition of the high-density residential scenario. It is therefore proposed as a matter of professional judgement to reduce the standard residential ingestion rates by a factor of two, rounded to the nearest five units. The child soil ingestion rate then becomes 25 mg/day and the adult rate 15 mg/day.

Soil ingestion – parks / recreation scenario

Section 3.2.2, in exploring exposure frequency for the parks / recreation scenario, noted that it is impossible to discuss exposure frequency (or contact rate) without discussing soil ingestion, as likely subscenarios tended to be combinations of low ingestion rates and high frequency, or high ingestion rates and low frequency. This approach is repeated here.

As noted in section 3.2.2, Australia and the Netherlands have scenarios similar to the proposed parks / recreation scenario. Australia uses default exposure ratios to increase the standard residential guideline value for other scenarios, as described above. In the Australian parkland scenario, the factor is two (NEPC, 1999a): this in effect factors down the residential soil ingestion rate of 100 mg/day to 50 mg/day if the contact rate is assumed to be the same.

Brand et al (2007), describing proposed revisions to the Dutch CSOIL model, proposed 20 mg/day and 10 mg/day for a child and adult, respectively, for the 'Greens' scenario. These rates are one-fifth of the standard Dutch residential with garden rates. It is notable, however, that the Dutch contact rates (days/year) are much less than typically employed elsewhere, with the combined ingestion and contact rate for Greens being a 25th of the residential combined ingestion and contact rate.

Two existing New Zealand documents, the 'Gasworks Guidelines' (MfE, 1997) and 'Sheep-dip Guide' (MfE, 2006a) have parklands scenarios. Both use 50 mg/day for a child and 10 mg/day for an adult, with exposure for most days (350) of the year. This translates to a combined contact and ingestion rate of about half that for the standard residential scenario, similar to the Australian approach.

Section 3.2.2 examined what are considered more realistic exposure scenarios for active recreational activities. An exposure frequency of 350 days per year is excessive for typical active recreational activities where a child might get dirty; upper-bound ingestion rates that might be typical of playing contact sports such as rugby will occur much less often than 350 days per year, even for the keen sportsperson. As a matter of professional judgement, for childhood play activities in green spaces near home (eg, grass-covered road berms and suburban green spaces) it seems reasonable to use an average childhood soil ingestion rate of one-third of the standard residential rate (rounded to the nearest 5 units) with an exposure rate of 200 days per year for threshold substances. This results in soil ingestion rate of 15 mg/day for children. As an alternative scenario for non-threshold substances, again as a matter of professional judgement, a reasonable scenario is an adult exposed for 150 days per year (including practices) at an ingestion rate of 75 mg/day to reflect potentially high ingestion rates during high-contact sports such as rugby. The combinations of ingestion rates and exposure frequencies are between about one-quarter and one-fifth of the residential combination.

Soil ingestion – commercial / industrial scenario

The commercial / industrial indoor worker has no soil ingestion, but has moderate exposure on most working days (up to 230 days per year) while carrying out maintenance activities. This is considered conservative for workers on an unpaved site and also covers occasional (a few times a year) excavation activities associated with site maintenance at higher exposure rates.

Current New Zealand guidelines do not have a similar scenario. The excavation / maintenance scenario in the 'Timber Treatment', 'Gasworks' and 'Oil Industry Guidelines' (MfE and MoH, 1977; MfE, 1977, 1999) is for relatively high soil ingestion of 100 mg/day for 50 days per year. This ingestion rate is from GRI (1988) and is the same as the default recommended for outdoor workers in US EPA (2002a). The rationale for the value in US EPA (2002a) is not given other than it is to reflect higher exposure than for indoor workers, who are assumed to ingest 50 mg/day from indoor dust containing soil tracked in from outside. The latter appears conservative for typical indoor areas. The Dutch assume 10 mg/day on a few days per year for their proposed scenario covering buildings, infrastructure and industry (Brand et al, 2007).

The standard New Zealand scenario for commercial / industrial sites, whether indoor, or outdoor is currently 25 mg/day (sourced from ANZECC, 1992).

The recommended soil ingestion for adult workers involved in maintenance activities that involve routine contact with soil (eg, gardening) is 50 mg/day. This is twice the current New Zealand value but reflects the assumption of greater routine soil contact; it is half the current 100 mg/day excavation / maintenance ingestion rate which assumes regular excavation work. The higher rate could be used for site-specific studies where long-term excavation work is being carried out. The values are based on professional judgement, rather than on scientific studies.

Summary of soil ingestion rates

A summary of the default soil ingestion rates is set out below.

Table 7: Default soil ingestion rates for child and adult (mg/day)

Scenario	Rural residential and residential		High-density residential		Parks / recreation		Commercial / industrial outdoor
Receptor	Child	Adult	Child	Adult	Child	Adult	Adult
Central tendency for generic guidelines	45	25	25	15	15	75	50
High-end estimate for site-specific assessment	100	–	50	–	35	100	100

The recommended values for the residential and parks / recreational scenarios are lower than used in the current guidelines. The general effect will be to increase SGVs, ie, be less conservative, although the effect is complicated by revising the toxicological values employed in the derivations for particular contaminants. Some toxicological values have increased and others reduced: this sometimes results in reduced soil ingestion values being outweighed by a reduction in the allowable daily intake of that contaminant. In addition, the effect of the changed soil ingestion rate may be emphasised or muted: this depends on whether the soil ingestion pathway is the major pathway, or whether produce ingestion or dermal absorption are also significant. Given that the parameters affecting the other pathways have also been revised, there is no direct relationship between SGVs calculated with the soil ingestion rate of 100 mg/day employed in current New Zealand guidelines, and $SGVs_{(health)}$ calculated with the revised parameters.

For scenarios where home-grown produce consumption is a major pathway, **and** for contaminants that tend to be taken up into plants, the soil ingestion rate has relatively little influence on the SGVs. This is particularly so for scenarios with a high proportion of home-grown produce (see next section). Where the produce ingestion pathway is particularly dominant (eg, for cadmium at low pH), the soil ingestion rate has no influence. However, for the high-density residential scenario without produce consumption, the derived SGV is generally directly or nearly directly proportional to the soil ingestion rate. This applies unless the dermal pathway dominates, in which case the soil ingestion pathway also has little or no influence on the final value.

5.4.2 Produce consumption

Produce consumption can be a significant exposure pathway in residential scenarios, depending on how much of a particular substance is taken up by edible home-grown plants and how much home-grown produce is consumed. A simple calculation can be made using the common value of home-grown produce taking up one per cent of the soil concentration of a contaminant in dry weight terms. If a young child obtains 10 per cent of a typical 10 g/day dry-weight vegetable consumption from this produce, then this is the equivalent of ingesting an additional 10 mg of contaminated soil per day. If 50 per cent of daily produce consumption is home-grown, the same one per cent contaminant uptake equates to 50 mg/day of soil ingestion. These rates are significant relative to the proposed 45 mg/day residential soil ingestion rate for a child.

The produce consumption pathway must consider four parameters:

- the total amount of produce consumed
- the proportion of that consumption from home-grown produce
- the vegetable-type-specific soil loading on the outside of produce
- the chemical-specific uptake factors for contaminants taken up into the produce.

The first three of these parameters is considered below. A general discussion on contaminant-specific produce uptake factors (or bioconcentration factors) is provided in section 5.5.1, while the basis for the specific bioconcentration factors used in the $SGV_{(health)}$ derivations is given in the relevant sections for each contaminant in section 6.

Produce consumption rates

Produce consumption rate is the most country-specific pathway in relation to the amount and type of produce consumed, and the proportion of home-grown produce. Current values for the amount of produce consumed in different countries (where used for deriving guideline values) range from 65 to 151 g/day for a child (table 8).

For adults, current industry-based guidelines in New Zealand use values for the amount and type of produce consumed from the Australian National Dietary Survey. Estimates based on daily nutrient requirements for toddlers (one to three years old) are provided in Langley (1993). These values are also shown in table 8.

Table 8: Produce consumption rates (g FW/day) used in international protocols

Receptor	New Zealand[a]		Australia	US	Canada	Netherlands	UK
	TTG, OIG, GWG	SDG					
Child	130 [b]	77	na	na	125	151	65 [c]
Adult	450 [b]	254	na	na	250	295	97 [c]

a TTG = 'Timber Treatment Guidelines' (MfE and MoH, 1997), OIG = 'Oil Industry Guidelines' (MfE, 1999), GWG = 'Gasworks Guidelines' (MfE, 1997), SDG = 'Sheep-dip Guide' (MfE, 2006a), na = not applicable.

b Divided into above-ground, roots and fruit.

c Values calculated from fresh weight data in EA (2008a) assuming body weights of 15 kg and 70 kg for 2–4 year toddler and adult, respectively.

In the 'Timber Treatment Guidelines' (MfE and MoH, 1997) these amounts are divided into leafy vegetables, root vegetables and fruit in the proportion 31:29:40 (100 per cent total). Uptake into fruit is considered to be negligible. In the 'Gasworks Guidelines' (MfE, 1997) the amount of fruit consumption is ignored on the basis that fruit is grown by only a very small proportion of people, and the balance of produce consumption is divided 50:50 into leafy and root vegetables.

The 'Sheep-dip Guide' (MfE, 2006a) updates these consumption rates by considering a variety of vegetables and age-related consumption rates for root and leafy vegetables to produce a weighted-average rate. The approach was adopted from Cavanagh and Proffitt (2005), in which soil guideline values were derived for the Sandilands subdivision in Christchurch. The 'Sheep-dip Guide' continues the assumption that very few people grow their own fruit and therefore this contributes a negligible produce exposure, on average. This assumption would need to be revisited on a site-specific basis in the event of a large amount of home-grown fruit.

The consumption rates in the 'Sheep-dip Guide' are based on data from the 2003/2004 New Zealand Total Diet Survey (Vannoort, 2003). This survey provides consumption rates for a variety of common vegetables based on fresh weights. These have been converted to dry weights; weighted-average dry-weight produce consumption rates for leafy and root vegetables were then calculated for both children and adults. This approach is also used for the derivations in this document, except that root vegetables are divided into true roots (eg, carrot) and tubers (eg, potatoes) for which further details are provided in Appendix 3. Table 9 is a summary of the values used in the $SGV_{(health)}$ calculations. Note that in the specific case of dioxins, the only vegetable considered is the cucurbit family (eg, marrows and pumpkins).

Table 9: Default produce consumption rates: weights in grams with percentage of total vegetables given in parenthesis

Produce type	Wet weight (g/day) and percentage		Dry weight (g/day) and percentage	
	Average adult	Average child	Average adult	Average child
Tuber vegetable	92 (36%)	33 (435)	18.9 (56%)	6.6 (63%)
Root vegetables	18 (7%)	9 (12%)	1.9 (6%)	1.0 (10%)
Above-ground vegetables (not including cucurbits)	119 (47%)	25 (33%)	10 (30%)	2.4 (23%)
Cucurbits (eg, pumpkin)	14 (6%)	6 (8%)	1.4 (4%)	0.46 (4%)
Subtotal	**243**	**73**	**32.2**	**10.46**
Unlikely to be grown at home	10 (4%)	3.6 (5%)	1.4 (4%)	0.57 (5%)
Total	**253**	**76.6**	**33.6**	**11.03**

In dry-weight terms, the percentages of vegetable types likely to be grown at home have been rounded to tuber: 60 per cent; root: 10 per cent; and above-ground including cucurbits: 30 per cent. Where it has been important to differentiate cucurbits, their contribution has been taken as 4 per cent.

Proportion of home-grown produce

For the countries that take consumption of home-grown produce into consideration for their soil guideline values, the proportion of home-grown produce consumed ranges from 10 to about 28 per cent. This difference is largely related to whether average consumption is assumed, or whether the proportion of home-grown produce consumed is based on the small proportion of homes that do grow a significant proportion of their own vegetables. For example, survey data in the Netherlands show home-grown vegetables contribute 54.8 per cent and 13 per cent of the total consumption of vegetables and potatoes, respectively, for about 18 per cent of residents. Expressed as an average of the entire population, home-grown produce contributes 7.1 per cent of the total consumption, which is similar to the value of 10 per cent used in the CSOIL model (Otte et al, 2001; Brand et al, 2007).

A survey in the UK found that a similar proportion (15 per cent) of the population grew their own vegetables. However, in the old CLEA model (Defra and EA, 2002a), the proportion of home-grown produce is based on the six vegetables most commonly consumed by people growing their own. This equates to around 28 per cent of produce consumed for that group, which determines the soil guideline value for all as a high-end value. The updated CLEA model (EA, 2008a) divides the proportion of various produce types grown at home into subgroups by type of vegetable or fruit: leafy, root and tuber vegetables, and herbaceous and shrub fruit. Using survey data, EA (2008a) assumes that between two and six per cent of vegetables, depending on type, and six per cent of herbaceous fruit (eg, courgette, pumpkin, tomato) are grown in a typical home garden situation. However, for those households with allotments, EA (2008a) assumes greater proportions of 13 to 40 per cent for vegetables (depending on type) and 40 per cent for herbaceous fruits.

The *Exposure Factors Handbook* (US EPA, 1997) reports that about 40 per cent of households had a vegetable garden in 1986. Further data on the types of produce grown and consumed by the United States population is also given. This information is intended for use on a site-specific basis as opposed to a generic scenario (US EPA, 1997).

The applicability to New Zealand of produce consumption proportions from overseas is dubious, given the different cultures and opportunities for gardening. However, there is limited information on the proportion of residents who grow their own produce in New Zealand; what information there is comes from specific studies undertaken by local or regional councils. For example, surveys in Hamilton and Christchurch found between about one-fifth and one-third of households grew their own vegetables. One survey, conducted in five new substantial subdivisions near the periphery of Christchurch, found that garden areas where home-grown produce was cultivated ranged from 1 to 64 square metres, with the average slightly under 3 m^2: 90 per cent were smaller than 10 m^2. Estimates from garden guides and personal knowledge suggests that the household garden size required to produce 50 per cent and 10 per cent of home-grown vegetables is of the order of 45–50 m^2 and 9–10 m^2, respectively. These areas are consistent with those reported in Defra (2006a) required to grow potatoes for a family.

A similar survey was carried out by the Hastings District Council. Residents of 121 out of 300 properties surveyed in Havelock and Hastings responded to a questionnaire requesting information on area of vegetable garden, types of produce grown, over what period crops were harvested, and an estimate of the percentage vegetables grown at home (Philip McKay, Hastings DC, pers. comm). While not a scientifically robust survey, the combination of area and harvest gave some check on residents' estimates. Sixty per cent of households claimed to grow vegetables, growing on average 23 per cent of their produce at home (although harvesting was not throughout the whole year). However, adjustment for their area of garden often showed there was unlikely to be sufficient garden area to produce the volume of vegetables claimed. After adjusting for garden area, the survey suggested perhaps 6 or 7 per cent of vegetable consumption was grown at home.

Current New Zealand guidelines use proportions of home-grown produce of 10 and 50 per cent for standard and rural residential, respectively. There is no substantial basis for selecting these proportions and further data should be collected. In the absence of this data, estimates of the proportion of home-grown vegetables are largely subjective. Based on limited data and anecdotal information, it is unlikely that more than 10 per cent of the produce consumed by the average urban resident is home-grown. Lifestyle blocks and farms enable a greater proportion of produce to be home-grown and consumed, yet it is debatable whether the 50 per cent proportion used in some earlier guidelines is representative of the current use of lifestyle blocks: it probably tends towards a high-end estimate. The 100 per cent home-grown produce in the agricultural scenario in the 'Timber Treatment Guidelines' (MfE and MoH, 1997) is excessive, being rarely if ever achieved in practice.

In the absence of more definitive data, it is considered appropriate to continue to use a fraction of 10 per cent produce for home-grown produce in residential scenarios. This is essentially a policy decision rather than a science-based decision.

Given the considerable uncertainty as to the proportion of produce that might be grown in a rural-residential situation, a particular proportion of home-grown produce cannot be recommended. For the purposes of illustration, 50 per cent has been used to derive SGVs as probably being conservative for most situations. In practice, the site assessor should make a judgement based on site information as to what proportion is appropriate: whether 10 per cent as for the standard residential scenario, 50 per cent as provided for conservative illustrative purposes, or some other proportion. If the latter, a site-specific derivation should be carried out.

For the rural residential scenario, a 50 per cent home-grown produce value is considered sufficiently conservative to also cover uptake into home-grown eggs, except for lipophilic contaminants (eg, dioxins). For sites where such contaminants are at significant concentrations, exposure should be considered on a site-specific basis for home-grown products such as eggs, poultry and dairy.

Soil attached to produce

Soil attached to produce may be one source of inadvertent soil ingestion and should be included in the SGV derivation in some circumstances. There is no need to include attached soil if the contaminant-specific values of the bioconcentration factor have been derived from empirical studies or measurements of contaminants in the produce and soil – whether for metals or organics. In that case the attached soil will be 'built in' to the derived BCF. However, consideration of attached soil needs to be included when BCFs are theoretically derived using partitioning relationships and the like, and therefore have not automatically considered attached soil. The following discussion relates to the latter case.

Soil may adhere to the skin of root vegetables (eg, carrots and potatoes) and on leafy vegetables, the latter from direct contact, rain splash or dust deposited on exposed surfaces. Although peeling and/or washing vegetables will reduce the amount of attached soil, it is likely that a residual amount remains. For example, Sheppard and Evenden (1992, cited in Defra and EA, 2002a) estimated the soil loading on thoroughly washed beet leaves was 2 mg soil per gram fresh weight beet (mg soil/g FW).

The old UK CLEA model (Defra and EA, 2002a) used soil loadings from 0.1 mg soil/g FW for root vegetables up to 1 mg soil/g FW produce for stem vegetables for the six vegetables considered in the exposure model. Conversion to dry weight results in soil loadings values of 0.1 per cent for root vegetables (which are assumed to have been peeled) and 1 per cent for leafy vegetables. The revised UK CLEA model (EA, 2008a) assumes dry weight soil loadings of 0.1 per cent for all produce types and then applies dimensionless 'preparation factors' ranging from 0.2 (green vegetables) to 1.0 (roots and tubers) to account for washing and peeling. The Dutch CSOIL model, following revision in 2001, assumes attachment of soil to leafy vegetables due to 'rain splash' amounting to one per cent of the vegetable dry weight. However, this is only applied for organic contaminants, as the empirically derived bioconcentration factors for metals are considered to include an allowance for attached soil (Otte et al, 2001; Brand et al, 2007).

Applying the current UK values to New Zealand produce consumption rates results in the equivalent of an additional soil ingestion rate of about 38 mg/day for an adult and 8 mg/day for a child, if 100 per cent of produce is grown at home (an extreme case), and correspondingly less for lower produce percentages. When converted to a body weight basis the incremental amount is virtually identical for adults and children. The amount of additional soil is significant for high percentages of home-grown produce but only marginally significant for 10 per cent produce (3.8 mg/day for an adult and 0.8 mg/kg for a child).

5.4.3 Dermal exposure

Dermal exposure to soil contaminants can result in acute effects (eg, dermatitis) on the skin, or it may contribute to cumulative exposure. The latter is the most common way to assess dermal exposure for exposure assessments. However, where acute effects occur, it may be appropriate to consider the dermal exposure pathway separately and derive soil guideline values to prevent these effects from occurring.

Three factors are typically used to estimate exposure via dermal absorption:

- the area of skin exposed
- the amount of soil that adheres to the skin
- the absorption rate of the individual contaminant.

The first two are discussed in more detail below and the contaminant-specific absorption factor is discussed in section 5.5.2.

Area of skin exposed

The area of skin exposed depends on the receptor considered (ie, child or adult) and on assumptions about what parts of the body are exposed. Table 10 provides a summary of the area of skin exposed, and the assumed exposed body parts used in national and international protocols. Most protocols assume exposure of the face or head, hands, forearms and lower legs of child residents, and exposure of fewer body parts for adult residents and workers.

The value for exposed skin surface area used in current New Zealand industry-based guidelines, except the *Sheep-dip Guide*, is based on Langley (1993), which in turn was based on ATSDR data (ATSDR, 1992). The ATSDR data assumes that 30 per cent of the average total skin surface area for a child aged 1–11 years is exposed, while 24 per cent of the skin surface area of an adult male is exposed. Older US EPA guidance (US EPA, 2001a) used a value of 2800 cm^2 for the exposed skin surface area for children, which corresponds to exposure of about 43 per cent of the total skin surface area for children aged 0–6 years (6560 cm^2, average of 50th percentile values for males and females). This value is based on data provided in the *Exposure Factors Handbook* (US EPA, 1997) for the skin surface area of children aged 0–6 years, although this value overestimates the exposed skin surface area because the skin surface area of a three-year-old child is used to estimate the skin surface area for children aged less than three years. Skin area used in existing New Zealand and international protocols is summarised in table 10.

A revised approach to dermal exposure has more recently been taken by US EPA (2004c), where the amount of skin exposed depends upon the exposure scenario. Clothing is expected to limit the extent of the exposed surface area in cases of soil contact. All skin area estimates are 50th percentile values to correlate with average body weights used for all scenarios and pathways. Skin area is closely correlated with body weight.

Body-part-specific skin areas are presented in US EPA (2004c) calculated for an adult (>18 years old) and a child (<1 to <6 years old), based on data from the *Exposure Factors Handbook* (US EPA, 1997). No New Zealand data exists for skin area. Given skin area is related to weight and it has been decided to use a 15 kg child and a 70 kg adult, the US EPA data may be used to calculate skin areas. Any error between New Zealanders and Americans will be small compared with errors elsewhere in the dermal exposure pathway (such as the contaminant-specific absorption factors).

Table 10: Area of exposed skin and body parts considered to be exposed for different receptors, used in national and international protocols

Pathway	Child		Adult – resident		Adult – worker	
	Exposed skin surface area (cm^2)	Area exposed	Exposed skin surface area (cm^2)	Area exposed	Exposed skin surface area (cm^2)	Area exposed
New Zealand (based on Langley, 1993)	2,625	30% of total body surface area for ages 1–11	4,700	24% of total body surface area for males	4,700	24% of total body surface area
Environmental Risk Management Authority New Zealand	14,000	100% skin surface of 10-year-old child	4,500	25% of total body surface area averaged for females and males: assumes head, hands, forearms, lower legs and exposure under clothing	NA	
US EPA (2001a)	2,800	Head, hands, forearms, lower legs, feet	5,500	Forearms, head, hands, lower legs	3,300	Forearms, head, hands
Canada	2,600	Head, arms, hands, lower legs	4,300	Head, arms, hands	4,300	Head, arms, hands
UK[1]	466	23% hands, forearms, lower legs	293	Hands 5%	293	Hands 5%
Netherlands	2,800 50	Outdoor Indoor	1,700 90	Outdoor Indoor	NA	

1 Assumes that soil adheres to only one-third of the total exposed skin surface (Defra and EA, 2002a).

NA = not applicable.

US EPA (2004c) assumes a child resident will be wearing a short-sleeved shirt and shorts, but no shoes. Therefore, the exposed skin is limited to face, hands, forearms, lower legs, and feet. This is probably a high-end estimate for year-round exposure, more clothes being worn in the winter. Arguably, it may be appropriate to reduce the average skin area by some factor (eg, two) to allow for more clothes in winter; however, some people may tend to wear fewer clothes all year round in the warmer parts of New Zealand. The dermal exposure route is generally minor and attempting to modify the skin areas for different times of the year is probably an unnecessary refinement.

Table 11 shows the body-part-specific skin areas for a child aged 1–2 and 2–3 years, the two age ranges closest to a 15 kg weigh, taken from US EPA (2004c). The mean of the two ranges will sufficiently approximate a 15-kg child. The total, 1900 cm^2, is proposed as the skin area for a child in both residential and recreational settings. This is smaller by about 30 per cent than all the existing New Zealand guidelines.

Table 11: Body parts exposed for a child and associated skin areas (cm^2)

Age	Face	Forearms	Hands	Lower legs	Feet	Total
1–2	325	346	336	544	371	1,921
2–3	280	314	313	550	418	1,874
Mean	302	330	324	547	394	1,897

US EPA (2004c) assumes the adult resident wears a short-sleeved shirt, shorts and shoes; therefore, the exposed skin surface is limited to the head, hands, forearms and lower legs. The rationale for including the head area is not given. The average skin area for males and females for the face, hands, forearms and lower legs is given as 4850 cm^2 in US EPA (2004c), based on data from US EPA (1997). It is proposed to use this value for the adult resident.

The adult commercial / industrial worker was assumed to wear a short-sleeved shirt, long pants, and shoes. Therefore, the exposed skin surface was limited to the face, hands and forearms. It is probable that in New Zealand an outdoors worker would commonly wear shorts in summer. It is therefore proposed that the skin area be the total of face, hands, forearms and half the area of the lower legs, the latter to allow for the lower legs being exposed half the year. For the outdoor worker the area will therefore be 3670 cm^2. This skin area has also been adopted for adults playing outdoor sports throughout the year, on the assumption that wearing shorts is common while participating in active sports.

Soil adherence

The amount of soil that adheres to the skin will influence dermal exposure to soil contaminants. This factor differs for different activities and skin surfaces (eg, hands, forearms, lower legs). The soil adherence factor currently used in existing New Zealand guidelines, except the 'Sheep-dip Guide' for the residential scenarios (0.5 mg/cm^2), is originally sourced from Hawley (1985) and is based on the amount of soil adhering to a child's hands. A greater amount of soil adheres to hands and feet compared to other parts of the body (eg, face, forearms, legs) so extrapolating the soil adherence factor based on hands to the rest of exposed skin overestimates soil adherence. For this reason, recent US EPA guidance for estimating dermal exposure has proposed revised soil adherence factors expressed as surface-area-weighted values.

US EPA (2004c) presents the data from a number of studies of children playing in soil and in childcare centres, and for adults carrying out various activities: mean and 95th percentile soil adherences have been calculated for various body parts. Body-part-weighted soil adherences for children and adults have then been calculated, using the body parts exposed as in the previous section. The central tendency and 95th percentile values are reproduced in table 12 for standard residential and industrial scenarios.

The 95th percentile values for adults and children from US EPA (2004c) were used for the existing 'Sheep-dip Guide' (MfE, 2006a). However, as a policy decision it had been decided to use central estimates for these factors in that protocol. The resulting skin adherences for adult resident and worker are 0.01 mg/cm^2 and 0.02 mg/cm^2, respectively, and for a child 0.04 mg/cm^2.

These values are not necessarily appropriate for all the proposed scenarios, particularly the recreational and outdoor worker scenarios: here soil adherence is expected to be significantly greater than the residential or the US EPA industrial scenario. US EPA (2004c) provides weighted-average values for various outdoor and sports activities. A selection of the activities is also shown in table 12.

Table 12: Estimated soil adherence factors (mg/cm^2)

Receptor	Central tendency	95th percentile
Residential adult	0.01	0.07
Industrial	0.02	0.2
Grounds keepers	0.02	0.1
Gardeners	0.1	0.5
Utility workers	0.2	0.1
Rugby player	0.1	0.6
Soccer player	0.01	0.08
Residential child	0.04	0.2

Source: US EPA (2004c).

The residential value for a child is proposed for all scenarios involving a child, except the high-density residential scenario – for which an arbitrary halving of the residential value is proposed. The high-density residential skin adherence value for adults is also arbitrarily half the standard residential value.

For the adult recreational scenario, an average between summer and winter is needed. Using a rugby player as the basis and assuming an adherence of 0.01 mg/cm^2 for summer, an average soil adherence of 0.06 mg/cm^2 has been adopted.

For most substances, dermal exposure is a minor pathway and the exact soil adherence values will make little difference to the SGVs for the residential and child recreational scenarios.

The proposed outdoor worker scenario is expected to cover a caretaker carrying out a variety of routine outdoor maintenance work, including gardening and occasional excavation; it will be conservative for a worker on an unpaved commercial / industrial site. A combination of soil adherence values for grounds keepers (most of the time), gardeners (once a week), and utility workers (five times per year) results in a weighted-average value of 0.04 mg/cm^2. The proposed values are summarised in table 13.

Table 13: Soil adherence values (mg/cm^2)

Scenario	Adult	Child
Rural and residential	0.01	0.04
High-density residential	0.005	0.02
Parks / recreational	0.06	0.04
Outdoor worker	0.04	–

5.4.4 Inhalation of particulates and volatiles

As noted earlier, exposure from inhalation of particulates is so small relative to the soil ingestion pathway that it is not worth calculating for most generic $SGVs_{(health)}$. Using the various parameters from the existing 'Timber Treatment Guidelines' (MfE and MoH, 1997) for illustrative purposes, the high dust environment of the commercial / industrial maintenance scenario contributes only 1.6 per cent to the combined soil ingestion and inhalation pathways. However, the exposure contribution from the inhalation pathway should be checked where the inhalation reference health standard is orders of magnitude lower than the oral reference health standard. In addition, there may be particularly dusty situations where inclusion of the inhalation pathway in site-specific assessment is appropriate. Inhalation rates are therefore summarised here for these situations.

Inhalation rates are also important for assessing the effects of inhalation of volatile compounds. While $SGVs_{(health)}$ for volatile compounds are not being calculated in the current document, the inhalation rates given here are appropriate for that purpose.

Inhalation rates

Inhalation rates for existing guidelines for various countries are summarised in table 14. A direct comparison between the guidelines is difficult because of the way the different models and methodologies use the values: as a combination of intake values, worst case for child and adult, combining indoor and outdoor, or treating indoor and outdoor separately for volatiles.

The *Exposure Factors Handbook* (US EPA, 1997) provides a review of inhalation studies. From that review a summary of inhalation rates for children of various ages and adults in different levels of activity was developed. The summary is reproduced in table 15. Daily rates are presented for long-term exposures, and hourly rates for short-term activities, as might be expected in an outdoor occupational setting when carrying out physical work.

The long-term average daily rates are applicable for residential setting, and are lower than the default values adopted in the *Soil Screening Guidance* (US EPA, 1996a, 1996b, 2002a) for adults as well as children (assuming the 15-kg child falls within the 1–2 year age range). The US EPA (1997) acknowledges this and recommends the summary values to be used in site-specific assessment; and also recommends that mean values, rather than high estimates, are employed. Taking the average of the male and female long-term rates as appropriate for a 70-kg adult, a rate of 13.3 m^3/day is obtained, whereas 6.8 m^3/day is appropriate for a 15-kg child in a residential setting.

The former value is lower than rates used in current New Zealand guidelines. The 20 m^3/day used for the calculation of outdoor inhalation of volatiles in the *Gasworks* and 'Oil Industry Guidelines' (MfE, 1997, 1999) is particularly unrealistic. Based on the US EPA figures, it suggests that on average, adults would spend a good part of their daylight hours outside carrying out moderate to heavy activities; whereas the reality is that an average adult is more likely to spend most time either inside the house or off-site at work.

It is notable that the Dutch CSOIL model assumes that an adult spends only an average of one hour outside each day at home, whereas the UK model does not consider adult inhalation exposure: a child is considered the critical receptor instead.

The US EPA young child rate of 6.8 m^3/day is nearly twice that currently used in all the New Zealand guidelines except the *Sheep-dip Guide*, which uses a rate identical to the Dutch rate of 7.6 m^3/day (although no source is cited for the value). The UK child inhalation rate falls between the New Zealand rates. It is proposed that a child rate of 6.8 m^3/day be used for child inhalation in a residential setting.

When the methodology for calculating soil guidelines for exposure to volatiles is reviewed, the apportionment of the inhalation rate to indoor and outdoor residential activities will need to be considered.

For adult occupational exposure, the *Exposure Factors Handbook* (US EPA, 1997) recommends a 1.3 m^3/hour average for outdoor workers. Using this rate, an occupational exposure of eight hours gives a daily inhalation rate of 10.4 m^3/day. This is similar to the 9.6 m^3/day values used in existing New Zealand industry-based guidelines. An indoor worker is likely to have an inhalation rate of between 0.5 and 1 m^3/day (table 15). Taking the higher rate as conservative, results in an indoor worker inhalation rate of 6 m^3/day.

Inhalation of either particulates or volatiles is not likely to be a significant factor for the parks / recreation scenario and therefore no generic inhalation rate needs to be determined. Site-specific assessment of volatiles is recommended for the rare occasions when particulates or volatiles might be important in a parks / recreational setting.

Table 14: Summary of daily inhalation rates in existing New Zealand and international guidelines (m^3/day)

	New Zealand[a]				USA	Canada	Netherlands	UK
	TTG	GWG	OIG	SDG	SSL	SQG	CSOIL [b]	CLEA [c]
Residential								
Child indoor	3.8 [d]	3.8 [d]	3.8 [d]	7.6 [d]	10 [d]	–	6.7	5
Child outdoor							0.9	
Adult indoor	20 [d]	15	15	25 [d]	20 [d]	–	19	–
Adult outdoor		20	20			23	1	–
Parkland / recreation								
Child	–	1.1	–	7.6	–	5	0.32	–
Adult		2.4		25		23	0.83	
Commercial Industrial								
Adult indoor	9.6 [d]	10 [e]	10 [e]	10.4 [d]	20 [e]	23 [e]	5	4.7 [f]
Adult outdoor		10	10		20	23	0.8	0.95

a TTG = 'Timber Treatment Guidelines' (MfE and MoH, 1997), GWG = 'Gasworks Guidelines' (MfE, 1997), OIG = 'Oil Industry Guidelines' (MfE, 1999), SDG = 'Sheep-dip Guide' (MfE, 2006a).

b CSOIL model calculates a combination of child and adult and indoor and outdoor exposure.

c Calculated using means values of probability distributions of inhalation rates and exposure times.

d No differentiation between indoor and outdoor.

e Soil guideline values for indoor and outdoor exposure calculated as separate scenarios.

f Calculated as combined indoor and outdoor exposure.

Table 15: Inhalation rates recommended in *Exposure Factors Handbook*

Short-term exposures		Long-term exposures		
Population	Mean rates (m^3/hour)	Population		Mean rates (m^3/day)
Adults		**Children**		
Rest	0.4	Infants <1 year		4.5
Sedentary activities	0.5	1–2 years		6.8
Light activities	1.0	3–5 years		8.3
Moderate activities	1.6	6–8 years		10
Heavy activities	3.2	9–11 years	Male	14
			Female	13
Children		2–14 years	Male	15
Rest	0.3		Female	12
Sedentary activities	0.4	15–18 years	Male	17
Light activities	1.0		Female	12
Moderate activities	1.2			
Heavy activities	1.9			
Outdoor workers		**Adults**		
Hourly average	1.3	19–65+ years	Female	11.3
Slow activities	1.1		Male	15.2
Moderate activities	1.5			
Heavy activities	2			

Source: US EPA, 1997.

Particle emission factor

Existing New Zealand guidelines use particle emission factors based on US EPA studies and Australian measurement of dust concentrations (eg, see MfE and MoH, 1997 – chapter 5, Appendix C). The assumptions in MfE and MoH (1997) around what proportion of respirable dust comes from the contaminated site appear to have little basis. For example, it is assumed that 20 per cent of respirable dust is from soil sources, based on Australian measurements; this ignores the probability that for all but the largest sites, much of the soil in air-borne dust will be from uncontaminated off-site sources.

Consideration of the particulate inhalation pathway is only necessary for site-specific assessment of dusty sites. Rather than use generic particulate emission factors, it is therefore more appropriate to directly measure site-specific dust concentrations and the contaminant concentrations within the dust, and use these to estimate the intake of dust-borne contaminants.

5.5 Contaminant-specific factors

Plant uptake factors

Perhaps the greatest uncertainty in determining uptake of a contaminant in produce is selecting the plant uptake factors, otherwise known as bioconcentration factors (BCF).

Plants can accumulate contaminants via a number of pathways, the most important of which is typically absorption by roots (Kabata-Pendias and Pendias, 2000). Uptake of organic contaminants and metals occurs predominantly from the soil solution. Normally the concentration of a contaminant measured in the soil solution represents only a fraction of the total contaminant present in the soil. The ratio of the concentration in soil solution to the total in soil depends on a number of factors including soil pH, redox potential, soil organic matter, and soil texture (Kabata-Pendias and Pendias, 2000). In soils and sediments where the clay content is relatively low, the availability of organic contaminants is strongly related to the fraction of organic carbon present.

The uptake of contaminants from soil and transport within plant tissues also differs for different plant species, as well as for different contaminants and different soil types. Ideally, because of the complexity of soil-plant systems, the concentration of soil-derived contaminants in vegetables would be based on site-specific measured data. However, this is not practical for deriving generic assessment criteria, and there is a need to provide mechanisms for estimating plant uptake to enable the derivation of such criteria. A general review of the various BCF derivation methods is provided here, while specific discussion on the contaminant-specific BCFs used for each $SGVs_{(health)}$ derivation is provided against each contaminant in section 6.

The simplest empirical relationship for estimating plant uptake of soil contaminants follows the form:

$$C_P = BCF \times C_s$$

where C_P = concentration in produce (mg/kg)

C_s = concentration in soil (mg/kg)

BCF = bioconcentration factor – typically expressed as the ratio of the contaminant content in produce (mg/kg dry weight) and soil (mg/kg dry weight); if either produce or soil concentration is other than dry weight, the BCF units will reflect this.

Typically the BCF for metals is based on field or laboratory experiments, while the BCF for organics is more often estimated from the octanol-water partition coefficient (K_{ow}).[9]

For metals where field or laboratory data is available, point estimates of the BCF for a given metal have frequently been used by regulatory agencies. A number of different point estimates may be used including:

- geometric mean / median / mean of all available data
- geometric mean / median / mean of available data for selected vegetables of interest
- separate point (geometric mean) estimates for root and stem vegetables
- weighted-average BCF based on the consumption rates of different vegetables.

Which point estimate is used depends on the available data and the methodological approach used to estimate contaminant intake via consumption of home-grown produce. For example, the UK bases estimates of contaminant intake from produce consumption on six vegetable and fruit groups that are commonly grown and consumed in the UK (EA, 2008a). In contrast, a consumption-weighted BCF that is also dependent on soil properties has been proposed for use in the Dutch CSOIL model (Otte et al, 2001).

For organics, there is a paucity of data on plant uptake, hence estimation of the plant uptake of organic contaminants is typically undertaken using models based on the octanol-water partition coefficient (K_{ow}) of the individual contaminants. These models tend to over-predict the bioconcentration factors compared to real data (Otte et al, 2001). The models include the relatively simple model of Travis and Arms (1988), which is an empirical relationship between the bioconcentration factor for organic contaminants in above-ground plants parts and the K_{ow}:

$$\log B_v = 1.588 – 5.78 \log K_{ow}$$

where: B_v = bioconcentration factor for above-ground plant parts (dry weight basis)
K_{ow} = octanol-water coefficient.

More complex models, such as that of Trapp and Matthies (1995), consider the sorption of contaminants to plant fats. See Rikken et al (2001) and EA (2006) for further discussion on plant uptake models for organic contaminants. While a considerable amount of research on plant uptake of organic contaminants has been undertaken, there is generally a lack of agreement with measured data (EA, 2006).

EA (2006) reviewed a number of models for the uptake of organic compounds in plants, comparing their predictions with actual measurements in the literature of uptake into common vegetable types from contaminated soil in both field and laboratory studies. Unfortunately, many of these studies were limited to persistent chemicals such as polycyclic aromatic hydrocarbons, polychlorinated biphenyls and dioxins / furans. Little data was available on common industrial chemicals, including petroleum hydrocarbons and low-molecular-weight chlorinated compounds. The authors were surprised to find they had to reject many studies because these did not report whether the results were on a fresh-weight or dry-weight basis, or whether the measurements were of roots, shoots, fruits or tubers. This indicates a general need for care if attempting to select representative uptake factors from the literature for a particular substance.

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9 The octanol-water partition coefficient is a measure of the relative affinity an organic compound has to bind to organic material (eg, in soil) or dissolve in water.

EA (2006) concluded that model performance was highly variable. All except one of the six models reviewed over-predicting root uptake by a least an order of magnitude. The exception, Travis and Arms (1988), only applies to above-ground parts.

Given this, the recommended approach is to simply use BCFs based on available data, and only resort to models when measured values are not available. As noted above, bioconcentration factors for the particular contaminants considered in this protocol, and justifications for the chosen values, are given for each $SGV_{(health)}$ derivation in section 6. Two elements excepted, all the $SGV_{(health)}$ derivations in this document use BCFs estimated from experimental data specific to the contaminant; BCFs were not used for the derivation of the copper and boron $SGVs_{(health)}$.

5.5.2 Skin absorption factors

Dermal uptake of soil contaminants is largely dependent on the physico-chemical properties of individual contaminants, although it can be modified by soil loading on the skin and exposure duration (eg, McKone, 1990).

An absorption factor (expressed as a percentage or fraction) is generally used to estimate dermal uptake when deriving generic soil guideline values. Factors specific to the individual priority contaminants are given in MfE (2010b), where more detailed discussion is provided for the individual priority contaminants. A brief general discussion is provided below.

Absorption rates need to be determined on the basis of available information, which is often limited. Further, dermal absorption studies are often only available for animals, and the dermal permeability of different animals can be different to that of humans. Pig and guinea pig are suggested to be most representative of absorption across human skin (McKone, 1990). Expert judgement is required to interpret available dermal studies. Adsorption on soil particles may reduce the dermal uptake of some contaminants: eg, Hawley (1985) suggested that only 15 per cent of contaminants adsorbed onto soil would be dermally absorbed. Yet for some contaminants this reduction in dermal uptake may be insignificant. For example, pentachlorophenol (PCP) from contaminated soil was absorbed at a similar rate to PCP in acetone, with 24 per cent of PCP absorbed over a 24-hour period (Wester et al, 1993).

Detailed models for estimating the dermal absorption of (primarily organic) contaminants (eg, McKone, 1990; US EPA, 2001a) are available. These models take into account factors such as skin thickness, chemical properties of the contaminants (eg, octanol-water partition coefficient, K_{ow}), and soil loadings. Generally, these models are unnecessarily complex in terms of developing a generic soil guideline value. Nevertheless in the absence of data for specific contaminants, the model results can provide a useful insight into potential dermal uptake of contaminants. For example, based on model estimates from McKone (1990) for compounds with a log K_{ow} of 6 and below and a K_h (Henry's Law Coefficient) below 0.001, it is reasonable to assume 100 per cent absorption in 12 hours. For compounds with a K_h of 0.01 and above, the uptake fraction is unlikely to exceed 40 per cent in 12 hours; for contaminants with a K_h of 0.1 and above, uptake of no more than 3 per cent in 12 hours is expected.

In the case of $SGVs_{(health)}$ derived in this document, dermal absorption factors were determined for seven of the 12 contaminants. Of these, the dermal pathway had no significant effect on the derived SGVs for two of the contaminants (arsenic and cadmium), a marginal effect on the SGVs for four of the contaminants (benzo(a)pyrene, DDT, dieldrin and dioxins), and a significant effect on the derived SGV for PCP. In all cases, dermal absorption factors were obtained from the literature, rather than resorting to models.

5.6 Summary of exposure factors

A summary of the recommended general and scenario-specific exposure parameters is presented in table 16.

Table 16: General and scenario-specific exposure parameters

Generic factors							
Body weight (child):	15 kg		Averaging time (non-threshold):	75 years			
Body weight (adult):	70 kg		Averaging time (threshold):	6 years			
Scenario-specific factors	**Lifestyle block**	**Standard residential**	**High-density residential**	**Parks / recreational[1]**	**Commercial / industrial indoor worker**	**Commercial / industrial outdoor worker**	**Unit**
Exposure frequency	350	350	350	200 / 150	230	230	day/year
Exposure duration (child)	6	6	6	6			years
Exposure duration (adult)	24	14	14	14	20	20	years
Soil ingestion rate (child)	45	45	25	15			mg/day
Soil ingestion rate (adult)	25	25	15	75	0	50	mg/day
Age-adjusted ingestion factor	26.6	23.0	13.0	6.0 / 15.0	0	14.3	mg year/kg day
Inhalation rate (child)	6.8	6.8	6.8	6.8			m^3/day
Inhalation rate (adult)	13.3	13.3	13.3	20	8	10.4	m^3/day
Age-adjusted inhalation rate	7.3	5.4	5.4	2.7 / 4.0	2.3	3.0	m^3 year/kg day
Particulate retention	0	0	0	0	0	0	dimensionless
Particle emission factor							m^3/kg
Skin area (child)	1,900	1,900	1,900	1,900			cm^2
Skin area (adult)	4,850	4,850	4,850	3,670	3,670	3,670	cm^2
Soil adherence (child)	0.04	0.04	0.02	0.04			mg/cm^2
Soil adherence (adult)	0.01	0.01	0.005	0.06	0	0.04	mg/cm^2
Age-adjusted dermal exposure factor	47.0	40.1	20.1	30.4 / 44	0	41.9	dimensionless
Produce ingestion (child)	0.0105	0.0105					kg/day (DW)
Produce ingestion (adult)	0.0322	0.0322					kg/day (DW)
Proportion of above-ground produce	0.3	0.3	0.0	0.0	0.0	0.0	dimensionless
Proportion of root (not tuber) produce	0.1	0.1	0.0	0.0	0.0	0.0	dimensionless
Proportion of tuber produce	0.6	0.6					dimensionless
Age-adjusted produce ingestion	0.0152	0.0106	0.0				kg year/kg day

1 The parks / recreational scenario has alternate scenarios for a child and an adult, calculated separately, with the worst case becoming the guideline value. In this case child and adult parameters are shown as child / adult.

DW = Dry weight.

6 Soil Guideline Values for Selected Priority Contaminants

Soil guideline values have been derived for a group of priority contaminants, specifically arsenic, boron, cadmium, chromium, copper, inorganic lead, inorganic mercury, benzo(a)pyrene, DDT, dieldrin, dioxin (as 2,3,7,8-tetrachlorodibenzo-*p*-dioxin) and dioxin-like polychlorinated biphenyls (PCBs), and pentachlorophenol. All calculations use the updated contaminant-specific toxicological, background intake and skin absorption factors presented in MfE (2010b) and produce bioconcentration factors presented in each contaminant-specific section below.

The details of the derivations are provided in appendix 1 (and appendix 2 for additional calculations for cadmium); summary results for each contaminant are provided against each contaminant below and in an overall summary for all contaminants in section 7.

In the $SGV_{(health)}$ summaries, values have generally been rounded down to two significant digits, with some exceptions for pragmatic reasons. If the value was close to a round number, the value has been rounded up or down to that number, eg, 69 to 70. Values in the hundreds have generally been rounded to the nearest 10 unless close to a multiple of a hundred, eg, 410 becomes 400. For values in the hundreds of thousands, rounding is generally to the nearest 10,000 but again, if it was close to a multiple of 100,000 that value has been chosen, whether up or down. Some intermediate values have been derived as greater than the pure substance (ie, >1,000,000 mg/kg). This is not possible and such values have been shown as 'no limit' (NL).

For values between an arbitrary 30 and 100 mg/kg, rounding has been to the nearest 5 mg/kg. Values below 0.1 mg/kg have been rounded to 1 significant digit. Dioxins are an exception, as these are in units of µg/kg and have been rounded to two significant digits.

A 'no limit' value is shown for the commercial / industrial indoor worker scenario in the $SGV_{(health)}$ summary table for each contaminant. These are not derived values, are but based on the assumption that there is no exposure.

6.1 Arsenic

6.1.1 Arsenic bioconcentration factor

The US EPA (2007) reports a median bioconcentration factor (BCF) of 0.03752 for arsenic, determined from 122 data points with a range of 0.00006 to 9.0741 (Bechtel-Jacobs, 1998). The 'Timber Treatment Guidelines' (MfE and MoH, 1997) use a bioconcentration factor of 0.01 for root vegetables, and 0.002 based on the assumption that bioaccumulation in vegetative parts is one-fifth of that in the roots. In contrast, Baes et al (1984) found a higher BCF in vegetative parts (0.04) as compared to non-vegetative parts (0.006, tubers, seeds, fruits). Dutch agencies have used a BCF of 0.021 in the derivation of the current intervention value, although this has been revised and a BCF of 0.009 has now been proposed for use (Lijzen et al, 2001). This value is a consumption-weighted BCF, however, and the median value of the revised data set is 0.025. The Environment Agency (EA, 2009a) provides the most recent summary of plant BCFs. Their report reviews the existing literature and provides recommendations for BCFs (based on fresh weight) for the edible portion of six produce types, which are used in the CLEA model for the derivation of soil guideline values. The recommended BCFs are the geometric mean of the available data for each produce type.

The BCFs recommended in EA (2009a) are lower than those provided in Bechtel-Jacobs (1998) and used in Otte et al (2001). Close inspection of the Bechtel-Jacobs data indicates that most of it relates to non-vegetables, and thus has arguable application to the current work. The data in Otte et al (2001) appears to be based only on three data points. For the purpose of deriving soil guideline values protective of human health, the recent EA (2009a) review could be used in the derivation of New Zealand soil guideline values. However, there is some New Zealand-specific data (Gaw et al, 2008), which is not included in the EA (2009a) review which indicates a higher BCF to be appropriate. As such, the original references used in EA (2009a) were gathered where possible, and this data along with that of Gaw et al (2008) was used to derive BCFs. The derived BCFs are shown in table 17 along with BCFs determined from other sources.

Table 17: BCF for arsenic from different sources

MfE and MoH, 1997		Dutch (Otte et al, 2001)		US EPA (2007)	UK (EA, 2009) Fresh weight (dry weight)[2]			This study		
Root	Leafy	CW[1]	Median		Green (n=46)	Root (n=26)	Tuber (n=6)	Green (n=29)	Root (n=16)	Tuber (n=2)
0.01	0.002	0.009	0.025	0.0375	0.00043 (0.0043)	0.0004 (0.004)	0.00023 (0.001)	0.011	0.011	0.001

1 Consumption-weighted BCF

2 Calculated using conversion factors of 0.096 kg DW/kg FW, 0.103 kg DW/kg FW and 0.21 kg DW/kg FW for green, root and tuber vegetables, respectively (EA, 2009b).

It should also be noted that Swartjes et al (2007) recently concluded that use of a fixed value for concentration of arsenic in vegetables, that is independent from arsenic soil concentrations and soil properties, is most appropriate for assessing the human health risks due to the consumption of produce from contaminated soil. This is because no significant relationship between vegetable concentration and soil concentration exists.

6.1.2 Arsenic guideline calculations

The soil guideline value calculations for arsenic use the contaminant-specific parameters in table 18 and the derived $SGVs_{(health)}$ are set out in table 19. Arsenic has been treated as a non-threshold substance and therefore a risk-specific dose has been employed. A single BCF value was used for both root and leafy parts of vegetables, with a separate factor for tubers; these are combined into a single consumption-weighted mean value, using weighting factors of 0.4 and 0.6 (dry weight), respectively.

Table 18: Contaminant-specific parameters for arsenic soil guideline value derivation

Risk-specific dose: oral		0.000 008 6 mg/kg BW/day
Background exposure		not applicable (non-threshold substance)
Dermal absorption factor		0.005
Plant bioconcentration factor	Green	0.011
	Root	0.011
	Tuber	0.001
	Consumption-weighted mean	0.005

Table 19: Arsenic soil guideline value (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	25	20[a]	10
Standard residential	29	24[a]	14
High-density residential	50		
Recreational	100		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	70		

a Different rural residential and residential exposure durations result in different SGVs because non-threshold substance SGV derivation uses age-adjusted exposure rates.

NL = No limit.

The controlling pathway for arsenic is soil ingestion. Produce consumption has a significant influence for residential scenarios at high home-grown produce proportions but only a moderate influence at the default proportion of 10 per cent. Site-specific assessment should be considered where high home-grown produce is suspected.

6.2 Boron

6.2.1 Boron bioconcentration factor

The 'Timber Treatment Guidelines' (MfE and MoH, 1997) calculate SGVs using a BCF_{root} of 3 based on a measured range of 1–10 sourced from ECETOC (1990) and a BCF_{stem} of 0.6, the latter derived assuming stem concentrations are 20 per cent of root concentrations. No other regulatory agencies appear to consider boron and plant uptake of boron in the derivation of soil guideline values. The 'Timber Treatment Guidelines' also use a threshold for plant toxicity of boron of 3 mg/kg (as water-soluble boron) sourced from UK Interdepartmental Committee on the Redevelopment of Contaminated Land (ICRCL, exact reference not specified) as the basis for guideline values adopted for boron. These values are no longer considered appropriate.

It has not been possible to develop bioconcentration factors for boron. Reviewing the literature shows that boron uptake into plants is highly variable between species with no relationship with soil concentration or other soil parameters. Boron is an essential element for plant growth, but what may be optimal boron for one species may be toxic or insufficient for other species (Blevins and Lukaszewski, 1998; Nable, 1997). In addition, potential human health effects arising from ingestion of produce containing boron are generally considered to be protected by the soil-plant barrier, where toxicity to the plant will occur at concentrations far lower than what would affect human health (Chaney, 1980 in Langmuir et al, 2004). It is therefore appropriate that a maximal concentration of boron in produce is used in preference to a BCF.

Sensitive crop plants, considered to be cereals and cotton, may be affected by boron concentration in soil solution at 1 mg/L while 5 mg/L may be tolerated by various plant species, include most vegetables, and up to 15 mg/L by tolerant species (Blevins and Lukaszewski, 1998). Nable (1997) indicates that soil containing more than 5 to 8 mg/L hot water soluble boron may require special revegetation requirements.

Langmuir et al (2004) consider that 75 mg/kg represent phytotoxic levels in plants, however Nable (1997) indicates that in species that accumulate boron in their leaves, leaves can contain 250 mg/kg (dry weight) when boron in the soil approaches toxic levels and may exceed 1000 mg/kg (dry weight) in extreme conditions of boron toxicity. In species that do not accumulate boron in their leaves under conditions of toxicity, boron concentrations greater than 300 mg/kg (dry weight) may indicate that boron toxicity is present (Nable et al, 1997). In some species, concentrations of boron in plant tissue that hasn't resulted in toxicity may range up to 4800 mg/kg dry weight in corn (Kabata-Pendias and Pendias, 2000), although other studies using the same species indicate toxicity (yield reduction) at plant concentrations of 100 mg/kg.

Determining the significance of plant uptake of boron to human exposure is difficult, given the wide ranging and overlapping concentrations that determine boron essentiality and toxicity in various species. Nonetheless, it appears that 300 mg/kg is a reasonable upper limit of non-toxic plant boron concentrations and thus can be used as the reasonable maximum amount of boron likely to be taken up in home-grown vegetables. Beyond that point vegetables are unlikely to be harvestable. Alternatively, a hot water-soluble boron concentration of 8 mg/L could be considered as an upper limit of non-toxic concentrations for plants although the relationship between hot water-soluble boron and total boron concentrations is unclear, and likely to be highly variable.

6.2.2 Boron guideline calculations

The soil guideline value calculations for boron use the contaminant-specific parameters in table 20 and the derived $SGVs_{(health)}$ are set out in table 21. Boron has been treated as a threshold substance.

As noted above, BCF values could not be determined for boron and the $SGVs_{(health)}$ calculation has been carried out differently from the other guidelines with respect to the produce consumption pathway, because a produce pathway value cannot be determined as a function of soil concentration. This means that a soil guideline combining soil ingestion and the produce pathway cannot be calculated in the usual way. Instead, a soil ingestion guideline has been calculated for the residential-with-produce scenarios by subtracting a further notional background intake to take into account the amount of produce that could theoretically be consumed if the produce was at the phytotoxic limit of 300 mg/kg tissue concentration. The modified background is subtracted off the TDI in the usual fashion.

To obtain the additional background intake, a child's produce consumption (0.010 5 kg DW/day) was multiplied by 300 mg/kg and divided by the child body weight of 15 kg to obtain the maximum additional background daily intake for 100 per cent of produce being home-grown. This was then multiplied by the home-grown produce percentage relevant to the particular $SGVs_{health)}$.

As the dermal and inhalation intakes are insignificant there is no contribution from these pathways and the result with the modified background becomes the $SGV_{(health)}$ value.

This method breaks down for home-grown produce consumption percentages greater than about 55 per cent because the 'produce background' exceeds the TDI minus the normal background intake, resulting in nonsensical negative SGVs. For a site where greater home-grown produce consumption is a possibility, site-specific assessment will need to be carried out by measuring boron concentrations in relevant plants. This can be used to assess human exposure from plants in addition to soil.

Table 20: Contaminant-specific parameters for boron guideline value derivation

Tolerable daily intake: oral		0.2 mg/kg BW/day
Background exposure	Child Adult	0.08 mg/kg BW/day 0.017 mg/kg BW/day
Dermal absorption factor		0
Plant bioconcentration factors	Root Tuber Leafy	Not applicable Not applicable Not applicable
Additional child background for given produce percentage at produce concentration of 300 mg/kg	10% 50%	0.021 mg/kg BW/day 0.105 mg/kg BW/day

Table 21: Boron soil guideline value (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	42,000	34,000	5,200
Standard residential	42,000	34,000	5,200
High-density residential	75,000		
Recreational	220,000		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	400,000		

Note: NL = No limit.

The controlling pathway for boron is soil ingestion. It should be noted that the derived SGV values for standard residential and non-produce consumption scenarios, while safe for human exposure, are well above the phytotoxic threshold. Plants are unlikely to survive at such high soil concentrations. The concentrations are also well above what would normally be encountered on contaminated sites in New Zealand. However, if high concentrations are encountered on a site, the risk assessor will need to consider whether this could affect the use to which a site could be put. This is unlikely to be a consideration for most industrial or commercial uses.

6.3 Cadmium

6.3.1 Cadmium bioconcentration factors

Cadmium uptake by plants is a function of cadmium concentration in the soil solution, although plant species and cultivars differ widely in their ability to absorb and accumulate cadmium (Kabata-Pendias and Pendias, 2000). Many researchers have found conflicting evidence for the relativity of uptake between different garden vegetables. Cadmium absorption has been shown to depend strongly on soil pH, and to a lesser degree on hydrous oxide and soil organic matter content (Alloway, 1995). Cadmium absorption (ie, plant uptake of cadmium) increases with decreasing pH. This means that in acid soils, produce consumption can represent a significant exposure route for cadmium (Swartjes et al, 2007).

Various regression equations have been developed for regulatory agencies to describe the plant uptake of cadmium in relation to various soil parameters. For example in the Netherlands, Otte et al (2001) and Swartjes et al (2007) used a regression equation describing plant concentrations in relation to four soil parameters: clay, organic carbon, total soil metal concentration and pH. Bechtel-Jacobs (1998) and McBride (2002) developed regression equations for plant uptake based on soil concentrations and pH; in the UK, Defra and EA (2002c) used regression equations for the behaviour of BCF in relation to soil pH for root and leafy vegetables.

Soil pH appears to be a dominant influence of plant uptake of cadmium, with greater uptake at lower pH. Use of the UK equation (Defra & EA, 2002c) in the derivation of soil guideline values in New Zealand is particularly attractive, as it uses only soil pH to describe the BCF. However, the original report describing this approach has now been withdrawn and a revised report on the derivation of soil guideline values for cadmium has recently been released which does not use the BCF-soil-pH relationship (EA, 2009b). Instead the geometric means of BCFs determined from the literature for the edible portion of six produce types are used. The applicability of the Dutch equations to New Zealand soils is debatable as it is based on data from Dutch soils, and the Bechtel-Jacobs relationship was developed for all plant types, not just vegetables.

A review of the primary literature was undertaken and the relationships between plant uptake of cadmium and soil concentrations re-examined using collated data including those for New Zealand. Specifically, only data from field studies was used, ie, studies where cadmium was already present in the soils (as opposed to experimental studies where solutions of cadmium salts are added to the soil: plant uptake of salts often overestimates uptake, particularly for cadmium – Effroysom et al, 2004). Further, only studies that reported soil pH, soil cadmium concentration and cadmium concentrations in the edible portions of plants were used. This resulted in 108 data points from 13 studies, with 51 data points for leafy vegetables and 33 for root and tuber vegetables. These data were used to examine the relationship between plant cadmium concentrations and soil pH. Based on the previous research described above, two relationships were examined using regression analyses:

$$\text{Ln (BCF)} = a + b\ \text{(soil pH)} \qquad \text{Eqn 23}$$

$$\text{Ln (plant Cd)} = a + b\ [\ln \text{(soil Cd)}] + c\ \text{(soil pH)} \qquad \text{Eqn 24}$$

Table 22 provides a summary of the equations developed and the percentage of the variability in the data explained (R^2).

Table 22: Coefficients determined for equations 23 and 24, and the percentage of the variability in the data

Vegetable type	Ln (BCF) = a + b (soil pH)		
	A	b	R^2
All data	6.42	–0.991	23.7
Leafy	6.16	–0.844	23
Root / tuber	7.04	–1.14	40.8

Vegetable type	Ln (plant Cd) = a + b ln(soil Cd)+c pH			
	A	B	C	R^2
All data	5.462	0.6981	–0.859	59.8
Leafy	5.66	0.8783	–0.779	65.9
Root / tuber	4.61	0.6215	–0.777	77.7

As a greater proportion of the data is explained using equation 24, it is considered this general form of equation is appropriate for determining vegetable uptake of cadmium. Further, the equations developed for the different vegetable types (leafy, root / tuber) were used as these equations have a better fit than those developed using all vegetable types.

Further analyses was undertaken using an extended data set and restricted to only leafy and root / tuber data. The equations developed from this data are shown in table 22. The leafy and root / tuber values were used in the SGV derivation.

Table 23: Coefficients determined for equation 24, and the percentage of the variability in the data explained using an extended data set

Vegetable type	Ln (plantCd) = a + b [ln(soilCd)] + c (soil pH)			
	A	b	c	R^2
All data	4.982	0.7279	–0.764	58.0
Leafy	4.79	0.7591	–0.626	68.1
Root / tuber	4.73	0.5995	–0.838	58.6

The BCFs derived using the parameters shown in table 23 are plotted in figures 3 and 4 for leafy and root / tubers, respectively, for soil concentrations ranging from 0.1 to 4 mg/kg at soil pH 5, 5.5, 6 and 6.5. Studies on the properties of New Zealand soils indicate that soils under various land uses are acidic, with typical pH values for individual land types ranging from 5.2 to 6.2 (Sparling et al, 2000; Sparling and Schipper, 2002, 2004).

6.3.2 Cadmium guideline calculations

Cadmium has been treated as a threshold substance. The dependence of plant uptake on both soil pH and soil concentration requires the calculation of SVGs over a range of pH values, with an iterative derivation being employed. Guidelines have been calculated at 0.5 pH unit intervals for the range pH=5–8. As noted previously, New Zealand soils tend to be acidic.

The calculation procedure for a particular soil pH requires an estimated starting soil concentration in order to calculate the BCF. The resultant trial SGV was then compared with the starting concentration: if not within one per cent, a value midway between the starting value and the resultant value was fed back into the calculation (interval halving). This was carried out successively until convergence to a single value was achieved. This typically occurred in fewer than 10 iterations provided the starting value was chosen carefully.

The review of TDI values by MfE (2010b) notes that the WHO/FAO Joint Experts Committee on Food Additives (JECFA) had last determined the TDI for cadmium in 2004. This value (0.001 mg/kg BW/day) is widely used by international agencies and has been adopted by New Zealand for food safety. Recently, however, the European Food Safety Authority (EFSA) reviewed the science and decided to recommend a lower value. In the UK, the Environment Agency has used the new EFSA TDI (0.000 36 mg/kg BW/day) to derive the most recent UK versions of the cadmium SGV (EA, 2009b). JECFA is due to review the cadmium TDI in 2010. SGVs have been derived using the current JECFA TDI as the recommended value, but this is likely to be reviewed if JECFA decides to change its cadmium TDI.

Figure 3: Variation in plant bioconcentration factors (BCF, dry weight) for leafy vegetables with soil concentration and pH

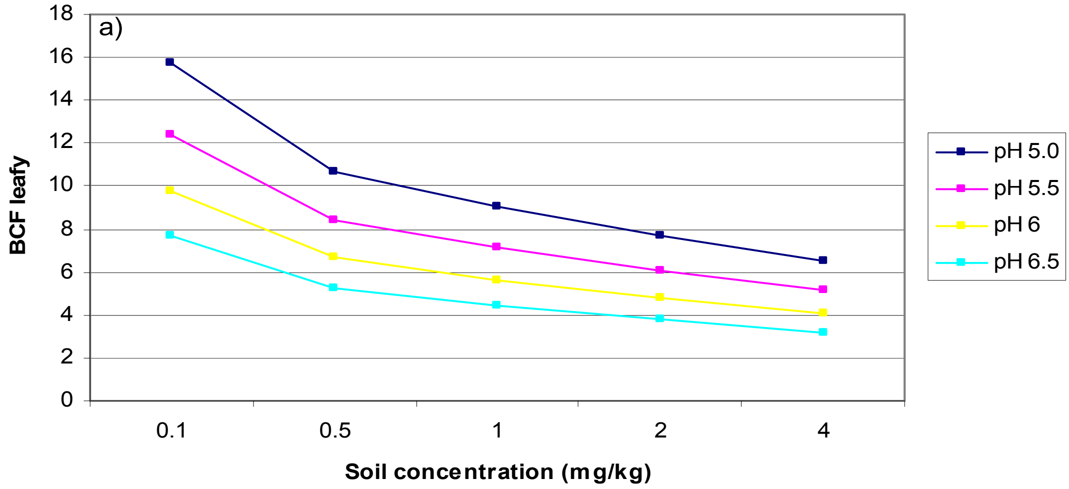
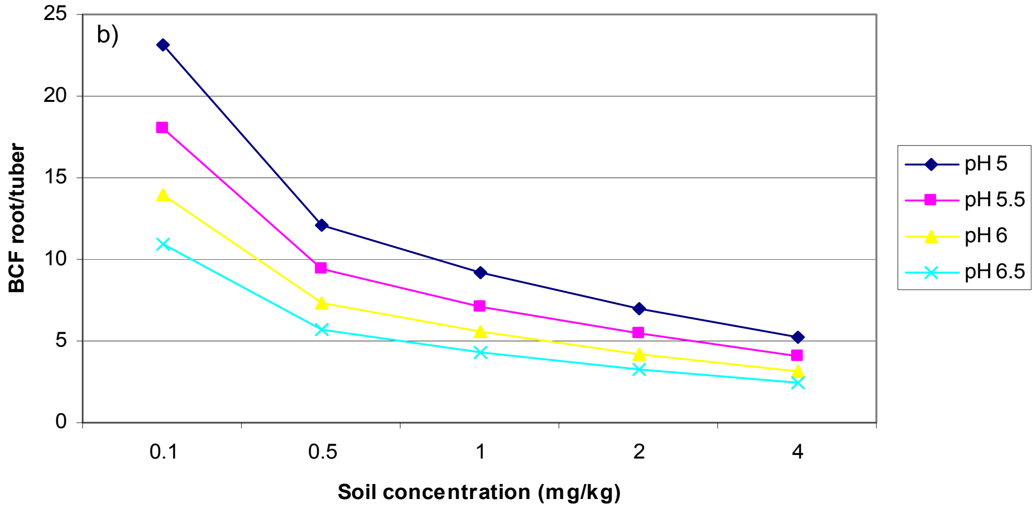


Figure 4: Variation in plant bioconcentration factors (BCF, dry weight) for root and tuber vegetables with soil concentration and pH



The contaminant-specific parameters values for the SGV calculations for cadmium are in table 24 and the derived SGVs for the different pHs are set out in table 25. The relationships with soil pH and produce percentage are shown in figure 5. The detailed calculations from which the values in table 25 are derived are given in appendices 1 and 2. The BCF values are unique to each residential SGV value and are not shown in table 24 but are given in appendix 2.

Table 24: Contaminant-specific parameters for cadmium soil guideline value derivation

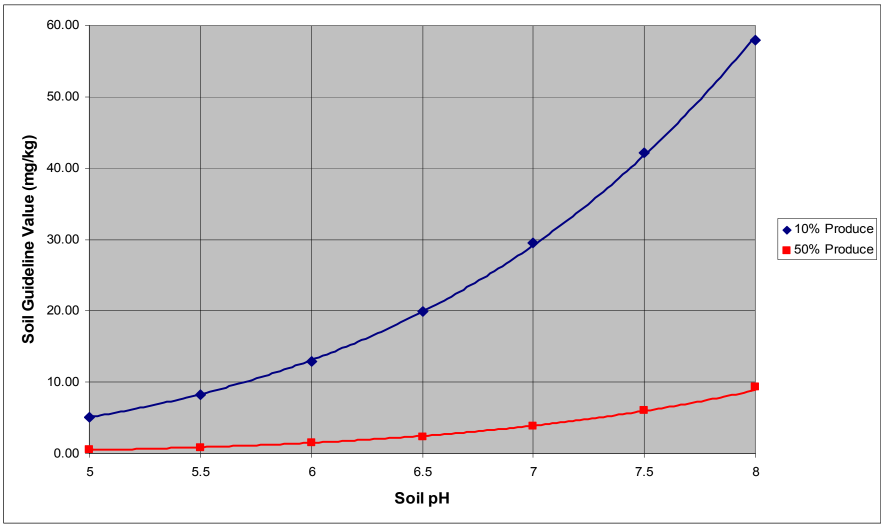
Tolerable daily intake: oral		0.001 mg/kg BW/day
Background exposure	Child	0.000 41 mg/kg BW/day
	Adult	0.000 26 mg/kg BW/day
Dermal absorption factor		0.001
Plant bioconcentration factors		Depend on pH and soil concentration. See separate calculations, appendix 2

Table 25: Cadmium soil guideline values (mg/kg)

Scenario		Combined SGV values		
		No produce	10% produce	50% produce
Rural residential / lifestyle block	pH 5	200	5	0.51
	pH 5.5		8	0.86
	pH 6		13	1.4
	pH 6.5		20	2.3
	pH 7		29	4
	pH 7.5		40	6
	pH 8		60	9
Standard residential	pH 5		5	0.51
	pH 5.5		8	0.86
	pH 6		13	1.4
	pH 6.5		20	2.3
	pH 7		29	4
	pH 7.5		40	6
	pH 8		60	9
High-density residential		370		
Recreational		1,100		
Commercial / industrial indoor worker		NL		
Commercial / industrial outdoor worker		1,600		

Notes: NL = No limit.

Figure 5: Dependence of cadmium soil guideline value on pH



The soil ingestion rate has little or no influence on the SGVs derived for the standard and rural residential scenarios, with the produce consumption pathway being dominant. Where there is no produce consumption the derived SGVs are much larger. Dermal absorption is insignificant.

New Zealand soils are typically acidic. Unless there is information on the soil pH, the default rural and standard residential SGVs are for pH 5. Given the significant increase in SGV for small changes in pH, it is preferable that soil sample pH is measured during laboratory analysis for an investigation targeting cadmium (or for which a metal suite has been specified).

SGVs calculated for the site-specific situation of high home-grown produce consumption for acidic soils may be less than the background concentration of cadmium for the locality. The background concentration is typically in the range 0.02 to 0.5 mg/kg in New Zealand. The background cadmium concentrations should be determined by taking samples of soil not expected to be affected by site activities, including cadmium enrichment from the use of superphosphate fertiliser. Samples should be taken from unused or undeveloped parts of the site, from adjacent sites that are not expected to have been contaminated, and/or, for sites that may have been fertilised (most agricultural land) from a depth below which cadmium enrichment is not expected to be significant (typically below 300 mm from the surface).

6.4 Chromium

6.4.1 Chromium bioconcentration factor

Concentrations of chromium in plant-available form are extremely low in most soils, and this is reflected in low concentrations of the element in plants (McGrath, 1995). Anthropogenic sources of chromium are believed to be responsible for elevated concentrations of this metal in plants, although overall there have been few studies of its uptake and accumulation (Kabata-Pendias and Pendias, 2000).

Chromium typically occurs in two valence states, chromium III and chromium VI; the former is the only state normally found in aerobic soils. In theory, chromium VI is likely to be more available to plants for uptake (because of its greater mobility in soils), but there have been few studies to support this (Kabata-Pendias and Pendias, 2000).

The 'Timber Treatment Guidelines' (MfE and MoH, 1997) uses BCFs of 0.015 for root and 0.003 for leafy vegetables for chromium III (table 26) based on the assumption that bioaccumulation in vegetative parts is one-fifth of that in the roots. For chromium VI a BCF of 0.24 for root vegetables is used, with a BCF of 0.048 for leafy vegetables also being one-fifth of the root value. These values are nominally sourced from ECOTOC (1990).

The US EPA (2007) reports a median BCF of 0.041, determined from 28 data points with a range of 0.021 to 0.48 (Bechtel-Jacobs 1998). Defra and EA (2002d) use BCFs of 0.02 and 0.055 for root and leafy vegetables, respectively, based on a literature review of the uptake of total and chromium III.

A summary of international values is shown in table 26. Given that there is little evidence for preferential uptake of chromium VI, and the lack of information on the source of data provided in the 'Timber Treatment Guidelines' (MfE and MoH,1997), it is proposed that the BCF for chromium is based on a simple average[10] of all values shown in bold in table 26. This gives a BCF of 0.0324, and is considered to be applicable to all vegetables.

10 Produce-type consumption-weighted mean BCFs have generally been used in this document, however, this refinement was not considered appropriate for the available data.

Table 26: Summary of BCFs for chromium from different sources

Substance	MfE and MoH (1997, Table A1)		Dutch (Otte et al, 2001)		US EPA (2007)	Defra and EA (2002d)		Mean
	Root	Leafy[a]	CW[b]	Median		Root	Leafy	
Cr III	0.015	0.003						
Cr VI	0.24	0.048						
Cr			0.011	**0.0135**	**0.041**	**0.02**	**0.055**	**0.0324**

a Assuming leafy five times less than root.

b CW = consumption weighted BCF, calculated using plant-soil relationships and produce consumption data.

6.4.2 Chromium guideline calculations

Soil guidelines have been calculated for both chromium III and chromium VI as threshold substances. The soil guideline value calculations for chromium use the contaminant-specific parameters in table 27 and the derived SGVs are set out in tables 28 and 29 for chromium III and VI, respectively.

A single BCF is used, applicable to both root/tuber and leafy parts of vegetables. The same background intake and BCF were used for both valance states. There is no data for the background intake of chromium VI. In accordance with MfE (2010b) the background intake is taken as five per cent of the TDI.

Table 27: Contaminant-specific parameters for chromium soil guideline value derivation

		Chromium III	Chromium VI
Tolerable daily intake: oral		1.5 mg/kg BW/day	0.003 mg/kg BW/day
Background exposure	Child	0.0012 mg/kg BW/day	No data
	Adult	0.000 53 mg/kg BW/day	No data
Dermal absorption factor		0	
Plant bioconcentration factor	Leafy	0.0324	
	Root	0.0324	
	Tuber	0.0324	

Table 28: Chromium III soil guideline value (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	500,000	280,000	100,000
Standard residential	500,000	280,000	100,000
High-density residential	890,000		
Recreational	NL		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	NL		

Note: NL = No limit.

Table 29: Chromium VI soil guideline value (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	1,000	560	210
Standard residential	1,000	560	210
High-density residential	1,800		
Recreational	5,200		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	6,300		

Note: NL = No limit.

It is notable that the calculated values for chromium III are all very high, some in excess of pure chromium, which is impossible. The results indicate no practical limit to allowable soil concentrations. However, acute effects for a pica child (consuming 5 to 10 grams of soil in a day) have not been checked.

Apart from possibly acute effects for children, phytotoxicity will occur well before the indicated soil concentrations are reached. This suggests that the calculated intake from produce consumption will not occur because produce will not reach a harvestable condition.

In practical terms a concentration of chromium III of several hundred thousand mg/kg (> 100 g/kg) is not likely.

6.5 Copper

6.5.1 Copper bioconcentration factor

The 'Timber Treatment Guidelines' (MfE and MoH, 1997) use a BCF for copper of 0.28 for roots, and assume that the BCF for leafy vegetables is 20 per cent of this. However, as discussed in Cavanagh (2004b), the equation used to estimate BCF is incorrectly stated to be root vegetables, as opposed to above-ground vegetables. This means that BCF_{leafy} should be 0.28, and BCF_{root} should be 1.4.

The US EPA (2007) reports a median bioconcentration factor of 0.124 from 180 data points and a range of 0.0011 to 7.4. Baes et al (1984) have shown that plant uptake of copper is dependent on the copper soil concentration. Copper is phytotoxic at relatively low concentrations, and plant uptake of copper is limited by its toxic effect on plants. A tissue copper concentration of 15–20 mg/kg (dry weight) is considered to be representative of excessive tissue concentration in agronomic species, while a 10 per cent growth yield decrease is most likely at 10–30 mg/kg (dry weight) tissue copper concentrations (Kabata-Pendias and Pendias, 2000).

Given the variable uptake of copper by plants from soil, and the known phytotoxic effects of copper, it is recommended that a maximal concentration of copper in produce is used in preference to a BCF. Based on the discussion above it is further recommended that a produce concentration of 30 mg/kg (dry weight) is used as the maximum amount of copper likely to be taken up in home-grown vegetables. Vegetables containing greater than this concentration would be so stunted and deformed that harvesting would be unlikely.

6.5.1 Copper guideline calculations

The soil guideline value calculations for copper use the contaminant-specific parameters in table 30 and the derived SGVs are set out in table 31. Copper has been treated as a threshold substance.

A BCF was not used to calculate the SGV for scenarios with produce consumption. As noted above, vegetables are unlikely to be harvested with tissue concentrations greater than 30 mg/kg dry weight. Any concentrations well in excess of this would theoretically accumulate in the plants at soil concentrations that are safe for the soil ingestion pathway, so a produce pathway soil guideline value cannot be determined as a function of soil concentration. Instead, a soil ingestion guideline has been calculated for the residential-with-produce scenarios by subtracting a further notional background intake to take into account the amount of produce that could theoretically be consumed if the produce was at the phytotoxic limit of 30 mg/kg tissue concentration. The modified background is subtracted from the TDI in the usual fashion.

To obtain the additional background intake, a child's produce consumption (0.0105 kg DW/day) was multiplied by 30 mg/kg and divided by the child's body weight of 15 kg to obtain the maximum additional background daily intake for 100 per cent of produce being home-grown. This was then multiplied by the homegrown produce percentage relevant to the particular SGV.

As the dermal and inhalation intakes are insignificant there is no contribution from these pathways and the result with the modified background becomes the SGV value.

Table 30: Contaminant-specific parameters for copper soil guideline value derivation

Tolerable daily intake: oral		0.15 mg/kg BW/day
Background exposure	Child Adult	0.056 mg/kg BW/day 0.02 mg/kg BW/day
Dermal absorption factor		0
Plant bioconcentration factors	Root Tuber Leafy	na na na
Additional child background for given produce percentage at produce concentration of 30 mg/kg	10% 50%	0.0021 mg/kg BW/day 0.0105 mg/kg BW/day

na = not applicable.

Table 31: Copper soil guideline value (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	33,000	32,000	29,000
Standard residential	33,000	32,000	29,000
High-density residential	60,000		
Recreational	170,000		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	290,000		

NL = No limit.

6.6 Inorganic lead

6.6.1 Lead bioconcentration factor

Inorganic lead is generally considered to be relatively immobile in soil and has limited plant uptake. Lead BCFs for most plants typically range from 0.001 to 0.03 (Jones and Johnston, 1991). Various regulatory agencies have examined plant uptake of lead for use in deriving soil guideline values. The US EPA (2007) used a relationship describing plant foliage lead concentrations as a function of soil lead concentrations derived by Bechtel Jacobs (1998) in the derivation of ecological soil screening levels (Eco-SSLs).

The Ontario Ministry for the Environment and Energy (OMEE) used a similar BCF of 0.039 (dry weight) determined from a study of the uptake of lead from Canadian soils by common backyard vegetables (OMEE, 1994). This is mentioned in CCME environmental guidelines on lead in the section on derivation of environmental guidelines, but does not appear to be used in the derivation of guidelines for the protection of human health. The Dutch use plant concentrations as a function of soil properties (Otte et al, 2001)

As there have been no recent summaries of the plant uptake of lead, a literature search was undertaken and data compiled to determine the geometric mean for leafy, root (carrot and radish) and tuber (potato) vegetables. Seventy-eight data points from 10 studies were found. These derived geometric means are recommended for use in deriving soil guideline values in New Zealand and are shown in table 32, along with BCFs determined by other authors for a similar purpose.

Table 32: Recommended BCF (dry weight) and BCF (dry weight) for lead from different sources

Recommended			Dutch (Otte et al, 2001)		Bechtel-Jacobs (1998)	OMEE (1994)
Leafy / above ground (n=41)	Root (n=13)	Tuber (n=4)	CW[a]	Median[b]	Geometric mean	
0.019	0.015	0.005	0.017	0.015	0.038	0.039

a CW = consumption weighted BCF, calculated using plant-soil relationships and produce consumption data.

b Median of all measured data for vegetables.

6.6.2 Lead guideline calculations

Soil guidelines have been calculated for lead as a threshold substance. The calculations use the contaminant-specific parameters in table 33 and the derived SGVs are set out in table 34.

Three BCF values were used, one each for leafy, root and tuber vegetables, combined into a single consumption-weighted mean value, using weighting factors of 0.3, 0.1 and 0.6 (dry weight), respectively.

Table 33: Contaminant-specific parameters for lead soil guideline value derivation

Tolerable daily intake: oral		0.003 57 mg/kg BW/day
Background exposure	Child	0.000 97 mg/kg BW/day
	Adult	0.000 41 mg/kg BW/day
Dermal absorption factor		0
Plant bioconcentration factors	Leafy	0.019
	Root	0.015
	Tuber	0.005
	Consumption weighted mean	0.0102

Table 34: Lead soil guideline values (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	900	730	400
Standard residential	900	730	400
High-density residential	1,600		
Recreational	4,700		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	7,000		

NL = No limit.

The derived SGVs for lead are dominated by the ingestion pathway. The dermal pathway has no influence. For the residential with produce scenarios, the produce pathway has a significant influence.

6.7 Inorganic mercury

6.7.1 Mercury bioconcentration factor

There is limited data available on mercury uptake into home produce. Three sources have examined plant uptake of mercury for the purpose of developing regulatory guideline values. Bechtel-Jacobs (1998) provide a summary of plant uptake data from a range of plants including vegetables. Using regression analyses, this data was used to develop relationships between concentrations in plants and soil that are used in the derivation of US ecological soil screening levels (Eco-SSLs (US EPA, 2007 (in attachment 4.1). The geometric mean and median values provided in Bechtel-Jacobs (1998) are shown in table 35.

Versluijs and Otte (2001) developed a series of equations relating mercury (and other metal) concentrations in the edible parts of various vegetables to soil mercury concentrations and soil parameters using multiple regression. Using a standard soil, these equations are used to determine a consumption-weighted bioconcentration factor that is used to develop Dutch intervention values. This value is reported in Lijzen et al (2001) and Otte et al (2001); the latter also provides a summary of the actual plant BCFs from the collected data. The consumption-weighted BCF is shown in Table 35, along with the BCF used for calculating the current Dutch mercury intervention value.

The Environment Agency (EA, 2009c) provides the most recent and relevant summary of plant BCFs. This report reviewed the existing literature and provides recommendations for BCFs (based on fresh weight) for the edible portion of six produce types, which are used in the CLEA model for the derivation of soil guideline values. The recommended BCFs are the geometric mean of the available data for each produce type.

The BCFs recommended by Environment Agency (EA, 2009c) are lower than provided by Bechtel-Jacobs (1998) and used in Otte et al, 2001. Close inspection of the Bechtel-Jacobs (1998) reveals a high proportion of non-vegetables that have significantly higher BCFs than vegetables provided in the same dataset. The vegetable BCFs in the Bechtel-Jacobs data set are similar to that provided by the Environment Agency (EA, 2009c), and also similar to that previously used in the derivation of Dutch intervention values. In contrast, the revised BCF determined from derived soil-plant relationships is higher than that determined from the literature. As the review undertaken by EA (2009c) is recent and for the purpose of deriving soil guideline values for the protection of human health, these values are recommended for the derivation of New Zealand soil guideline values. Conversion of the uptake factors to dry weight has been undertaken using the dry weight conversion factors specified in EA (2009a).

Table 35: BCF for mercury from different sources

Dutch (Otte et al, 2001)			Bechtel-Jacobs (1998)	UK (EA 2009c) Fresh weight (dry weight)[c]			Recommended values		
CW[a]	GM[b]	Previous	GM	Green (n=52)	Root (n=52)	Tuber (n=13)	Green (n=52)	Root (n=52)	Tuber (n=13)
0.15	0.28	0.02	0.35	0.0038 (0.04)	0.0069 (0.07)	0.0043 (0.02)	0.04	0.07	0.02

a CW = consumption weighted BCF, calculated using plant-soil relationships and produce consumption data.

b Geometric mean of all measured data for vegetables.

c Dry weight (DW) calculated using conversion factors of 0.096 kg DW/g FW, 0.103 kg DW/kg fresh weight (FW) and 0.21 kg DW/kg FW for green, root and tuber vegetables respectively (EA, 2009a).

6.7.1 Mercury guideline calculations

Soil guidelines have been calculated for inorganic mercury as a threshold substance. The calculations use the contaminant-specific parameters in table 36 and the derived SGVs are set out in table 37.

Three BCF values were used: one each for leafy, root and tuber vegetables, combined into a single consumption-weighted mean value, using weighting factors of 0.3, 0.1 and 0.6 (dry weight), respectively.

Table 36: Contaminant-specific parameters for inorganic mercury soil guideline value derivation

Tolerable daily intake: oral		0.002 mg/kg BW/day
Background exposure	Child	0.00005 mg/kg BW/day
	Adult	0.000065 mg/kg BW/day
Dermal absorption factor		0
Plant bioconcentration factors	Leafy	0.04
	Root	0.07
	Tuber	0.02
	Consumption weighted mean	0.031

Table 37: Inorganic mercury soil guideline values (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	660	380	140
Standard residential	660	380	140
High-density residential	1,200		
Recreational	3,500		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	4,200		

Note: NL = No limit.

Produce consumption has a significant influence in the SGVs for inorganic mercury for the residential-with-produce scenarios. For scenarios without produce consumption, soil ingestion is the dominant pathway. Dermal absorption is insignificant.

The inorganic mercury SGV is not intended to be applied to a site contaminated with elemental mercury or methyl mercury.

6.8 Benzo(a)pyrene (BaP)

The SGV for benzo(a)pyrene (BaP) is intended to represent the several polycyclic aromatic hydrocarbons (PAHs) thought to be carcinogenic (MfE, 2010b). PAHs typically occur as complex mixtures in which one or more carcinogenics may be present. The application of the SGV, using potency equivalency factors (PEFs) to calculate a BaP equivalence, is explained at the end of this subsection.

6.8.1 BaP bioconcentration factor

Limited data is available on plant uptake of BaP and a number of sources have used various models to determine plant uptake. For example, in New Zealand, the 'Oil Industry Guidelines' (MfE, 1999) use the method of Ryan et al (1988), whereas the 'Gasworks Guidelines' (MfE, 1997) use the method of Travis and Arms (1988). Both methods are based on the log K_{ow} of the organic contaminant, and slightly different values are used in the two guideline documents – the 'Oil Industry Guidelines' use a log K_{ow} = 6.04 whereas the 'Gasworks Guidelines' use K_{ow} = 9.55×10^5, which gives a log K_{ow} = 5.98. The Travis and Arms (1988) relationship determines a plant bioaccumulation factor on a dry-weight basis in above-ground parts. In the *Gasworks Guidelines*, this is converted to a fresh-weight plant uptake factor (PUF) using the assumption that dry weight is 20 per cent of fresh weight for all vegetables. It should be noted that the US EPA (2003) have criticised the Travis and Arms relationship as being based on few data, some of which are at variance with the source documents cited by Travis and Arms.

The Ryan et al (1988) model determines fresh-weight PUF for roots and leaves directly from concentrations in the pore water. However, the 'Oil Industry Guidelines' do not indicate how the pore water concentrations relate to soil concentrations provided in the document.

Dutch authors use the model of Briggs et al (1982, 1983; in Lijzen et al, 2001) to estimate uptake of organic contaminants in leafy vegetables and the model of Trapp and Matthies (1995, in Lijzen et al, 2001) with modified parameters to estimate uptake of organic contaminants by root vegetables.

In contrast, the US EPA (2007) reports a median bioaccumulation factor (BAF) for plant foliage of 0.1 from 15 data points with a range of 0.039–0.2; it uses a relationship describing plant concentration as a function of soil concentration based on measured data to derive ecological soil screening levels (Eco-SSLs).

The preferred approach in the current work is to use BCFs determined from measured data. As such, literature data on plant uptake of BaP was compiled and used to determine the BCF for leafy, root and tuber vegetables. Root and tuber vegetables were kept separate as the available data indicates a difference in uptake between the two vegetable types. It should be noted that uptake of vapour-phase BaP (or any PAH) is primarily from ambient air as opposed to volatilisation from soil. This is often implicated as being the most significant pathway of plant uptake leading to accumulation in all plant parts, even in plant roots (eg, Kipopoulou et al, 1999; Wild et al, 1992). This suggests that BCFs, particularly for leafy vegetables, are likely to be overestimated.

6.8.2 BaP guideline calculations

The soil guideline value calculations for BaP use the contaminant-specific parameters in table 38 and the derived SGVs are set out in table 39. BaP has been treated as a non-threshold substance and therefore a risk-specific dose has been employed. Three BCF values were used: one each for leafy, root and tuber vegetables, combined into a single consumption-weighted mean value` using weighting factors of 0.3, 0.1 and 0.6 (dry weight), respectively.

MfE (2010b) provides two skin absorption factors, a 'worst case' of 0.06 and 0.026 for 'aged' contamination. The worst-case value has been used for the SGV derivation, although as dermal absorption has little influence on the SGV value, the difference between the two values is small.

Table 38: Contaminant-specific parameters for BaP soil guideline value derivation

Risk-specific dose:oral		0.000 043 mg/kg BW/day
Background exposure		Not applicable (non-threshold substance)
Dermal absorption factor		0.06
Plant bioconcentration factors	Leafy (n=10)	0.005
	Root (n=17)	0.031
	Tuber (n=3)	0.004
	Consumption weighted mean	0.007

Table 39: BaP soil guideline value (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	110	85[a]	40
Standard residential	130	100[a]	55
High-density residential	240		
Recreational	440		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	300		

a Different rural residential and residential exposure durations result in different SGVs because non-threshold substance SGV derivation uses age-adjusted exposure rates.

NL = No limit.

The controlling pathway for BaP is soil ingestion but produce ingestion has a significant influence on the derived SGV for residential scenarios, even at the default produce proportion of 10 per cent. Dermal absorption has minimal influence.

As noted earlier, BaP is used to represent the carcinogenic PAHs. To enable an estimate of the potential carcinogenicity of polycyclic aromatic hydrocarbon mixtures, potency equivalence factors have been used previously in New Zealand guidance (MfE, 1997, 1999), and are recommended for the revised SGV to calculate a BaP equivalence concentration (BaP_{eq}).[11] As recommended in MfE (2010b), the PEFs developed by Kalberlah et al (1995 cited in WHO, 1998) are to be used. The PEFs are given in table 40. The PEFs cover a wider range than used in current New Zealand guidance documents (MfE, 1997, 1999).[12]

Table 40: PEFs for use in assessing potential carcinogenicity of PAH mixtures

Polycyclic aromatic hydrocarbon	Potency equivalency factors
Benz(a)anthracene	0.1
Benzo(b)fluoranthene	0.1
Benzo(j)fluoranthene	0.1
Benzo(k)fluoranthene	0.1
Benzo(a)pyrene	1.0
Chrysene	0.01
Dibenz(a,h)anthracene	1.0
Fluoranthene	0.01
Indeno(1,2,3-c,d)pyrene	0.1

Source: WHO, 1998.

6.9 DDT

The SGV for DDT has been derived to represent DDT and its metabolites DDD and DDE. The sum of the six isomers[13] is commonly referred to as ∑DDT.

6.9.1 DDT bioconcentration factor

Limited data on the plant uptake of DDT are available, although two recent studies provide some data on the uptake of DDT and its primary degradation products, DDE, in vegetables from historically contaminated soils (Gaw et al, 2008; Mikes et al, 2009). Gaw et al (2008) examined uptake into lettuce and radish, whereas Mikes et al (2009) examined uptake into radishes. Using data from the edible portions of the vegetables (lettuce leaf, radish root) from these studies, the geometric mean for the BCFs for *p,p'*-DDT and *p,p'*-DDE in the two vegetable types are shown in table 41. These values compare with a median plant uptake factors for grasses of 0.136 (n=3) for DDE and 0.037 (n=6) for DDT in US EPA (2007).

11 The equivalent BaP concentration is calculated as the sum of each of the detected concentrations of nine carcinogenic PAHs (benz(a)anthracene, benzo(b)fluoranthene, benzo(j)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, fluoranthene and indeno(1,2,3-cd) pyrene), multiplied by their respective potency equivalency factors (table 40).

12 FAO/WHO (2006) identified 13 PAHs as carcinogenic, of which eight, together with fluoranthene, are included in table 40. Consideration is being given to expanding table 40 to include PEFs for dibenzo(a,e)pyrene, dibenzo(a,h)pyrene, dibenzo(a,i)pyrene and dibenzo(a,l)pyrene.

13 The six isomers are *p,p'*-DDT, *o,p'*-DDT, *p,p'*-DDE, *o,p'*-DDE, *p,p'*-DDD, *o,p'*-DDD.

A plant uptake factor for DDE as opposed to DDT is more appropriate, as DDE is the primary metabolite of DDT and is the compound most commonly found in highest concentrations in contaminated soils.

Table 41: BCF (geometric mean) for *p,p'*-DDT and *p,p'*-DDE in root and leafy vegetables determined from the literature

Parameter	Root		Leafy	
	BCF	n	BCF	n
p,p'-DDE	0.038	10	0.012	9
p,p'-DDT	0.022	9	0.003	7

6.9.2 DDT guideline calculations

The soil guideline value calculations for DDT use the contaminant-specific parameters in table 42 and the derived SGVs are set out in table 43. DDT has been treated as a threshold substance. Two BCF values were used for root / tuber and leafy parts of vegetables, combined into a single consumption-weighted mean value, using weighting factors of 0.7 and 0.3 (dry weight), respectively.

Table 42: Contaminant-specific parameters for DDT soil guideline value derivations

Tolerable daily intake: oral		0.0005 mg/kg BW/day
Background exposure	Child	0.0000511 mg/kg BW/day
	Adult	0.0000193 mg/kg BW/day
Dermal absorption factor		0.018
Plant bioconcentration factors	Leafy	0.012
	Root / tuber	0.038
	Consumption weighted mean	0.0302

Table 43: DDT soil guideline value (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	150	90	35
Standard residential	150	90	35
High-density residential	270		
Recreational	750		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	1,000		

Note: NL = No limit.

Produce consumption has a significant influence in the SGV for DDT for the residential-with-produce scenarios. For scenarios without produce consumption, soil ingestion is the dominant pathway. Dermal absorption has only a minor influence on the combined value.

As noted early, the SGV is for the sum of DDT, DDD and DDE. The value is compared with the sum of the concentrations from the laboratory analysis of all six isomers of these compounds.

6.10 Dieldrin

6.10.1 Dieldrin bioconcentration factor

Limited data on the plant uptake of dieldrin are available. A plant uptake factor of 0.41 was used for dieldrin, based on the median value for nine observations provided in US EPA (2007). These data were determined for the above-ground portions of three plants; in the absence of any further information, this value is considered to be applicable to all vegetable parts.

6.10.2 Dieldrin guideline calculation

The soil guideline value calculations for dieldrin assume it is a threshold substance. The contaminant-specific parameters used in the calculations are in table 44 and the derived SGVs are set out in table 45. A single BCF value was used for root, tuber and leafy parts of vegetables.

Table 44: Contaminant-specific parameters for dieldrin soil guideline value derivations

Tolerable daily intake: oral		0.0001 mg/kg BW/day
Background exposure	Child	0.0000036 mg/kg BW/day
	Adult	0.0000014 mg/kg BW/day
Dermal absorption factor		0.1
Plant bioconcentration factors	Leafy	0.41
	Root / tuber	0.41
	Consumption weighted mean	0.41

Table 45: Dieldrin soil guideline value (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	28	3.1	0.67
Standard residential	28	3.1	0.67
High-density residential	50		
Recreational	110		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	160		

Note: NL = No limit.

Produce consumption has a significant influence in the SGVs for dieldrin for the residential-with-produce scenarios. For scenarios without produce consumption soil ingestion is the dominant pathway, although dermal absorption has some influence with dieldrin.

6.10.3 Applicability of the dieldrin SGV to aldrin

Aldrin was last used in New Zealand almost 50 years ago (MFE, 2006a). As aldrin degrades to dieldrin in the environment, with reported half lives in soil of 20–100 days (FAO, 2000), only small amounts of aldrin are expected to be detected, and most likely in conjunction with dieldrin, which would be at higher concentrations.

As the TDI for dieldrin is also applicable to aldrin (MfE, 2010b) the SGV is applicable to dieldrin or aldrin separately, or to the sum of aldrin and dieldrin if both are involved.

6.11 Dioxin and dioxin-like PCBs

The term 'dioxins' encompasses a group of 75 polychlorinated dibenzo-*p*-dioxin (PCDD) and 135 polychlorinated dibenzofuran (PCDF) congeners. PCDDs and PCDFs are formed during incomplete combustion processes. They also occur as contaminants during various industrial processes, eg, the chemical manufacture of some chlorinated compounds and chlorine bleaching of paper pulp.

The toxicity of individual dioxin congeners differs considerably. The congeners that are toxicologically important have chlorine atoms substituted in each of the 2-, 3-, 7- and 8-positions. Thus, from 210 theoretically possible congeners, only 17 are of toxicological concern. These compounds have a similar toxicological profile to that of the most toxic congener, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD).

Twelve of the 209 possible polychlorinated biphenyl (PCB) congeners also exhibit 'dioxin-like' toxicity. This document only considers dioxin-like PCB congeners and does not consider other PCB congeners. The toxicity of other PCB congeners will need to be considered separately in any assessment where PCBs are of concern.

Dioxins and PCBs do not occur as pure compounds but as complex mixtures of many congeners. When considering the toxicity of mixtures, the total toxicity is assessed as a toxic equivalency (TEQ) to 2,3,7,8-TCDD using toxic equivalency factors (TEF). The TEQ is defined as the sum of the products of the concentration of each compound multiplied by the value of its TEF; it is an estimate of the total 2,3,7,8-TCDD-like activity of the mixture.

A number of TEF schemes have been developed, the most recent from the 2005 World Health Organization re-evaluation by Van den Berg et al (2006). These TEFs are given in table 46 and are the values recommended for calculating the TEQ to be compared with dioxin and dioxin-like PCB SGVs.

Table 46: WHO (2005) TEFs for dioxins and dioxin-like PCBs

Compound	Abbreviation	WHO (2005)
Polychlorinated dibenzodioxins		
2,3,7,8-Tetrachlorodibenzodioxin	TCDD	1
1,2,3,7,8-Pentachlorodibenzodioxin	1,2,3,7,8-PeCDD	1
1,2,3,4,7,8-Hexachlorodibenzodioxin	1,2,3,4,7,8-HxCDD	0.1
1,2,3,6,7,8-Hexachlorodibenzodioxin	1,2,3,6,7,8-HxCDD	0.1
1,2,3,6,7,9-Hexachlorodibenzodioxin	1,2,3,6,7,9-HxCDD	0.1
1,2,3,4,6,7,8-Heptachlorodibenzodioxin	1,2,3,4,6,7,8-HpCDD	0.01
Octachlorodibenzodioxin	OCDD	0.0003
Polychlorinated dibenzofurans		
2,3,7,8-Tetrachlorodibenzofuran	2,3,7,8-TCDF	0.1
1,2,3,7,8-Pentachlorodibenzofuran	1,2,3,7,8-PeCDF	0.03
2,3,4,7,8-Pentachlorodibenzofuran	2,3,4,7,8-PeCDF	0.3
1,2,3,4,7,8-Hexachlorodibenzofuran	1,2,3,4,7,8-HxCDF	0.1
1,2,3,6,7,8-Hexachlorodibenzofuran	1,2,3,6,7,8-HxCDF	0.1
1,2,3,7,8,9-Hexachlorodibenzofuran	1,2,3,7,8,9-HxCDF	0.1
2,3,4,6,7,8-Hexachlorodibenzofuran	2,3,4,6,7,8-HxCDF	0.1
1,2,3,4,6,7,8-Heptachlorodibenzofuran	1,2,3,4,6,7,8-HpCDF	0.01
1,2,3,4,7,8,9-Heptachlorodibenzofuran	1,2,3,4,7,8,9-HpCDF	0.01
Octochlorodibenzofuran	OCDF	0.0003
‘Non-ortho’ polychlorinated biphenyls		
3´,4,4´-Tetrachlorobiphenyl (PCB 77)	3,3´,4,4´-TCB	0.0001
3,4,4´,5,-Tetrachlorobiphenyl (PCB 81)	3,4,4´,5-TCB	0.0003
3,3´,4,4´,5-Pentachlorobiphenyl (PCB 126)	3,3´,4,4´,5-PeCB	0.1
3,3´,4,4´,5,5´-Hexachlorobiphenyl (PCB 169)	3,3´,4,4´,5,5´-HxCB	0.03
‘Mono-ortho’ polychlorinated biphenyls		
2,3,3´,4,4´-Pentachlorobiphenyl (PCB 105)	2,3,3´,4,4´-PeCB	0.00003
2,3,4,4´,5-Pentachlorobiphenyl (PCB 114)	2,3,4,4´,5-PeCB	0.00003
2,3´,4,4´,5-Pentachlorobiphenyl (PCB 118)	2,3´,4,4´,5-PeCB	0.00003
2,3´,4,4´,5’-Pentachlorobiphenyl (PCB 123)	2,3´,4,4´,5´-PeCB	0.00003
2,3,3´,4,4´,5-Hexachlorobiphenyl (PCB 156)	2,3,3´,4,4´,5-HxCB	0.00003
2,3,3´,4,4´,5´-Hexachlorobiphenyl (PCB 157)	2,3,3´,4,4´,5´-HxCB	0.00003
2,3´,4,4´,5,5´-Hexachlorobiphenyl (PCB 167)	2,3´,4,4´,5,5´-HxCB	0.00003
2,3,3´,4,4´,5,5´-Heptachlorobiphenyl (PCB 189)	2,3,3´,4,4´,5,5´-HpCB	0.00003

6.11.1 Dioxin and dioxin-like PCB bioconcentration factors

Plant uptake of dioxins and dioxin-like PCBs is suggested to primarily occur through atmospheric deposition, eg, uptake of a gaseous fraction or particle-bound contaminants via the leaves (Hulster and Marschner, 1993; McLachlan, 1997; Meneses et al, 2002; Collins et al, 2006). Root-uptake / translocation of PCDD/PCDFs from the soil, or volatilisation of PCDD/CDFs from the soil and their subsequent adsorption onto the vegetation is negligible (McLachlan, 1997 and references contained therein; Jones and Duarte-Davidson, 1997).

Plant uptake via soil for dioxins and dioxin-like PCBs is only considered for the cucurbits, particularly those in the genus *Cucurbita* (zucchini and pumpkin): these are the only plants for which uptake via soil has been conclusively demonstrated (Hulster et al, 1994; Inui et al, 2008). This is suggested to be due to the production of root exudates in these species, although differences in uptake also exist between different cultivars (Inui et al, 2008). For other vegetables, uptake from soil is not considered a relevant exposure pathway.

Transfer of dioxin-like PCBs is indicated to be greater than that of PCDD/PCDFs, with penta- and hexa-chloro-biphenyls showing the highest BCFs (fresh weight): up to about 0.13 in high-accumulator cultivars. Average BCF for PCBs was 0.045, and for PCDDs and PCDFs, 0.01. BCFs in low-accumulator cultivars were typically less than 0.001, although up to 0.003 for tetrachloro-biphenyl and 0.002 for the penta- and hexachloro-biphenyls.

Hulster et al (1994) provide BCFs for different PCDD/PCDF-homologue groups for zucchini fruit. The values from Inui et al (2008) and Hulster et al, 1994 are shown in table 47. The recommended values for cucurbits are the geometric mean values determined from Inui et al (2008) and Hulster et al (1994), and are also show in table 47.

Table 47: BCF for dioxins and dioxin-like PCBs from Inui et al (2008), Hulster et al (1994) and recommended BCFs

Polychlorinated dibenzodioxins	Inui et al (2008)		Hulster et al (1994) (dry weight)		Recommended BCF value (dry weight)[b]
	Fresh weight	Dry weight[a]			
TCDD	0.065	0.65	0.25	0.08	**0.24**
1,2,3,7,8-PeCDD	0.035	0.35	0.2	0.09	**0.18**
1,2,3,4,7,8-HxCDD	0.01	0.1	0.17	0.04	**0.09**
1,2,3,6,7,8-HxCDD	0.01	0.1			
1,2,3,6,7,9-HxCDD	0.01	0.1			
1,2,3,4,6,7,8-HpCDD	<0.001	<0.01	0.03	0.01	**0.017**
OCDD	<0.001	<0.01	<0.005	<0.005	**<0.005**
2,3,7,8-TCDF	0.045	0.45	0.14	0.09	**0.18**
1,2,3,7,8-PeCDF	0.022	0.22	0.14	0.011	**0.07**
2,3,4,7,8-PeCDF	0.02	0.2			
1,2,3,4,7,8-HxCDF	0.005	0.05	0.1	0.04	**0.06**
1,2,3,6,7,8-HxCDF	0.005	0.05			
1,2,3,7,8,9-HxCDF	<0.001	<0.01			
2,3,4,6,7,8-HxCDF	0.05	0.5			
1,2,3,4,6,7,8-HpCDF	<0.001	<0.01	0.03	0.01	**0.017**
1,2,3,4,7,8,9-HpCDF	<0.001	<0.01			
OCDF	<0.001	<0.01	0.01	0.05	**0.02**
Total PCBs	0.045	0.45			**0.45**

a Calculated using a conversion factor of 0.1 kg DW/kg FW.

b Geometric mean of data provided in Inui et al (2008) and Hulster et al (1994).

6.11.2 Dioxin and dioxin-like PCB guideline calculation

The most common occurrence of dioxins in New Zealand contaminated land is that associated with the use of pentachlorophenol (PCP) as a wood preservative. The dioxin contamination in PCP is dominated by the octa- and hepta-chlorinated congeners (in TEQ terms). Dioxin contamination associated with the manufacture of trichlorophenol and 2,4,5-T herbicide is dominated by 2,3,7,8-TCDD. Combustion-derived dioxin mixtures will fall between these extremes.

Because the lower chlorinated congeners are taken up in cucurbits considerably more than the octa- and hepta-congeners, it can be expected that the produce pathway will be more significant for dioxin mixtures dominated by the lower-chlorinated congeners. Similarly, the produce pathway is significant for dioxin-like PCB mixtures. Because of this, SGVs have been separately calculated for OCDD- and TCDD-dominated mixtures and dioxin-like PCBs for residential scenarios with produce consumption (appendix 4). As noted above, the only vegetable type with significant dioxin uptake is the cucurbit family. The proportion of cucurbits relative to total vegetable consumption has been taken to be 0.04 for calculating the consumption-weighted BCFs.

It is necessary to choose representative BCFs from table 47 to calculate the residential-with-produce consumption SGVs. For TCDD-dominant mixtures, the BCF value of TCDD was chosen as conservative. For OCDD- and HpCDD-dominated mixtures, the BCF for HpCDD was chosen rather than the lower value for OCDD, again to be conservative.

It is also necessary to choose representative skin absorption factors. PCDFs contribute little to the TEQ in most PCDD/PCDF mixtures encountered, therefore the PCDD skin absorption factor 0.02 recommended in MfE (2010b) has been used for all dioxin SGV derivations, but the larger factor 0.07 that is specific to dioxin-like PCBs has been used for that derivation.

The soil guideline value calculations for dioxin and dioxin-like PCBs use the contaminant-specific parameters in table 48 and the derived SGVs are set out in table 49. Dioxins and dioxin-like PCBs have been assumed to be threshold substances in the derivation.

Only the SGVs for TCDD and dioxin-like PCBs for residential scenarios are presented in table 49. The calculated SGVs with produce consumption for OCDD and TCDD are similar for 10 per cent homegrown produce, as the uptake into cucurbits has only a small influence on the final value (appendix 1). The TCDD values have therefore been chosen as the default value to cover all dioxin mixtures. However, the SGVs for derived dioxin-like PCBs are sufficiently different from the TCDD values to warrant separate default values for PCB mixtures.

It may be advantageous to derive site-specific SGVs for OCDD-dominated or other mixtures, rather than using the TCDD defaults, particularly if the occupants of the particular site grow a high proportion of their vegetables. Use of other than the defaults would have to be demonstrated on a case-by-case basis.

Table 48: Contaminant-specific parameters for dioxin and dioxin-like soil guideline value derivations

Tolerable daily intake: oral		0.000001 µg/kg BW/day		
Background exposure	Child	0.00000033 µg/kg BW/day		
	Adult	0.00000033 µg/kg BW/day		
		TCDD	**OCDD**	**PCBs**
Dermal absorption factor		0.02	0.02	0.07
Plant bioconcentration factors	Leafy	0	0	0
	Root	0	0	0
	Tuber	0	0	0
	Cucurbits	0.24	0.017	0.45
	Consumption weighted mean	0.0096	0.00068	0.018

Table 49: Dioxin and dioxin-like PCB soil guideline values (µg TEQ/kg)

Scenario	Combined soil guideline values					
	No produce		10% produce		50% produce	
	Dioxins	PCBs	Dioxins	PCBs	Dioxins	PCBs
Rural residential / lifestyle block	0.23	0.21	0.19	0.15	0.11	0.07
Standard residential	0.23	0.21	0.19	0.15	0.11	0.07
High-density residential	0.41	0.38				
Recreational	1.1	0.90				
Commercial / industrial indoor worker	NL	NL				
Commercial / industrial outdoor worker / maintenance	1.4	1.2				

Note: NL = No limit.

For scenarios without produce consumption, soil ingestion is the dominant pathway for dioxins and dioxin-like PCBs, but dermal absorption affects the final SGV by up to about 10 per cent, depending on the congener mix / compound type. Skin absorption is greater for PCDFs than for PCDDs and absorption for dioxin-like PCBs is greater again (MfE, 2010b). The produce consumption pathway is a significant pathway for those residential scenarios with home-grown produce consumption, and becomes dominant for high home-grown produce percentages. The produce consumption pathway has the greatest significance for the dioxin-like PCBs.

Note: The SGV values for dioxins have units of µg TEQ/kg, that is, 1000 times lower than the other SGVs presented in this document. The guidelines are applicable to dioxin or PCB concentrations after converting to TEQ (toxic equivalents) and should not be applied to individual congener concentrations or a simple sum of dioxin or PCB concentrations. The PCB guideline is not applicable to the many non-dioxin-like PCBs. The guidelines should also not be applied to dioxin 'screens' for which all the 2,3,7,8 congeners have not been analysed, without considering whether the analytical screen provides a sufficiently good estimate for the TEQ. For example, the commonly used OCDD screen[14] may underestimate the TEQ by a factor of two to three for a PCP-contaminated site. The PCB screens currently available from New Zealand laboratories include all the dioxin-like PCBs.

6.11.3 Dioxin and dioxin-like PCB uptake into home-produced eggs

Eggs may accumulate lipophilic organic compounds, if the hens producing the eggs forage in an area contaminated with such compounds. Dioxins and dioxin-like PCBs are lipophilic. Several studies have noted that free-range eggs have higher concentrations of dioxins than commercial eggs (eg, Van Overmeire et al, 2006 and references contained in Van Overmeire et al, 2009). The contact with the outdoor environment, in particular soil and soil organisms, is believed to contribute to egg contamination of free-range hens (Stephens et al, 1995; Schuler et al, 1997; Harnley et al, 2000; Van Overmeire et al, 2009).

Keeping hens for egg production is a reasonably common activity on rural properties and some urban-dwellers also keep hens. This may be a more common activity for households who already grow a significant proportion of their vegetable consumption. It is therefore appropriate

[14] The OCDD screen provides concentrations of OCDD and 1,2,3,4,6,7,8-HpCDD.

to consider an SGV incorporating both consumption of home-grown produce and home-produced eggs. Accordingly, SGVs have been derived for illustrative purposes – but these should not be used for site assessment without consideration of the site-specific circumstances. Three produce scenarios and two egg-consumption scenarios have been considered: 0, 10 and 50 per cent of vegetables consumed being home-grown for each of average egg consumption and twice the average egg consumption (the latter arbitrarily chosen as being a more likely estimate for those households that run hens).

According to Statistics New Zealand (see www.stats.govt.nz), New Zealanders consume an average of about 200 eggs per year, or a little over half an egg per day. This is somewhat higher than the average 20 g/day[15] assumed for a 25+ adult (averaged across males and females) in the simulated diets of the New Zealand Total Diet Survey. The survey also assumes a 1- to 3-year-old toddler consumes an average of about 8 g/day (Brinsdon, 2004).

Dioxins are taken up into the egg lipids, therefore it is the egg-lipid content of eggs that is important in determining exposure. Egg lipids are 10–11 per cent of the total egg weight (Van Overmeire et al, 2009), or, using the consumption values from Brinsdon (2004), the equivalent of about 2.2 grams lipid per day for an adult and 0.9 grams lipid per day for a toddler (taking the higher lipid percentage and rounding to one decimal place). When normalised by standard body weights, this converts to 0.03 and 0.06 g lipid/kg BW/day. The child is therefore the critical receptor with respect to eggs, as it is for soil ingestion and produce consumption.

There is no statistical information on egg consumption of New Zealand toddlers, so the egg lipid consumption value derived from Brinsdon (2004) has been used for the SGV calculation.

To calculate dioxin intake from eggs and hence an SGV, a relationship between soil concentration and egg lipid concentration is necessary. Some studies have found a correlation between egg and soil concentrations (Harnley et al, 2000; Van Overmeire et al, 2009), others have not (Schuler et al, 1997; Pirard and de Pauw, 2004). This lack of a relationship is likely a reflection of the amount of soil and/or soil organisms ingested by the chickens (Schuler et al, 1997; Van Overmeire et al, 2009). This in turn will be influenced by foraging activity, the number of chickens in a given area and the extent of bare soil exposed, and therefore the amount of soil and/or soil organisms available to be ingested per chicken.

Schuler et al (1997) provides an estimate of egg lipid/soil concentration factors based on soil concentrations in a hen yard for different PCDD/PCDF congeners. Additionally, crude egg lipid/soil ratios can be derived from the median egg lipid and soil concentrations provided in Van Overmeire et al (2009), who assessed PCDD/PCDF concentrations in eggs and soil from 10 private hen owners. The egg lipid/soil ratios from Schuler et al (1997) and Van Overmeire et al (2009) are similar in the sense that both are based on egg lipid with an unknown / variable amount of soil ingestion, and yield similar ratios (table 50).

Few studies have examined the uptake of dioxin-like PCBs into eggs (Van Overmeire et al, 2009 and references contained therein), although PCBs in eggs may contribute significantly to the total toxic equivalent concentrations. For example, Van Overmeire et al (2009) found dioxin-like PCBs contributed, on average, 47 per cent to the total TEQs in eggs compared with 14 per cent in soil.

[15] 20 g/day equates to about 0.4 egg/day if it is assumed the edible portion of a ‘typical’ egg is about 50 g. The most common egg sizes available for purchase are sizes 6 and 7, which must be a minimum of 53 and 62 g, respectively. If it is assumed a typical egg lies between these sizes, then such an egg would weigh approximately 60 g. The egg shell makes up 11–13 per cent of the egg, suggesting the edible part is around 50 g.

The egg lipid / soil ratios for dioxin-like PCBs determined from median egg lipid and soil concentration data in Van Overmeire et al (2009) are shown in table 50. These ratios may be used to provide crude estimates of potential contamination of eggs as a result of running hens on soils with elevated PCDD/PCDF and dioxin-like PCB concentrations. A better approach, if some information about a given site is known, would be to use a model similar to that developed by Waegeneers et al (2009), which accounts for different sources of intake, bioavailability of dioxins and variable intake of soil depending on hen-run conditions.

Table 50 shows greater uptake into eggs for the less chlorinated congeners and for dioxin-like PCBs. This is similar to the plant uptake behaviour.

For the purposes of illustrative calculation, egg-lipid concentration factors of 1.9 for TCDD dominated mixtures, 0.7 for OCDD/PeCDD-dominated mixtures (average of the OCDD and PeCDD ratios given in table 50) and 17 for dioxin-like PCBs[16] were used.

The detailed calculations (appendix 4) show egg consumption dominates all other pathways. The same combined SGV is arrived at regardless of home-grown produce consumption percentage. The results are shown in table 51.

[16] Dioxin-like PCBs occur in small quantities within commercial PCB mixtures such as the Aroclor brand manufactured by General Electric. Aroclor 1254 and Aroclor 1260 were two mixtures commonly used in capacitors. Frame et al (1996) provide typical compositions of Aroclor mixtures. These have been used to calculate weighted average egg lipid/soil ratios, which for both Aroclor 1254 and 1260 was about 17 and for Aroclor 1242 about 14. Actual concentrations should be used to calculate a weighted average egg lipid/soil ratio for a site-specific analysis.

Table 50: Egg lipid / soil ratios for PCDD/PCDFs and dioxin-like PCBs

Dioxin	Schuler, 1997	Derived from Van Overmeire et al, 2009	Average
TCDD	1.2	2.7	1.9
1,2,3,7,8-PeCDD	2.4	4.1	3.2
1,2,3,4,7,8-HxCDD	1.5	2.5	2.0
1,2,3,6,7,8-HxCDD	1.6	2.7	2.2
1,2,3,6,7,9-HxCDD	0.8	1.4	1.1
1,2,3,4,6,7,8-HpCDD	0.4	0.9	0.6
OCDD	0.1	0.5	0.3
2,3,7,8-TCDF	3.3	2.4	2.9
1,2,3,7,8-PeCDF	4.4	1.8	3.1
2,3,4,7,8-PeCDF	0.8	1.7	1.2
1,2,3,4,7,8-HxCDF	0.9	0.9	0.9
1,2,3,6,7,8-HxCDF	1	0.8	0.9
1,2,3,7,8,9-HxCDF	0.1	1.0	0.6
2,3,4,6,7,8-HxCDF	0.6	0.7	0.6
1,2,3,4,6,7,8-HpCDF	0.2	0.2	0.2
1,2,3,4,7,8,9-HpCDF	0.1	1.0	0.5
OCDF	0.1	0.2	0.15
PCB77		6.4	
PCB81		3.1	
PCB126		7.9	
PCB169		6.8	
PCB105		15.8	
PCB114		18.5	
PCB118		17.2	
PCB123		13.3	
PCB156		17.2	
PCB157		16.1	
PCB167		18.4	
PCB189		17.8	

Table 51: Dioxin and dioxin-like PCB soil guideline values for egg consumption pathway (μg TEQ/kg)

Scenario	Combined soil guideline values					
	Average egg consumption			Twice average egg consumption		
	TCDD	OCDD/ PeCDD	Dioxin-like PCBs	TCDD	OCDD/ PeCDD	Dioxin-like PCBs
Rural and standard residential	0.006	0.015	0.000 7	0.003	0.008	0.000 3

The calculations show that running hens on a dioxin-contaminated property is generally not advisable without precautions against hens foraging in contaminated areas. However, a number of assumptions have been made in these calculations. Assessment of such a site should be carried out using site-specific information, in particular the actual amount of eggs consumed, the actual dioxin or dioxin-like PCB concentrations in the soil or, better still, the concentrations within the home-produced eggs.

6.12 Pentachlorophenol

6.12.1 PCP bioconcentration factor

A number of factors influence plant uptake of PCP. For example, if PCP does not persist in the soil for a sufficiently long period (eg, over the growth period of the plant), a significant quantity is unlikely to be taken up into a plant (Anon, 1987). The form in which PCP exists in the soil is also a critical factor in determining plant uptake. For example, in a non-ionised form a limited amount of PCP will be taken up into plants through dissolution in soil pore water and passive diffusion into the roots (Anon, 1987).

A number of models describing the accumulation of *non-ionised* organic compounds in plants exist (eg, Travis and Arms, 1988; Ryan et al, 1988). However, at pH 6.7, as much as 99 per cent of PCP is ionised and exists as pentachlorophenate anion – compared to 1 per cent ionisation at pH 2.7 (Crosby, 1981). Pentachlorophenate is highly soluble in water and leaches readily. However, while ionisation potentially increases the uptake of PCP into a plant (due to increased solubility in water), no model is available to adequately describe plant uptake of *ionised* organic compounds. Furthermore, PCP can be metabolised by plants, which will also reduce the concentrations present in the plant (Weiss et al, 1982 in Anon., 1987; Casterline et al, 1985; Haque et al, 1988; Bellin and O'Connor 1990). In fact, based on the weight of evidence of uptake, metabolism and elimination of PCP, the Canadian Council of Ministers of the Environment concluded that bioaccumulation of PCP in plants would not be significant (CCME, 1997).

In contrast, US EPA (2007) determined a BCF of 5.93 for PCP in plant foliage, which was the median of 10 data points from four studies, of which only one was based on field data. The field study (Bellin and O'Connor 1990) reported BCFs in fescue of less than 0.000 72 and 0.000 1 (dry weight), in contrast to the laboratory studies which resulted in calculated BCFs ranging from 2.3 to 46. Only one of these studies included verification of intact PCP in plant tissue (Casterline et al, 1985). Calculated BCFs for PCP (in spinach and soybean) from this study ranged from 2.3 to 7.8. Bellin and O'Connor (1990) suggested the higher BCFs determined by Casterline et al (1985) are due to the persistence of PCP in the soil. Other authors (Scheunert et al, 1986) suggest that radio-labelled PCP metabolites, including CO_2, arising from the degradation of radiolabelled PCP used in the experiments were responsible for the observed radioactivity in plant tissue – giving rise to 'erroneously' high BCF values for PCP based on that radioactivity.

Dutch authors use models for estimating plant uptake of PCP, and suggest it forms a significant pathway of exposure.

The 'Timber Treatment Guidelines' (MfE and MoH, 1997) consider plant uptake of PCP and nominally use a BCF(stem) of 0.09 (wet weight, 0.4 dry weight) for PCP: this is based on the Travis and Arms (1988) relationship and a log K_{ow} for PCP at pH 7 of 3.3 (appendix A1 in MfE and MoH, 1997). However, in the tables for calculating the guideline values (tables 5.10 and 5.12 in MfE and MoH, 1997) a soil / stem concentration of 63.5 or a BCF (stem) of 0.016 and BCF (root) of 0.078 is apparently used (the latter derived by assuming stem concentrations are 20 per cent of root concentrations). The basis for these BCFs is unclear.

However, it is considered that plant uptake of PCP is not a significant pathway of exposure – given that:

- PCP is known to be metabolised by plants (and hence there is over-prediction of plant uptake by models predicting plant uptake of organic contaminants)
- BCFs reported in a field-based studies are low
- recent papers on plants and PCP in soil focus on phytoremediation (through enhanced microbial activity associated with plant roots: eg, He et al, 2005; Lin et al, 2006) as opposed to plant uptake.

6.12.2 PCP guideline calculations

The soil guideline value calculations for PCP use the contaminant-specific parameters in table 52 and the derived SGVs are set out in table 53. PCP has been assumed to be a threshold substance. The BCF for all produce types has been set to zero, indicating no plant uptake. This results in SGVs for residential scenarios with produce uptake being the same as scenarios without produce uptake.

Table 52: Contaminant-specific parameters for PCP soil guideline value derivations

Tolerable daily intake: oral		0.000 3 mg/kg BW/day
Background exposure	Child	0.000 02 mg/kg BW/day
	Adult	0.000 02 mg/kg BW/day
Dermal absorption factor		0.24
Plant bioconcentration factor		0

Table 53: PCP soil guideline values (mg/kg)

Scenario	Combined soil guideline values		
	No produce	10% produce	50% produce
Rural residential / lifestyle block	70	70	70
Standard residential	70	70	70
High-density residential	130		
Recreational	230		
Commercial / industrial indoor worker	NL		
Commercial / industrial outdoor worker / maintenance	360		

NL = No limit.

The SGV for PCP is dominated by the soil ingestion pathway, however, dermal absorption is also significant.

It should be noted that, as technical grade PCP was contaminated with dioxins, consideration should be given to investigating dioxins even if SGVs for PCP are complied with. Investigation of New Zealand sawmill sites (T&T and SPHERE, 2008) has shown that at PCP concentrations between about 0.3 and 70 mg/kg (the latter being the residential SGV for PCP), roughly 50 per cent of samples will also exceed the residential SGV for dioxins of 0.19 µg TEQ/kg.

7 Summary of Soil Guideline Values

Table 54: Summary of soil guideline values for inorganic substances (mg/kg)

	Arsenic	Boron[1]	Cadmium (pH 5)[2]	Chromium[1]		Copper[1]	Inorganic lead	Inorganic mercury compounds[3]
				III	VI			
Rural residential / lifestyle block no produce	25	42,000	200	500,000	1,000	33,000	900	660
Rural residential / lifestyle block 10% produce	20	34,000	5	280,000	560	32,000	730	380
Rural residential / lifestyle block 50% produce	10	5,200	0.5	100,000	210	29,000	400	140
Residential no produce	29	42,000	200	500,000	1,000	33,000	900	660
Residential 10% produce	24	34,000	5	280,000	560	32,000	730	380
Residential 50% produce	14	5,200	0.5	100,000	210	29,000	400	140
High-density residential	50	75,000	370	890,000	1,800	60,000	1,600	1,200
Recreational	100	220,000	1,100	NL	5,200	170,000	4,700	3,500
Commercial / industrial outdoor worker / maintenance	70	400,000	1,600	NL	6,300	290,000	7,000	4,200

1 SGVs for boron, chromium III and copper are much greater than the soil concentration at which plant health will be affected. Plant and other environmental effects may need to be considered separately.

2 Default value is for pH 5. See table 25 for SGVs at other soil pH values.

3 The inorganic mercury SGV does not apply to elemental (pure) mercury.

Note: Shading indicates SGV used for the purpose of the NES

Table 55: Summary of soil guideline values for organic compounds

Scenario	BaP[1] (mg/kg)	DDT (mg/kg)	Dieldrin[2] (mg/kg)	PCP[3] (mg/kg)	Dioxin (µg/kg TEQ)[4]	
					TCDD	Dioxin-like PCBs[5]
Rural residential / lifestyle block no produce	110	150	28	70	0.23	0.21
Rural residential / lifestyle block 10% produce	85	90	3.1	70	0.19	0.15
Rural residential / lifestyle block 50% produce	40	35	0.67	70	0.11	0.07
Residential no produce	130	150	28	70	0.23	0.21
Residential 10% produce	100	90	3.1	70	0.19	0.15
Residential 50% produce	55	35	0.67	70	0.11	0.07
High-density residential	240	270	50	130	0.41	0.38
Recreational	440	750	110	230	1.1	0.90
Commercial / industrial outdoor worker / maintenance	300	1,000	160	360	1.4	1.2

1 SGV to be compared with the equivalent BaP concentration calculated as the sum of each of the detected concentrations of the nine PAHs listed in table 40 multiplied by the respective PEF.

2 SGV for dieldrin also applies to aldrin separately, or to the sum of aldrin and dieldrin where both are present.

3 Consideration should be given to investigating dioxins for PCP concentrations in excess of 0.3 mg/kg, see last paragraph of section 6.

4 TCDD (WHO, 2005) TEQ calculated as the sum of each of the 17 PCDDs and PDDFs, or 12 PCBs listed in table 46, multiplied by the respective 2005 WHO TEF (table 46).

5 The SGV applies to only the 12 dioxin-like PCBs. The ‘ordinary’ toxicity of the simple sum of the concentrations of these and all other detected PCBs must be considered separately.

Note: Shading indicates SGV used for the purpose of the NES

Appendix 1: Detailed Soil Guideline Value Calculations

Note: The combined $SGVs_{(health)}$ shown in the following calculations are not rounded. For rounded $SGVs_{(health)}$ see tables 54 and 55.

Table A1.1: General and scenario-specific exposure parameters

Generic factors							
Body weight (child): 15 kg			Averaging time (non-threshold): 75 years				
Body weight (adult): 70 kg			Averaging time (threshold): 6 years				
Scenario-specific factors	**Lifestyle block**	**Standard residential**	**High-density residential**	**Parks/ recreational[1]**	**Commercial/ industrial indoor worker**	**Commercial/ industrial outdoor worker**	**Unit**
Exposure frequency	350	350	350	200 / 150	230	230	day/year
Exposure duration (child)	6	6	6	6			years
Exposure duration (adult)	24	14	14	14	20	20	years
Soil ingestion rate (child)	45	45	25	15			mg/day
Soil ingestion rate (adult)	25	25	15	75	0	50	mg/day
Age-adjusted ingestion factor	26.6	23.0	13.0	6.0 / 15.0	0	14.3	mg year/kg day
Inhalation rate (child)	6.8	6.8	6.8	6.8			m^3/day
Inhalation rate (adult)	13.3	13.3	13.3	20	8	10.4	m^3/day
Age-adjusted inhalation rate	7.3	5.4	5.4	2.7 / 4.0	2.3	3.0	m^3 year/kg day
Particulate retention	0	0	0	0	0	0	dimensionless
Particle emission factor							m^3/kg
Skin area (child)	1,900	1,900	1,900	1,900			cm^2
Skin area (adult)	4,850	4,850	4,850	3,670	3,670	3,670	cm^2
Soil adherence (child)	0.04	0.04	0.02	0.04			mg/cm^2
Soil adherence (adult)	0.01	0.01	0.005	0.06	0	0.04	mg/cm^2
Age-adjusted dermal exposure factor	47.0	40.1	20.1	30.4 / 44	0	41.9	dimensionless
Produce ingestion (child)	0.0105	0.0105					kg/day (DW)
Produce ingestion (adult)	0.0322	0.0322					kg/day (DW)
Proportion of above-ground produce	0.3	0.3	0.0	0.0	0.0	0.0	dimensionless
Proportion of root (not tuber) produce	0.1	0.1	0.0	0.0	0.0	0.0	dimensionless
Proportion of tuber produce	0.6	0.6					dimensionless
Age-adjusted produce ingestion	0.0152	0.0106	0.0				kg year/kg day

1 The parks/recreational scenario has alternate scenarios for a child and adult, calculated separately, with the worst case becoming the guideline value. In such cases, child and adult parameters are shown above as child / adult.

NL = No limit.

na = Not applicable.

Health-based arsenic soil guideline values (mg/kg)		Non-threshold	
Oral RHS (mg/kg BW/day)	0.0000086	Skin absorption factor	0.005
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.011
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.011
Background intake child (mg/kg BW/day)	na	Bioconcentration factor tuber	0.001
Background intake adult (mg/kg BW/day)	na	Mean bioconcentration factor	0.005

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.0000086	0.0000086	na	0.0000086					
RHS-adult background (mg/kg BW/day)	0.0000086	0.0000086	na	0.0000086					
Rural residential / lifestyle block	25	2,861	na	na	88	18	25	20	10
Standard residential	29	3,355	na	na	126	25	29	24	14
High-density residential	52	6,710	na	na			51		
Parks / recreation	105	7,128	na				103		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	72	4,881	na				71		

NL = No limit.

na = Not applicable.

Health-based boron soil guideline values (mg/kg)				Threshold			
Oral RHS (mg/kg BW/day)		0.2		Skin absorption factor		0	
Dermal RHS (mg/kg BW/day)		na		Bioconcentration factor leaf		na	
Inhalation RHS (mg/kg BW/day)		na		Bioconcentration factor root		na	
Background intake child (mg/kg BW/day)		0.08		Bioconcentration factor tuber		na	
Background intake adult (mg/kg BW/day)		0.017		Max conc. in produce (mg/kg DW)		300	
10% produce additional background intake		0.021					
50% produce additional background intake		0.105					
Scenario	**Soil Ingestion**	**Dermal**	**Inhalation**	**Combined SGV**			
RHS-child background (mg/kg BW/day)	0.120	na	na	**No produce**	**10% produce**	**50% produce**	
RHS-adult background (mg/kg BW/day)	0.183	na	na				
Rural residential / lifestyle block	41,714	na	na	41,714	34,414	5,214	
Standard residential	41,714	na	na	41,714	34,414	5,214	
High-density residential	75,086	na	na	75,086			
Parks / recreation	219,000	na	na	219,000			
Commercial / industrial indoor worker	NL	NL	na	NL			
Commercial / industrial outdoor worker	406,578	na	na	406,578			

Note: BCF not used for produce pathway. Additional background intake subtracted from TDI on assumption that maximum plant tissue boron concentration = 30 mg/kg dry weight.

NL = No limit.

na = Not applicable.

Health-based cadmium soil guideline values – JECFA TDI (mg/kg)					Threshold					
Oral RHS (mg/kg BW/day)			0.001		Skin absorption factor		0.001			
Dermal RHS (mg/kg BW/day)			na		Bioconcentration factor leaf		Depend on pH and soil conc. See separate calculations			
Inhalation RHS (mg/kg BW/day)			na		Bioconcentration factor root					
Background intake child (mg/kg BW/day)			0.00041		Bioconcentration factor tuber					
Background intake adult (mg/kg BW/day)			0.00026		Mean bioconcentration factor					
Scenario		**Soil ingestion**	**Dermal**	**Inhalation**	**Produce ingestion**			**Combined SGV**		
					No produce	**10% produce**	**50% produce**	**No produce**	**10% produce**	**50% produce**
RHS-child background (mg/kg BW/day)		0.00059	0.00059	na	0.00059					
RHS-adult background (mg/kg BW/day)		0.00074	0.00074	na	0.00074					
Rural residential / lifestyle block	pH 5	205	121,438	na	na	5.2	0.51	205	5	0.51
	pH 5.5	205	121,438	na	na	8.5	0.86	205	8	0.86
	pH 6	205	121,438	na	na	14	1.4	205	13	1.4
	pH 6.5	205	121,438	na	na	22	2.4	205	20	2.3
	pH 7	205	121,438	na	na	34	3.8	205	29	3.7
	pH 7.5	205	121,438	na	na	52	6.1	205	41	5.9
	pH 8	205	121,438	na	na	80	9.7	205	57	9.3
Standard residential	pH 5	205	121,438	na	na	5.2	0.51	205	5	0.51
	pH 5.5	205	121,438	na	na	8.5	0.86	205	8	0.86
	pH 6	205	121,438	na	na	14	1.4	205	13	1.4
	pH 6.5	205	121,438	na	na	22	2.4	205	20	2.3
	pH 7	205	121,438	na	na	34	3.8	205	29	3.7
	pH 7.5	205	121,438	na	na	52	6.1	205	41	5.9
	pH 8	205	121,438	na	na	80	9.7	205	57	9.3
High-density residential		369	242,876	na	na			369		
Parks / recreation		1,077	212,516	na				1,071		
Commercial / industrial indoor worker		NL	NL	na				NL		
Commercial / industrial outdoor worker		1,644	559,975	na				1,639		

NL = No limit.

na = Not applicable.

Health-based chromium III soil guideline values (mg/kg)		Threshold	
Oral RHS (mg/kg BW/day)	1.5	Skin absorption factor	0
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.0324
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.0324
Background intake child (mg/kg BW/day)	0.0012	Bioconcentration factor tuber	0.0324
Background intake adult (mg/kg BW/day)	0.00053	Mean bioconcentration factor	0.0324

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.0000086	0.0000086	na	0.0000086					
RHS-adult background (mg/kg BW/day)	0.0000086	0.0000086	na	0.0000086					
Rural residential / lifestyle block	495,357	na	na	na	655,234	131,047	495,357	282,094	103,631
Standard residential	495,357	na	na	na	655,234	131,047	495,357	282,094	103,631
High-density residential	891,643	na	na	na			891,643		
Parks / recreation	2,600,625[a]	na	na				2,600,625[a]		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	3,165,978[a]	na	na				3,165,978[a]		

Note: Background intake less than 5% of the TDI, therefore background taken as 5% of the TDI.

NL = No limit.

na = Not applicable.

a calculated values greater then 1,000,000 are impossible in reality and indicate no limit.

Health-based chromium VI soil guideline values (mg/kg)		Threshold	
Oral RHS (mg/kg BW/day)	0.003	Skin absorption factor	0
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.0324
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.0324
Background intake child (mg/kg BW/day)	No data	Bioconcentration factor tuber	0.0324
Background intake adult (mg/kg BW/day)	No data	Mean bioconcentration factor	0.0324

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00285	na	na	0.00285					
RHS-adult background (mg/kg BW/day)	0.00285	na	na	0.00285					
Rural residential / lifestyle block	991	na	na	na	1,310	262	991	564	207
Standard residential	991	na	na	na	1,310	262	991	564	207
High-density residential	1,783	na	na	na			1,783		
Parks / recreation	5,201	na	na				5,201		
Commercial / industrial indoor worker	NL	na	na				NL		
Commercial / industrial outdoor worker	6,332	na	na				6,332		

Note: There is no data on the background intake of CrVI. In accordance with MfE (2010b) the background intake is taken to be 5 per cent of the tolerable daily intake.

NL = No limit.

na = Not applicable.

Health-based copper soil guideline values (mg/kg)			Threshold			
Oral RHS (mg/kg BW/day)		0.15	Skin absorption factor		0	
Dermal RHS (mg/kg BW/day)		na	Bioconcentration factor leaf		na	
Inhalation RHS (mg/kg BW/day)		na	Bioconcentration factor root		na	
Background intake child (mg/kg BW/day)		0.056	Bioconcentration factor tuber		na	
Background intake adult (mg/kg BW/day)		0.02	Max conc. in produce (mg/kg DW)		30	
10% produce additional background intake		0.0021				
50% produce additional background intake		0.0105				
Scenario	**Soil Ingestion**	**Dermal**	**Inhalation**	**Combined SGV**		
RHS-child background (mg/kg BW/day)	0.0940	na	na	**No produce**	**10% produce**	**50% produce**
RHS-adult background (mg/kg BW/day)	0.1300	na	na			
Rural residential / lifestyle block	32,676	na	na	32,676	31,946	29,026
Standard residential	32,676	na	na	32,676	31,946	29,026
High-density residential	58,817	na	na	58,817		
Parks / recreation	171,550	na	na	171,550		
Commercial / industrial indoor worker	NL	NL	na	NL		
Commercial / industrial outdoor worker	288,826	na	na	288,826		

Note: BCF not used for produce pathway. Additional background intake subtracted from TDI on assumption that maximum plant tissue boron concentration = 300 mg/kg dry weight.

NL = No limit.

na = Not applicable.

Health-based inorganic lead soil guideline values (mg/kg)		Threshold	
Oral RHS (mg/kg BW/day)	0.00357	Skin absorption factor	0
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.019
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.015
Background intake child (mg/kg BW/day)	0.00097	Bioconcentration factor tuber	0.005
Background intake adult (mg/kg BW/day)	0.00041	Mean bioconcentration factor	0.0102

Scenario	**Soil ingestion**	**Dermal**	**Inhalation**	**Produce ingestion**			**Combined SGV**		
				No produce	**10% produce**	**50% produce**	**No produce**	**10% produce**	**50% produce**
RHS-child background (mg/kg BW/day)	0.00260	na	na	0.00260					
RHS-adult background (mg/kg BW/day)	0.00316	na	na	0.00316					
Rural residential / lifestyle block	904	na	na	na	3,798	760	904	730	413
Standard residential	904	na	na	na	3,798	760	904	730	413
High-density residential	1,627	na	na	na			1,627		
Parks / recreation	4,745	na	na				4,745		
Commercial / industrial indoor worker	NL	na	na				NL		
Commercial / industrial outdoor worker	7021	na	na				7,021		

NL = No limit.

na = Not applicable.

Health-based inorganic mercury soil guideline values (mg/kg)		Threshold	
Oral RHS (mg/kg BW/day)	0.002	Skin absorption factor	0
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.04
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.07
Background intake child (mg/kg BW/day)	0.00005	Bioconcentration factor tuber	0.02
Background intake adult (mg/kg BW/day)	0.000065	Mean bioconcentration factor	0.031

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00190	na	na	0.00190					
RHS-adult background (mg/kg BW/day)	0.00190	na	na	0.00190					
Rural residential / lifestyle block	660	na	na	na	913	1,836	660	383	143
Standard residential	660	na	na	na	913	1,836	660	383	143
High-density residential	1,189	na	na	na			1,189		
Parks / recreation	3,468	na	na				3,468		
Commercial / industrial indoor worker	NL	na	na				NL		
Commercial / industrial outdoor worker	4,221	na	na				4,221		

Note: Background intake less than 5 per cent of the TDI, therefore background taken as 5 per cent of the TDI.

NL = No limit.

na = Not applicable.

Health-based BaP soil guideline values (mg/kg)		Non-threshold	
Oral RHS (mg/kg BW/day)	0.000043	Skin absorption factor	0.06
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.005
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.031
Background intake child (mg/kg BW/day)	na	Bioconcentration factor tuber	0.004
Background intake adult (mg/kg BW/day)	na	Mean bioconcentration factor	0.007

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.000043	0.000043	na	0.000043					
RHS-adult background (mg/kg BW/day)	0.000043	0.000043	na	0.000043					
Rural residential / lifestyle block	127	1,192	na	na	315	63	114	84	41
Standard residential	146	1,398	na	na	452	90	132	102	54
High-density residential	259	2,796	na	na			237		
Parks / recreation	523	2,970	na				445		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	358	2,034	na				305		

NL = No limit.

na = Not applicable.

Health-based ∑DDT soil guideline values (mg/kg)		Threshold	
Oral RHS (mg/kg BW/day)	0.0005	Skin absorption factor	0.018
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.012
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.038
Background intake child (mg/kg BW/day)	0.0000511	Bioconcentration factor tuber	0.038
Background intake adult (mg/kg BW/day)	0.0000193	Mean bioconcentration factor	0.0302

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.0004489	0.0004489	na	0.000449					
RHS-adult background (mg/kg BW/day)	0.0004750	0.0004750	na	0.000475					
Rural residential / lifestyle block	156	5,133	na	na	221	44	151	90	34
Standard residential	156	5,133	na	na	221	44	151	90	34
High-density residential	281	10,266	na	na			273		
Parks / recreation	819	8,983	na				751		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1,055	19,969	na				1,002		

NL = No limit.

na = Not applicable.

Health-based dieldrin soil guideline values (mg/kg)		Threshold	
Oral RHS (mg/kg BW/day)	0.0001	Skin absorption factor	0.1
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.41
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.41
Background intake child (mg/kg BW/day)	0.0000036	Bioconcentration factor tuber	0.41
Background intake adult (mg/kg BW/day)	0.0000014	Mean bioconcentration factor	0.41

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.000095	0.000095	na	0.000095					
RHS-adult background (mg/kg BW/day)	0.000095	0.000095	na	0.000095					
Rural residential / lifestyle block	33	196	na	na	3.45	0.69	28	3.1	0.67
Standard residential	33	196	na	na	3.45	0.69	28	3.1	0.67
High-density residential	59	391	na	na			52		
Parks / recreation	173	342	na				115		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	211	719	na				163		

Note: Background intake less than 5 percent of the TDI, therefore background taken as 5 per cent of the TDI.

NL = No limit.

na = Not applicable.

Health-based dioxin (TCDD/PeCDD dominant) soil guideline values (µg TEQ/kg)		Threshold	
Oral RHS (µg/kg BW/day)	0.000001	Skin absorption factor	0.02
Dermal RHS (µg/kg BW/day)	na	Bioconcentration factor leaf	0
Inhalation RHS (µg/kg BW/day)	na	Bioconcentration factor root	0
Background intake child (µg/kg BW/day)	0.00000033	Bioconcentration factor tuber	0
Background intake adult (µg/kg BW/day)	0.00000033		0.24
		Mean bioconcentration factor	0.0096

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067					
RHS-adult background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067					
Rural residential / lifestyle block	0.233	6.9	na	na	1.04	0.208	0.23	0.19	0.11
Standard residential	0.233	6.9	na	na	1.04	0.208	0.23	0.19	0.11
High-density residential	0.419	14	na	na			0.41		
Parks / recreation	1.22	12	na				1.1		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1.49	25	na				1.4		

NL = No limit.
na = Not applicable.

Health-based dioxin (OCDD/HpCDD dominant) soil guideline values (µg TEQ/kg)		Threshold	
Oral RHS (µg/kg BW/day)	0.000001	Skin absorption factor	0.02
Dermal RHS (µg/kg BW/day)	na	Bioconcentration factor leaf	0
Inhalation RHS (µg/kg BW/day)	na	Bioconcentration factor root	0
Background intake child (µg/kg BW/day)	0.00000033	Bioconcentration factor tuber	0
Background intake adult (µg/kg BW/day)	0.00000033		0.017
		Mean bioconcentration factor	0.00068

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067					
RHS-adult background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067					
Rural residential / lifestyle block	0.233	6.9	na	na	14.7	2.94	0.23	0.22	0.21
Standard residential	0.233	6.9	na	na	14.7	2.94	0.23	0.22	0.21
High-density residential	0.419	14	na	na			0.41		
Parks / recreation	1.22	12	na				1.1		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1.49	25	na				1.4		

NL = No limit.

na = Not applicable.

Health-based dioxin-like PCB soil guideline values (µg TEQ/kg)		Threshold	
Oral RHS (µg/kg BW/day)	0.000001	Skin absorption factor	0.07
Dermal RHS (µg/kg BW/day)	na	Bioconcentration factor leaf	0
Inhalation RHS (µg/kg BW/day)	na	Bioconcentration factor root	0
Background intake child (µg/kg BW/day)	0.00000033	Bioconcentration factor tuber	0
Background intake adult (µg/kg BW/day)	0.00000033		0.45
		Mean bioconcentration factor	0.018

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067					
RHS-adult background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067					
Rural residential / lifestyle block	0.233	2.0	na	na	0.555	0.111	0.21	0.15	0.07
Standard residential	0.233	2.0	na	na	0.555	0.111	0.21	0.15	0.07
High-density residential	0.419	3.9	na	na			0.38		
Parks / recreation	1.22	3.4	na				0.90		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1.49	7.2	na				1.2		

NL No limit.

na Not applicable.

Health-based pentachlorophenol soil guideline values (mg/kg)		Threshold	
Oral RHS (mg/kg BW/day)	0.0003	Skin absorption factor	0.24
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0
Background intake child (mg/kg BW/day)	0.00002	Bioconcentration factor tuber	0
Background intake adult (mg/kg BW/day)	0.00002	Mean bioconcentration factor	na

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00028	0.00028	na	0.00028					
RHS-adult background (mg/kg BW/day)	0.00028	0.00028	na	0.00028					
Rural residential / lifestyle block	97	240	na	na	na	na	69	69	69
Standard residential	97	240	na	na	na	na	69	69	69
High-density residential	175	480	na	na			128		
Parks / recreation	511	420	na				231		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	622	883	na				365		

NL = No limit.

na = Not applicable.

Appendix 2: Detailed SGV$_{(health)}$ Calculations for Cadmium

Note: The combined SGVs$_{(health)}$ shown in the following calculations are not rounded. For rounded SGVs$_{(health)}$ see tables 54 and 55.

Health-based cadmium soil guideline value (mg/kg) at pH 5 – JECFA TDI		Threshold		
Oral RHS (mg/kg BW/day)	0.001	Homegrown produce %	10%	50%
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	3.56	6.18
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.90	2.24
Background exposure child (mg/kg BW/day)	0.00041	Bioconcentration factor tuber	0.90	2.24
Background exposure adult (mg/kg BW/day)	0.00026	Mean bioconcentration factor	1.70	3.42
Skin absorption factor	0.001			

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00059	0.00059	na	0.00059					
RHS-adult background (mg/kg BW/day)	0.00074	0.00074	na	0.00074					
Rural residential / lifestyle block	205	121,438	na	na	5.2	0.5	205	5.1	0.51
Standard residential	205	121,438	na	na	5.2	0.5	205	5.1	0.51
High-density residential	369	242,876	na	na			369		
Parks / recreation	1,077	212,516	na				1,071		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1,644	559,975	na				1,639		

NL = No limit.

na = Not applicable.

Health-based cadmium soil guideline value (mg/kg) at pH 5.5 – JECFA TDI		Threshold		
Oral RHS (mg/kg BW/day)	0.001	Homegrown produce %	10%	50%
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	2.32	3.99
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.49	1.20
Background exposure child (mg/kg BW/day)	0.00041	Bioconcentration factor tuber	0.49	1.20
Background exposure adult (mg/kg BW/day)	0.00026	Mean bioconcentration factor	1.04	2.04
Skin absorption factor	0.001			

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00059	0.00059	na	0.00059					
RHS-adult background (mg/kg BW/day)	0.00074	0.00074	na	0.00074					
Rural residential / lifestyle block	205	121,438	na	na	8.5	0.9	205	8.1	0.86
Standard residential	205	121,438	na	na	8.5	0.9	205	8.1	0.86
High-density residential	369	242,876	na	na			369		
Parks / recreation	1,077	212,516	na				1,071		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1,644	559,975	na				1,639		

NL = No limit.

na = Not applicable.

Health-based cadmium soil guideline value (mg/kg) at pH 6 – JECFA TDI		Threshold		
Oral RHS (mg/kg BW/day)	0.001	Homegrown produce %	10%	50%
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	1.52	2.58
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.27	0.64
Background exposure child (mg/kg BW/day)	0.00041	Bioconcentration factor tuber	0.27	0.64
Background exposure adult (mg/kg BW/day)	0.00026	Mean bioconcentration factor	0.64	1.23
Skin absorption factor	0.001			

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00059	0.00059	na	0.00059					
RHS-adult background (mg/kg BW/day)	0.00074	0.00074	na	0.00074					
Rural residential / lifestyle block	205	121,438	na	na	13.7	1.4	205	13	1.4
Standard residential	205	121,438	na	na	13.7	1.4	205	13	1.4
High-density residential	369	242,876	na	na			369		
Parks / recreation	1,077	212,516	na				1,071		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1,644	559,975	na				1,639		

NL = No limit.

na = Not applicable.

Health-based cadmium soil guideline value (mg/kg) at pH 6.5 – JECFA TDI		Threshold		
Oral RHS (mg/kg BW/day)	0.001	Homegrown produce %	10%	50%
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	1.00	1.68
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.15	0.35
Background exposure child (mg/kg BW/day)	0.00041	Bioconcentration factor tuber	0.15	0.35
Background exposure adult (mg/kg BW/day)	0.00026	Mean bioconcentration factor	0.41	0.75
Skin absorption factor	0.001			

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00059	0.00059	na	0.00059					
RHS-adult background (mg/kg BW/day)	0.00074	0.00074	na	0.00074					
Rural residential / lifestyle block	205	121,438	na	na	21.7	2.4	205	20	2.3
Standard residential	205	121,438	na	na	21.7	2.4	205	20	2.3
High-density residential	369	242,876	na	na			369		
Parks / recreation	1,077	212,516	na				1,071		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1,644	559,975	na				1,639		

NL = No limit.

na = Not applicable.

Health-based cadmium soil guideline value (mg/kg) at pH 7 – JECFA TDI		Threshold		
Oral RHS (mg/kg BW/day)	0.001	Homegrown produce %	10%	50%
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.67	1.09
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.08	0.19
Background exposure child (mg/kg BW/day)	0.00041	Bioconcentration factor tuber	0.08	0.19
Background exposure adult (mg/kg BW/day)	0.00026	Mean bioconcentration factor	0.26	0.46
Skin absorption factor	0.001			

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00059	0.00059	na	0.00059					
RHS-adult background (mg/kg BW/day)	0.00074	0.00074	na	0.00074					
Rural residential / lifestyle block	205	121,438	na	na	34.0	3.8	205	29	3.7
Standard residential	205	121,438	na	na	34.0	3.8	205	29	3.7
High-density residential	369	242,876	na	na			369		
Parks / recreation	1,077	212,516	na				1,071		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1,644	559,975	na				1,639		

NL = No limit.

na = Not applicable.

Health-based cadmium soil guideline value (mg/kg) at pH 7.5 – JECFA TDI		Threshold		
Oral RHS (mg/kg BW/day)	0.001	Homegrown produce %	10%	50%
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.45	0.72
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.05	0.10
Background exposure child (mg/kg BW/day)	0.00041	Bioconcentration factor tuber	0.05	0.10
Background exposure adult (mg/kg BW/day)	0.00026	Mean bioconcentration factor	0.17	0.29
Skin absorption factor	0.001			

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00059	0.00059	na	0.00059					
RHS-adult background (mg/kg BW/day)	0.00074	0.00074	na	0.00074					
Rural residential / lifestyle block	205	121,438	na	na	52	6.1	205	41	5.9
Standard residential	205	121,438	na	na	52	6.1	205	41	5.9
High-density residential	369	242,876	na	na			369		
Parks / recreation	1,077	212,516	na				1,071		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1,644	559,975	na				1,639		

NL = No limit.

na = Not applicable.

Health-based cadmium soil guideline value (mg/kg) at pH 8 – JECFA TDI		Threshold		
Oral RHS (mg/kg BW/day)	0.001	Homegrown produce %	10%	50%
Dermal RHS (mg/kg BW/day)	na	Bioconcentration factor leaf	0.30	0.47
Inhalation RHS (mg/kg BW/day)	na	Bioconcentration factor root	0.03	0.06
Background exposure child (mg/kg BW/day)	0.00041	Bioconcentration factor tuber	0.03	0.06
Background exposure adult (mg/kg BW/day)	0.00026	Mean bioconcentration factor	0.11	0.18
Skin absorption factor	0.001			

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Combined SGV		
				No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (mg/kg BW/day)	0.00059	0.00059	na	0.00059					
RHS-adult background (mg/kg BW/day)	0.00074	0.00074	na	0.00074					
Rural residential / lifestyle block	205	121,438	na	na	80	9.7	205	57	9.3
Standard residential	205	121,438	na	na	80	9.7	205	57	9.3
High-density residential	369	242,876	na	na			369		
Parks / recreation	1,077	212,516	na				1,071		
Commercial / industrial indoor worker	NL	NL	na				NL		
Commercial / industrial outdoor worker	1,644	559,975	na				1639		

NL = No limit.

na = Not applicable.

Appendix 3: Determination of the Amount of Produce Consumed and its Dry Weight

Estimating the intake of soil contaminants via the home-grown produce pathway requires as a basic parameter an estimate of the average child and adult's produce consumption. The estimate presented here is slightly modified from appendix 2 of Cavanagh (2005b), which in turn was a modification from Cavanagh and Proffitt (2005). The major modification from Cavanagh (2005b) is dividing vegetable types into root, tuber and above-ground (including cucurbits), rather than just below-ground and above-ground vegetables.

Typically, only the consumption of vegetables is taken into account in the derivation of soil guideline values (eg, the Netherlands, Canada), although the UK also takes fruit into account. In New Zealand, consideration will be given only to the consumption of vegetables. This approach is valid for two reasons. Firstly, fruit is not widely grown in residential gardens. Secondly, contaminant uptake and translocation to fruit (in fruit trees) is considered to be negligible (MfE and MoH, 1997; MfE, 1999).

For calculation purposes, table A3.1 shows the amount of produce consumed by different age and gender groups based on simulated diets (from Brinsdon, 2004). Quantities of fruit, vegetables and totals have been calculated for the average adult and the average child.

Table A3.1: Amount of fruit and vegetables consumed (grams per day) by different age-gender groups based on simulated diets for the New Zealand 2003/04 Total Diet Surveys

Produce	Young male 19–24 years	Male 25+ years	Female 25+ years	Average adult[1]	Child 5–6 years	Child 1–3 years	Child 6–12 months	Average child[2]
Vegetables	224	294	232	254	115	63	42.5	77
Fruit	110	138	141	136	140	77	62.5	95
Total	336	432	373	390	256	140	105	173
Body weight[3]	70	80	65	70	20	13	10	15

Source: Adapted from Brinsdon (2004).

1 Adjusted for difference in body weight before averaging and converted to the amount consumed by a 70-kg adult.

2 Adjusted for difference in body weight before averaging and converted to the amount consumed by a 15-kg child.

3 Vannoort et al, 2000.

Vegetable types

The extent of uptake of contaminants by different vegetables is dependent on the individual vegetable. In the development of soil guideline values, vegetables are most frequently separated into root (eg, carrots, potato) and above-ground (eg, lettuce, peas) vegetables. The vegetables considered in the development of the simulated diets were grouped into root vegetables, tubers, 'above-ground' vegetables (including the cucurbits), and vegetables unlikely to be grown at home, as shown in table A3.2.

Table A3.2: Vegetables considered in simulated diets, and their grouping

'Above-ground' vegetables	Root vegetables	Unlikely to be grown at home
Bean	Carrot	Avocado
Broccoli / cauliflower	Kumara (tuber)	Celery
Cabbage	Potato (tuber)	Mushrooms
Capsicum		
Courgette (cucurbit)		
Cucumber (cucurbit)		
Lettuce		
Onion		
Peas		
Pumpkin (cucurbit)		
Silverbeet		
Tomato		

Dry weight contents of individual vegetables were determined from the average of the values provided in US EPA (1996b; 1997) and Alloway et al (1988). The exception was avocado, whose dry weight content was derived from the average dry weight of the edible portion of avocado over a growing season (Hofshi et al, 2000).

Table A3.4 provides a summary of the amount of vegetables consumed by an average adult and child, using the above grouping. Plant uptake of contaminants is typically expressed on a dry-weight basis, and therefore requires conversion from the wet weights typically used to express the amount of produce consumed. However, the dry-weight content of different vegetables is also variable. The dry weight proportion of individual vegetables (table A3.3) and the amount of each vegetable consumed was used to determine the amount of produce consumed on a dry-weight basis (table A3.4). This also enables derivation of consumption-weighted dry weights for the different types of vegetables, and for different receptors. The consumption-weighted dry-weight content does not vary much for the different receptors (table A3.4), with root vegetables having an average dry-weight content of about 0.18, and above-ground vegetables of 0.09.

Table A3.3: Fresh to dry conversion factors for vegetables

Vegetable	Conversion factor
Avocado	0.25
Beans	0.094
Broccoli / cauliflower	0.089
Cabbage	0.083
Capsicum	0.15
Carrot	0.110
Celery	0.116
Courgette	0.035
Cucumber	0.079
Kumara	0.21
Lettuce	0.046
Mushrooms	0.082
Onion	0.124
Peas	0.145
Potatoes	0.210
Potatoes, with skin	0.167
Pumpkin	0.084
Silverbeet (spinach)	0.073
Tomato	0.058

Table A3.4: Amounts of different vegetable groups consumed by an average adult and average child, with percentage of total vegetables given in parenthesis

Produce type	Wet weight (g/day) and percentage		Dry weight (g/day) and percentage		Consumption-weighted dry-weight content (g DW/g FW)	
	Average adult	Average child	Average adult	Average child	Adult	Child
Tuber vegetable	92 (36%)	33 (43%)	18.9 (56%)	6.6 (63%)	0.21	0.20
Root vegetables	18 (7%)	9 (12%)	1.9 (6%)	1.0 (10%)	0.11	0.11
Above-ground vegetables (not including cucurbits)	119 (47%)	25 (33%)	10 (30%)	2.4 (23%)	0.08	0.10
Cucurbits (courgette, pumpkin)	14 (6%)	6 (8%)	1.4 (4%)	0.46 (4%)	0.10	0.08
Subtotal	243	73	32.2	10.5	0.13	0.14
Unlikely to be grown at home	10 (4%)	3.6 (5%)	1.4 (4%)	0.57 (5%)	–	–
Total	253	76.6	33.6	11.03	0.13	0.14

Source: Adapted from Brinsdon (2004).

Appendix 4: Dioxin SGV$_{(health)}$ Calculations with Egg Consumption

Health-based dioxin (TCDD/PeCDD-dominant) soil guideline value (µg TEQ/kg)		Threshold	
Oral RHS (µg/kg BW/day)	0.000001	Skin absorption factor	0.02
Dermal RHS (µg/kg BW/day)	na	Bioconcentration factor leaf	0
Inhalation RHS (µg/kg BW/day)	na	Bioconcentration factor root	0
Background exposure child (µg/kg BW/day)	0.00000033	Bioconcentration factor tuber	0
Background exposure adult (µg/kg BW/day)	0.00000033	Bioconcentration factor cucurbits	0.24
Egg lipid BCF	1.9	Mean bioconcentration factor	0.0096

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Average egg ingestion	2 x average egg ingestion	Combined SGV no eggs			Combined SGV average eggs			Combined SGV with 2 x average eggs		
				No produce	10% produce	50% produce			No produce	10% produce	50% produce	No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067			0.00000067	0.00000067									
RHS-adult background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067			0.00000067	0.00000067									
Rural residential / lifestyle block	0.233	6.9	na	na	1.04	0.208	0.006	0.003	0.225	0.185	0.108	0.006	0.006	0.006	0.003	0.003	0.003
Standard residential	0.233	6.9	na	na	1.04	0.208	0.006	0.003	0.225	0.185	0.108	0.006	0.006	0.006	0.003	0.003	0.003
High-density residential	0.419	14	na	na					0.407			0.407			0.407		
Parks / recreation	1.22	12	na						1.11			1.11			1.11		
Commercial / industrial indoor worker	NL	NL	na						NL			NL			NL		
Commercial / industrial outdoor worker	1.49	25	na						1.41			1.41			1.41		

NL = No limit.

na = Not applicable.

Health-based dioxin (OCDD/HpCDD-dominant) soil guideline value (µg TEQ/kg)		Threshold	
Oral RHS (µg/kg BW/day)	0.000001	Skin absorption factor	0.02
Dermal RHS (µg/kg BW/day)	na	Bioconcentration factor leaf	0
Inhalation RHS (µg/kg BW/day)	na	Bioconcentration factor root	0
Background exposure child (µg/kg BW/day)	0.00000033	Bioconcentration factor tuber	0
Background exposure adult (µg/kg BW/day)	0.00000003	Bioconcentration factor cucurbits	0.017
Egg lipid BCF	0.7	Mean bioconcentration factor	0.00068

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Average egg ingestion	2 x average egg ingestion	Combined SGV no eggs			Combined SGV average eggs			Combined SGV with 2 x average eggs		
				No produce	10% produce	50% produce			No produce	10% produce	50% produce	No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067			0.00000067	0.00000067									
RHS-adult background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067			0.00000067	0.00000067									
Rural residential / lifestyle block	0.233	6.9	na	na	14.7	2.9	0.017	0.008	0.23	0.222	0.209	0.015	0.015	0.015	0.008	0.008	0.008
Standard residential	0.233	6.9	na	na	14.7	2.9	0.017	0.008	0.23	0.222	0.209	0.015	0.015	0.015	0.008	0.008	0.008
High-density residential	0.419	14	na	na					0.41			0.41			0.41		
Parks / recreation	1.22	12	na						1.1			1.1			1.1		
Commercial / industrial indoor worker	NL	NL	na						NL			NL			NL		
Commercial / industrial outdoor worker	1.49	25	na						1.4			1.4			1.4		

NL = No limit.

na = Not applicable.

Health-based dioxin-like PCBs soil guideline value (µg TEQ/kg)		Threshold	
Oral RHS (µg/kg BW/day)	0.000001	Skin absorption factor	0.07
Dermal RHS (µg/kg BW/day)	na	Bioconcentration factor leaf	0
Inhalation RHS (µg/kg BW/day)	na	Bioconcentration factor root	0
Background exposure child (µg/kg BW/day)	0.00000033	Bioconcentration factor tuber	0
Background exposure adult (µg/kg BW/day)	0.00000033	Bioconcentration factor cucurbits	0.45
Egg lipid BCF	17	Mean bioconcentration factor	0.018

Scenario	Soil ingestion	Dermal	Inhalation	Produce ingestion			Average egg ingestion	2 x average egg ingestion	Combined SGV no eggs			Combined SGV average eggs			Combined SGV with 2 x average eggs		
				No produce	10% produce	50% produce			No produce	10% produce	50% produce	No produce	10% produce	50% produce	No produce	10% produce	50% produce
RHS-child background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067			0.00000067	0.00000067									
RHS-adult background (µg/kg BW/day)	0.00000067	0.00000067	na	0.00000067			0.00000067	0.00000067									
Rural residential / lifestyle block	0.233	2.0	na	na	0.55	0.11	0.0007	0.0003	0.21	0.151	0.072	0.001	0.001	0.0007	0.0003	0.0003	0.0003
Standard residential	0.233	2.0	na	na	0.55	0.11	0.0007	0.0003	0.21	0.151	0.072	0.001	0.001	0.0007	0.0003	0.0003	0.0003
High-density residential	0.419	3.9	na	na					0.38			0.38			0.38		
Parks / recreation	1.22	3.4	na						0.90			0.90			0.90		
Commercial / industrial indoor worker	NL	NL	na						NL			NL			NL		
Commercial / industrial outdoor worker	1.49	7.2	na						1.2			1.2			1.2		

NL = No limit.

na = Not applicable.

Appendix 5: International Comparison of Soil Guideline Value Derivation

A5.1 Introduction

New Zealand has four industry-based guideline documents providing soil guideline values for a variety of contaminants (MfE and MoH, 1997; MfE, 1997, 1999, 2006a). The United States risk assessment procedures set out in US EPA (1989a) provided the basis for the derivation of the soil guideline values in these documents. In addition, New Zealand has typically looked to the United States and a number of other countries, in particular Australia, Canada, the Netherlands and the United Kingdom, to provide soil guideline values where New Zealand guidelines do not exist. A compilation of guidelines from these countries formed Contaminated Land Management Guideline No. 2 (MfE, 2003). Each of these overseas countries has well-developed contaminated sites risk assessment frameworks, with guideline documents readily available, including Dutch documents available in English.

This section explores the derivation methods and differences of the New Zealand documents and documents from the five overseas jurisdictions as background to proposing a consistent methodology for New Zealand, drawing on and updating earlier work for the Ministry for the Environment by Cavanagh (2003) and Cavanagh and O'Halloran (2003).

Given that the US has provided the basis for New Zealand's guideline, the US EPA guidance is considered first, then the other overseas jurisdictions, and finally the existing New Zealand guidelines.

A5.2 Regulatory context

The derivation of human-health numeric values largely focuses on the exposure of a given human receptor in designated scenarios that typically relate to different land uses. But the various choices in soil guideline derivation, including the selection of designated scenarios and exposure parameters, are influenced by the particular policy and legislation of each country. The various legislative and policy environments overseas may not (and frequently do not) accord with the New Zealand situation and therefore many overseas choices do not suit New Zealand.

Carlon (2007) recently compared the reasons for the differences in SGVs for 12 European countries. While only two of the countries studied are considered here (the Netherlands and the UK), his findings are generally applicable. Carlon identified five, often overlapping, categories of reasons for differences in SGVs:

1. **Geographical and biological**: associated with the environmental variability between countries (across Europe in his case).
2. **Socio-cultural**: associated with the variability of social behaviours and land use between countries.
3. **Regulatory**: associated with regulatory requirements, such as constitutional aspects or commonalities with existing laws.
4. **Political**: associated with the prioritisation of environmental and economic values, as made by policy makers and regulators, or forced by societal views.
5. **Scientific**: associated with arguments of different scientific views.

This document largely focuses on the last of these, with some input from socio-cultural aspects such as the New Zealand lifestyle (eg, ownership of rural 'lifestyle' properties and vegetable growing habits). However, the lack of New Zealand-specific data does not always allow proper accounting of how New Zealand's lifestyle and work habits differ from other countries.

Geographical and biological differences have a limited role in New Zealand. Its climate is sufficiently consistent across the country that the large differences in summer and winter lifestyles forced by heavy winter snows in some countries do not have to be taken into account in New Zealand guidelines. On the other hand, the lack of good New Zealand data and/or the ready accessibility of overseas data (particularly from the US) means that New Zealand typically draws on overseas data to describe common activities and the physical characteristics of the 'average' person.

This document has attempted to avoid 'importing' regulatory or policy decisions from other countries in making recommendations, other than the inevitable 'scientific' policy embedded in the relatively generic risk assessment frameworks common to most countries. An example of country-specific policies is Canada choosing to assign only one-fifth of the acceptable daily intake of a substance to the soil compartment, dividing the remainder amongst air, water, consumer products and food. There is no scientific justification for this – it is purely a policy decision.

The differences in approach can make large difference in final soil guideline values. In his analysis of values across the European Union, Carlon (2007) found a difference of one to two orders of magnitude between values for particular metals and metalloids; and an even larger range of two to three orders of magnitude for some organic compounds commonly found on contaminated sites.

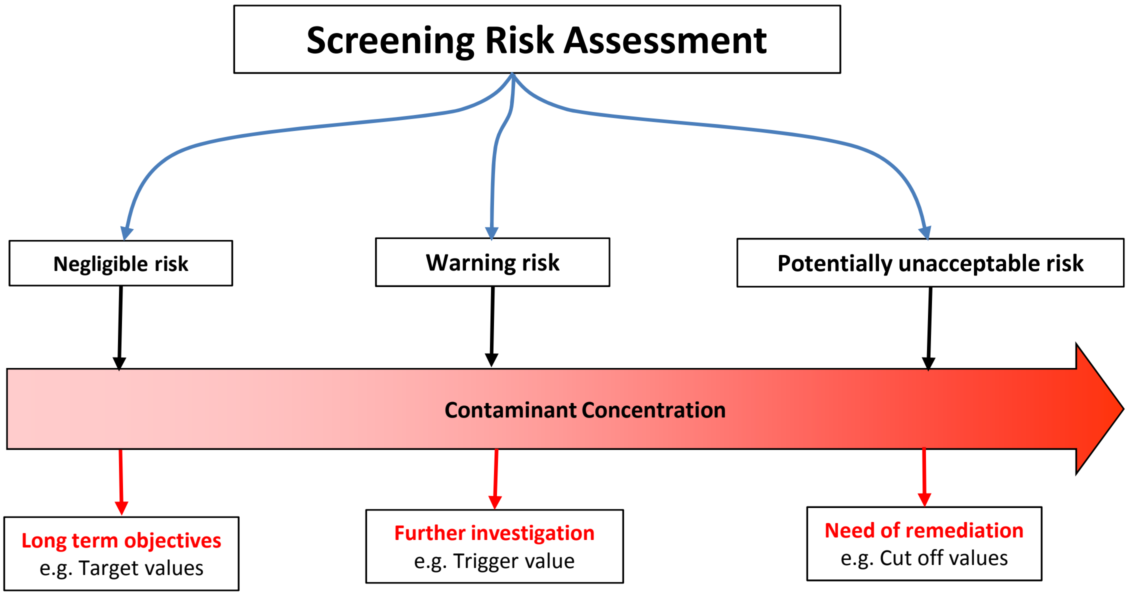
Two important policy issues require mentioning at this point: the purpose to which soil guideline values are to be put, and whether the values are to protect human health only, or human and ecological receptors.

Soil guideline values are generally derived for one of the following purposes:

- investigation – if the stated value is exceeded, further site investigation is required
- intervention – if the stated value is exceeded, remediation (which may include management actions or imposition of a barrier to contact) or assessment of the urgency of remediation, is required. Typically an intervention value is higher than an investigation value
- remediation goals – soil concentrations to which contaminated soil should be remediated or some other management action taken to minimise exposure. Such values may be higher than the equivalent intervention value.

Carlon (2007) set this down in diagrammatic form, as shown in figure A5.1. This shows that for there to be a negligible risk to all receptors, very low soil values have to be set (called target values in the figure). These tend to be aspirational, rather than realistic goals, and are not very useful for assessing whether a contaminated site is safe for its present use (with or without management) or whether further assessment is necessary. By their very nature, many sites would fail on the basis of screening using target values; using such values as remediation goals would result in many sites requiring expensive clean-ups with limited, if any, overall human-health benefit.

Figure A5.1: Derivation of screening values based on different risk levels and applications



Source: after Carlon, 2007.

In the middle of the continuum are values that provide a warning of potential risk – the orange light to use the traffic light analogy. Measured values below the designated screening value require no further action while concentrations above the value require further investigation. Such values are often called 'trigger', 'screening' or 'investigation' values. Exceeding such a value does not necessarily indicate a risk – particularly as most sites will not perfectly match the conservative generic assumptions behind the guideline value – but there is sufficient potential for risk that further enquiries should be made.

Screening values can move up or down the concentration axis depending on how conservative the policy and regulatory authorities wish to be (influenced by the various socio-cultural and other contexts). Lower values are more protective but potentially a greater unnecessary economic burden on a society, while higher values will be less protective, with the potential for some sites that actually present a risk slipping through the net.

The highest values are remediation or 'intervention' values. Such values indicate a risk is a distinct possibility and some form of intervention to moderate that risk is required. Site-specific assessment could result in even higher values being set, based on the actual contaminant exposure assessed for the site.

Most countries use more or less conservative values in the middle as screening values, as shown in table A5.1 (modified from Cavanagh, 2003). The art in setting such values is having them high enough to not trigger too much unnecessary further investigation, but not so high as to result in a risk for some sites. The latter is generally unlikely, because the toxicological values generally have large factors of safety given the uncertainty of the science. Not setting them too low will also result in less remediation and greater ongoing use of marginally contaminated sites: screening values are inevitably used as clean-up goals, either because a small site is not worth investigating further or because of regulatory expediency. The burden of unnecessarily low screening levels will typically fall on the small property owner; owners of larger, more valuable sites will more often have the resources to investigate further and possibly avoid remediation.

For this report, the purposes of soil guideline values, as enunciated in the MfE position paper (MfE, 2007), are to:

- serve as Tier 1 or screening criteria to assess whether there is a potential risk to human health
- when the criteria are exceeded, serve as conservative clean-up targets for many situations, ie, where further investigation or site-specific risk assessment is not warranted or economic
- inform onsite management actions to reduce the potential for adverse effects
- trigger further investigation to determine site-specific criteria.

The soil guidelines values represent 'clean down to levels' at contaminated sites and not 'pollute up to levels' for less contaminated sites, that is, the guideline values must not be considered as permission to contaminate up to a certain level. They are also not intended to be used to manage pristine sites (CCME, 2006).

Table A5.1: Summary of the name / terminology used, purpose, and basis for derivation of soil guideline values in different countries

Country		Terminology used in country / by agency	Purpose	Derivation basis
New Zealand	Timber treatment	Soil acceptance criteria	Investigation	Human health / phytotoxicity
	Gasworks	Soil acceptance criteria	Investigation	Human health
	Oil industry	Soil acceptance criteria	Investigation	Human health
	Sheep-dip sites	Soil guideline values	Investigation	Human health
Australia		Investigation levels	Investigation	Ecological Human health
United States	Federal Soil Screening Guidance	Soil screening level (SSL)	Investigation	Ecological Human health
	US EPA Region 6	Media-specific human health screening levels	Investigation	Human health
	US EPA Region 9	Preliminary remediation goals	Investigation	Human health
Canada		Soil quality guidelines	Remediation goal	Integrated
United Kingdom		Soil guideline values	Investigation	Human health
The Netherlands		Intervention value	Intervention	Integrated
		Remediation values	Remediation goal	
		Target value	Long-term remediation goal	Ecological

A largely philosophical decision also needs to be made as to whether soil numeric values should be protective of ecological or human receptors. In some countries, one or the other is considered; in other countries, both types of receptors are considered together to create 'integrated guidelines', eg, in Canada and the Netherlands. Some New Zealand guidelines, eg, some of the values in the 'Timber Treatment Guidelines' (MfE and MoH, 1997) have mixed consideration of human health and phytotoxicity (plant health), while other values are purely for human health. The underlying premise in existing New Zealand industry-based guidelines is that protection of on-site ecosystems is only required to the extent necessary to facilitate the use of the land.

This report only considers human health, and therefore the ecological component of the guidelines reviewed in the following section has not been considered.

A5.3 United States

A5.3.1 Legislative framework

The United States has a considerable body of legislation and guidance, both federal and state, that controls contaminated land assessment, management and remediation. The legislation is too voluminous to review in any detail here. As New Zealand has adapted parts of the US EPA's 'Superfund' guidance in developing its existing guidelines, the legislation controlling this programme will be reviewed.

The Federal Superfund programme was established under the Comprehensive Environmental Response Compensation and Liability Act (CERCLA) in 1980 and is administered by the US EPA. This law created a tax on the chemical and petroleum industries and provided broad federal authority to respond directly to releases or threatened releases of hazardous substances that may endanger public health or the environment. CERCLA and subsequent amendments established prohibitions and requirements concerning closed and abandoned hazardous waste sites. They provided for the liability of persons responsible for release of hazardous waste, and established a multi-billion dollar trust fund to provide for the clean-up when no responsible party could be identified. This is funded by taxes on the chemical and petroleum industries. The law authorises two kinds of response actions (US EPA, 2006b):

- short-term removals, where actions may be taken to address releases or threatened releases requiring prompt response
- long-term remedial response actions, that permanently and significantly reduce the dangers associated with releases or threats of releases of hazardous substances that are serious, but not immediately life threatening. These actions can be conducted only at sites listed on EPA's National Priorities List (NPL). There are about 1500 NPL sites.

CERCLA also enabled the revision of the National Contingency Plan (NCP). The NCP provided the guidelines and procedures needed to respond to releases and threatened releases of hazardous substances, pollutants or contaminants. The NCP also established the NPL.

The Hazard Ranking System (HRS) is the principal mechanism the US EPA uses to place uncontrolled waste sites on the NPL. It is a numerically based screening system that uses information from initial, limited investigations – the preliminary assessment and the site inspection – to assess the relative potential of sites to pose a threat to human health or the environment. Any person or organisation can petition the EPA to conduct a preliminary assessment.

The federal government plays a strong role in site-specific clean-up decisions and action. There are national guidelines for assessment of risk and decision making (see below), with the risk targets common to all clean ups.

Operating waste facilities fall within the Resource Conservation and Recovery Act (RCRA Corrective Action). The RCRA primarily provides that hazardous waste is properly managed so it does not contribute to future contamination, and under its corrective action programme, the clean up of existing contamination at operating industrial facilities is addressed. About 3800 sites were undergoing corrective action as of 2006, many more than on the Superfund NPL (US EPA, 2006b). Clean-up expectations are similar to that for Superfund. Thirty-eight states are authorised to run their own programmes. Federal oversight is limited.

A further federal programme is the clean up of underground storage tanks (UST), established under RCRA in 1984. US EPA's Federal UST regulations require that contaminated UST sites must be cleaned up to restore and protect groundwater resources and create a safe environment for those who live or work around these sites. The programme is intended to oversee clean-ups by responsible parties and to pay for clean ups at sites where the owner or operator is unknown, unwilling, or unable to respond, or which require emergency action. A trust fund, which receives US$70 million per year from a fuel tax, has been established to enable this. There are 125,000 contaminated underground storage tank facilities to address under the programme, with thousands of new petroleum releases each year. The federal programme provides only general guidance, with states defining their own programmes. Risk-based decision making is encouraged but there is no specific protocol or expectations for clean up.

A5.3.2 Superfund risk assessment process

New Zealand has based much of its past soil guideline derivation on US guidance, and in particular the multi-volume Risk Assessment Guidance for Superfund (RAGS). This has been developed by the US EPA as a requirement of both CERCLA and RCRA[17]. Key guidance is RAGS Volume I (and its various parts in separate documents) covering human health risk assessment and Volume II covering ecological risk assessment. More recently, RAGS Volume III, Part A (US EPA, 2001b), on probabilistic risk assessment has been developed. Only Volume I is considered here.

RAGS Volume I has multiple parts, including supplementary guidance, developed over several years (US EPA, 1989a, 1991a, 1991b, 1991c, 1996a, 1996b, 2001a, 2001c, 2002a, 2004b), the most important of which for the New Zealand context are:

- Risk Assessment Guidance for Superfund (RAGS): Volume I. Human Health Evaluation Manual (HHEM) (Part A, Baseline Risk Assessment) (US EPA, 1989a)
- RAGS Volume I, Part E: Supplemental Guidance for Dermal Risk Assessment (US EPA, 2004c)
- Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites (US EPA, 2002a).

The risk assessment guidance is supported by the multi-volume *Exposure Factors Handbook*, first published in 1989 (US EPA, 1989b) as a three-volume set covering general factors, food intake factors and activity factors. The latest version was published in 1997 (US EPA, 1997) as a single 1200-page volume and available online at http://www.epa.gov/ncea/efh/. The handbook is intended to serve as a support document to *Guidelines for Exposure Assessment* (US EPA, 1992), the original version of which was published in 1986, and developed to promote consistency among the various exposure assessment activities carried out by the US EPA.

Chapter 6 of RAGS Volume 1, Part A (US EPA, 1989a) provides the basic scientific basis of exposure assessment and presents the basic equations to be used. Chapter 7 of this document covers the toxicological basis of risk assessment. RAGS Volume 1, Part B (US EPA, 1991a) originally provided additional information and equations on determining preliminary remediation goals (PRGs), but has since been superseded by the supplementary guidance of US EPA (2002a) for non-residential exposure and for the dermal exposure pathway by US EPA (2004c).

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[17] Comprehensive Environmental Response, Compensation, and Liability Act -- otherwise known as CERCLA or Superfund; RCRA, Resource Conservation and Recovery Act

A PRG is an initially developed chemical concentration for an environmental medium that is expected to be protective of human health and ecosystems. Risk-based PRGs, either at scoping or later on, are initial guidelines. They do not establish that clean up to meet these goals is warranted, that is, they are not clean-up values (US EPA, 1989a, 1991a). The risk assessment process described in RAGS Volume 1, part A and B is, in effect, a site-specific risk assessment process and involves consideration of many exposure pathways.

The process described in the various RAGS documents is quite complex, time-consuming and expensive, and is only warranted for the typically large and complex sites that are named as Superfund sites. Soil screening levels (SSLs) have been developed to standardise and accelerate the evaluation and clean up of sites, initially in 1996 (US EPA, 1996a; US EPA, 1996b) with an updated procedure for non-residential exposure in 2002 (US EPA, 2002a).

It is the generic set of SSLs that are of interest for New Zealand, certainly with respect to the derivation methods. However, the generic values are potentially more conservative than would be derived in New Zealand. In the US context, SSLs are not national clean-up standards; instead, they are used as a first screening tool to identify areas, chemicals and pathways of concern at federally listed sites that need further investigation. They are but one step in a long and public statutory process of investigation, initial screening, site-specific risk assessment, establishment of clean-up criteria and selection of remediation.

In the US context, soil screening levels can be used as preliminary remediation goals (but not necessarily the final clean-up levels) provided conditions found during subsequent investigations at a specific site are the same as the conditions assumed in developing the SSLs. However, the conservative assumptions built into the generic SSLs, while appropriate for a screening analysis, may be overly conservative for setting PRGs and, ultimately, site clean-up levels (US EPA, 2002a). It is important to understand this context, as New Zealand practitioners frequently refer to SSLs when New Zealand guidelines are not available. The obvious conclusion from the US framework is that SSLs are rather more conservative than soil guideline values envisaged in the proposed framework for New Zealand (MfE, 2007).

Generic human-health SSLs based for residential land use were first developed in the Soil Screening Guidance (SSG) (US EPA, 1996a) with the technical background given in US EPA (1996b). The more recent supplemental guidance (US EPA, 2002a) widened the application of generic SSLs to commercial / industrial land use, for both indoor and outdoor workers, and to the construction scenario, with the latter having site-specific exposure durations but otherwise standard factors. A summary of pathways of concern for derivation of SSLs for the residential, commercial / industrial and construction settings is shown in table A5.2 (US EPA, 2002a). This list is not exhaustive but will depend on the particular site.

The 2002 supplement also provided new SSL equations for combined exposures via ingestion and dermal absorption, updated dispersion modelling data for the residential air exposure model, and new methods to develop SSLs for the migration of volatiles from subsurface sources into indoor air. These changes are important for New Zealand, because New Zealand guidance developed in the late 1990s was based on the earlier US EPA guidance. If the US EPA guidance is still to be followed, at least these updates should be considered.

Table A5.2: Summary of US EPA exposure pathways of concern for residential and commercial / industrial land uses, and construction for deriving SSLs

Scenario1	Residential	Non-residential (commercial / industrial)		Construction
Receptor	On-site resident	Outdoor worker	Indoor worker	Construction worker
Pathways of concern	Ingestion (surface and shallow subsurface soils) Dermal absorption (surface and shallow subsurface soils) Inhalation (fugitive dust, outdoor vapours) Inhalation (indoor vapours) Migration to groundwater	Ingestion (surface and shallow subsurface soils) Dermal absorption (surface and shallow subsurface soils) Inhalation (fugitive dust, outdoor vapours) Migration to groundwater	Inhalation (indoor vapours) Ingestion (indoor dust) Migration to groundwater	Ingestion (surface and subsurface soil) Dermal absorption (surface and subsurface soil) Inhalation (fugitive dust, outdoor vapours)

The US EPA guidance is too extensive to attempt to summarise all the equations here. The basic equations used to derive generic SSLs are as used in the 'Timber Treatment Guidelines' (MfE and MoH, 1997) with a number of subsidiary equations to derive input values into these equations. However, the following are of note:

- Residential exposure:
  - The SSLs apply to the top 2 cm of soil, being the soil that people are mostly exposed to and generates the dust that may be inhaled or migrate into homes.
  - The default exposed group are children aged 1–6 and older children and adults aged 7–30.
  - For threshold substance the critical receptor is a child.
  - For non-threshold substances age-adjusted exposure rates are calculated to account for exposure over the complete exposure duration, as an adult and child, with averaging over a default lifetime of 70 years.
  - Soil ingestion and dermal exposure are combined to arrive at the SSL.
  - Home-grown produce consumption is not considered, being left to site-specific consideration.
  - Migration of volatiles to indoor air is not considered, because of difficulties identifying suitable default values for such things as building dimensions and the distance between the building foundation and the contamination; however, volatiles to outdoor air and inhalation of fugitive dust outdoors are allowed for. These pathways are not combined with ingestion and dermal exposure.
- Two commercial / industrial scenarios are considered, for indoor and outdoor workers, to recognise the different exposure these two groups would have to soil.
  - Indoor workers have no direct exposure to soil, but may contact indoor dust, and have exposure to volatiles. This scenario covers people such as full-time factory and shop workers.
  - Outdoor workers are those involved full-time in everyday outdoor maintenance activities involving moderate digging and landscaping (eg, the site caretaker). Such a worker is expected to have an elevated soil ingestion rate, dermal exposure and inhalation of dust or vapours.
  - Migration of volatiles to indoor air is not considered but volatiles to outdoor air and inhalation of fugitive dust outdoors has been allowed for (based on the Johnson and Ettinger (1991) model and associated (since updated) guidance (US EPA, 2002c, 2004c).

- Generic construction scenario SSLs have not been calculated because of the difficulty of defining standardised default exposure assumptions.

The RAGS process and calculation of SSLs by US EPA is based on an estimate of the reasonable maximum exposure (RME) expected to occur under both current and future land-use conditions. The RME is defined as the highest exposure that is reasonably expected to occur at a site. The intent of the RME is to estimate a conservative exposure case, that is, well above the average case, but still within the range of possible exposures. RMEs are estimated for individual pathways. If a population is exposed via more than one pathway, the combination of exposures across pathways also must represent an RME.

All soil screening level equations in the 1996 soil screening guidance were designed to be consistent with the concept of RME in the residential setting. Accordingly, the US EPA used reasonably conservative defaults for intake and exposure duration, combined with values for site-specific parameters that reflect average or typical site conditions, to develop the risk-based SSLs. The SSLs were based on RME assumptions rather than central tendency conditions because this approach results in a conservative (though not a worst-case) estimate of long-term exposure that is protective of the majority of the population (US EPA, 2002a).

In deriving the generic SSLs the bioavailability of substances is assumed to be 100 per cent (except for lead, for which a different derivation process based on a soil–blood lead model is employed) and background exposure is not subtracted from the acceptable daily intake for the contaminant concerned.

The US EPA reviewed and confirmed its use of RMEs in response to criticism that the concept has so overemphasised conservatism that most risk estimates are false or meaningless (US EPA, 2004a). The US EPA's policy position is that it uses defaults that guard against underestimating risk while also being scientifically plausible given existing uncertainty. Further, the policy is to examine and report on the upper end of a range of risks or exposures when there is uncertainty about where the particular risk lies – in other words, a precautionary approach.

The US EPA rejects the argument that combining several values results in excessive overestimates of risk (eg, combining two 95th percentile defaults results in an estimate above the 99th percentile and combining three 95th percentile defaults results in an estimate above the 99.9th percentile) as being too simplistic. It has pointed out that just multiplying the numbers as implied will not necessarily lead to the answers above, but will depend on the variability of the data and the shape of the input distributions, with different parameters having little 'influence' (eg, for narrow distributions there is little difference between high-end and central estimates). If all the input variables show the same variability, shape, etc, then the multiplicative reasoning with respect to compounding values is true, but otherwise not (US EPA, 2004a). The US EPA further notes that it is rarely the case that distributions are the same in actual situations.

An example is when three variables are involved and one has a wide distribution while the other two have narrow distributions: taking the mean of the wide distribution, instead of the 95th percentile, while taking the 95th percentile of the other two variables, will result in an outcome that is less than the intended 95th percentile (or some similarly high number) protection because of the overriding influence of the wide distribution. Following this logic, it is necessary to consider the likely variability of each exposure variable used in the risk assessment. Thus it may be appropriate to use central estimates (means) for most variables, but the 95th percentile for a variable that has a wide distribution (eg, exposure duration).

The SSL derivation has been used by a number of US EPA regional offices (eg, regions 3, 6 and 9) to produce their own sets of SSLs (called PRGs in Region 9), but these have been recently harmonised into a single set of regional guidelines (US EPA, 2008). In addition, an on-line PRG calculator (the Risk Assessment Information System – RAIS) developed by the United States Department of Energy's Oak Ridge National Laboratory using the RAGS guidance and the SSL equations, is available at http://rais.ornl.gov/prg/prg_document.shtml. The scenarios considered are:

- construction worker / excavation land use – exposure to contaminants in soil
- industrial land use (indoor and outdoor worker) – exposure to contaminants in soil
- industrial land use (indoor and outdoor worker) – exposure to contaminants in groundwater
- recreational land use – exposure to contaminants in soil
- recreational land use – exposure to contaminants in water
- residential land use – exposure to contaminants in soil
- residential land use – exposure to contaminants in water
- agricultural land use – exposure to contaminants in soil, groundwater and homegrown produce.

The online tool goes further in developing generic values than assumed by US EPA (2002a), in particular agricultural and recreational scenarios are considered, and multiple direct (ingestion, dermal) and indirect (inhalation of dust and vapours) pathways are combined. The documentation with the online tool notes that the combination of pathways will result in lower (more conservative) PRGs than the SSLs calculated in the Soil Screening Guidance.

The agricultural and recreational scenarios use site-specific exposure factors developed for the assessment of the Oak Ridge Reservation, in Tennessee (USDOE, 1999). The documentation warns that the default agricultural and recreational assumptions may not be applicable to other sites. The usefulness of these PRGs as generic values must therefore be questioned.

Most, if not all, states in the US have developed generic guideline values based on the US EPA guidance.

A5.4 Australia

A5.4.1 Legislative framework

Federal management of contaminated sites in Australia is provided by the National Environmental Protection (Assessment of Site Contamination) Measure (NEPM) (NEPC, 1999b). Its purpose is to establish a nationally consistent approach to assessment of site contamination. The measure was developed by the National Environment Protection Council (NEPC), a body comprising environment ministers from the Australian Government and each state and territory. The Council was brought into being by the National Environment Protection Council Act 1994, enacted by the Commonwealth Government following the Intergovernmental Agreement on the Environment in 1992. The Act provides for the NEPC making national environment protection measures in a number of defined areas, including general guidelines for the assessment of site contamination.

Various activities are subject to Commonwealth law, including activities of the Commonwealth in a participating jurisdiction. The Australian Government has implemented law to make the Commonwealth subject to relevant NEPM implementation law in the states and territories.

Responsibility for contaminated land rests with states and territories unless the site is owned by the Commonwealth, and individual states and territories have implemented specific legislation and produced guidance for contaminated site management. Detailed reporting of the various state provisions is beyond the scope of this study, but some brief examples are given here.

In New South Wales (NSW) the Environmental Planning and Assessment Act 1979 provides for the management of contaminated land to ensure that it is not put to inappropriate use; and requires local authorities to consider land contamination when making rezoning or development decisions. Councils are required to provide information on land contamination when issuing planning certificates. Land remediation is facilitated and controlled through State Environmental Planning Policy 55 – Remediation of Land (DUAP and EPA, 1998).

The NSW Environmental Protection Authority has issued a number of guidelines under the Contaminated Land Management Act 1997 and has adopted the framework of the contaminated land NEPM within these guidelines.

Victoria gazetted the State Environmental Protection Policy (Prevention and management of contaminated land) in 2002 under the Environment Protection Act 1970 (Victoria Government, 2002). The policy sets out, amongst other things, land-use categories (the same as in the NEPM) and the adoption of the investigation levels and the procedures for deriving such levels, set out in the NEPM.

Victoria initiated the use of environmental auditors within a statutory process to oversee assessment and remediation of contaminated land. New South Wales and Western Australia have adopted similar schemes.

The NEPM has recently been reviewed, with many priority and lesser priority proposals for its modification coming out of that review (NEPC, 2006). The recommendations included:

- Revise the NEPM policy framework and Schedule A (the assessment process) to improve clarity and understanding of the fundamental site assessment principles and emphasise the appropriate use of the NEPM, in particular to address the misuse of investigations levels, eg, use of investigation levels for clean up.
- Revise the existing Health-based Investigation Levels (HILs) in the light of current knowledge, leading to more accurate and often less conservative numbers, and the derivation of additional HILs for priority substances.
- Follow-up review of worldwide models and field methods for the assessment of volatiles, and adopting as interim guidance a model(s), analytical approaches and field methods, from a 'best-fit' scenario most suited to Australian conditions.
- Review current bioavailability approaches, methods and limitations to improve the basis for their application in site assessment.

The widespread practice of invoking screening or investigation levels as a clean-up standard, contrary to the intent, has led in most jurisdictions to widespread reliance upon the 'dig and dump' strategy as the most common remediation method (Fowler, 2007). This has been reinforced by environmental authorities in a number of jurisdictions transferring responsibility to environmental auditors for determining remediation approaches. Auditors have tended to act cautiously by recommending removal to landfill in most cases rather than exploring alternative approaches, largely out of a concern to avoid any possible future personal liability.

A5.4.2 The NEPM derivation methodology

The methodology for deriving HILs is mainly described in NEPM schedules B(7a) and B(7b) (NEPC, 1999a, 1999c), the former providing the derivation methodology and the latter a description of standard exposure scenarios (see below). Three exposure pathways are considered: soil ingestion, inhalation of particulates, and dermal absorption.

Exposure via home produce consumption is discussed but not provided for in the scenarios for which HILs are calculated, with site-specific assessment recommended where home produce consumption is found to be greater than 10 per cent of fruit and vegetable consumption. Exposure via consumption of home-grown produce was considered to be too variable and uncertain (due to site-specific differences and plant type) to be included on a generic basis (NEPC, 1999b). Implicitly, home produce consumption of less than 10 per cent of total produce consumption is considered an insignificant contribution to a resident's contaminant intake. None of the HILs presented in the NEPM include exposure via consumption of home-grown produce in their derivation.

The derivation provided in NEPC (1999c) uses similar equations to those of other jurisdictions, being based on estimating the total exposure to a given substance:

$$\text{Exposure} = BE + (S_{ing} \times C_{ing} \times B_{ing}) + (S_{inh} \times C_{inh} \times B_{inh}) + (S_{skin} \times C_{skin} \times B_{skin})$$

where: BE = background exposure
S_{ing} = amount of soil ingested
C_{ing} = concentration of substance in ingested soil
B_{ing} = bioavailability of substance when ingested
S_{inh} = amount of soil/dust inhaled and retained
C_{inh} = concentration of substance in soil/dust inhaled and retained
B_{inh} = bioavailability of soil/dust inhaled and retained
S_{skin} = amount of soil on skin
C_{skin} = concentration of substance in soil on skin
B_{skin} = bioavailability of substance when on skin

The bioavailability of substances is assumed to be 100 per cent if specific information is not available (NEPC, 1999a) (a conservative default assumption for dermal absorption for most substances); but it is also noted that different levels of bioavailability will occur between soil ingested, inhaled or in contact with skin (NEPC, 1999d).

Total exposure includes that from background sources, principally food and water, and therefore less than the allowable intake of a contaminant is assigned to contaminated soil sources. The total exposure must not exceed the provisional tolerable weekly intake (PTWI), acceptable daily intake (ADI) or Guideline Dose (GD – for cancer toxic effects) of the contaminant concerned.

A specific equation is not provided in the Schedule B(7a) for the derivation of an HIL. But implicitly, the same contaminant concentration is taken to be soil that is ingested, inhaled and attached to the skin, with this concentration solved for when the total exposure is equated to the allowable intake (PTWI, ADI or GD), or some fraction of this value. However, no subsidiary equations are provided for calculation of the inhaled dust intake or dermal absorption.

Allowable intakes are from WHO/FAO sources; or in the case of guideline doses for carcinogenic compounds, as set by national health advisory bodies.

Health investigation levels have been produced for four exposure scenarios, while two other exposure scenarios are discussed but left for site-specific assessment. The four exposure scenarios for which values are given are (as listed in NEPC, 1999a):

A. standard residential (<10 per cent consumption of produce grown on-site; no poultry) – includes children's daycare centres, kindergartens, preschools and primary schools

D. residential with minimal opportunities for soil access – includes dwellings with fully and permanently paved yard spaces such as high-rise apartments and flats

E. parks, recreational open spaces and playing fields – includes secondary schools

F. commercial / industrial – includes premises such as shops and offices as well as factories and industrial sites.

The other two scenarios, for which values are not given because investigation levels need to be determined on a site-specific basis, are:

B. residential with substantial vegetable garden (contributing 10 per cent or more of vegetable and fruit intake) and/or poultry providing any egg or poultry intake

C. residential with substantial vegetable garden (contributing 10 per cent or more of vegetable and fruit intake); poultry excluded.

Values for each scenario are not calculated in detail; rather, those for scenarios D, E and F are simple factorings up from the residential scenario. These factors represent default assumptions of reduced exposure relative to the standard residential exposure, based on expert judgement and are irrespective of exposure pathway. The default exposure ratios (DER) are 0.25 for residential with minimal soil access, 0.5 for park and recreational open spaces, and 0.2 for commercial / industrial use, resulting in a factoring up of HILs of 4, 2 and 5 respectively. The use of DERs is a simplification which is out of step with most other jurisdictions, where specific consideration is typically given to exposure rates for each exposure scenario / exposure pathway combination.

While a methodology to derive HILs is nominally provided in the NEPM, the majority of the HILs have not followed this methodology, and have been inconsistently derived. Health investigation levels for the residential scenario are nominally determined using a two-and-half-year-old child as the critical receptor. However, the NEPM provides no compilation of parameters that should be used in deriving the HILs. Some parameters (soil ingestion rates, inhalation and dermal exposure parameters) are provided in NEPM Schedule 4 (NEPC, 1999d). Other values are provided in the derivation of HILs for specific contaminants described in the proceedings of national workshops on health risk assessment and management of contaminated land (El Saadi and Langley, 1991; Langley, 1993; Langley et al, 1996, 1998).

A summary of the parameters is provided in table A5.1, section A5.2 above.

As noted in the previous section, the NEPM methodology is undergoing review. The emphasis is on:

- revising the existing HILs in the light of current knowledge
- reviewing worldwide models and field methods for the assessment of volatiles with a view to adopting those that best fit Australian conditions
- reviewing current bioavailability approaches.

A5.5 Canada

A5.5.1 Legislative framework

Within Canada, individual provinces are responsible for their own policies and regulation of contaminated sites, independent of the other provinces and the federal government. Each province has specific regulation on land contamination, but some regulatory regimes are more developed than others. Quebec and British Columbia are more active than the other provinces. Each province has a set of land-use-based generic criteria but the values differ. Each province also allows risk-based management decisions, but requirements are not the same (Beaulieu, 2007).

Many provinces have public lists (internet-based) of known contaminated sites, but the lists are not compiled in the same way. Three provinces have put in place a network of private acknowledged experts to supervise some of the assessment and remediation work. There is a trend to tackle governmental environmental liabilities, with the federal government more active than the provinces.

The larger provinces have developed their own policies and guidance which may or may not draw on national guidance developed by the Canadian Council of Ministers of the Environment. CCME as an intergovernmental body has developed a number of documents (see next section) intended to be applied Canada-wide, but the CCME guidelines are sometimes adopted, sometimes transformed and sometimes ignored by the provinces (Beaulieu, 2007). The provinces are free to develop their own regulation, policies, criteria and priorities independently of the other provinces and the federal government.

There is no overarching federal legal framework for contaminated land in Canada (Fowler, 2007), but a variety of federal legislation exists governing soil contamination, with the Canadian Environmental Protection Act (CEPA), Fisheries Act, and Canadian Environmental Assessment Act (CEAA) all containing provisions that relate to contaminated sites management. In 2003, the federal government established the Federal Contaminated Sites Accelerated Action Plan (FCSAAP), a new contaminated sites initiative to help identify, assess and manage the risks at contaminated properties under the custodial care of Canadian federal government departments.

A5.5.2 Guideline derivation

The Canadian Council of Ministers of the Environment has developed separate soil quality guidelines for the protection of environmental and human health, as generic guidance for the federal and provincial governments (although as noted earlier, not all provinces have adopted them). Guidelines are developed based on four defined land-use scenarios: agricultural, residential / parkland, commercial and industrial. Other land-use scenarios may be defined by provincial jurisdictions or on a site-specific basis.

The guidelines have been produced in response to growing public concern over the potential ecological and human-health effects associated with exposure to contaminated sites in Canada. To promote consistency and provide guidance in assessing and remediating contaminated sites under the National Contaminated Sites Remediation Program, initiated by CCME in 1989, an interim set of numerical environmental quality guidelines was released in 1991 (CCME, 1991). These adopted existing criteria for soil and water used by various jurisdictions in Canada. However, many of the interim criteria for soil were based on professional judgement. To ensure that revised guidelines are scientifically defensible, a derivation protocol was developed in 1996, with an update released in 2006 (CCME, 1996, 2006).

The Canadian land-use definitions are (CCME, 2006):

- **Agricultural**: where the primary land use is growing crops or tending livestock. This also includes agricultural lands that provide habitat for resident and transitory wildlife and native flora.
- **Residential / parkland**: where the primary activity is residential or recreational activity; parkland is defined as a buffer between areas of residency, and also includes campground areas, but excludes wildlands such as national or provincial parks.
- **Commercial**: where the primary activity is commercial and not residential or manufacturing. This does not include zones where food is grown. The toddler was chosen as the critical receptor for children, as commercial facilities (such as a shopping mall) could have childcare facilities and children would have unrestricted access to the complex.
- **Industrial**: where the primary activity involves the production, manufacture, or construction of goods.

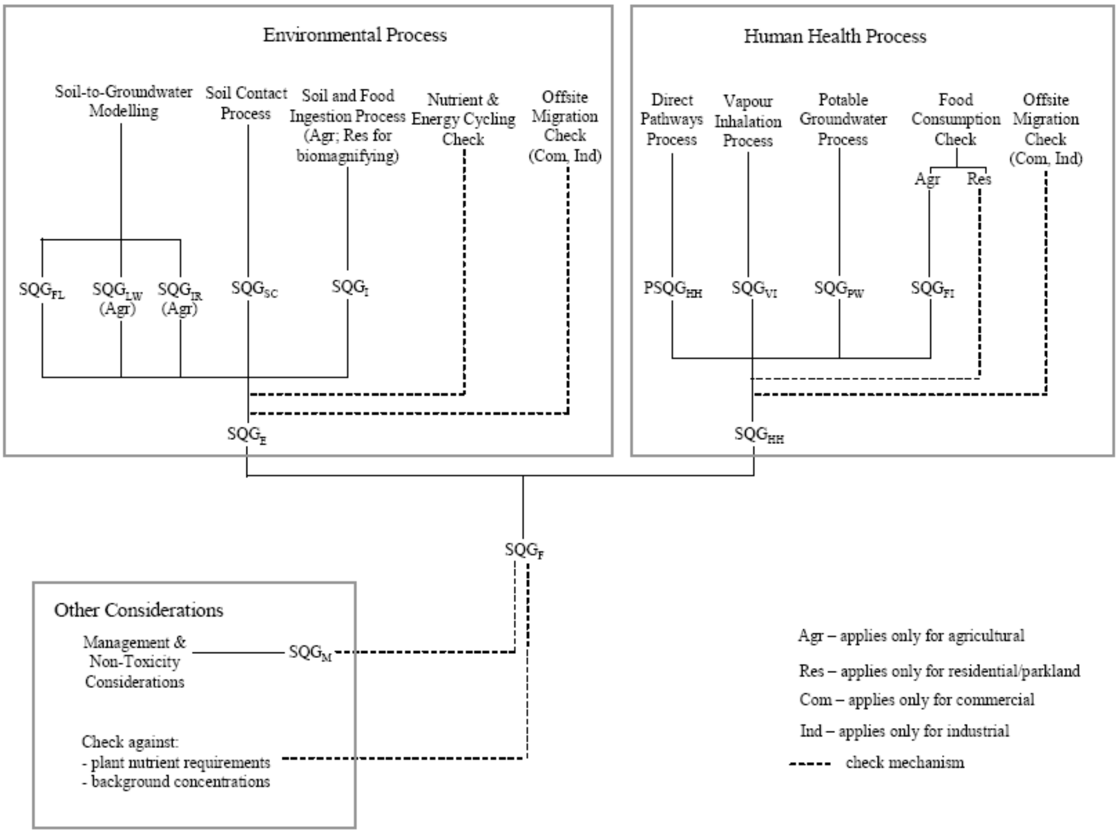
Soil quality guidelines for each chemical are developed for both ecological and human receptors. For each of the four land uses, to protect both human health and the environment, the most protective guideline is chosen as the recommended soil quality guideline. This is done after following through a complex set of checking a number of direct and indirect pathways for both ecological and human receptors. This is shown diagrammatically in figure A5.2.

Only the human health guideline derivation process will be considered here. The human health soil quality guideline (SQGHH) is determined by evaluating direct soil exposure (soil ingestion, dermal contact, and particulate inhalation), transport of contaminants through groundwater to potential potable water sources, intrusion of contaminant vapours into buildings, and human consumption of contaminated food. The specific exposure scenario is dependent on the land use, with some of the exposure pathways not evaluated for all land uses or contaminant types. The lowest of the soil concentrations deemed protective of each of these potential exposure pathways becomes the SQGHH. This contrasts with the approach taken in many other jurisdictions to combine exposure from the various pathways on the assumption the risk is additive.

The development of the soil quality guideline is a two step process (CCME, 2006). The first step considers all direct soil exposure pathways, including the ingestion of soil/dust, dermal contact, and inhalation of soil particles into lungs (as a combined value), as well as the primary indirect pathways. The primary indirect pathways that are included in the first step are dependent on the contaminant type, but may include inhalation of vapours migrating into indoor air (for volatile contaminants), ingestion of groundwater used as potable water (for soluble organic contaminants) and consumption of produce (for substances that biomagnify). The latter pathway applies principally to agricultural land use. The second step is to assess two ‘check mechanisms’: exposure from ingestion of food grown on contaminated soils (if not already applied in the initial step for substances which biomagnify); and the off-site migration via wind and water erosion of contaminants from commercial or industrial sites to more sensitive neighbouring properties.

The basic equations used for the derivation are similar to those used in other derivations and will not be repeated here. The equations in CCME (2006) are as presented in Health Canada (2004) and use the various exposure parameters presented in that document.

Figure A5.2: Overview of steps for derivation of a soil quality guideline in Canada



Source: CCME, 2006.

In common with many jurisdictions, for threshold contaminants the background intake (estimated daily intake, EDI) is subtracted from the tolerable daily intake (TDI) to obtain a residual TDI before calculating the soil guideline value. An unusual aspect is that the residual TDI is then, as a matter of policy, equally allocated five ways between soil, consumer products, air, water and food. This has the effect of double-counting exposure to food and water. The allocation of the residual is shown diagrammatically in figure A5.3.

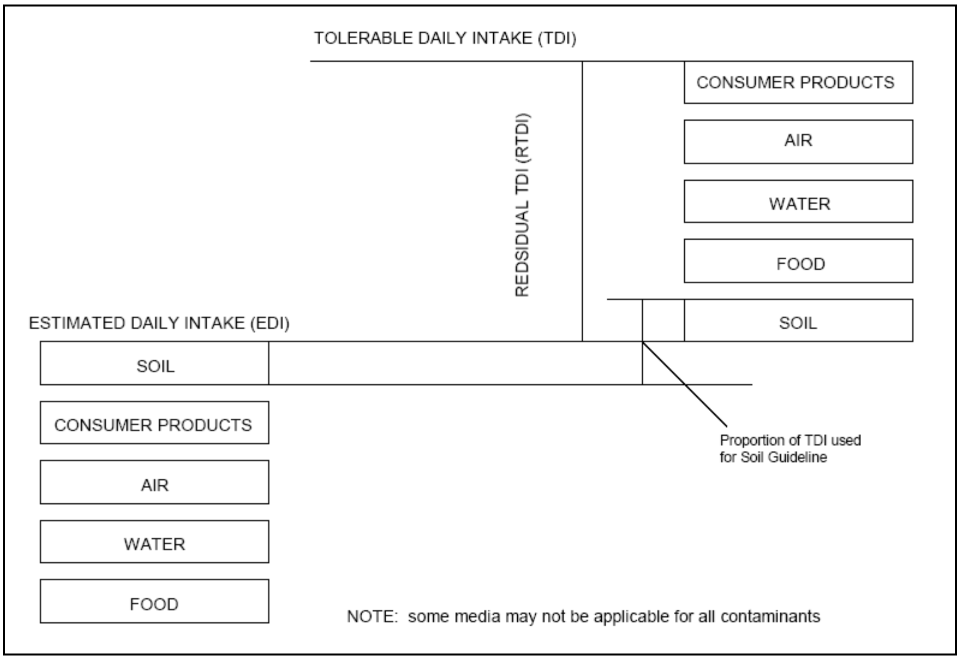
Having only a small part of the residual TDI allocated to soil potentially results in quite conservative soil guideline values for some contaminants. As a result, the 2006 revision of the 1996 protocol now permits adjustment of the soil allocation factor for chemicals that can defensibly shown to be not present (or not in significant concentrations) in all media. When the EDI is greater than the TDI (residual TDI = 0), theoretically the population cannot be safely subjected to any increased exposure. In these circumstances, the provisional soil quality guideline should be set at the background soil concentration or practical quantitation limit for that contaminant (CCME, 2006).

For threshold contaminants an excess cancer risk approach is taken, with guideline values calculated for both 1×10^{-5} and a 1×10^{-6} incremental risk, with the choice being left up to the individual jurisdictions as to what risk value to use.

The protocol recognises two generic soil types to minimise the uncertainty in guideline derivation introduced by soil variability: coarse-textured soils (soils containing predominantly sand and gravel sizes) and fine-textured soils (soils containing predominantly silt and clay sizes).

The protocol does not specify the depth to which the generic soil guidelines apply, although it notes that most direct human and ecological exposure pathways apply to soil located at or near the surface and suggests: *Surface soils are often defined as those within the uppermost 1.5 m of the soil profile*.

Figure A5.3: Assumed soil allocation factor from the residual tolerable daily intake



Source: CCME, 2006.

A5.6 The United Kingdom

Note: Since this section was written the situation in the United Kingdom has changed. The legislative framework remains the same but the derivation of SGVs has changed from the probabilistic model presented in Defra and EA (2002a) to a deterministic methodology as set out in EA (2008a), supported by a revised methodology for determining toxicological criteria in EA (2008b), both published in August 2008. The existing SGVs calculated in 2002 have been withdrawn. Several new SGV documents have been published (eg, EA, 2009a, 2009b, 2009c) with more in preparation.

The revisions follow a review of the methodology (Defra, 2006a) and the outcomes of that review (Defra, 2008a). The change to a deterministic model was foreshadowed in Defra (2006a). Other revisions include reconsideration of the generic land-use scenarios and default assumptions used in the CLEA model to derive SGVs including improvements in clarity, internal consistency, and practical usability of the approach. The basic exposure equations remain the same, albeit used in a deterministic fashion with single-point estimates for the various parameters. New guidance on the definition of contaminated land has also been published (Defra, 2008b).

As key decisions had already been made on the new New Zealand methodology by the Technical Advisory Group by the time the authors became aware of the changes to the UK approach and as the changes are not so significant as to require reconsideration of the New

Zealand approach (and arguably reinforce some of the choices made), the decision was made not to update this section.

A5.6.1 Legislative framework

In the United Kingdom, contaminated land is regulated under the Environmental Protection Act introduced in 1990. The act initially mainly focused on preventing new contaminated land being created. It wasn't until the Environment Act was introduced in 1995 that contaminated land was specifically addressed. Section 57 of the 1995 legislation inserted Part 2A into the Environmental Protection Act 1990, although it did not come into force in England until April 2000. Part 2A has also been implemented in Wales and Scotland, with minor differences. Part 2A largely replaced and modernised regulatory powers that existed under much older public health legislation under which polluted land could be abated as a statutory nuisance.

The legislation was intended to improve the focus and transparency of the controls, to enable all problems resulting from contamination to be handled as part of the same process, to increase the consistency of approach taken by different authorities; and to provide a more tailored regulatory mechanism – including liability rules that are better able to reflect the complexity and range of circumstances found on individual sites.

Part 2A states that:

> 'Contaminated land' is any land which appears to the local authority in whose area it is situated to be in such a condition, by reason of substances in, on or under the land, that –
>
> significant harm is being caused or there is a significant possibility of such harm being caused; or pollution of controlled waters is being, or is likely to be, caused.

The Environmental Protection Act is administered by the Department for Environment, Food and Rural Affairs (Defra) which has issued guidance on the Part 2A in the form of Circular 01/2006 'Contaminated land' (Defra, 2006b), which replaced an earlier 2000 circular. Regulations under the Act specify detailed provisions, including special sites, remediation notices, compensation, appeals and public registers. Section 78R requires enforcement authorities to keep a public register of all regulatory action taken by the enforcing authority in respect of the remediation of contaminated land, and will include information about the condition of land (Defra, 2006b). However, when Part 2 came into force, Section 143 was repealed (this would have required local authorities to compile registers of land which may be contaminated).

The policy objectives of Part 2A with respect to contaminated land are threefold:

a. to identify and remove unacceptable risks to human health and the environment
b. to seek to bring damaged land back into beneficial use
c. to seek to ensure that the cost burdens faced by individuals, companies and society as a whole are proportionate, manageable and economically sustainable.

These three objectives underlie the 'suitable for use' approach to the remediation of contaminated land, which focuses on the risks caused by the contamination. The approach recognises that the risks presented by any given level of contamination will vary greatly according to the use of the land and a wide range of other factors, such as the underlying geology. Risks therefore need to be assessed on a site-by-site basis. The 'suitable for use' approach consists of three elements:

a. Ensuring that land is suitable for its current use by identifying any land where contamination is causing unacceptable risks to human health and the environment, assessed on the basis of the current use and circumstances, and returning such land to a condition where such risks no longer arise, ie, remediating the land.

b. Ensuring that land is made suitable for any new use, as planning permission is given for that new use by assessing the potential risks from contamination, on the basis of the proposed future use, before permission is given for the development and, where necessary, remediating the land before the new use commences.

c. Limiting requirements for remediation to the work necessary to prevent unacceptable risks to human health or the environment in relation to the current use or future use of the land for which planning permission is being sought. In other words, recognising that the risks can be satisfactorily assessed only in the context of specific uses of the land (whether current or proposed), and that any attempt to guess what might be needed at some time in the future for other uses is likely to result either in premature or unnecessary work.

Local authorities have the primary regulatory role under both planning legislation and Part 2A. Land contamination is a material planning consideration within the planning regime. This means that a planning authority has to consider the potential implications of contamination both when it is developing ‘structure’ or ‘local’ plans and when it is considering individual applications for planning permission. Under Part 2A, a local authority must ensure that their area of responsibility is inspected to identify contaminated land, to determine whether any particular site is contaminated land, and, provided the site is not a ‘special site’, establish liability for the costs of inspection and remediation, to decide, after consultations, what remediation is required and ensure remediation takes place. The responsibility for establishing liability and enforcing remediation of designated as special sites lies with the Environment Agency (EA). There are four kinds of special sites; where ‘controlled’ water is being polluted, particular industrial sites which pose special problems or are subject to national legislation of some kind, defence sites, and land contaminated by radioactive material (Defra, 2006b).

In addition to acting as the enforcement authority for special sites, the Environment Agency is required to assist local authorities in identifying contaminated land, particularly in cases where water pollution is involved, report periodically on contaminated land and provide site-specific guidance to local authorities. In response to the need to assist and provide guidance, the Environment Agency and Defra have developed risk-based procedures for assessing harm from contaminated sites to humans and ecosystems. Examples are the research and development publications CLR 7–11 on human health risk assessment (Defra and EA, 2002a, 2002b, 2002e, 2002f, 2004). These set out, among other things, the requirements for risk assessment, a set of priority contaminants for development of SGVs, the framework for toxicity assessment and the model (CLEA) used for deriving generic SGV values and for carrying out site-specific assessments.

To date only 10 generic SGVs have been published and development of further SGVs has now been suspended pending the completion of a review of the underlying assumptions for the CLEA model (Defra, 2006a). A number of issues have arisen, including criticism that some values are too conservative and uncertainty as to how the generic SGVs fit under the Part 2A regime. In 2005, Defra issued a statement advising caution on applying the SGVs to determine whether land was contaminated under Part 2A (Defra, 2005). The statement included the following:

> ... it should be a matter for careful consideration by local authorities whether concentrations of substances in soil equal to, or not significantly greater than, an SGV would meet the

> legal test … it is apparent that there is a wide body of opinion that such concentrations would not necessarily satisfy that legal test.

The problem identified was that soil concentrations below SGVs provide are ‘acceptable’ but do not necessarily indicate that concentrations at or just above the SGV will be ‘unacceptable’ in the legal context. Exceedance of SGVs only indicates that further assessment or remedial action may be required (Defra, 2005). This appears to call into question whether the CLEA model can achieve one of its key objectives, ie, to determine whether land is ‘contaminated’ under Part 2A.

A5.6.2 Derivation of soil guideline values

The United Kingdom uses the CLEA model (Defra and EA, 2002a) to derive soil guideline values. The model is used to estimate average daily human exposure (ADE) to soil contamination based on the conceptual exposure models for three standard land uses. These are:

- **residential** – covers a wide variety of dwellings including detached, semi-detached and terraced properties up to two storeys high, and takes into account several different house designs including buildings based on suspended floors and ground-bearing slabs. Residents are assumed to have private gardens and/or access to community open space close to the home and exposure has been estimated with and without a contribution from eating home-grown vegetables
- **allotment** – allows for the use of communal open space, commonly provided by the local authority, for local people to grow fruit and vegetables for their own consumption
- **commercial / industrial** – assumes that work takes place in a permanent single-storey building, factory or warehouse where employees spend most time indoors involved in office-based or relatively light physical work. This land use is not designed to consider those sites involving 100 per cent hard cover (such as car parks), because of the implausibility of exposure from ingestion or skin contact that the scenario assumes.

The ADE is defined as the average daily amount of a contaminant to which a critical human receptor is exposed over the duration of exposure, calculated using the equation below and is reported as a function of body weight to enable direct comparison with relevant health criteria values (HCV). As noted in Defra and EA (2002a), the equation is based on a standard methodology that has been adopted internationally for such assessments. For example, for an exposure scenario with three exposure pathways – oral and inhalation intake and dermal contact – the equation is:

$$\mathrm{ADE} = \frac{\mathrm{IR_{inh}} \times \mathrm{EF_{inh}} \times \mathrm{ET_{inh}}}{\mathrm{AT} \times \mathrm{BW}} + \frac{\mathrm{IR_{oral}} \times \mathrm{EF_{oral}} \times \mathrm{ET_{oral}}}{\mathrm{AT} \times \mathrm{BW}} + \frac{\mathrm{IR_{dermal}} \times \mathrm{EF_{dermal}} \times \mathrm{ET_{dermal}}}{\mathrm{AT} \times \mathrm{BW}}$$

where: the subscripts inh, oral and dermal refer to the inhalation, ingestion and dermal contact routes respectively, and

ADE = average daily human exposure to a chemical from soil (mg/kg BW/day)
IR = chemical exposure rate (mg/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = human body weight (kg)
AT = averaging time (day)

The chemical exposure rate is a function of the concentration of contaminant in the relevant medium and the daily human exposure rate to that medium, which in turn is a function of land use and the receptor(s) exposed.

In the usual way, ADE is equated with the relevant HCV which is either an index dose value for a non-threshold substance (see MfE, 2010b for a further explanation of an index dose) or, for a threshold substance, a tolerable daily soil intake (TDSI), which is the TDI less the background intake from sources other than soil. An index dose is set specifically for exposure to soil and does not have the background intake deducted.

The conventional approach to risk assessment has been the use of deterministic models, in which single values are inserted into the simple equations defining exposure. However, the UK uses a probabilistic model, in which some of the single-value parameters in the exposure assessment are replaced with a family of values selected from defined probability distributions (Defra and EA, 2002a). This is intended to avoid the problem in a deterministic model of having to deal with parameter uncertainty and variability by selecting values representative of a worst-case exposure scenario, a practice which Defra and EA (2002a) notes as a common practice. While this has the assumed comfort of being more protective against an unforeseen situation or risks to sensitive individuals, the problem with this approach can be that such choices, however defensible they might be individually, tend to be implausible collectively (Defra and EA, 2002a). It has the further disadvantage that the model is complex and unable to be understood by any but the most expert individual.

Eight parameters are treated probabilistically in the CLEA model, on the basis that:

- the exposure estimate is sensitive to a change in its value
- variability in the parameter is well characterised
- there is no correlation between any two independently modelled probabilistic parameters.

The eight parameters treated probabilistically are:

- body weight
- total body surface area as a function of body weight
- respiration rate as a function of body weight
- mean daily soil ingestion rate by children aged 1–6 years
- estimated ratio of the concentration of a contaminant in chosen vegetables to the contaminant concentration found in the soil
- daily vegetable consumption rate
- fraction of homegrown garden vegetables as part of daily vegetable consumption rate
- fraction of exposed skin area in contact with soil.

The output of the CLEA model, for a particular contaminant and land-use scenario is a probability distribution of average daily exposure, calculated through many iterations of inserting values from the distributions of each of the exposure parameters into the exposure equation. A value then has to be chosen to equate with the HCV to derive the SGV. The point chosen by Defra and the Environment Agency is the 95th percentile of that distribution, so as to arrive at a 'reasonable worst case'. Note that this is not the same as being protective of the 95th percentile of the population, as some practitioners in the UK have assumed (Defra, 2006a).

The CLEA model can incorporate up to 10 different environmental pathways based on the concentration of the chemical in soil, with the choice dependent on the particular land-use category and specific considerations on the fate and transport, and toxicological properties of the contaminant of concern (eg, a vapour pathway would be chosen in the case of a volatile organic compound but not in the case of a heavy metal).

The 10 exposure pathways are:

- ingestion of soil
- ingestion of household dust
- ingestion of contaminated vegetables
- ingestion of soil attached to vegetables
- dermal contact with soil
- dermal contact with household dust
- inhalation of fugitive soil dust
- inhalation of fugitive household dust
- inhalation of vapours outside
- inhalation of vapours inside.

The model also uses 18 age intervals (or age classes) to break down the exposure characteristics of a human lifetime, allowing the model flexibility to consider exposure periods of a year or more. The first 16 intervals correspond to the first 16 years of life, the 17th interval is typical of an adult working life (age 16–59), and the 18th represents retirement (age 60–70). The intervals have been chosen to represent those stages in life where the most significant differences in site use are likely to occur. In deriving SGVs for the standard residential and allotment land-uses, a young female child from birth to six years is assumed to be the critical receptor. For the standard commercial and industrial land-use, a working adult is assumed to be the critical receptor. To date, 10 soil guideline values have been published.

Because of the probabilistic approach, it is difficult to compare the exposure parameter values with those adopted in deterministic models. However, of note is that exposure frequencies are not a constant across all exposure pathways, as is common with some deterministic models. For example, dermal exposure to indoor dust and outdoor soil are treated separately, with indoor dust exposure for 365 days a year, but outdoor soil exposure is treated as a less frequent occurrence. In addition, indoor dust has a lower concentration of chemical than outdoor soil, to reflect the fact that the indoor dust is only partly made up of outdoor soil tracked inside. This sort of refinement is not included in, for example, Australian and New Zealand guideline derivations.

A number of updates to the model have been issued as Contaminated Land Briefing Notes (EA, 2004a, 2004b, 2005a, 2005b) in respect of the dermal exposure route, vapour intrusion into buildings and combining exposure pathways.

For the dermal exposure route, the Environment Agency took into account work by the US EPA (2001a). Skin areas exposed to contaminants and soil adherence factors were revised (including differentiating between indoor and outdoor activities). In addition, dermal exposure was changed from being explicitly time-based (the default had been 12 hours per day) to an exposure per event, using a contaminant-specific absorbed fraction per event. These changes had no effect on the SGVs published to that point (arsenic and cadmium) because dermal intake was considered to be negligible (EA, 2005a).

The changes to the vapour intrusion algorithm within the CLEA model were triggered by a report commissioned by the Environment Agency in 2002. The original model considers vapour intrusion for both concrete slab-on-grade and concrete and wooden suspended floor construction. This report (Evans et al, 2002) examined 10 models for modelling vapour intrusion for the slab-on-grade situation and selected four for detailed assessment against case study results. The report concluded that the existing vapour intrusion models should be replaced by the Johnson and Ettinger (1991) model. This model assumes partitioning of vapours from the soil to the soil gas, migration of these vapours up through the soil to the underside of the slab, and then migration into the building through dust-filled cracks around the perimeter of the building. As a result of the recommendation, the CLEA model was changed and at the same time a review of typical British construction details resulted in modifications to the default parameters in the model (EA, 2004a, 2004b). However, the adoption of the Johnson and Ettinger approach meant that the previous suspended floor options were dispensed with in the CLEA model. The details of the adopted algorithms are beyond the scope of this report.

The CLEA model has been the subject of criticism within the contaminated-land assessment community in Britain, and the model and its underpinning policy is currently undergoing review. A discussion document (Defra, 2006a) was released in late 2006 which noted that concerns had been expressed about the limited number of SGVs; and that the SGVs that existed were not proportionate or realistic. There was an overall perception that the values were too stringent. The discussion document proposed making some immediate changes to the derivation method and proposed further study on possible changes. Immediate recommended changes included:

- replacing the child's soil ingestion parameter from a probability distribution with means of 100 mg/day and 95th percentile of 300 mg/day to a single-point estimate of 100 mg/day
- changing home-grown produce consumption from being based on a self-sufficient subgroup of the population to being based on a reasonable worst case for the whole population
- reviewing plant uptake models, with the objective of taking the conclusions of that review into the CLEA model
- improving guidance for estimating vapour intrusion into buildings – the perception was that the models overestimated vapour intrusion and were not consistent with British building practices
- changing the CLEA model to being fully deterministic.

With respect to plant uptake and vapour intrusion, UK practitioners had expressed concern during the review that these pathways are very difficult to predict on a generic basis as a result of both scientific uncertainty and site variability. This was noted as consistent with the opinion of the US EPA, although most other countries still include estimation of these routes within their guidance because there are few practical alternatives (Defra, 2006a). Excluding these pathways was considered potentially problematic as they often represent significant mechanisms for human exposure for some chemicals.

For plant uptake, CLEA considers both uptake into roots and above-ground-parts of vegetables, and divides produce intake into different types of above-ground and below-ground vegetables with uptake factors for each; a single consumption-weighted factor is derived for each group of above-ground and below-ground vegetables. Factors are derived for each contaminant on a case-by-case basis. The following hierarchy is used in selecting uptake factors (Defra, 2006a):

- empirical studies of plant uptake for the specific vegetables and chemicals of concern (that is, measured uptake factors under typical growing conditions)
- generic values for plant uptake recommended by authoritative bodies from the UK and other countries (often includes measured uptake factors from a range of plants and under varying soil and climate conditions)
- generic screening models based on good scientific principles.

For plant uptake of organics, the current CLEA model uses the relationship of Ryan et al (1988) which built on earlier work of Briggs et al (1982, 1983) to predict root and shoot concentrations of organic compounds (Defra and EA, 2002a; EA, 2006). The relationship assumes that plant uptake is proportional to the partitioning of an organic chemical between the soil and soil solution (pore water), based on experimental work on barley in growth solutions using a small range of organic compounds. The partitioning is a function of the organic matter present and the relative hydrophobicity of the compound, as measured by the octanol-water partition coefficient, K_{ow}. The greater the fraction of soil organic carbon (f_{oc}) and the higher the hydrophobicity, the lower the plant uptake. The Ryan et al (1988) approach applies to non to weakly polar and relatively hydrophilic compounds rather than hydrophobic organic contaminants (log K_{ow} 0 to 4). Care should be taken for log K_{ow} outside the range for which the relationship was developed and it should not be used for compounds that ionise in the soil (Defra and EA, 2002d; EA, 2006)

Defra (2006a) notes that the Environment Agency recognised the limitations of the Briggs and Ryan relationships and was reviewing available models for uptake of organics. This review (EA, 2006) was released at about the same time as the Defra review. The review concluded that none of the plant uptake models was adequate as a general screening tool, with all resulting in over-estimates of root uptake, in some cases by up to five orders of magnitude. The situation with uptake into above-ground parts was more confused, with individual models both under- and over-predicting. The simplest models (eg, Travis and Arms, 1988) were just as effective as the more complex models. The report concluded with recommendations for further study.

A5.7 The Netherlands

A5.7.1 Legislative and policy framework

The main piece of legislation pertaining to soil contamination in the Netherlands is the Soil Protection Act 1987. This Act aims to prevent new pollution and requires that soil contaminated after 1987 is remediated as much as can be reasonably achieved so the land can be returned to 'multifunctional use', ie, for all land uses. For soil contaminated before 1987, the seriousness of contamination is determined and the necessary action identified. Two generic risk-based soil screening values are used to determine the seriousness of contamination: intervention values (IVs) and target values (TVs). Both screening values are based on potential risks under standardised conditions. Target values are based on the potential risks to ecosystems, while intervention values are based on the potential risk to humans and ecosystems.

Intervention values are used to determine cases of serious soil contamination, while target values are viewed as sustainable-soil-quality objectives. Additionally, an intermediate value, which is arbitrarily set as the average of the target value and intervention value, is used to assist the process of site investigation (Swartjes, 1999; Carlon, 2007). These values are applied independent of soil use, whether residential, industrial or other use, and have statutory standing through the Soil Protection Act.

From a site investigation, which is conducted following standardised procedures, four outcomes are determined:

1. If the average soil concentration is below target values, no action is required.
2. If average soil concentrations exceed target values but fall below intermediate values, no further investigation is required but minor restrictions on land use are put in place.

3. If average soil concentrations exceed the intermediate values but are below intervention values, further investigation is required to confirm these concentrations, after which restrictions on land use are put in place.
4. For soils in which the average soil concentrations in 25 m^3 of soil exceed the intervention value, remediation will be necessary in principle, but the urgency of remediation has to be determined.

Remediation urgency is determined using a standardised computer-based methodology (CSOIL), to distinguish between urgent and non-urgent cases of serious soil contamination. Non-urgent cases are taken up in the provincial soil remediation programme without a defined time for starting the remediation. The determination of remediation urgency is based on actual (ie, site-specific) risks to human health, the ecosystem and risk due to contaminant migration.

Before 1997, all soils in the Netherlands were required to be remediated to multifunctional use (essentially to the target values) unless there were site-specific reasons not to do so. However, this 'strict' remediation goal result in a standstill in site remediation operations due to the perceived expense and necessity in relation to benefit and subsequent land use; it also prevented the development of urban land, revitalisation of business sites and sale of companies (VROM, 1999). Because of this, the Dutch Government reviewed the soil remediation policy to establish how the impediments to soil remediation could be removed.

In addition to the screening values, reference values have been derived for acceptable soil quality after remediation (Swartjes and Walthaus, in Carlon, 2007). Reference values represent sustainable soil quality of the upper soil layer, being the top layer of soil from 0.5 to 1 metre depth depending on land use. A distinction is made between land-use specific national reference values and local reference values, which can be derived on a site-specific basis. National Reference Values have been derived for several specific land uses, for immobile contaminants only. Mobile contaminants should be removed, as far as this is economically possible (Carlon, 2007).

A5.7.2 Calculation of guideline values

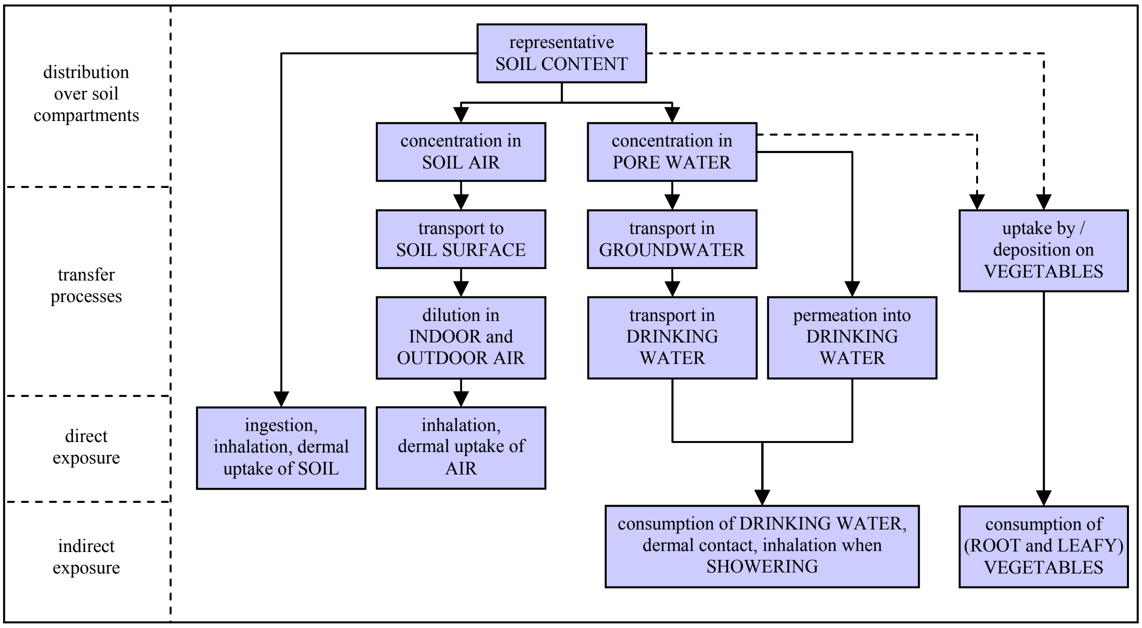
The target values for soil are determined for negligible risk to ecosystems, which is assumed to be 1 per cent of the maximal permissible risk level for ecosystems (MPR_{eco}). MPR_{eco} is defined as the hazardous concentration for 5 per cent of the species in the ecosystem (HC5), ie, 95 per cent protection. The HC5 is derived on an empirical basis by statistical interpretation of observed NOECs (no observed effect concentrations) and LOECs (lowest observed effect concentrations). For metals the added risk approach was followed for the derivation of target values by adding the natural background concentration in soils to the risk-based concentration as calculated above.

The human exposure model CSOIL is used to derive intervention values for soil and groundwater (Swartjes, 1999). The exposure is calculated using a standardised scenario based on all possible exposure pathways for a residential situation. The exposure routes are shown in figure A5.4. CSOIL is also used for derivation of remediation objectives, determination of the urgency of remediation and calculation of site-specific exposure.

Current IVs and indicative values are available in VROM (2000) although more recent review has resulted in proposals for new IVs (Bars et al, 2001; Lijzen et al, 2001) for what are known as first-series compounds (a group of most important compounds). However, these values have yet to be adopted into law (the Soil Protection Act). There are also second, third and fourth series of compounds currently being reviewed. These were originally developed by Crommentuijn et al (1994), Kreule et al (1995) and Kreule and Swartjes (1998), and still apply as legal intervention values for these contaminants.

The current version of CSOIL is described in detail in Brand et al (2007).

Figure A5.4: Schematic lay-out of the CSOIL exposure model



Source: Brand et al, 2007.

Calculations by Otte et al (2001) determined that three of the exposure routes are responsible for at least 90 per cent of the total exposure for almost all compounds. These routes are the human exposure:

- via the ingestion of contaminated soil particles
- to volatile compounds in the indoor air
- via the consumption of contaminated crops.

The other most significant pathways, depending on contaminant are:

- dermal uptake via soil contact (1–7 per cent for 18 compounds)
- drinking-water intake due to permeation through plastic pipes (1–13 per cent for 29 compounds)
- dermal uptake during bathing (1–5 per cent for 20 compounds).

Although not every exposure route has a significant contribution to the total human exposure, the basic principle of the model is that all possible exposure routes are taken into account.

The model concept consists of roughly three parts:

1. the description of the behaviour of the compound in the soil and the partitioning over the soil phases

2. the transfer processes and parameterisation of the different exposure routes (direct and indirect)
3. the quantification of the lifetime average exposure (Otte et al, 2001).

Details of the model are too complex to describe here, however, in common with all such models, there are a number of input parameters such as:

- compound-specific input parameters; mainly physicochemical properties
- site and soil properties, related to potential exposure
- exposure parameters which describe the receptor characteristics and behaviour such as breathing volume or ingestion frequency (Otte et al, 2001).

A5.8 New Zealand

A5.8.1 Policy and legislative framework

The New Zealand policy and legislative framework is outlined in MfE (2006b) and will not be repeated here. The document is available on MfE's website at: http://www.mfe.govt.nz/issues/hazardous/contaminated/direction-land-management.html.

A5.8.2 Industry-based guidelines

Four industry-based guideline documents have been developed by the Ministry for the Environment to provide good practice guidance for risk assessment of four types of site considered as a priority because of their prevalence in New Zealand. The guidelines have not been developed with any statutory basis, but as a matter of administrative expediency: to assist local government and contaminated-site practitioners to assess the risk posed by sites. Adoption of the guidelines is voluntary, unless referred to in operative district or regional plans, in which case they have regulatory force. The guidelines are ostensibly industry-based. Nevertheless they can be, and routinely are, used to assess the particular contaminants within each guideline on other sorts of sites that may happen to have the contaminants. The four guidelines are:

- *Health and environmental guidelines for selected timber treatment chemicals* (MfE and MoH, 1997), (the 'Timber Treatment Guidelines'). These present soil and water criteria for the timber preservation chemicals copper, chromium, arsenic, boron and pentachlorophenol. An interim guideline for dioxins was also adopted (as represented by 2,3,7,8- tetrachlorodibenzo-p-dioxin) but no details were given to support its derivation.
- *Guidelines for assessing and managing contaminated gasworks sites in New Zealand* (MfE, 1997) (the *Gasworks Guidelines*), which present human-health soil acceptance criteria for selected polycyclic aromatic hydrocarbons (PAHs), the mono-aromatic hydrocarbons benzene, ethyl benzene, toluene and xylene (collectively, BTEX), the phenol compounds phenol and cresol, and inorganic cyanide compounds in both free and complexed forms.
- *Guidelines for assessing and managing petroleum hydrocarbon contaminated sites in New Zealand* (MfE, 1999), (the 'Oil Industry Guidelines'). This presents human health-guideline values for soil and groundwater and, indirectly for indoor and outdoor air, for petroleum hydrocarbons (represented as three carbon ranges of mixed aliphatic hydrocarbons), the BTEX group of mono-aromatics and the PAHs naphthalene, benzo(a)pyrene and pyrene. Guidelines are presented for several different generic soil texture types, to account for the different vapour migration behaviour with different soil texture (clay, silt, sand, etc).

- *Identifying, investigating and managing risks associated with former sheep-dip sites: A guide for local authorities* (MfE, 2006a) (the 'Sheep-dip Guide'), which presents human-health guideline values for arsenic (as derived in the TTG, above) and the chlorinated pesticides DDT (as the sum of DDE, DDD and DDT – ∑DDT), dieldrin (including aldrin) and lindane.

All these guidelines draw on principles set down by the US EPA (US EPA 1989a; 1991a; 1996a; 1996b) in developing values primarily for the protection of human health. The 'Timber Treatment Guidelines' methodology, perhaps because it was the first and generally more detailed document in terms of derivation methodology of the four guidelines, is generally considered by practitioners to be current New Zealand policy with respect to derivation. The other New Zealand guidelines have similar derivations but with some significant differences that would result in different values for the same substances (and do result in different values for a few organic compounds that are in common). The detail of the 'Timber Treatment Guidelines' is presented below, with differences of the other guidelines presented in separate sections, where relevant.

Background exposure is generally not explicitly included in the methodology provided in the existing New Zealand guidelines, and typically 100 per cent of the TDI is assigned to exposure from soil sources, following US EPA practice. An exception is for copper (*Timber Treatment Guidelines*), where only 10 per cent of the TDI is assigned to soil sources "due to the relatively high intake of copper from other sources" (MfE and MoH, 1997). This appears to be overly conservative in relation to exposure from soil sources.

The underlying premise in existing New Zealand industry-based guidelines is that protection of on-site ecosystems is only required to the extent necessary to facilitate the use of the land, eg, plant growth and livestock grazing. The 'Timber Treatment Guidelines' provide numeric values for the protection of plant life and livestock health. Some of the values that have been adopted (eg, for copper) are specifically based on protection of plant life as opposed to human health. The 'Oil Industry' and 'Gasworks Guidelines' consider that the nature of the contaminants (ie, volatile, readily degradable) is such that soil guideline values based on human health will also protect plant and livestock health.

The exposure scenarios used for the derivation within existing New Zealand guidelines is summarised in table A5.3 and the detailed description are given in following sections.

Table A5.3: Exposure scenarios in existing New Zealand industry-based guidelines

Guideline	Scenario
Timber Treatment Guidelines	Agricultural / horticultural Residential Industrial – paved, unpaved Subsurface maintenance workers
Gasworks Guidelines	Agricultural / horticultural Standard residential High-density residential Commercial / industrial Parkland / recreational Maintenance workers
'Oil Industry Guidelines'	Agricultural / horticultural Residential Commercial / industrial Maintenance
Sheep-dip Guide	Lifestyle block Standard residential High-density urban residential Parks / recreation Commercial / industrial

For a given exposure scenario, the estimated exposure – and therefore the derived soil numeric value – is dependent on the exposure pathways applicable for that scenario, and the parameters (eg, soil ingestion rate, exposure frequency) selected to define the extent of exposure via those pathways.

Timber treatment guidelines

The 'Timber Treatment Guidelines' (MfE and MoH, 1997) provide for four exposure scenarios with the residential and commercial / industrial pathway each having two subscenarios. These allow for different proportions of home-grown produce consumption and paved and unpaved commercial / industrial sites. The scenario definitions are:

- **Agricultural** – includes all agricultural and horticultural uses, particularly those involved in the production of food for human consumption. The general public is protected by ensuring that soil contamination would not give rise to concentrations of contaminants in produce that would pose a concern to public health. The health of residents at any farm property is also considered, assuming that residents may be exposed via the consumption of home-grown livestock and produce, and through direct contact with the contaminated soil, eg, ingestion of contaminated soil.
- **Residential** – includes low-density residential use and rural residential use, where a considerable proportion of the total amount of produce consumed may be grown at the site. If livestock for human consumption are kept at a site then it should be assessed against the agricultural criteria, in the first instance. The small size of many residential developments within urban areas limits the amount of produce that may be grown, reducing the potential exposure for some contaminants. Recommended acceptance criteria have been derived for two rates of home produce consumption, reflecting the differences between urban residential use and rural residential use.

- **Commercial / industrial –** reflects exposure conditions at a largely unpaved industrial site where workers may come into direct contact with contaminated soil. This scenario is not designed to include consideration of workers actively involved in excavation or similar activities, for which separate criteria are derived. Where a site is largely paved, higher contaminant concentrations may be acceptable, as outlined in the guidelines.
- **Maintenance –** for each of the above site uses, human exposure to ground contamination may be associated with subsurface maintenance works, eg, repair and replacement of services. While the duration of such works is generally much shorter than the other exposure scenarios considered, the rate of exposure is likely to be much higher and this may be significant where the work is undertaken routinely by the same person.

The 'Timber Treatment Guidelines' provide a set of equations to estimate the chronic daily intake of individual contaminants for four exposure pathways: soil ingestion, produce consumption, inhalation of particulates, and dermal absorption.[18] These equations are based on the same generic equations described previously and follow US EPA practice.

$$\text{Intake} = \frac{\text{concentration} \times \text{contact rate} \times \text{exposure frequency} \times \text{exposure duration}}{\text{averaging time} \times \text{body weight}}$$

The pathway-specific equations for threshold contaminants are, using the nomenclature of the document:

Soil ingestion: $$\mathrm{CDI} = \frac{C_S \times IR \times CF \times ED \times EF \times MF}{AT \times BW}$$

Produce ingestion: $$\mathrm{CDI} = \frac{C_P \times IP \times ED \times EF \times Pg}{AT \times BW}$$

Inhalation of dust: $$\mathrm{CDI} = \frac{C_S \times IH \times ED \times EF \times MF \times R}{AT \times BW \times PEF}$$

Dermal absorption: $$\mathrm{CDI} = \frac{C_S \times CF \times AR \times AH \times AF \times ED \times EF}{AT \times BW}$$

Where:

CDI	=	chronic daily intake in terms of body weight (mg/kg BW/day)
C_S	=	contaminant concentration in soil (mg/kg)
IR	=	soil ingestion rate (mg/day)
CF	=	conversion factor (10^{-6} kg/mg)
ED	=	exposure duration (years)
EF	=	exposure frequency (days/year)
MF	=	matrix factor, typically set to 1
AT	=	averaging time ED × 365 days for a threshold substance, or 70 × 365 days for non-threshold
BW	=	body weight (kg)
C_P	=	concentration in produce (mg/kg)
IP	=	produce ingestion rate expressed as dry weight (kg DW/day)
Pg	=	proportion of home-grown produce (dimensionless)
IH	=	inhalation rate (m^3/day)
R	=	proportion retained in lungs (dimensionless)
PEF	=	particle emission factor (m^3/kg)

[18] The equations in the 'Timber Treatment Guidelines' are in error, as presented in Chapter 5 of the document, but the correct equations are used in the derivation calculations. This error is separate from other errors in the produce pathway calculations. Details are given in Cavanagh and Proffitt (2005).

AR = exposed skin surface area (cm^2)
AH = soil adherence factor (mg/cm^2)
AF = absorption factor (dimensionless)

For threshold substances, there is a single critical receptor and the exposure duration in days is the averaging time, ie, averaging time = ED × 365. ED in the numerator and denominator of each equation cancel and the contact rate is just that for the critical receptor, with, for example, the equation for the soil ingestion pathway becoming:

$$CDI = \frac{C_S \times IR \times CF \times EF \times MF}{365 \times BW}$$

The equations for the other three exposure pathways are similar.

For non-threshold substances there is no critical receptor, as exposure at any time could cause an adverse effect. Using one or more receptor groups to better represent varying intake rates, body weights and other age-dependent parameters, the total chronic intake is obtained by summing over a total period of exposure for the different receptor groups.

For each exposure pathway the total intake becomes:

$$CDI = CDI_1 + CDI_2 + CDI_3 + \ldots CDI_n = \sum CDI_i$$

where i represents each receptor age group.

It is possible to have all the various parameters being age-dependent but in the US EPA protocol that New Zealand adopted at the time, exposure frequencies for each age range are taken to be common across all age ranges. Similarly, dermal soil adherence has been taken as common to all age groups (no longer the case in US EPA practice). Thus, by substituting the pathway exposure equations into Equation 3.9 and collecting common terms, only the exposure rate, exposure duration and body weight remain in the summation, as an 'age-adjusted' rate, with, using soil ingestion as an example, the total chronic intake rate becoming:

$$CDI = \frac{C_S \times IR_{adj} \times CF \times EF \times MF}{AT}$$

With IR_{adj} being represented by:

$$IR_{adj} = \sum \frac{I_i \times ED_i}{BW_i}$$

where $\sum$ signifies summation over receptor groups i = 1 to n
Ir_i = soil ingestion (or produce ingestion rates, inhalation rates or skin surface areas, as appropriate for each of the equations, for receptor group i
BW_i = body weight for receptor group i.

Following US EPA practice for generic guidelines, by convention the averaging time for non-threshold substances is a 70-year lifetime and there are two age ranges: a child aged 1–6 and an adult aged 7–30. Total exposure therefore becomes 30 years and this duration is divided by 70.

The ‘Timber Treatment Guidelines’ combine the pathway-specific guideline values using an equation of the following form. This follows from the total exposure from each pathway being additive.

$$\text{Soil guideline value} = \frac{1}{\left(\dfrac{1}{\text{soil guideline value}_1} + \dfrac{1}{\text{soil guideline value}_2} + \dfrac{1}{\text{soil guideline value}_3} + \cdots\right)}$$

The generic equations above, except the final equation for combining pathways, are used in all New Zealand guideline documents, although there can be slight differences in their application. This is discussed in subsequent sections.

Subsidiary equations are required for each of the produce exposure and inhalation equations, to determine contaminant concentrations in produce in terms of soil concentration and to determine a particle emission factor for calculating dust inhalation. Vapour inhalation is not considered in the *Timber Treatment Guidelines*.

Calculation of produce concentrations uses bioconcentration factors to relate soil concentration to produce concentration.

$$C_P = BCF \times C_S$$

Where: C_P = concentration in produce

BCF = bioconcentration factor, expressed as the ratio of the contaminant content in produce (mg/kg dry weight) and soil (mg/kg dry weight)

C_S = concentration in soil

In addition, the ‘Timber Treatment Guidelines’ divide produce into root and leafy vegetables, with the root BCF taken to be five times that of the ‘Timber Treatment Guidelines’ leafy BCF. Fruit are ignored on the basis that contaminants do not significantly translocate to fruit. In the calculations, fruit is ignored not by ignoring the fruit component of the typical diet – the totally daily produce consumption (kg/day) is specified with fruit included – but by only using the proportions of the total for leafy and root vegetables. The two proportion factors sum to less than 1, in effect reducing the total daily produce consumption. The same effect could have been achieved by specifying only the daily vegetable intake, with the two proportions summing to 1.

The detail of the equations is not provided in the ‘Timber Treatment Guidelines’, but it is apparent from the ‘Oil Industry Guidelines’ (which adopt a similar approach) that an average concentration in produce is calculated from:

$$C_P = P_{leafy} \times C_{P\ leafy} + P_{root} \times C_{P\ root}$$

Where: the subscripts leafy and root refer to above-ground edible vegetation and roots, respectively

P is the proportion of leafy or root vegetables

$C_{P\ leafy}$ and $C_{P\ root}$ are related to $C_{S\ by}$ bioconcentration factors (BCF) for leafy parts and roots, respectively, ie, using $C_{P\ leafy} = BCF_{leafy} \times C_S$, and similar for roots

Substituting in the BCFs and rearranging in order to represent C_P in terms of C_S should yield:

$$C_P = C_S\,(BCF_{root} \times P_{root} + BCF_{leafy} \times P_{leafy})$$

However, as Cavanagh (2004b) pointed out, the 'Timber Treatment' and 'Gasworks' guidelines incorrectly used an equation equivalent to:

$$C_P = C_S (BCF_{root} \times P_{leafy} + BCF_{leafy} \times P_{root})$$

The actual effect of this error is small because P_{leafy} and P_{root} have similar values, being 31 per cent for leafy vegetables (referred to as 'stem' in the guideline) and 29 per cent for root vegetable based on 1985 Australian dietary surveys (MfE and MoH, 1997).

Typically the BCF for metals is based on field or laboratory experiments, while the BCF for organics is often estimated from the octanol-water partition coefficient (K_{ow}). For metals, the 'Timber Treatment Guidelines' relate BCF_{root} to a soil-plant distribution coefficient, K_d (unit: ml/mg):

$$0.85 \ln BCF_{root} = 3.02 – \ln K_d$$

This is attributed to ECETOC (1990); however, as Cavanagh (2004b) has noted, ECETOC (1990) indicates this equation is applicable to above-ground parts not root. As BCF_{root} is assumed to be five times that of BCF_{leafy} in the *Timber Treatment Guidelines*, the effect is an underestimation of both BCFs by a factor of five (assuming the equation above is valid for the particular metals). A further error pointed out by Cavanagh (2004b) is that the BCF (and produce concentration) are expressed as dry weight, but the amount of produce consumed is expressed as wet weight, requiring BCFs expressed on a dry-weight basis to be converted to a wet-weight basis (or alternatively, the amount of produce consumed to be converted to dry weight). This correction does not appear to have been undertaken for the inorganic contaminants.

For organic compounds (only pentachlorophenol is considered) the 'Timber Treatment Guidelines' use an empirical relationship developed by Travis and Arms (1988) to describe the uptake into above-ground parts, based on the octanol-water partition coefficient (K_{ow}). It should be noted that the US EPA (2003) has criticised the Travis and Arms relationship as being based on few data, some of which are at variance from the source documents they cite.

$$\log BCF_V = 1.588 – 0.578 \log K_{ow}$$

where: BCF_V = bioconcentration factor for vegetation
K_{ow} = octanol-water partition coefficient

BCF_V is based on the dry weight of vegetation, but as noted early, the guidelines assume the fresh weight concentration can be estimated using a moisture content of 80 per cent (ie, divide by a factor of 5). No justification is given for this moisture content.

Finally, the values derived as above are not necessarily the values adopted in the *Timber Treatment Guidelines*. Protection of plant health is also considered and the value adopted is the lowest for protection of human health or plant health. Plant health values were only adopted for the agricultural and residential scenarios, but in some cases this results in very much lower guidelines than would have been the case if only human health had been considered. The plant health values have been taken from the literature as a worst-case value for the onset of phytotoxicity for acid (sandy) soils.

The human-health guideline values for copper are unusually low relative to international values because only 10 per cent of the TDI was assigned to soil sources (in other words, 90 per cent was assigned to background intake). This has caused some controversy within the contaminated-land assessment industry (and additional remediation expenditure in some cases, although generally arsenic has driven the assessment for timber treatment sites and orchard land, the guidelines often being applied to the latter site type also).

Gasworks and oil industry guidelines

Two other guidelines, the 'Gasworks' and 'Oil Industry' guidelines, were developed shortly after the 'Timber Treatment Guidelines'. These two guidelines use essentially the same exposure scenarios as before, with an additional scenario, parkland / recreation in the *Gasworks Guidelines*. The new definitions are repeated below, as are clarifications to the agricultural / horticultural and residential scenarios (clarifications shown in italics below) contained within these documents.

- **Agricultural / horticultural –** deemed to include all agriculture and horticulture, particularly those related to food production. The general public is protected by ensuring that soil contamination does not give rise to a concentration in produce *that exceeds a published maximum residue level (MRL), although MRLs have not been nominated for most contaminants of concern. Consideration is given to the risk associated with consuming 100 per cent of produce from a contaminated source*. Consideration is also given to protecting the health of residents at any farm property, assuming that residents may be exposed by consuming home-grown livestock and produce, and through direct contact with contaminated soil. It is assumed that residences do not incorporate basements.
- **Standard residential** – based on low-density residential use, including rural residential use, where a considerable proportion of the total amount of produce consumed is grown at the site. *No consideration is given to livestock uptake of contaminants. It is assumed that residences do not incorporate basements.*
- **High-density residential** – assumes there are limited soil access opportunities, therefore there is significantly less soil and dust exposure by ingestion compared with a standard residential site. This scenario does not include consuming of produce grown at the site. (Values derived in 'Gasworks Guidelines' only, but commentary provided in 'Oil Industry Guidelines' for deriving values from generic pathway values).
- **Commercial / industrial –** as for 'Timber Treatment Guidelines'.
- **Maintenance workers –** as for 'Timber Treatment Guidelines'.
- **Parkland / recreational** ('Gasworks Guidelines' only) – reflects shorter exposure times but potentially on a regular basis. Opportunities for contact with soil will arise and children are the key concern in these areas.

While the base exposure scenarios are essentially the same, the 'Oil Industry Guidelines' examine a number of more complex pathway scenarios for each exposure scenario than the 'Gasworks Guidelines', involving the inhalation pathway to indoor and outdoor air for different soil types. This results in very many more soil guideline values for each contaminant, and makes for quite a complex document. In addition, the 'Oil Industry Guidelines' contain soil guideline values for the protection of groundwater. Examination of this pathway is beyond the scope of this report.

Previous reviews (Cavanagh, 2004b, 2004c, 2004d, 2005a; Cavanagh and Proffitt, 2005) have identified differences and inconsistencies between the earlier 'Timber Treatment Guidelines' and the subsequent 'Gasworks' and 'Oil Industry' guidelines. Rather than deriving a combined value from each of the relevant human health exposure pathways, as described in the previous section, both the 'Gasworks' and 'Oil Industry' guidelines take the lowest value derived from the individual human health pathways; however, this is not adhered to consistently with the

‘Gasworks Guidelines’ as some values appear to have been derived by combining pathways. Not combining pathways is a less conservative approach than combining the values and it is not clear why this method has been chosen; it may have been because the inhalation pathway for some volatile organic compounds and the produce ingestion pathway for some polycyclic aromatic hydrocarbons produce very low values. Combining these with the soil ingestion would have resulted in even lower, and potentially untenable, values. In other instances, there is clearly a much lower value for a particular pathway; with the other pathways have only a small effect if pathways had been combined.

Ecological receptors are considered separately in a qualitative way in both of these guidelines.

As mentioned above, the ‘Oil Industry Guidelines’ provide guideline values for several different soil types. However, soil type only influences the values derived for the inhalation pathway for volatile substances. Where vapour inhalation is not an exposure route, or not critical, the adopted guideline is the same regardless of soil type.

Further differences occur in the derivation of soil guideline values related to the produce consumption pathway and partitioning of volatiles to indoor air. For the produce consumption pathway, the ‘Gasworks Guidelines’ use the same methodology (including the errors) and assumptions used in the ‘Timber Treatment Guidelines’ to derive soil guideline values for the produce-consumption pathway, with the exception that 50 per cent of the produce consumed are vegetables and all are assumed to be root crops. The source of percentage of vegetables consumed is Langley (1993) using Australian data. Plant uptake factors for organics are predicted using the Travis and Arms (1988) relationship.

The ‘Oil Industry Guidelines’ are also different from the ‘Timber Treatment Guidelines’ with respect to produce uptake. Notably, the equation for determining soil concentrations from produce concentration is correct (MfE 1999: appendix 4F), and instead of Travis and Arms (1988), the model of Ryan et al (1988) is used to estimate plant uptake of contaminants. This model determines a fresh-weight plant uptake factor for roots and stems from contaminant concentrations in pore water. However, the ‘Oil Industry Guidelines’ do not indicate the relationship between pore water concentrations and soil concentrations, which become the soil acceptance criteria. The ‘Oil Industry Guidelines’ also assume a different composition of root and leafy vegetables to that in both the ‘Gasworks’ and ‘Timber Treatment’ guidelines, notably 10 per cent are assumed to be root vegetables, 50 per cent leafy (stem) vegetables or fruit, and 40 per cent tree-fruits, where tree-fruits are assumed not to take up contaminants.

All three guidelines assumed that 100 per cent of produce consumed was home-grown for the agricultural scenario to be protective of farming families and also, by default, consumers of farm produce. This was later considered to be unrealistic for the agricultural scenario, and was explicitly changed by the time the ‘Sheep-dip Guide’ was issued. Guideline values were derived for two standard residential scenarios: for 50 per cent and for 10 per cent home-grown produce, the latter suggested for larger urban areas. The higher percentage was routinely adopted in the early days of the ‘Timber Treatment Guidelines’ but by the time the later two guidelines had been issued, the 10 per cent home-grown produce was generally accepted as the standard urban scenario regardless of location. For example, while both 10 and 50 per cent produce values are presented in route-specific tables within the ‘Oil Industry Guidelines’, in the most commonly used residential ‘All Pathways’ tables it is the 10 per cent home-grown produce values that are given when the produce pathway is critical, rather than the 50 per cent values. This suggests an official acceptance that the 10 per cent values were the residential default.

Few details are given within the 'Gasworks Guidelines' with respect to the vapour intrusion into buildings, other than that a volatilisation factor was determined by modelling. Two scenarios were considered, surface soil (<1 m) and subsurface soil, and one soil type a sand / sandy loam, on the presumption that that would be conservative. Mention is made of the 'Oil Industry Guidelines' within the section on volatile modelling and it is presumed a similar approach was adopted, although lower values have been derived within the 'Oil Industry Guidelines' for the same soil type and contaminant.

The 'Oil Industry Guidelines' used a modified form of the Jury et al (1983, 1984) model for migration of vapours from the soil to the outdoor air. For migration into buildings the Johnson and Ettinger (1991) model was used. As noted previously, this model was developed for migration into buildings with concrete slab-on-grade floor, or into basements. It is not applicable to the common New Zealand construction of a house on piles with a crawl space below the floor, although Davis et al (2008) report that the model has recently been modified to cope with crawl-space construction.

The completely underground basement, a common residential feature in the United States, is uncommon in New Zealand for standard low-density residences. Further, the assumption within the Johnson and Ettinger model is of a perimeter crack between the floor and walls. However, in New Zealand it is common for the concrete floor to be poured integrally with the perimeter footings (typically just a thickening of the floor) with no perimeter crack being present, with timber framed walls being fixed to the concrete. In such construction vapours would have to go to the outside of the building before migrating through walls and/or under the wall/floor connection, or migrate through floor penetrations, the latter not apparently allowed for in the Johnson and Ettinger model. Further, the 'Oil Industry Guidelines' assume a crack to floor area ratio of 0.01 (1 per cent).

In comparison, the US EPA default recommendation of a 1 mm crack width (US EPA, 2004b) translates to a crack ratio of 0.38 percent for a typical slab-on-grade house in the US, the 2 mm default crack width in UK's CLEA model (EA, 2004a, 2004b), translates to a crack ratio of 0.1 per cent, only one-tenth of the assumption in the 'Oil Industry Guidelines', while the Dutch VOLASOIL model - concerning soils contaminated with volatile compounds - (Waitz et al, 1996) uses a default crack ratio of 0.01 per cent for a 'bad' floor and 0.001 per cent for a 'normal' floor. Using the 'Oil Industry Guidelines' value, a 1 per cent crack ratio for a 10 m × 15 m building (150 m^2) is 1.5 m^2 of floor openings and would mean a perimeter crack 30 mm wide, clearly unrealistic for even poor construction.

Sheep-dip guide

The general methodology provided in the 'Timber Treatment Guidelines' (MfE and MoH, 1997) and the 'Oil Industry Guidelines' (MfE, 1997) was used for soil guideline values in the 'Sheep-dip Guide' (MfE 2006a).

In contrast to the earlier guidelines, the 'Sheep-dip Guide' explicitly includes lifestyle-block land use as a typical New Zealand land use, rather than the agricultural or horticultural scenarios. This scenario assumes that 50 per cent of the produce consumed by residents is grown on site, but consumption of meat, milk and eggs of animals raised on site is excluded. Previously this land use has been a subset of residential land use (high home-grown produce in the *Timber Treatment Guidelines*). The standard residential land-use category within the 'Sheep-dip Guide' assumes that 10 per cent of the produce consumed by residents is grown on site, while the remaining categories do not consider consumption of produce grown on site.

The following five land-use categories were adopted.

- **Lifestyle block** – residential property where 50 per cent of vegetables consumed are assumed to be grown on site. The consumption of products (eggs, milk, meat) from animals raised on site is excluded and should be considered on a site-specific basis.
- **Standard residential** – low-density residential property with home-grown vegetables contributing 10 per cent of the total intake.
- **High-density urban residential** – residential with minimum opportunity for exposure to soil; no produce consumption; includes daycare centres, kindergartens, preschools and primary schools, where no gardens are present.
- **Parks / recreation** – parks, recreational open space, playing fields; includes secondary schools.
- **Commercial / industrial** (unpaved) – unpaved commercial and industrial properties. Where paving is present, its integrity and likely effectiveness in reducing exposure must be considered on a site-specific basis. No consideration of the protection of plant life has been included.

The exposure scenarios considered are largely based on those provided in the *Gasworks Guidelines*, while the parameter values used are based on those contained in both the *Timber Treatment* and *Gasworks Guidelines*. However, different parameters were used for the dermal exposure and produce consumption pathways. The equations used to derive the soil guideline values were as provided in Cavanagh and Proffitt (2005), which are essentially corrected versions of those appearing in the *Timber Treatment Guidelines*. In addition, the produce consumption values were derived using a weighted average of the consumption of different types of vegetables, using fresh weight to dry weight conversions for each. This approach, while not making a large difference to the derived soil guideline values, is technically more robust and uses New Zealand estimates for daily vegetable consumption, rather than Australian estimates.

The 'Sheep-dip Guide' is important in clarifying that the standard urban residential scenario had only a small percentage of home-grown produce consumption (chosen to be 10 per cent), whereas with the 'Timber Treatment Guidelines' there was initial (and perhaps continuing) confusion amongst practitioners as to whether a large percentage (50 per cent) or a small percentage (10 per cent) should be chosen as the urban residential default. In addition, the dropping of the agricultural scenario (which used 100 per cent home-grown produce), which was intended to represent a rural residential scenario (eg, a farm), in favour of the lifestyle block scenario, resulted in a home-grown produce percentage that was more likely to be representative of an average to high-end for most rural residents. Dropping the agricultural scenario also removes consideration of having to protect the productive capacity of the land (by setting phytotoxicity based values) and not exceeding food maximum residual levels; both are considerations that go beyond protection of site users.

A5.8.3 Summary

The basic methodology of the New Zealand guidelines compares well with the overseas guidelines reviewed. This is not surprising given their US EPA origins. While less complex than both the Dutch and UK derivation methodology, the added complexity provides dubious advantages given the uncertainty of calculating some of the primary pathways (eg, plant uptake and indoor inhalation of volatiles). The supposed additional refinement offered for some pathways and/or the addition of minor pathways will tend to be overshadowed by the uncertainties in the main pathways. The bigger advantage is possibly to remain relatively simple, for generic guideline value derivation, with the greater complexity reserved for site-specific assessment if warranted on a case-by-case basis.

It is appropriate to persevere with the US EPA-based general 'Timber Treatment Guidelines' methodology, as recommended by the Technical Review Group for the NES (MfE, 2005). However, it is clear that the inconsistencies and errors need to be corrected and a greater use be made of New Zealand-specific scenarios and parameters where possible.

The findings of this comparison have been incorporated into the *Draft Methodology for Deriving Soil Guideline Values Protective of Human Health*.

Abbreviations and Glossary

AD_{adj}	Age-adjusted dermal absorption factor
ADE	Average daily human exposure
ADI	Acceptable daily intake
AF	Contaminant-specific dermal absorption factor
AH	Soil adherence factor
AR	Skin area
AT	Averaging time
ATSDR	Agency for Toxic Substances and Disease Control (US)
BAF	Bioaccumulation factor
BaP	Benzo(a)pyrene
BaP_{eq}	BaP equivalence concentration
BCF	Bioconcentration factor
BI	Background intake
BTEX	Benzene, ethyl benzene, toluene and xylene
BW	Body weight
CCME	Canadian Council of Ministers of the Environment
CEAA	Canadian Environmental Assessment Act
CEPA	Canadian Environmental Protection Act
CERCLA	Comprehensive Environmental Response Compensation and Liability Act (US)
CLEA	Contaminated Land Exposure Assessment soil guideline derivation model (UK)
CSOIL	Contaminated soil exposure model (the Netherlands)
C_P	Concentration in produce
C_S	Concentration in soil
DDD	Dichloro diphenyl-dichloroethane
DDE	Dichloro diphenyl-dichloroethylene
DDT	Dichloro diphenyl-trichloroethane
Defra	Department for the Environment, Food and Rural Affairs (UK)
DER	Default exposure ratio
DW	Dry weight
EA	The Environment Agency (UK)
ECO-SSL	Ecological soil screening limit (USA)
ED	Exposure duration (years)
EDI	Estimated daily intake
EF	Exposure frequency (days/year)
EPA	Environmental Protection Agency/Authority (US)

ERMA	Environmental Risk Management Authority (NZ)
EFSA	European Food Safety Authority
FAO	Food and Agriculture Organization
FW	Fresh weight
GD	Guidance dose for cancer toxic effects
GRI	Gas Research Institute
GWG	Gasworks guidelines
HHEM	Human health evaluation manual
HIL	Health-based investigation levels (Australia)
HpCDD	Heptachloro dibenzo-*p*-dioxin
HQ	Hazard quotient
HRS	Hazard ranking system
IH	Inhalation rate
IH_{adj}	Age-adjusted inhalation rate
IP	Produce ingestion rate
IP_{adj}	Age-adjusted produce ingestion rate
IR	Soil ingestion rate
IR_{adj}	Age-adjusted soil ingestion rate
IV	Intervention value (The Netherlands)
JECFA	Joint Experts Committee on Food Additives
K_h	Henry's Law coefficient
K_{ow}	Octanol-water partition coefficient
LEED	Linked employer-employee data
ln	Natural logarithm (logarithm to the base e)
LOEC	Lowest observed effect concentration
log	Logarithm to the base 10
MfE	Ministry for the Environment
MoH	Ministry of Health
MPR_{eco}	Maximum permissible risk level for ecosystems (The Netherlands)
MRL	Maximum residue level
NCP	National contingency plan
NEPC	National Environmental Protection Council (Australia)
NEPM	National environmental protection (assessment of site contamination) measure (Australia)
NES	National environmental standard
NOEC	No observed effect concentration
NPL	National priorities list (US)
OCDD	Octachloro dibenzo-*p*-dioxin
OIG	'Oil Industry Guidelines' (also known as *Petroleum Industry Guidelines*)

PAH	Polycyclic aromatic hydrocarbons
PCB	Polychlorinated biphenyl
PCDD	Polychlorinated dibenzo-*p*-dioxin
PCDF	Polychlorinated dibenzofuran
PCP	Pentachlorophenol
PeCDD	Pentachloro dibenzo-*p*-dioxin
PEF	Particle emission factor or Potency equivalency factor
P_g	Proportion of home-grown produce
pH	Measure of acidity and alkalinity
p_i	Proportion of total vegetable consumption of vegetable type $_i$
PRG	Preliminary remediation goals
PTWI	Provisional tolerable weekly intake
RAGS	Risk assessment guidance for superfund (US)
RAIS	Risk assessment information system (US)
RCRA	Resource Conservation and Recovery Act (US)
RHS	Reference health standard
RME	Reasonable maximum exposure (US)
SDG	*Sheep-dip Guide*
SGV	Soil guideline value
$SGV_{(health)}$	Soil guideline value protective of human health
SL	Soil loading factor
SQGHH	Human health soil quality guideline (Canada)
SSL	Soil screening level (US)
TCDD	Tetrachlorodibenzo-*p*-dioxin or 2,3,7,8-tetrachlorodibenzo-*p*-dioxin
TDI	Tolerable daily intake
TDSI	Tolerable daily soil intake
TEF	Toxic equivalence factor
TEQ	Toxic equivalency (for dioxins)
TTG	*Timber Treatment Guidelines*
TV	Target value
UK	United Kingdom
US EPA	United States Environmental Protection Agency
UST	Underground storage tanks (US)
WHO	World Health Organization
∑DDT	Total DDT (sum of DDT, DDE, DDD)

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