Health and Environmental Guidelines for Selected Timber Treatment Chemicals

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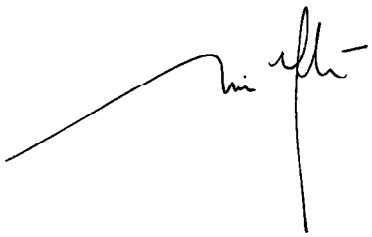
Wellington

PREFACE

The *Health and Environmental Guidelines for Selected Timber Treatment Chemicals* have been prepared to assist with the assessment and management of contaminants on sawmill and timber treatment sites in New Zealand.

The Guidelines represent the first use of detailed risk assessment procedures to derive soil and water acceptance criteria for the assessment, management and remediation of contaminated land in New Zealand.

The Guidelines are a product of extensive cooperation and effort involving experts from central and local government, industry, scientists and consultants. The assistance and cooperation of all those involved is gratefully acknowledged.



Hon Simon Upton
Minister for the Environment

Hon Neil Kirton
Associate Minister of Health

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Draft Health and Environmental Guidelines for Selected Timber Treatment Chemicals were first published in September 1993. Public submissions were received. The guidelines were further developed and revised by the National Steering Committee under the co-ordination of the Ministry for the Environment and the Ministry of Health.

National Steering Committee

Howard Ellis	Ministry for the Environment (Convenor)
Mark de Bazin	Carter Holt Harvey Ltd
Dr Beat Huser	Waikato Regional Council
Chris Shaw	Ministry of Health
Jim Waters	Ministry of Health

Regional Council and Industry Representatives

Campbell Boyd	Koppers Hickson Timber Protection (NZ) Ltd
Dr Kerry Laing	Tasman Lumber Co. (now Fletcher Challenge Forests)
Peter McLaren	Bay of Plenty Regional Council
John Sherriff	Canterbury Regional Council
Alex Singer	Tasman Lumber Co. (now Fletcher Challenge Forests)

Technical Advisers

Dr Alistair Bingham	Institute of Environmental Science and Research Ltd, Auckland
Stu Clarke	Opus Ltd (formerly Works Consultancy Services), Wellington
Stu McConnell	Egis Consulting Australia Pty Ltd, Melbourne*
Dr Peter Nadebaum	Egis Consulting Australia Pty Ltd, Melbourne*

Acknowledgement is also made of scientists from CRIs who provided expert comment, as well as those who made submissions to the draft document.

* Formerly known as CMPS&F Pty Ltd.

GLOSSARY

Acceptance criteria Levels of contaminants which are not considered to pose an unacceptable risk to human health or to the environment.

Acute An exposure or response which operates over a short term.

Background levels Levels of substances or chemicals that are commonly found in the local environment.

Bioaccumulation A general term for the process by which an organism stores a higher concentration of a substance within its body than is found in its environment.

Bioavailability The availability of a chemical in the surrounding environment for uptake by organisms.

Biodegradation Decomposition of substances into more elementary compounds by the action of micro-organisms.

Biomagnification The serial accumulation of a chemical by organisms in the food chain, with higher concentrations of the substance in each succeeding trophic level.

Carcinogen Cancer-causing agent.

Chronic An exposure or response which operates over a long term.

Clean-up The removal, treatment or containment of soil contaminated with chemicals at unacceptable concentrations.

Conservative In risk assessment or management, an analysis or a course of action which overestimates the risk to human health or the environment.

Contaminated A condition or state which represents or potentially represents an adverse health or environmental impact because of the presence of potentially hazardous substances.

Dermal Of, through or by the skin.

Ecosystem An area of nature including living organisms and non-living substances interacting to produce an exchange of material between the living and non-living parts. The term ecosystem implies interdependence between the organisms comprising the system.

Ecotoxicity	The property of being harmful to an ecosystem or to the wider environment.
Environmental risk assessment	The process of estimating the potential impact of a chemical or physical agent on a specified ecological system under a specific set of conditions.
Epidemiology	The study of the distribution and determinants of disease frequency in humans.
Exposure	Contact with a chemical, physical or biological agent.
Exposure assessment	The estimation (qualitative or quantitative) of the magnitude, frequency, duration, route and extent of exposure to a chemical substance or contaminant.
Fate and transport	Chemical, physical and biological processes that modify the concentration of a chemical through transformation (e.g. degradation), transfer between environmental media (e.g. soil to groundwater) and transport (e.g. moving with groundwater).
Genotoxic	Damaging to DNA and thereby capable of causing mutations or cancer.
Hazard	The capacity to produce a particular type of adverse health or environmental effect.
Hazard index	Sum of hazard quotients for exposure to more than one chemical simultaneously.
Hazard quotient	Ratio of exposure to tolerable daily intake for a single chemical.
Health risk assessment	The process of estimating the potential impact of a chemical or physical agent on a specified human population under a specific set of conditions.
Lowest observed adverse effect level (LOAEL)	The lowest exposure level at which there are statistically or biologically significant increases in the frequency or severity of adverse effects between the exposed population and its appropriate control group.
No observed adverse effect level (NOAEL)	An exposure level at which there are no statistically or biologically significant increases in the frequency or severity of adverse effects between the exposed population and its appropriate control.
Phytotoxicity	The property of being harmful to a type of plant.
Potable water	Water destined for human consumption.

Quality assurance	A system of activities which provides assurance that defined standards are met for the product or service concerned.
Quality control	The day-to-day operations which deliver the quality standards specified.
Receptor	An organism, plant, human or physical structure which may be exposed to a chemical or other hazardous agent.
Remediation	The clean-up or mitigation of contamination.
Risk	The probability of an adverse outcome in a person, a species, a group, or an ecosystem that is exposed to a hazardous agent. Risk depends on both the level of toxicity of the hazardous agent, and the level of exposure.
Risk assessment	The process of estimating the potential impact of a chemical or physical agent on an ecosystem or specified human population under a specific set of conditions.
Slope factor	The slope of the dose-response curve in the low-dose region, used to relate the probability of getting cancer to the chemical exposure.
Superfund	A system in the United States whereby specific industries contribute to a fund to be used for the clean-up of contaminated land.
Teratogenic	Producing malformation in embryos.
Threshold	The dose or exposure below which a significant adverse effect is not expected.
Tolerable daily intake (TDI)	An estimate of the amount of a substance in food or drinking water, expressed on a body weight basis, that can be ingested daily over a lifetime without appreciable health risk.
Toxicity	The quality or degree of being poisonous or harmful to plant, animal or human life.
Uncertainty	Imperfect knowledge concerning the present or future state of the system under consideration; a component of risk resulting from imperfect knowledge of the degree of hazard or its spatial and temporal distribution.

ABBREVIATIONS

AAS	Atomic Absorption Spectrophotometry
ACGIH	American Conference of Governmental Industrial Hygienists
ACS	American Chemical Society
ADI	Acceptable Daily Intake
AF	Absorption Factor
AH	Soil Adherence
AH_{adj}	Age Adjusted Soil Adherence
ANZECC	Australian and New Zealand Environment and Conservation Council
APHA	American Public Health Association
AR	Area
AR	Analytical Reagent Grade
ARMCANZ	Agricultural and Resource Management Council of Australia and New Zealand
As	Arsenic
As(III)	Trivalent Arsenic
As(V)	Pentavalent Arsenic
ASTM	American Society for Testing and Materials
AT	Average Time
ATSDR	Agency of Toxic Substances and Disease Registry
AWWA	American Water Works Association
B	Boron
BCD	Base Catalysed Dechlorination
BCF	Bioconcentration Factor
BCF_{FD}	Bioconcentration Factor for Foliar Deposition
BCF_{root}	Bioconcentration Factor for Roots
BH	Borehole
B_v	Bioconcentration Factor for Vegetation
BW	Body Weight

C	Concentration
C_A	Water Threshold Concentration - Aquatic Ecosystems
$CaCl_2$	Calcium Chloride
CAE	Centre for Advanced Engineering (University of Canterbury)
CAL	Calibration Standard
CCA	Copper Chrome Arsenate
CCME	Canadian Council of Ministers of the Environment
CCREM	Canadian Council of Resource and Environmental Ministers
C_D	Water Threshold Concentration - Drinking Water
CDI	Chronic Daily Intake
C_E	Soil Threshold Concentration - Environment
CF	Conversion Factor
CGSB	Canadian Government Standards Board
C_H	Soil Threshold Concentration - Human Health
COC	Chain of Custody
CP	Concentration in Plant
Cp_{FD}	Concentration in Plant due to Foliar Deposition
Cp_{root}	Concentration in Plant due to Root Uptake
Cr	Chromium
Cr(III)	Trivalent Chromium
Cr(VI)	Hexavalent Chromium
CRC	Canterbury Regional Council
Cs	Concentration in Soil
CSF	Camp Scott Furphy Pty Ltd
CSIRO	Commonwealth Scientific and Industrial Research Organisation
Cu	Copper
DASET	Department of Arts, Sports, Environment and Territories
DCM	Dichloromethane
DDT	Dichloro Diphenyl Trichloroethane
DF	Dilution Factor
DNA	Deoxy Ribosenucleic Acid
DQO	Data Quality Objective

DRo	Deposition Rate
DWG	Drinking Water Guidelines
DWSNZ	Drinking Water Standard for New Zealand
ECD	Electron Capture Detector
ECETOC	European Centre for Ecotoxicology and Toxicology of Chemicals
ED	Exposure Duration
EDTA	Ethylene Diamine Tetra-acetic Acid
EE	Equilibrium Extraction
EEC	European Economic Community
EF	Exposure Frequency
EP Tox	Extraction Procedure Toxicity Test
EPAV	Environment Protection Authority - Victoria
ETI	Environmental Toxicology International
EXSTD	External Standard
FAO	Food and Agriculture Organisation
FC	Fraction Contaminated
FCS	Field Control Sample
fei	Weathering Constant
FID	Flame Ionisation Detector
fin	Initial Fraction of Interception
frs	Fraction of Soil in Dust
GC	Gas Chromatography
GC-MS	Gas Chromatography - Mass Spectrometry
GF-AAS	Graphite Furnace Atomic Absorption Spectrophotometry
GI	Gastrointestinal
GRI	Gas Research Institute
GW	Groundwater
H_2SO_4	Sulphuric Acid
HA	Hand Auger

HASP	Health and Safety Plan
HAV	Highest Acceptable Value
HCl	Hydrochloric Acid
HESP	Human Exposure to Soil Pollutants
HI	Hazard Index
HMDS	Hexamethyldisilazane
HNO_3	Nitric Acid
HpCDD	Heptachlorodibenzodioxin
HpCDF	Heptachlorodibenzofuran
HPLC	High Performance Liquid Chromatography
HQ	Hazard Quotient
HR-MS	High Resolution Mass Spectrometry
IARC	International Agency for Research on Cancer
ICP-MS	Inductively Coupled Plasma Mass Spectrometry
ICRCL	Interdepartmental Committee on the Redevelopment of Contaminated Land
IH	Inhalation Rate
IH_{adj}	Age Adjusted Inhalation Rate
ILCP	Interlaboratory Comparison Program
IP	Produce Ingestion Rate
IP_{adj}	Age Adjusted Produce Ingestion Rate
IR	Ingestion Rate
IR_{adj}	Age Adjusted Ingestion Rate
IRIS	Integrated Risk Information System
ISO	International Standards Organisation
ISTD	Internal Standard
Kd	Distribution Co-efficient
KH_2PO_4	Potassium Dihydrogen Phosphate
K_{oc}	Organic Carbon/Water Partition Co-efficient
K_{ow}	Octanol Water Partition Co-efficient

LC_{50}	Median Lethal Concentration for 50% of Test Species
LEP	Leachate Extraction Procedure
LFB	Laboratory Fortified Blank
LFM	Laboratory Fortified Matrix
LIM	Land Information Memorandum
LOD	Limit of Detection
LOSP	Light Organic Solvent Phase
LPC	Laboratory Performance Check
LRB	Laboratory Reagent Blank
MAH	Monocyclic Aromatic Hydrocarbons
MAV	Maximum Acceptable Value (defined in Drinking Water Standards for New Zealand, 1995)
MBLP	Multiple Batch Leaching Procedure
MCC	Materials Characterization Centre
MCL	Maximum Contaminant Level
MDL	Method Detection Level (or Limit)
MEP	Multiple Extraction Procedure
MF	Matrix Factor
MfE	Ministry for the Environment
MOH	Ministry of Health
MRL	Maximum Residue Limit (Food Regulations 1984)
MS	Mass Spectrometry
MSW	Municipal Solid Waste
MTR	Maximum Tolerable Risk
MWEP	Monofill Waste Extraction Procedure
NA	Not Applicable
NaOH	Sodium Hydroxide
Na-PCP	Sodium Pentachlorophenate
NAPLH	Non Aqueous Phase Liquid Hydrocarbon
NATA	National Association of Testing Authorities
NATO	North Atlantic Treaty Organisation

ND	Not Detected
NECAL	National Environmental Chemistry and Acoustics Laboratory (now Mt Eden Science Centre, ESR)
NEHF	National Environmental Health Forum
NHMRC	National Health and Medical Research Council
NL	Not Limited
NOAEL	No Observed Adverse Effect Level
NOEL	No Observed Effect Concentration
NTG	National Task Group on Site Contamination from the Use of Timber Treatment Chemicals
OCDD	Octachloro Dibenzo Dioxin
OCN	Octachloronaphthalene
OECD	Organisation for Economic Co-operation and Development
OSH	Occupational Safety and Health Service of the Department of Labour
PAH	Polycyclic Aromatic Hydrocarbons
PCB	Polychlorinated Biphenyls
PCDD/PCDF	Polychlorinated dibenzodioxin/Polychlorinated dibenzofuran (dioxins)
PCP	Pentachlorophenol
PEF	Particle Emission Factor
Pg	Proportion of Produce Grown Onsite
PID	Photo Ionization Detection
PM_{10}	Concentration of particulates of less than 10 μm diameter
PPB	Parts per billion (1 in 10^9)
PPM	Parts per million (1 in 10^6)
PRG	Preliminary Remediation Goal
P_{RP}	Respirable Fraction
PTFE	Teflon
PTWI	Provisional Tolerable Weekly Intake
PVC	Poly Vinyl Chloride

QA	Quality Assurance
QAP	Quality Assurance Plan
QC	Quality Control
QCS	Quality Control Standard
R	Proportion Particulates Retained
RCRA	Resource Conservation and Recovery Act (USA)
RDL	Reliable Detection Level
RfC	Reference Concentration
RfD	Reference Dose
RfDc	Chronic Reference Dose
RMA	Resource Management Act
RME	Reasonable Maximum Exposure
RPD	Relative Percent Difference
RQL	Reliable Quantitation Level
RSD	Risk Specific Dose
SASG	Sulphuric Acid Silica Gel
SBE	Sequential Batch Extraction
SF	Slope Factor
SG	Specific Gravity
SLT	Standard Leaching Test
SPT	Standard Penetration Test
SWLP	Solid Waste Leaching Procedure
TCDD	Tetrachlorodibenzodioxin
TCLP	Toxicity Characteristic Leaching Procedure
TDI	Tolerable Daily Intake
TDS	Total Dissolved Solids
te	Crop Growth Period
TE	Toxic Equivalent (relates to toxicity of dioxin mixture)
TEF	Toxic Equivalence Factor
TELARC	Testing Laboratories Accreditation Council

TLV	Threshold Limit Value
TMS	Trimethylsilyl
TMS	Trimethylsilazane
TOC	Total Organic Carbon
TP	Testpit
TPAA	Timber Preservation Association of Australia
TRD	Technical Resource Document
TSCA	Toxic Substances Control Act (USA)
TSP	Total Suspended Particulates
TSS	Total Suspended Solids
USEPA	United States Environmental Protection Agency
VSS	Volatile Suspended Solids
WEF	Water Environment Federation
WET	Waste Extraction Test (California)
WHO	World Health Organisation
WPCF	Water Pollution Control Federation
WRU	Waste Research Unit
Yv	Vegetative Productivity
ZHE	Zero Headspace Extractor

CHAPTER 1
OVERVIEW

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1. OVERVIEW

1.1 BACKGROUND

1.1.1 The Purpose of these Guidelines

The Health and Environmental Guidelines for Selected Timber Treatment Chemicals provide owners, occupiers, regulators and assessors of sawmill and timber treatment sites with detailed, practical advice on site assessment and management.

The guidelines set out acceptance criteria for the timber treatment chemicals copper, chromium, arsenic, boron and pentachlorophenol.

The guidelines have been developed to protect human health and the environment, both on-site and off-site. They do this by promoting the following goals:

- protection of the health of site users, appropriate to the current or intended uses of the site;
- protection of the on-site environment consistent with the intended land use (e.g. protection of plant life on residential or agricultural sites);
- protection of the off-site environment, by specifying appropriate criteria for surface water and groundwater and for disposal of waste materials to landfills.

The guidelines are intended to provide a best-practice reference for parties involved in managing sites contaminated by timber treatment chemicals and are based on the best information that is currently available. Where this information is limited, the guidelines incorporate which have sought to combine:

- the need to protect human health and the environment;
- technical and scientific defensibility;
- practicality and pragmatism.

1.1.2 Timber Treatment Chemicals

Chemical treatment has been used for many years throughout New Zealand for the preservation of timbers, particularly softwoods. These guidelines focus on the following chemicals:

- Copper
- Chromium
- Arsenic
- Boron
- Pentachlorophenol.

Use of pentachlorophenol and its derivatives in timber preservation ceased several years ago whereas use of arsenic, chromium, copper and boron continues.

CCA Treatment

The CCA treatment process used water-soluble salts of copper, chromium and arsenic. CCA preservatives are mixtures of the following compounds:

- Chromium – sodium dichromate, chromic acid and chromic oxide
- Copper – copper sulphate and copper oxide
- Arsenic – arsenic pentoxide and sodium pyroarsenate.

Boron Treatment

Boron has been used as a water-soluble salt for timber preservation in conjunction with an anti-sapstain fungicide, such as sodium pentachlorophenate.

Boron is still widely used, although sodium pentachlorophenate has been replaced by other less environmentally persistent additives for anti-sapstain fungicide protection. Boron compounds used for anti-sapstain purposes are highly mobile in the aquatic environment.

Pentachlorophenol or PCP Treatment

Pentachlorophenol (PCP) was used in the form of sodium pentachlorophenate (NaPCP) as an anti-sapstain fungicide for short-term surface protection of sawn timber, often in conjunction with boron treatment. PCP was also used in an oil carrier, such as diesel, for permanent protection of timber. PCP and NaPCP mixtures typically contain other chlorinated phenols and polychlorinated dioxins and furans as unwanted by-products of manufacture.

1.2 STRUCTURE OF THE GUIDELINES

The guidelines deal with a wide range of issues related to the assessment and management of contaminated sites. They are organised as follows:

Chapter 1: Overview

Chapter 1 sets out the purpose of the guidelines and explains why certain chemical contaminants have been investigated. It also explains the risk-assessment approach and describes the status of the guidelines as Government policy. Finally, it outlines plans for revision of the guidelines.

Chapter 2: Environmental Sampling Strategy

Chapter 2 sets out a quality assurance strategy for site assessment. It then discusses requirements for gathering of background information and the design of the sampling and analytical programme. The chapter also discusses the requirements for soil sampling, groundwater sampling, surface water and sediment sampling, and dust sampling.

Chapter 3: Field Sampling Procedures and Quality Assurance Plan

The chapter gives detailed sampling procedures for soil, water, sediment and surface dust, including requirements to ensure the quality of information obtained from the sampling programme.

Chapter 4: Laboratory Methodologies for the Analysis of Soil and Water Samples

The chapter discusses laboratory methods for the analysis of soil and water samples for arsenic, copper, chromium, boron, pentachlorophenol, and dioxins and furans. It provides guidance on analytical method selection and on quality control procedures to help users obtain consistent analysis of samples.

Chapter 5: Soil Acceptance Criteria

Chapter 5 develops soil acceptance criteria for a range of potential future site uses. Health risk assessment techniques, contaminant fate modelling and food uptake are used to estimate acceptable soil concentrations for each land use scenario. The chapter includes interim acceptance criteria for arsenic and dioxins.

Chapter 6: Surface Water and Groundwater Acceptance Criteria

Chapter 6 develops acceptance criteria for surface water and groundwater based on a range of beneficial uses (e.g. drinking and ecosystem protection).

Chapter 7: Disposal of Timber Treatment Wastes to Landfills

This chapter develops acceptance criteria for disposal of contaminated timber treatment wastes to landfills. It includes an outline of the ANZECC Scheduled Waste Guidelines, landfill practices, leachate test requirements and threshold quantity and concentrations for disposal of wastes to landfill.

1.3 THE RISK-BASED APPROACH TO SITE ASSESSMENT AND MANAGEMENT

1.3.1 The Risk Assessment Methodology

The guidelines use a risk assessment methodology to determine acceptable levels of chemical residues for timber treatment sites. Risk assessment is a technique which allows the issue of site contamination to be addressed in an objective way. It is based on a defined process and uses the best available data. It requires that the assumptions used in deriving estimates of risk are made explicit and also provides a mechanism by which societal judgements about what is an acceptable level of risk can be incorporated into the process.

The methodology used enables the risk posed by any particular site to be expressed in quantitative terms. In these guidelines risk assessment is used to calculate levels of contaminants that do not pose an unacceptable level of risk to human health and/or the environment. These are referred to as acceptance criteria. These acceptance criteria allow decisions to be made about whether the condition of the site is acceptable for the current or any likely future intended land use.

The basis of the methodology is that the risk posed by any substance is a combination of two quite distinct factors. These are the hazardous nature of the substance, and the likely exposure of humans or the environment to that substance. The hazard factor is dealt with by using available toxicological or environmental impact studies to determine maximum permissible intake levels. The exposure factor is dealt with by using formulae which express the exposure to the substance in terms of the exposure to the site or to materials derived from the site.

The methodology assesses the risk posed by each site contaminant separately. It also assesses the risk to humans and the various ecosystem components (receptors) separately. For each chemical, the receptor that would result in the lowest acceptance criterion is then used to determine the soil or water acceptance criteria for that particular contaminant. It should be noted that ecological risk assessment techniques are still in their infancy, and as a consequence these guidelines focus mainly on the protection of human health.

1.4 ACCEPTABLE RISK

The acceptance criteria in the guidelines are primarily based on the requirement to protect human health. Very little New Zealand data is available for ecological risk assessment.

1.4.1 Human Health

In these guidelines, the level of health protection provided for human exposure to carcinogenic chemicals is such that the lifetime risk of additional cancer should be no greater than 1 in 100,000. This level is consistent with World Health Organisation guidelines and with other recently developed New Zealand health guidelines and standards.[1] For non-carcinogens, values which are believed to be protective of health are derived by standard procedures.

1.4.2 Uncertainty and Environmental Risk Management

Assessing the environmental impacts of chemical contamination requires expert judgement, particularly in areas of scientific uncertainty.

It is very difficult to derive acceptance criteria that are appropriate for sensitive land uses and that also reasonably reflect the uncertainties in the risk assessment. In these guidelines, precautionary assumptions about exposure have been adopted where uncertainties are high (appropriate to uncontrolled, sensitive land uses such as residential and agricultural uses). Less conservative assumptions about exposure might be justifiable where management controls can be readily implemented (as, for example, in ongoing industrial use).

If attainment of the recommended criteria is not practicable, other risk management options may need to be explored. These might include the use of a barrier such as pavement or a layer of clean soil to reduce the risk of exposure. The level of residual contamination and the action taken should be noted on the Land Information Memorandum[2] relating to the site.

1.4.3 Site-specific Modification of the Acceptance Criteria

The acceptance criteria presented in this document are based on specific exposure scenarios. Therefore scope exists for different acceptance criteria to be developed on a site-specific basis. If this is done, justification for the process, consistent with the level of detail presented in this document, should be provided.

1 Guidelines for Drinking Water Quality Management for New Zealand (Ministry of Health, 1995), Drinking Water Standards for New Zealand (Ministry of Health 1995).

2 A Land Information Memorandum that can be sought under Section 10 of the Local Government Official Information and Meetings Act 1987.

1.5 STATUS OF THE GUIDELINES

1.5.1 Government Policy

The guidelines are endorsed by the Minister of Health and the Minister for the Environment. As guidelines they do not have statutory force but it is recommended that local authorities make reference to these guidelines when preparing their plans and policy statements, and when granting consents under the resource Management Act 1991 (RM Act).

Earlier drafts of these guidelines have been a applied in resource consents since September 1993.

1.5.2 The Resource Management Act 1991

The guidelines are designed to facilitate the implementation of the RM Act in the management of timber treatment sites. They do this in two ways:

- The risk assessment process provides a mechanism by which the adverse effects of contaminants on the environment may be quantified.
- The acceptance criteria derived using risk assessment provide information which can be used in formulating resource consents and their conditions.

1.5.3 Other Relevant New Zealand Legislation

Other legislation is also relevant in this area: the Health Act 1956, the Building Act 1991 (Building Code F1), and the Health and Safety in Employment Act 1992 (for site clean-up[3]).

1.5.4 ANZECC and New Zealand Policy Objectives

The risk assessment/risk management framework used by these guidelines is consistent with the "Australian and New Zealand Guidelines for the Assessment and Management of Contaminated Sites" (ANZECC Guidelines; ANZECC/NHMRC, 1992). Guidance regarding investigation, assessment and management of sites, and disposal of site materials has been prepared in accordance with this risk-based approach.

3 "Health and Safety Guidelines on the Cleanup of Contaminated Sites", Occupational Safety and Health Service, Department of Labour, 1994.

The New Zealand goals for contaminated site assessment and clean-up (ANZECC Guidelines, p. 2) are:

- to render a site acceptable and safe for the long-term continuation of its existing use;
- to minimise environmental and health risks both on-site and off-site;
- where site clean-up is required, to achieve a standard that minimises risks to human health and the environment consistent with the existing and likely future use of the site, and in accordance with a system to inform future land owners that the clean-up has been conducted to an extent consistent with particular land uses.

The acceptance criteria in these guidelines achieve these goals.

The ANZECC Guidelines are currently under review, and draft revised guidelines are scheduled for release for public comment during 1997. The revision of the ANZECC Guidelines is likely to incorporate a number of land use scenarios that are broadly consistent with those developed and applied in these guidelines.

1.5.5 Application of the Guidelines in Other Industries

Although these guidelines address issues associated with the sawmilling and timber treatment industries, the guidance developed may be applicable (with expert judgement) in other situations. The elements for which guidance has been developed – copper, chromium, arsenic and boron – may also be of concern in other industries. For example, copper is a constituent of some fertilisers and pesticides; chromium is used in the electroplating and tanning industries; and arsenic and boron can arise from geothermal discharges.

1.5.6 Development of Other Guidelines

The guidelines are significant in that for the first time they set out a model of risk assessment adopted as Government policy for application to contaminated sites. Similar guidelines are being developed for other industry sectors to address the assessment and management of sites contaminated by hydrocarbons and gaswork chemicals.

1.6 SPECIFIC ISSUES REQUIRING FURTHER WORK

1.6.1 Dioxins

The Ministry for the Environment is currently undertaking a major review of specific organochlorine chemicals in the environment, including dioxins (The Organochlorines Programme). This review will develop acceptance criteria for dioxins in various environmental media. The acceptance criteria developed for the assessment of the Waipa Timber Processing Complex are referenced[4] in these guidelines and should be used in the interim for the assessment of dioxin contamination at sites where PCP has been used.

1.6.2 Arsenic

Considerable uncertainty remains regarding the bioavailability of arsenic on timber treatment sites where contamination is associated with current or historical use of copper chrome arsenate (CCA). The extent to which arsenic is taken up by plants is influenced by a number of factors, such as its chemical state, the presence of other contaminants, and soil characteristics such as composition, pH, organic content, nutrient status and porosity. Further research is needed to clarify the fate of arsenic in the soil resulting from industrial activity.

In these guidelines, only interim acceptance criteria are recommended for arsenic in agricultural and residential land use.

Due to the complexity of factors involved, consideration may need to be given to assessing the risk of exposure to arsenic on a site-specific basis.

1.6.3 Further Work on Ecosystems

The guidelines establish criteria for a range of beneficial uses of water and groundwater from the perspective of the protection of human health. In due course this work will extend to the development of New Zealand guidelines for the protection of ecosystems. In particular, future development of guidelines for the protection of aquatic ecosystems by the Ministry for the Environment[5] and the development of the National Environmental Risk Assessment Framework by ANZECC/NHMRC[6] may influence the management of sawmill and timber treatment sites. Methodologies for ecological risk assessment are still being developed internationally.

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4 Refer Chapter 5, section 5.9.2

5 "A Proposed Methodology for Deriving Aquatic Guideline Values for Toxic Contaminants", Ministry for the Environment, June 1996.

6 Draft Proc. 4th National Workshop Health Risk Assessment and Management of Contaminated Land, October 1996.

1.7 REVIEW OF THE GUIDELINES

Further work is required to address some of the issues in the guidelines, and risk assessment and management is a developing practice. It is thus proposed that these guidelines be reviewed in five years, or sooner if a significant issue arises or if new information indicates that amendment is necessary. It is expected that any review would be undertaken in the same manner as the development of these guidelines, by involving central government, regulatory authorities and the timber industry. Responsibility for any review lies with the Government.

It is likely that the interim acceptance values for arsenic and dioxins will be reviewed in advance of the revision of the guidelines as a whole.

CHAPTER 2
ENVIRONMENTAL SAMPLING STRATEGY

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2. ENVIRONMENTAL SAMPLING STRATEGY

2.1 INTRODUCTION

2.1.1 Background

It is essential that a site assessment be capable of providing reliable information on the nature, distribution and propensity for movement of contaminants present. Because of the cost of site assessment there is inevitably some compromise to the ideal goal of achieving a high level of confidence about location and concentration. At the same time, sufficient knowledge is needed to enable sensible decisions on site management. This chapter provides guidance on the adoption of a *targeted sampling approach* for the assessment of sites on which timber treatment chemicals have been used. The approach outlined is not the only one which may be appropriate.

Irrespective of the size or scope of the project, or the mode of assessment, it is important that data generated for the site be of known quality. Guidance is given in this chapter on quality assurance philosophy and framework.

2.1.2 Aim and Objectives

The principal aims of this chapter are:

- To provide direction to parties involved in the assessment of contaminated sites, on the adoption of a quality assurance strategy to assist in the production of data of a known quality.
- To provide guidance on the implementation of a specific sampling approach – the targeted sampling strategy.

2.1.3 Chapter Summary

Quality Assurance Strategy

A quality assurance philosophy and procedure appropriate to the assessment of sawmill and timber treatment sites is outlined. Practitioners are advised to:

1. Define the goal(s) of the assessment;
2. Define data quality objectives (DQOs) based on the desired quality of the data (e.g. accuracy, precision) appropriate to the nature of the assessment;
3. Formulate a quality assurance project plan for the site evaluation which is consistent with the requirements of the data quality objectives.

Targeted Sampling Strategy

A targeted sampling strategy is outlined:

- In the first phase pertinent site information is researched and reviewed to identify the likely contaminants from current and historical chemical use, the locations where contamination is likely to be found, and the potential for off-site migration of contaminant species. Check lists are provided relevant to each of these aspects.
- In the second phase a field investigation is conducted to evaluate the extent of site contamination. Advice is provided on the selection of sample sites and the choice of contaminant species for analysis for a number of field sample types including soils, groundwater, surface water, sediment and dust. Directions are given for the compilation of a contaminated land assessment report.

2.2 A QUALITY ASSURANCE/QUALITY CONTROL APPROACH TO SITE ASSESSMENT (CCME, 1991)

2.2.1 Overview

Regardless of the size or complexity of the site contamination or waste evaluation problem, management decisions need to be based on information of known quality. In essence this requires that "quality assurance" be an integral part of the overall site assessment process. The basis of a quality assurance/quality control (QA/QC)[1] programme is ensuring that data produced from any part of a study designed to evaluate the problem is sufficient to support the decision making process. The logical development of decision making is as much a part of QA/QC as the more commonly applied definitions of the quality of any single analytical result. Every "problem" evaluation should follow a pattern of development similar to that shown in Table 2.1.

[1] Quality assurance and quality control are concepts which have some degree of overlap. Quality assurance is seen as a system of activities that assures the producer or user of a product or a service that defined standards of quality with a stated level of confidence are met. Quality control differs in that it is an overall system of activities that controls the quality or a product or service so that it meets the needs of users. Simply, quality control consists of the internal day to day control and assessment of measurement whereas quality assurance is the management system that ensures that an effective quality control system is in place and working as intended.

Table 2.1
Steps Followed to Ensure the Decisions Made to Solve a Problem are Based on Data of Known Quality

No.	Step
1	Define the goal or purpose of the study and how it will be achieved
2	Define the data quality objectives that specify the quality of the data that is acceptable
3	Design a QA programme plan defining overall QA policy
4	Design a QA project plan detailing specific QA and QC requirements for the study
5	Undertake study based on the stipulations established in the previous steps
6	Evaluate data and make decisions

It is important to recognise that decision making may not necessarily require information of the best possible quality. For example, a preliminary investigation of a potentially contaminated site might involve the use of a low-cost screening analytical technique, which although sensitive, might respond simultaneously to a number of different species, including the one of immediate interest. The technique could be considered to be one of lower specificity and accuracy, with a tendency for positive bias (over-estimation of results). From the outset of the study the investigator should be aware of the limitations of the technique. Its application should be appropriate to the objectives of the study (e.g. the rapid, cost-effective assessment of a potentially contaminated site to establish if contaminant levels exceed those which are likely to give rise to an unacceptable human health risk).

A preliminary screening study would require the definition of data quality objectives which accept a degree of positive bias in the study results. The QA/QC project and programme plans would set in place an evaluation of the technique's bias by comparison with a reference method or the analysis of a standard reference material. Consequently the final evaluation of the study results would be based on a defined set of objectives and on data of known quality.

Similar considerations can be applied to sampling strategies, allowing cost-effective site investigations to be carried out to achieve defined objectives.

The individual steps shown in Table 2.1 are discussed in more detail in the following sections.

2.2.2 Defining the Goal or Purpose of the Study

Definition of the study goal or purpose should be the first activity that is carried out. The goal or purpose should be defined concisely but with sufficient detail to permit clear understanding by all parties involved.

In New Zealand most studies will be directed towards fulfilling the requirements of the Resource Management (RM) Act (1991) and, to a lesser extent, the Health Act (1956). The RM Act is based on the philosophy of sustainable management and is very much an effects based piece of legislation. The timbre of the RM Act might be summarised as requiring that processes (current or historical) shall not cause an actual or likely adverse effect on human health or on the environment downstream of the operation.[2]

Currently there are no regulatory levels in the RM Act which define "adverse effect". However, there are MFE guidelines in preparation (including this document). Guideline levels, where applicable, are clearly an essential component of any study and must be incorporated into study goal statements at an early stage. It is noted that studies undertaken as part of due diligence audits, transfer of land or in quantifying liability (whilst initiated in a legal, commercial context) must be designed with reference to the RM Act or the Health Act.

2.2.3 Data Quality Objectives

Data quality objectives (DQOs) are statements which describe the overall level of uncertainty that a decision-maker is willing to accept in results derived from environmental data. DQOs then allow for data of known quality to be generated as part of the study.

DQOs may be qualitative or quantitative in nature. Quantitative DQOs contain reference to specific quantitative terms such as standard deviations, percent recovery and concentration, whereas qualitative DQOs are descriptive in nature and may refer to specific actions that would be taken in a particular instance.

DQOs are developed for a study by stepwise consideration of a list of relevant issues. They might involve the following decision-making stages:

- State the problem to be resolved.
- Identify the decisions that need to be made.
- Identify the inputs to the decision.
- Narrow the boundaries of the study.
- Develop a decision rule.
- Develop uncertainty constraints.
- Optimise design for obtaining data.

[2] Note also, the RM Act S.107(g) refers to "Any significant adverse effects" on aquatic life.

One advantage of the DQO approach is clear communication at **the beginning of the study** between the teams involved with study management, sampling, analysis and data interpretation. The development of DQOs may involve completion of a mental checklist for a relatively simple site, or preparation of a separate scoping document for a large and complex investigation. They can be equated with good project management and become part of the record of due diligence.

Once programme goals and DQOs have been appropriately defined, a matching programme must be designed to meet them. QA and QC measures will be used to monitor the programme and to ensure that all data generated are suitable for their intended use.

An approach that has been found useful in developing a manageable structure for appropriate QA/QC measures is the preparation of separate QA programme and QA project plans which are described in Sections 2.2.5 and 2.2.6 respectively. Where possible reference has been made to later sections in this document which illustrate aspects related to specific points in the QA project plan.

2.2.4 Data Quality Indicators – The Link Between Data Quality Objectives and Quality Assurance Practice (Smith et al., 1988)

A data quality indicator is a property that can be used to assess data acquired in a sampling programme. Often conditions associated with certain data quality indicators are specified as data quality objectives. Accordingly data quality indicators form a means of assessing whether data quality objectives have been met. Quantifying or describing data quality indicators in effect dictates many of the quality assurance procedures that will be adopted during the sample design, collection and analysis programme. Data quality indicators therefore provide the conceptual bridge between specifying the data quality required and measuring it through quality assurance practices (such as the acquisition of blank samples, field replicate samples etc.).

The USEPA lists five data quality indicators that it considers important in contaminated site assessment: precision, bias, representativeness, completeness and comparability.

Precision: can be described as a "measure of mutual agreement among individual measurements of the same property". More simply here it can be thought of as a measure of how greatly an analytical result varies on repeated analysis of a sample. It is best expressed in terms of a standard deviation or variance. In contaminated site sampling components associated with sampling design, sample collection and analysis will contribute to the overall estimate of precision. It is not possible to estimate the contribution from sampling design. Combined sampling and analytical precision can be estimated by collection and analysis of duplicate (i.e. co-located) samples. Analytical precision alone can be measured by repeated analysis of laboratory replicated samples.

Bias: can be defined as "the degree of agreement of a measurement (or an average of measurements) with an accepted reference or true value". If "X" is the measurement value and "T" the true value then bias is often expressed as the difference between the two values (X-T), or a difference as a percentage of the reference or true value (100 [X-T]/T), or as a ratio (X/T).

For contaminated site evaluation, as with variance, the bias parameter may contain components from sample design, collection and analysis phases. Again the contribution from sampling design cannot be estimated. However, combined sampling and analytical steps bias can be estimated by using collected samples spiked in the field. In this process the field sample is sub-divided in the field, at least one fraction is spiked with a known quantity of the target analyte and each fraction is analysed. The percent recovery of the spike is calculated. By combining several such results an average percent recovery or bias is obtained (i.e. average percent recovery – 100%).

Representativeness: expresses the degree to which data accurately and precisely represent a characteristic of a population, parameter variations at a sampling point or an environmental condition.

For estimating an average concentration over some region, representativeness of a sample is assured by random sampling from the target population. Maximum concentration estimates over the same region require scientific judgement to choose sampling locations at or near the maximum. A strategy to achieve this for timber treatment sites is discussed in Section 2.3.3.

Completeness: is a measure of the amount of valid data obtained from a measurement system, compared to the amount that was expected to be obtained under correct normal conditions.

Use of the completeness parameter acknowledges that data may be lost by a number of different routes including specific sampling sites being inaccessible at the time of sample collection, breakage or spilling of sample during handling or shipping and sample holding time being exceeded before analysis.

Circumstances, such as where statistical parameter tests are used to assess data, may dictate a certain level of completeness requiring that contingency plans for resampling or reassessment of the sampling site be in place.

Comparability: expresses the confidence with which one data set can be compared to another.

Comparability between different monitoring exercises can be assessed by considering such variables as philosophy of sample site selection, how experimental results are reported (corrected to the same standard conditions e.g. dry weight, standard temperature and pressure etc.) and similarity of data quality measurement steps.

2.2.5 The QA Programme Plan

The QA programme plan is a document that commits the study overseers to a specific QA policy and sets forth the requirements for data needed to support programme objectives. The QA programme plan describes the overall policies, organisations, objectives and functional responsibilities for achieving data quality goals.

The five major parts of a QA programme plan are as follows:

- A statement of the purpose and importance of a QA plan.
- A description of the procedures that will be used to carry out the programme.
- A description of the resources committed to perform the QA work.
- An identification of the individual projects or packages of work in a study that require QA project plans.
- A description of how QA implementation will be evaluated.

2.2.6 The QA Project Plan

The QA project plan is a technical document that provides unified information on the project for all parties and provides details of specific QA and QC requirements. The QA project plan also specifies any QA/QC activities required to achieve the data quality goals of a project and describes how all data is to be assessed.

The QA project plan is readily divided into sections addressing different aspects of the assessment (e.g. sampling, analytical programme etc). Alternatively a number of generic stand-alone documents may be prepared, each addressing an aspect of the work, with a simple site specific work plan to be developed as part of each project.

A list of essential QA/QC activities and the area under which they would apply are presented in Appendix A.

2.2.7 Practical Implementation of the QA/QC Framework

Assessment phases at which QA/QC elements should be reviewed:

- on identifying the need for site assessment;
- on seeking proposals from consultants;
- on engaging a qualified consultant;
- on report back by the consultant;
- on deciding further action.

In the above context the timing and responsibility for each of the QA/QC tasks may be as follows:

- *Defining the goal or purpose of the study and how it will be achieved*
  This activity should be carried out by the site owner operator before engaging the consultant to undertake the investigation. It is one of the principal items in the brief provided to consultants. Whilst definition of the study goal should be undertaken by the owner/operator, inputs should be sought from regulatory authorities and consultants on the legislative or regulatory requirements.
- *Data quality objectives*
  The consultant needs to define the DQOs as an integral part of the quote for the study (refer to examples presented in Appendix B). The DQOs effectively define the scope of work which is necessary to cost such a study (e.g. how many samples to take, what analytical methods to use etc.).
- *QA programme plan*
  The QA programme plan is effectively a statement of the overall commitment to QA for the study and the outline of how this will be implemented. It would often be included by the consultant in their proposal or documentation accompanying the quotation.
- *QA project plan*
  Much of the information included in the QA project plan would normally be addressed as part of the following generic documentation:
  - internal company (consultant) quality assurance procedures, such as those complying with ISO 9000 (e.g. project organisation and responsibilities, project planning, management reporting of corrective action)
  - generic field sampling manuals or procedures developed by consultants as the documented procedures employed in site assessment field investigations. For an example of such procedures refer to Appendix C.
  - documented laboratory procedures (specific to each laboratory, and in accordance with relevant registration e.g. sample custody and storage, instrumentation).

In practice, it is also expedient to prepare a work plan that is site-specific. If an item that is normally addressed as part of the generic documentation needs to be altered (e.g. number of duplicate samples to be analysed by an independent laboratory) then this should be explicitly noted in the work plan. Other items that would normally be addressed as part of the work plan include: the chemicals of concern; QA objectives for experimental data (e.g. precision); experimental design and analytical procedures; use of statistical techniques for data evaluation; sampling network design and definition of sampling locations; and analytical detection limits.

2.3 TARGETED SAMPLING STRATEGY FOR THE ASSESSMENT OF TIMBER TREATMENT SITES

2.3.1 Introduction

If a significant assessment of contamination has not previously been undertaken, it is normal to approach a site assessment project in several stages:

1. identifying whether there is a problem;
2. fully defining the extent of the problem;
3. managing or cleaning up the problem.

This section provides guidance on Stage 1 of the assessment of soil and water contamination associated with timber treatment sites.

Given that judgemental or targeted sampling strategies are likely to take precedence over strict random or grid sampling strategies, at least for the first stage, this approach to assessment is discussed in greater detail.

The initial assessment of a timber processing site will usually consist of a two-phase approach:

Phase 1

A background study will first be carried out to identify the history of activities which could have resulted in contamination. The initial work generally consists of a site visit and a review of site history records and prior uses including, if possible, interviews with the present and previous site occupiers and employees.

Phase 2

A programme of field work will then be planned and carried out. This may include the collection of soil, groundwater and surface water samples, and their analysis. The extent of the investigation will be dependent on the type of site being evaluated, the exposure pathways and exposed population or environment; it will be based on the results of the background study and enable subsequent site characterisation.

The following sections of this report provide more detail on these two phases of work.

2.3.2 Objectives

The key goal/aim of a site contamination assessment programme is to identify:

- the general extent of soil, sediment, building dust, groundwater and surface water contamination and the potential adverse impact of such contamination on the health of workers on-site, the health and safety of the public, and the quality of the surrounding environment;

- sufficient information should be obtained to make an estimate of risk posed by contamination, to human health and the environment; and
- to determine whether remediation or mitigation measures are required in the context of current or likely future use of the site.

Information to assess specific contaminant migration pathways and environmental impacts (e.g. transport of contaminants in wind-blown dust) may also be required by the regulatory authorities.
As discussed in Section 2.2, the specific objectives of each site contamination assessment programme should be clearly and carefully defined at the start of each assessment.
The sampling and analysis strategy discussed in the following section provides guidance on how and where to sample, and what analyses should be undertaken. The strategy is designed to assist in the development of a detailed work plan for a particular site.

2.3.3 Basis for a Targeted Sampling Strategy

There are a range of sampling strategies that may be used as the basis for assessing a site. These strategies may be broadly classified as targeted (or judgemental), random or grid. Each of the strategies has advantages and disadvantages, and in practice usually a combination of these approaches is found to be most appropriate (e.g. sampling may be targeted toward the areas of primary concern, however, within each of these areas a small sampling grid may be established).

The overall aim of the first stage in the assessment process is to determine whether there is a problem (refer 2.3.2) as cost-effectively as possible. Data quality objectives would probably indicate that the most expedient approach (i.e. the least number of samples/locations) is to target the locations of those areas most likely to be contaminated. However, a targeted sampling strategy is only as good as the review of the site history on which it is based. If an area of potential contamination is not identified as part of the site history review, then it will not be addressed as part of a targeted sampling programme. Therefore, a targeted sampling programme should not be used where there is little or no site history to support the selection of sampling locations. In practice, taking into account possible uncertainties in site history information, a common approach is to take several samples at close grid spacings and within areas likely to be contaminated, and several samples at wider grid spacings across broader areas of the site where contamination is less likely.

It is difficult to statistically define the probability of recovering a contaminated sample from an area of significant contamination using a targeted sampling approach, but the number of samples required to achieve a similar notional level of confidence (or probability of detection) using a grid or random sampling approach is likely to be much greater- provided reasonable site history information is available. On this basis a targeted sampling approach is often more cost-effective than a grid or random strategy.

Noting the above constraints, a targeted sampling programme is suggested as the most appropriate approach for the first stage of the site assessment process.

2.4 BACKGROUND INFORMATION GATHERING – PHASE 1

2.4.1 General

All pertinent background information should be reviewed in order to identify the potential for on-site and off-site contamination. This phase of the work should be completed prior to commencing Phase 2.

The background information study should include:

(a) The chronological history of previous site uses and industries; the activities or processes carried out on the site with respect to the timber industry, particularly the location and historical usage of timber treatment chemicals and associated operations such as chemical mixing.

Interviews with site personnel, past workers at the site and local residents can be an invaluable source of such information. Other sources of site history information include:

- past and present owners of the site;
- aerial and ground photographs;
- local government records (e.g. history of complaints, discharge or building permits);
- trade and street directories;
- local literature (e.g. newspapers).

(b) Identification of equipment and areas where the likelihood of contamination resulting from historical or current work practices is high (e.g. accidental spillage of chemicals at mixing and treating facilities and at treated wood storage and conveying areas).

(c) Source information (e.g. current and past site management; engineers; workers) in order to establish raw material use, products, known chemical or treatment waste release history (spills, leaks, etc.) and waste disposal practices (i.e. on-site, off-site).

(d) Local hydrogeologic data including:

- the extent, interconnection and use of aquifers in the area;
- probable direction and rate of groundwater flow in each aquifer;
- information on the site geology and soils at the site; and

- local municipal drinking water supply sources, and the location of private or industrial wells or bores, especially those supplying drinking water.

(e) Surface water bodies (creeks, rivers, estuaries, wetlands) particularly where these may be adversely affected by contaminated groundwater or surface drainage from the site. Surface water bodies should be evaluated to determine environmental values, beneficial uses, sensitivity to change and physical, chemical and biological characteristics.

(f) Any published or known information which establishes whether adjacent property owners are or have been potential sources of contamination (of the site soil and groundwater).

(g) Location, age and construction material of above and underground chemical or fuel storage tanks on the site. If integrity testing of storage tanks has been undertaken, the results of such tests should be reviewed.

(h) Locations and construction details of underground services including the site stormwater system (they may have a potential impact on future remediation activities or can act as preferential drainage pathways).

(i) Present and likely future zoning of the site.

(j) Contour or topographic maps – location of filling/earthmoving.

(k) Likely future use of the site.

(l) Potential cultural issues, e.g. iwi, archaeological, etc.

2.4.2 Potential Contaminant Sources

As part of the background information study, specific consideration should be given to the following areas or potential sources of contamination:

Anti-sapstain treatment – Na-PCP, and other antisapstain chemicals (e.g. carbendazim)

- bulk storage tanks for Na-PCP;
- anti-sapstain preparation areas – mix rooms;
- dip baths;
- green chain;
- timber drying and storage areas (drippage from timber);
- transportation routes for freshly treated timber;

- sludge and residue storage and disposal sites; and
- dry chemical storage areas.

Possible contaminants include PCP, chlorinated phenols other than PCP, dioxins and furans, and others depending upon site specific review of anti-sapstain preparations used.

Boron diffusion

- bulk storage tanks;
- dip baths;
- drip pads;
- timber storage areas (drippage from timber);
- transportation routes for freshly treated timber;
- sludge and residue storage and disposal areas; and
- dry chemical storage areas.

There is a need to consider what additives were used in the boron diffusion preparations. Possible contaminants include: boron, PCP, chlorinated phenols other than PCP, dioxins and furans, and organochlorine pesticides.

CCA treatment

- bulk storage tanks;
- chemical preparation areas;
- pressure treatment vessels;
- drip pads and drip areas (i.e. where timber was put immediately after removal from the vessels);
- timber storage areas;
- sludge and residue storage and disposal;
- treated timber residues and ash; and,
- transportation routes for freshly treated timber.

There is a need to consider additives or contaminants that may be in CCA preparations and sludges (e.g. PCP). Possible contaminants include: copper, chromium (hexavalent and trivalent) and arsenic.

(Note: additives and contaminants in CCA preparations are only likely to be present in soils at low levels).

PCP in oil pressure treatment

- bulk PCP storage tanks;
- bulk oil (diesel storage tanks);
- PCP in oil preparation areas;
- pressure treatment vessels;
- drip pads or areas (i.e. where timber was placed immediately after removal from the treatment vessel);
- transportation routes for freshly treated timber;
- treated timber storage areas; and
- sludge and residue storage and disposal.

Possible contaminants include PCP, chlorinated phenols other than PCP, dioxins and furans and petroleum hydrocarbons.

Creosote treatment

- bulk storage tanks;
- chemical preparation areas;
- treatment areas;
- drip pads and areas;
- freshly treated timber transportation route;
- treated timber storage areas; and
- sludge and residue storage and disposal.

Possible contaminants include petroleum hydrocarbons, polycyclic aromatic hydrocarbons, phenols and cresols.

Miscellaneous

Possible contamination sources include:

- bulk chemical storage tanks, preparation areas, treatment areas and treated timber storage associated with:
  - organochlorine pesticide treatment (e.g. chlordane and lindane)
  - LOSP process (possible contaminants: tributyl tin, petroleum hydrocarbons/monocyclic aromatic hydrocarbons, naphthenates and chlordane)

  Note the LOSP is a relatively recent technology and contamination potential is reduced;

- underground and aboveground fuel storage tanks (possible contaminants: petroleum hydrocarbons, monocyclic aromatic hydrocarbons, polycyclic aromatic hydrocarbons);
- workshop and maintenance areas (possible contaminants: petroleum hydrocarbons, solvents, others);
- landfilled or stockpiled sawdust or wood chip;
- other waste pits on site;
- contaminated building and construction materials.

The potential for contamination associated with each of the above items should be evaluated as part of the investigation design. The sampling programme should be tailored to reflect the actual conditions, focussing on obtaining information about the areas of greatest risk, giving consideration to the available site history.
Note that in some cases consideration must be given to other sources or modes of transport of contaminants resulting in human or environmental exposure e.g. contamination of rain water supplies by contaminated dust fallout, transport of contamination in stormwater and stormwater sediments.

2.5 FIELD INVESTIGATION PROGRAMME – PHASE 2

The overall aim of the field investigation programme is to evaluate site contamination and provide sufficient information to assess possible site remediation, if found to be necessary.

The field investigation programme should be developed on a site-specific basis after the completion of the background study, and could include the following as appropriate:

- soil sampling, targeted to areas of likely contamination and some background locations;
- surface water and sediment sampling at locations to be determined following determination of site runoff patterns;
- groundwater sampling; and
- surface dust sampling from selected structures on-site.

Sampling of stored sludges, stockpiles or waste pits may also be required.
A site work plan should be prepared which sets out the requirements and objectives for the various field sampling activities and protocols for collection of samples. All field sampling and associated data collection must be supervised by a person experienced in the collection of environmental samples, and carried out in accordance with approved sampling procedures (Quality Assurance Plan (QAP), Chapter 3), and an approved site Health and Safety Plan (HASP).

Typical programmes for sampling soil, groundwater, surface water, sediments, and surface dust, and the associated analytical programmes, are outlined in the following sections.

2.6 SOIL SAMPLING

2.6.1 Objective

An assessment programme for the characterisation of soil contamination should:

- determine whether potential sensitive human receptors on and off-site (e.g. full and part time workers, maintenance workers, residents and recreational users) are possibly at risk from contact with contaminated soil;
- determine whether there are unsecured areas of contaminated soil which could be transported off-site as contaminated sediment in runoff or dust;
- determine whether the contamination is mobile within the soil environment, and with potential to leach to groundwater (off-site transport);
- determine the potential for other off-site impacts.

It may be appropriate to carry out the site investigation in stages so that information gained early is used to focus later investigations. Typically, up to three stages may be warranted:

1. Initial – a minimal level of sampling to indicate if there is likely to be any problem.
2. Indicative – an attempt to estimate the extent of contamination, vertically and laterally, based on contamination patterns identified in the initial investigations and the importance of exposure pathways identified during the initial investigation.
3. Quantitative – to determine volumes of soil and scope of remediation.

2.6.2 Field Sampling

Based on a review of site history, a targeted soil sampling programme can be implemented in areas considered likely to have been contaminated by past site activities. In addition, limited soil sampling may also be completed across areas considered more representative of the general site conditions. Typically, sampling across general site areas would be undertaken on a coarse grid basis[3], the aim of which would be to identify broad areas of contamination rather than to identify relatively localised hot spots with a high level of statistical confidence.

[3] Refer Gilbert (1987) for information on the design of grid and/or random sampling programmes.

For example, in order to identify a hot spot 12 metres in diameter with 95% confidence, a sampling density of 100 samples per hectare (or a 10 m grid) would be required (Standards Australia, 1994). Such a hot spot would clearly be significant for some site uses, however, identification using a grid sampling strategy would be prohibitively expensive. It is therefore necessary to use other techniques (e.g. site history, geophysics) to identify localised areas for targeted sampling, together with broad grid sampling to identify widespread contamination.

The site would be divided into investigation areas on the basis of site history and the nature of potential contamination. The sampling programme undertaken in each investigation area should be based on information gathered in the first assessment phase and should be a targeted approach (i.e. maximising the likelihood of identifying whether contamination is present, and to indicate likely contaminant distribution while minimising initial costs).

The initial programme must be designed on a site-specific basis, however, for a small site it typically may involve:

- between, say, four and eight sample locations from each investigation area (e.g. samples from four locations and, at least, two different depths around a Na-PCP mix room etc.), and from between, say, eight and sixteen locations across the general site[4];
- typically two depths (say 0-100 and 300-500 mm) at each location with surface only testing in areas where penetration at depth is less likely or where there is no requirement for information regarding the depth of penetration.

Additional soil sampling to depths in the order of four metres or greater may be required in the vicinity of any underground storage tanks and pipework (where the initial sampling indicates that contamination has migrated downward) or in areas of historic waste disposal or filling.

The above is given as an indication of possible sampling requirements. The actual sampling programme for each site should be developed based on a consideration of the available information on the possible location and extent of contamination.

The initial location and number of sampling points will be based on known site use patterns. Detailed procedures for sampling of soils by hand auger, mechanical auger or backhoe are discussed in Chapter 3.

There are a number of possible scenarios that would lead to variation of the typical sampling programme, including:

[4] Must be determined on a site-specific basis. Refer to Gilbert (1987) for advice regarding the statistical design of sampling programmes.

- specific consideration should be given to soil sampling at the groundwater interface where groundwater is relatively shallow. In particular, free phase hydrocarbons may accumulate at the groundwater interface.
- where little or no site history information is available, selected sampling locations may not reliably detect contamination and therefore it may be necessary to undertake a more detailed grid sampling programme across the site.

Compositing[5] of soil samples prior to analysis is a useful tool. It increases the area addressed by the sampling programme without greatly increasing analytical costs. However, there are limitations to the use of this approach. Compositing of soil samples assumes that a valid estimate of the population mean of the characteristic under consideration can be obtained from this single analysis of the composite sample: i.e. all samples which form the composite are drawn from the same population; each sample contributes equally to the composite.
In general, in areas where contamination is expected, samples may be composited provided there is some basis for expecting similar contaminant concentrations in each sample (e.g. at the end of a dip tank), or where average contaminant concentration is specifically sought (e.g. estimating the average exposure of site users). In areas where contamination is not expected, compositing may be undertaken to reduce analytical costs.
Compositing should be limited to, say, four sub-samples to ensure exceedance of the guidelines by any sub-sample can be reasonably detected if required.
It is recommended that composite samples be prepared in the analytical laboratory from discrete, documented site samples which have undergone appropriate sample preparation stages (refer Section 4.5.3). Accordingly if significant contaminant concentrations are detected in the composite, it is possible to reanalyse the individual sub-samples to assist in identifying the source of contamination.

In the design of the sampling programme, it should be remembered that clean-up guidelines are generally based on estimates of long term average exposures; further sampling and analysis may be required to estimate such exposures. **Analysis of single samples may be advisable when characterising the extent of contamination.**

Further information regarding the sampling of contaminated soils may be obtained from published standard guidance (e.g. USEPA, ASTM) such as USEPA (1991) "Description and Sampling of Contaminated Soils: A Field Pocket Guide".

2.6.3 Analytical Programme

Sub-samples may be composited by the primary laboratory for analysis for each investigation area as defined in the sampling plan (part of the QA Project Plan). Any variations in the sampling plan must be discussed in the report.

[5] Where a sample is made up from a number of sub-samples taken at different locations.

Primary and quality control analyses should be undertaken by laboratories that conform with the requirements for TELARC or equivalent (e.g. NATA) registration for the specific methods/compounds in question.

Samples should be analysed for those contaminants likely to occur in the area from which the sample was recovered, based on the results of the background information study.

Possible analytes include:

- copper
- chromium (hexavalent and trivalent)
- arsenic
- boron
- pentachlorophenol and other chlorophenols
- dioxins and furans (OCDD screen or full congener analysis)
- tributyl tin
- polycyclic aromatic hydrocarbons
- monocyclic aromatic hydrocarbons
- phenols and cresols
- organochlorine pesticides
- petroleum hydrocarbons
- other antisapstains
- others depending on chemicals used at a given site.

In practice, analyses should be directed to contaminants most likely to be present and of concern. It may be appropriate to omit other possible contaminants on the basis of a knowledge of the treatment chemicals used on the site, or to carry out preliminary screening on composite samples to exclude the need to carry out more extensive analyses.
Dioxin and furan contamination could be associated with areas subject to PCP contamination which has been ongoing over an extended period of time. In many cases, lower-cost screening methods of analysis may be sufficient for dioxin and furan characterisation, although an initial test to establish the full congener profile and the relevance of the screening method should be carried out. An example of such a dioxin screening method is the OCDD screen discussed in Chapter 4.

Where a site may be redeveloped for residential or agricultural use it is important that judicious use is made of the OCDD screen, given that dioxin concentrations may be limiting on some sites. Selected samples should be analysed for dioxins and furans, based on a review of the results of the PCP analyses. However, caution should be

exercised in this regard, given that elevated concentrations of dioxins have been reported in areas with relatively low PCP concentrations.

While PCP will be generally the contaminant which controls the overall health and environmental risk associated with contamination from PCP-based formulations, it is desirable to carry out an initial check of the range of chlorinated phenols present to ensure that PCP is the controlling chlorinated phenol.

Suggested maximum detection limits for various analytes are presented in Chapter 4. Analytical methods and practices appropriate for the assessment of soil and water contamination at timber processing sites are discussed in Chapter 4. For other parameters reference should be made to published methods (e.g. USEPA, ASTM and APHA).

2.7 GROUNDWATER SAMPLING

2.7.1 Objective

A preliminary groundwater investigation programme should be completed as part of the second phase investigation if groundwater is at a depth that may be affected by site contamination. Typically, if groundwater is at a depth of less than 10 to 15 m a groundwater monitoring programme should be considered, depending on site-specific factors including the nature of the overlying soils. The design of the groundwater investigations should be directed towards:

- determining the depth to groundwater, thickness of the near-surface aquifer, direction and rate of movement and probable discharge location of any contaminants to surface water (e.g. surface drains, streams, etc.); and
- determining whether contaminants are present in the groundwater and, if so, at what concentrations and in what form (both chemical and residues from timber wastes, such as leachate).

The groundwater monitoring programme will be designed to assess the impact of ground contamination and leachate (e.g. from filled areas or waste disposal sites) on the local groundwater quality, and the contribution of groundwater discharge on contaminant concentrations in surface water bodies.
Particular consideration should be given to the specific hydrogeological conditions at the site when designing the groundwater investigation programme. The possibility of contaminant migration along preferential flow paths and the need to use techniques other than conventional groundwater monitoring bores, as discussed below, should be considered on a site-specific basis.

2.7.2 Field Sampling

A limited groundwater investigation programme should be implemented where groundwater is at a depth that may be impacted by site contamination. Following a review of the available site history and a further review of the regional hydrogeology, the number and location of the groundwater monitoring bores should be determined. Typically, one bore should be installed upgradient of all known areas of major fill and significant contamination potential, and two to four bores may be located downgradient of areas of significant filling or contamination potential. This has the objective of measuring groundwater quality, direction and rate of movement on-site, and providing some indication of the contribution of groundwater discharge on surface water quality. The exact number and location of groundwater monitoring bores should be determined on a site-specific basis.

The groundwater monitoring bores should be installed under qualified supervision[6] by suitably qualified drilling contractors in accordance with the installation, design and procedures defined in the field sampling plan and the documented quality assurance procedures (refer Chapter 3). Consideration should be given to the recovery of soil cores during monitoring bore installation, to allow for later laboratory analysis. The specific location and reasons for recovery of such samples should be defined in the site specific sampling plan.

Note that in some areas there may be a need to obtain a bore permit prior to drilling, with information on the bore position and stratigraphy (borelog) to be forwarded to the relevant authority, particularly if the aquifer is used as a drinking water supply.

For details regarding requirements for preservation and filtration of groundwater samples, refer to Appendix E.

Installation, development and purging of the monitoring bores and groundwater samples should be performed in accordance with the procedures outlined in Chapter 3. All monitoring bores should be located with reference to permanent site features and the standpipe levelled or surveyed, allowing re-establishment of the bore locations should it be destroyed. The depth to groundwater should be measured relative to top of the standpipe casing, prior to purging and sampling. Note that the top of the standpipe casing should be surveyed to determine its elevation, referenced to a common datum e.g. mean sea level.

During preliminary groundwater investigations, selected bores should be subject to drawdown/recovery or similar tests to assist in the estimation of aquifer characteristics. This is also a check on the quality of installation. If test results do not represent what would be expected in the aquifer, the installation may have to be redeveloped or replaced.

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[6] Under the supervision of an experienced engineer/geologist/scientist.

2.7.3 Analytical Programme

Selected groundwater samples should be analysed for pH and total dissolved solids. Additional unfiltered samples may need to be recovered to facilitate the analysis of groundwater samples for these parameters.

In addition, samples may be analysed for some or all of the parameters measured in soils (Section 2.6.3) based on the results of the background information study. The required analytical detection limits are outlined in Chapter 4.

It may be desirable to undertake an anion/cation balance on some samples as a check on the sampling and analysis procedures, and to assist in characterising the groundwater system (e.g. differentiation between water types, recharge, flow paths, etc).

It is noted that whilst groundwater samples may be filtered prior to analysis, in the case of dioxins, limited additional analysis of unfiltered samples from well-developed bores (i.e. representative of the aquifer in terms of turbidity) may be warranted to assess the contribution by transparent or fine suspended particulates within the aquifer.

2.8 SURFACE WATER AND SEDIMENT SAMPLING

2.8.1 Objective

As part of the overall review of site discharges, a preliminary surface water and sediment sampling programme should be implemented if there is a surface water body in the vicinity of the site and there is potential for contamination of this water body to occur. The programme should provide an estimate of contaminants (i.e. load and concentration) leaving the site via drains, surface water runoff and groundwater discharge to surface water bodies. It may include recovery of water samples and sediment samples from both site drains and nearby surface water bodies. Sediment sampling is a useful source of qualitative information regarding off-site transport of contaminants as some substances will partition preferentially into the sediments. The recovery of sediment samples will also assist in the assessment of contaminant transport as suspended particles. Some of the primary contaminants of concern exhibit very low solubility in water and will attach preferentially to suspended particles.

2.8.2 Field Sampling

The surface water sampling locations should be determined following a detailed review of surface water flow patterns on-site and likely groundwater flow direction and discharge. Surface water samples should be recovered from at least one location upstream and one downstream of the site, and from one or more locations adjacent to the site. In addition, at least one sample should be recovered from any potentially

contaminated drain discharging from the site. Additional samples may be required on a site-specific basis.

At least two separate rounds of surface water sampling may be completed in order to provide estimates of water quality under wet and dry weather conditions. The sampling regime should be targeted towards characterising the first flush of runoff during a wet weather event, and surface water contamination resulting from groundwater inputs during dry weather.
All surface water samples collected at each sample location, for each weather condition, would normally be composited to form one sample for analysis (i.e. one sample per location, per weather conditions for analysis). In some situations it may be appropriate to analyse several 'grab' samples to facilitate evaluation of variation in contaminant concentrations with time (e.g. first flush of runoff after a dry period).

One representative sediment sample should be collected from each sample location. This may require the collection of several sub-samples from one location, followed by compositing. Additional samples may be recovered depending on requirements to analyse samples for other constituents. Sediment should be recovered during relatively dry weather flow conditions, in order to assess the conditions to which aquatic species would normally be exposed. Any sediment laid down during wet water flows would also remain in place for sampling during dry weather. Usually, wet weather flows would result in scouring or removal of sediment, rather than deposition.

2.8.3 Analytical Programme

The analysis of surface water samples includes the same parameters as specified for groundwater with the addition of total suspended solids (TSS). Sediment samples may be analysed for a range of parameters as listed for the analysis of soil samples.

2.9 DUST SAMPLING

2.9.1 Objective

A preliminary programme of sampling dusts deposited on site structures should be undertaken where there is potential for contaminated dust accumulations to occur. The usual objective of this programme is to assess potential risks to human receptors. Where residences are located on-site or immediately adjacent to the site, consideration should be given to the potential for contamination of dust within the residence.

2.9.2 Field Sampling

Dust samples should be collected from selected structure surfaces and from residences on-site or immediately adjacent to the site. Two types of dust samples may be recovered, as follows:

- dust samples recovered from the living spaces of residences at, or adjacent to, the site and from the main work areas on-site.
- dust samples recovered from undisturbed areas, such as ledges and the roof space of residences.

Samples recovered from living spaces and the main work areas on-site are most useful in the assessment of risk to current workers or residents. Exposure to dusts within living spaces of residences and main work areas may occur through inhalation or ingestion of dust, or by the absorption of contaminants through the skin. In the first instance, the soil acceptance criteria derived for these exposure routes may be used to provide an assessment of the risk to residents and workers (refer Chapter 5). Whilst sampling of dust from the main work areas on site may provide some information regarding the risk to workers at the site, conventional occupational exposure monitoring provides a more direct measure of occupational risk. Advice on this should be sought from the Occupational Safety and Health Service of the Department of Labour.

Samples recovered from undisturbed areas provide some qualitative information regarding the possible risk to workers or residents in the event of disturbance of the dust e.g. demolition of buildings. The quantity of accumulated dust can also provide useful information regarding the possible magnitude of the risk.

Sample locations should be confirmed on a site-specific basis; however, initially it is anticipated dust samples should be recovered from, say, at least two locations within on-site timber processing buildings. Additional samples may be required where there are residences on-site or immediately adjacent to the site.

2.9.3 Analytical Programme

The analytical programme for dust samples should be confirmed following the results of the background information study. The programme may include chlorinated phenols, copper, chromium, arsenic, boron, dioxins and furans and other parameters as listed in Section 2.6.3. In addition, structural surface dust samples should be analysed to determine the particle size distribution, allowing the potential for inhalation of particulates to be assessed.

Compositing of dust samples may be considered where several samples are collected from a single work or use area and the aim of the investigations is to characterise the average contaminant concentrations to which workers or other site users may be exposed.

2.10 CONTAMINATION ASSESSMENT AND REPORTING

A contaminated land assessment report for a site should include (but not be limited) to the following:

(a) a statement of the objectives, scope and limitations of the assessment and report;

(b) a detailed description of the land, including ownership and occupier details etc;

(c) a detailed history of the uses of the site. This should include a list which specifies the identities and locations on a premises of any known or suspected chemicals or any other substances which could be a hazard whether imminent or otherwise. Sources of information (documented and anecdotal) and validation of information should also be included;

(d) likely current and future use of the land;

(e) recording of any visual inspections of the site;

(f) details of the geology and hydrology of the area, including physical characteristics of the soil (for example: type, porosity and sorptivity, transmissivity, areas of fill, variation of such characteristics with depth) and groundwater (depth, rate of flow), regional groundwater quality, use of the groundwater in the area. Copies of all bore logs, soil profiles and other records of field observations and measurements should be provided;

(g) details of the condition and location on the site of buildings, sewer/drainage systems, natural water courses, underground storage tanks, waste disposal areas and other activities;

(h) a detailed site plan including scale, north point and all relevant site features and sampling locations;

(i) the sampling and analysis programme used to determine the extent and distribution of contamination, including:

- basis for selection of chemicals included in the analytical programme;
- rationale for sample locations and depths in each medium of concern (air, soil, groundwater, surface water);
- sampling methods;
- detection limits (levels chosen and how derived);
- quality assurance (procedures);
- quality control details; and
- laboratory and analytical methods used.

(This programme will consist usually of two stages, an initial evaluation to confirm the presence of contamination and then a more comprehensive evaluation to determine the nature and extent of contamination.)

(j) results of the sampling and analysis programme on which is based a conceptual model of how contaminants are moving on the site and their fate and transport characteristics in each media of concern.

(k) information about any contaminants of concern, selected on the basis of the results of the sampling programme. This information should include an evaluation of:

- the fate and transport of each chemical;
- the form or species present;
- physical characteristics;
- potential harm to humans, plants, animals and structures;
- aesthetic impairment;
- any detriment to any beneficial uses to be made of the site;
- potential for adverse off-site effects; and
- potential exposure pathways.

(l) the results of the field investigations should be discussed with reference to the guideline values nominated for various site uses in Chapter 5 of these guidelines. Particular attention should be given to site-specific factors which may require modification of the nominated values.

(m) recommendations including further activities required at the site to mitigate contamination, if necessary.

A typical table of contents for such a site assessment report is presented in Appendix F, for this chapter.

Particular care should be exercised when presenting and discussing the results of any sampling and analytical programmes. Wherever possible analytical data and field measurements should be presented in a tabular format or in graphs or site plans. Such presentation allows ready access to the available information and permits the reader to more easily visualise and comprehend the nature and extent of any contamination identified. Graphical presentation is particularly useful when examining variation in various analytes with time.

When assessing soil contamination it should be noted that the proposed soil acceptance criteria (refer Chapter 5) have been developed in the context of specific scenarios for each of the nominated land uses. In practice, the impact of soil contamination is influenced by a wide range of factors, some of which may not been considered in the development of the acceptance criteria. For example, where guidelines have been based on health risk considerations, the potential for contamination of groundwater or surface rainfall runoff should be considered on a site-specific basis, as this may determine the acceptable soil contamination concentration.

Site-specific factors which can affect the acceptability of contamination include:

- the extent and distribution of ground contamination;
- the extent of pavement or other ground cover limiting exposure of workers, residents etc. to ground contamination;

- off-site impact of contamination;
- site management and works practices;
- mixed waste sites for cases when contamination other than timber treatment chemicals is present.

The soil acceptance criteria (refer Chapter 5) have been based on the assumption of a largely unpaved, uniformly contaminated site. Where such an assumption is not valid, a site-specific review of the potential impact of ground contamination is required. In practice, limited areas of relatively higher-level contamination, together with more widespread areas of low-level contamination, is likely to be a typical pattern of contamination resulting from historical timber treatment activities.

Where chronic human exposure to soil contamination is the primary concern, it is appropriate to compare average contaminant concentrations (rather than the maximum concentration detected) with the acceptance criteria. In this regard, the area across which contaminant concentrations are averaged should be selected as being the typical area in which a person may spend most of his/her time exposed to soil contamination. In the case of a residential land use, the averaging area may be selected as the area of a typical backyard. The averaging area must be selected carefully, with reference to likely exposure patterns, to ensure that the significance of a localised hot spot is not obscured; e.g. it is unlikely that it would be appropriate to average contaminant concentrations across the entire site in an industrial context.

The health-based guidelines presented in Chapter 5 are based on estimates of the reasonable maximum exposure. The USEPA (1991b) indicates that a "conservative estimate of the media average concentration over the exposure period" should be used to estimate the reasonable maximum exposure. The average contaminant concentration used should be determined using appropriate statistical techniques, such as the 90^{th} percentile confidence interval for the sample mean, where samples were obtained from a geometric sampling arrangement. The approach of comparing an average contaminant concentration with the acceptance criteria may not be appropriate where the given criterion is set based on, say, the protection of plant life.

Criteria set on the basis of phytotoxicity may be reduced in some situations depending on the soil type and chemistry and the tolerance of various plant species. Such variation must be evaluated on a site-specific basis.

Care must be exercised when applying statistical methods to the assessment of data from a sampling and analytical programme. Where samples are recovered from a grid or from randomly selected locations, statistical analysis of the resulting data is a relatively well-defined process. It should be noted that site data is not usually normally distributed, and may require less common but well-established methods, e.g. non-parametric methodology. Generally grid sampling is undertaken when the aim is to characterise contamination across broad acre areas without defined point sources of contamination e.g. general treated timber storage yards.

More frequently, samples are recovered from selected, targeted locations focussing on known sources of contamination, e.g. antisapstain dip tanks, especially as part of the initial sampling programmes. Whilst statistical techniques can be applied to the analysis of data from such programmes, the techniques required are complex and the results of such analysis are frequently compromised by the lack of data points or samples. For initial sampling programmes or preliminary site assessments it is considered that the application of professional judgement by suitably qualified personnel is a more cost-effective approach to the data assessment task. Simple statistical techniques are not appropriate in such circumstances and care should be exercised when comparing reduced data (e.g. sample means) from targeted sampling programmes to the proposed guideline values. Statistical design and analysis of more detailed sampling programmes is, however, a useful tool and should be applied where appropriate, drawing on professionals experienced in the application of statistics to environmental sampling programmes.

A detailed description of applicable statistical guidelines is beyond the scope of these guidelines. For information of the application of statistical techniques, reference should be made to one of the texts on this subject, for example, Gilbert R.L. (1987) "Statistical Methods for Environmental Pollution Monitoring" Van Nostrand Reinhold, New York.

Particular care is required to ensure that compositing or averaging as part of a statistical evaluation does not obscure the presence of a significant hot spot.

2.11 REFERENCES

CCME (1991) "Guidance Manual on Sampling, Analysis and Data Management for Contaminated Sites. Volume 1: Main Report". Canadian Council of Ministers of the Environment, Report CCME EPC-NCS62E, Winnipeg, December 1993.

Gilbert, R.L. (1987) "Statistical Methods for Environmental Pollution Monitoring" Van Nostrand Reinhold, New York.

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Standards Australia (1994) "Draft Australian Standard, Analysis of Soil, Part 1: The sampling of potentially contaminated soil." Cl5/28/1/94-27.

USEPA (1991) "Description and Sampling of Contaminated Soils: A Field Pocket Guide" EPA/625/12-91/002.

USEPA (1991b) "Risk Assessment Guidance for Superfund, Volume 1, HHEM, Supplemental Guidance, Standard Default Exposure Factors".

2.12 FURTHER READING

Guidance Manual on Sampling, Analysis and Data Management for Contaminated Sites. Volume 1: Main Report. Canadian Council of Ministers of the Environment, Report CCME EPC-NCS62E, Winnipeg, December 1993.

Chapter 1 – Quality Control, Revision 0 (1986) and Revision 1 (November 1990) and Chapter 9 – Sampling Plan, Revision 0 (September 1986) in Test Methods for Evaluation of Solid Wastes, United States Environmental Protection Agency Office of Solid Waste and Emergency Response, SW-846, Washington DC, November 1986.

Chapter 9 – Statistical Methods in Environmental Sampling, Chapter 10 – Soil Sampling Quality Assurance and the Importance of an Exploratory Study, Chapter 11 – Quality Assurance for a Measurement Programme in Environmental Sampling for Hazardous Wastes, edited by G E Schweitzer and J A Santolucito, American Chemical Society, Washington DC, 1984.

Guidelines for the Sampling and Analysis of Contaminated Soils, Australia and New Zealand Environment and Conservation Council, Melbourne, December 1993, draft for comment.

"Identification and Assessment of Contaminated Sites; Improving Site History Appraisal" South Australian Health Commission, 1994.

"Description and Sampling of Contaminated Soils : A Field Pocket Guide", USEPA November, 1991. EPA/625/12-91/002.

APPENDIX A
ELEMENTS OF A QUALITY ASSURANCE PLAN

Overall Project Management

- Project description
- Project organisation and designated responsibilities
- Quality assurance objectives for the experimental data in terms of precision, accuracy, completeness, ruggedness and comparability
- Experimental design and analytical procedures
- Ensuring on-going quality assurance reports to management
- Corrective actions
- Defining statistical techniques for assessing the experimental data.

Field Sampling

- Sampling network design
- Selection of specific sampling sites
- Sampling methodology – detailing procedures to be used in the field
- Sampling devices, storage containers and preservatives
- Sample custody, transportation, preservation and storage
- Replicate sampling
- Documentation needed
- Special operating conditions (e.g. heat, light, reactivity etc.)
- Providing information on health and safety practices in sampling and field testing operations
- Providing accepted procedures designed to control and define errors associated with field measurements.

Laboratory Analysis

- Sample custody
- Sample storage
- Instrument selection and use
- Analytical methodology, and standard operating procedures
- Calibration procedures and frequency

- Reference standards and quality control standards
- Internal quality control checks and frequency
- Replicate analyses
- Blank and spiked samples
- Intra- and inter-laboratory QC procedures
- Specific routine measures to be used to assess data quality
- Data reduction, validation, verification and reporting.

APPENDIX B
DEVELOPMENT OF DATA QUALITY OBJECTIVES

Example of the Process of Developing Data Quality Objectives

- *State the problem to be resolved*
  For example, to determine whether there is the potential for a significant adverse effect on human health or the environment associated with soil groundwater contamination at a timber treatment site.

- *Identify the Decisions that Need to be Made*
  For example, does the site pose an immediate risk to human health or the environment? Is there a requirement for immediate remedial action? Is there potential for an adverse effect on human health or the environment in the longer term? Is there need for further, more detailed, investigation to define the extent of contamination, the current impact on human health and the environment and the specific requirements for any remedial action in the longer term?

- *Identify the Inputs to the Decisions*
  For example, the contaminants that may be present at the site may be at concentrations that are of concern in relation to guideline levels; the concentration of contaminants in soil, groundwater, surface water, dust that may have accumulated on surfaces of structures, and in the air; the effects the contaminants may have on human health and the environment, and the concentration in each of the media at which those contaminants have the potential to have a significant impact on human health and the environment; the level of protection required for human health and the environment, i.e. is it a pristine ecosystem or an urban environment.

- *Narrow the Boundaries of the Study*
  For example, to undertake a sampling programme targeted toward identifying contaminant concentrations in the areas most likely to be contaminated, in order to provide a cost-effective assessment of whether there is the potential for a significant adverse effect on human health or the environment.

- *Develop a Decision Rule*
  For example, if the identified concentrations of contaminants in environmental media exceed the guideline values nominated in the Health and Environmental Guidelines for Selected Timber Treatment Chemicals, then further more detailed investigation to determine the extent of contamination is required.

- *Develop Uncertainty Constraints*
  For example, that the Relative Percent Differences (RPD) shall be less than 30% for the results of QA/QC check analyses undertaken by an independent laboratory on duplicate samples; that the sampling programme will be designed such that there is a high level of confidence (notionally 95%) a significant area of potential contamination (say greater than 10 sq.m) would be sampled as part of the sampling programme, giving consideration to the quality of the site history and other background information (such confidence would be measured, in effect, by the independent review of the plan based on professional judgement of an experienced, senior professional in the site contamination area).

- *Optimise Design for Obtaining Data*
  For example, review the sampling plan to ensure all areas of significant potential contamination have been targeted, and that the sampling within an area of potential contamination is such that the level of uncertainty about whether an area of significant contamination may be missed is consistent with the above uncertainty constraint.

- *Example Data Quality Objectives*
  Example DQOs for a timber treatment site assessment are presented as follows:
  - That the investigation shall be sufficient to determine whether there is the potential for a significant adverse effect on human health or the environment.
  - That the data shall at least be representative of the higher contaminant concentrations that are likely to be encountered at the site, in order to determine whether a further detailed evaluation of the extent of contamination is required. (On this basis a targeted cost-effective sampling programme may be used to achieve this objective.)
  - That the level of confidence that a significant area of contamination shall be sampled shall be notionally greater than 90%.
  - If a contaminant concentration in a sample is reported as not detectable, the confidence that the actual concentration is less than one fifth the relevant acceptance criteria shall be greater than 90%.
  - That the reported concentration in a sample shall be representative (e.g. within +/- 50%) of the actual concentration in the media in-situ at the point of sampling (this can be notional only as it cannot be measured).
  - The RPD of duplicate samples analysed by independent laboratories shall be less than 30%.

APPENDIX C
SAMPLING PLAN AND PROTOCOL CHECKLISTS

Table C1
Sampling Plan Checklist[1]

What are your data quality objectives (DQOs)?

- What will you do if your DQOs are not met (i.e. resample or revise DQOs)?

Do programme objectives need exploratory, monitoring, or both sampling types?

Have arrangements been made to obtain samples from the sites?

- Have alternative plans been prepared in case not all sites can be sampled?

Is specialised sampling equipment needed and/or available?

Are samplers experienced in the type of sampling required available?

Have all analytes been listed?

- Has the level of detection (LOD) for each been specified?
- Have methods been specified for each analyte?
- What sample sizes are needed based on the method and desired LOD?

List specific good laboratory practice and federal, provincial or method QA/QC protocols required.

- Are there percentages or required numbers and types of QC samples?
- Are there specific instrument tuning or other special requirements?

What type of sampling approach will be used?

- Random, systematic, judgemental, or combinations of these?
- Will the type of sampling meet your DQOs?

What type of data analysis methods will be used?

- Geostatistical, control charts, hypothesis testing, etc.?
- Will the data analysis methods meet your DQOs?
- Is the sampling approach compatible with data analysis methods?

[1] Adapted from CCME Guidance Manual, Vol 1. – Sampling, Analysis, Data Management

How many samples are needed?

- How many sample sites are there?
- How many methods were specified?
- How many test samples are needed for each method?
- How many control site samples are needed?
- What types of QC samples are needed?
- Will the QC sample types meet your DQOs?
- How many of each type of QC samples are needed?
- Are these QC samples sufficient to meet your DQOs?
- How many exploratory samples are needed?
- How many supplementary samples will be taken?

Number of samples = Test + control + QC + Exploratory + Supplementary

- Test samples = Methods x Sample sites x Samples per site
- Control samples = Methods x Sample sites x Samples per site
- QC samples = Methods x Type of QC sample x % Needed to meet DQOs
- Exploratory samples = (Test samples + Control samples) x 5 to 15%
- Supplementary samples = (Test samples + Control samples) x 5 to 15%

Table C2
Sample Protocols Checklist[(1)]

What observations at sampling sites are to be recorded?

Has information concerning DQOs, analytical methods, LODs, etc. been included?

Have instructions for modifying protocols in case of problems been specified?

Has a list of all sampling equipment been prepared?

- Does it include all sampling devices?
- Does it include all sampling containers?
- Are the container compositions consistent with analytes?
- Are the container sizes consistent with the amount of samples needed?
- Does it include all preservation materials/chemicals?
- Does it include materials for cleaning the equipment?
- Does it include labels, tape, waterproof pens, and packaging materials?

- Does it include chain of custody forms and sample seals?
- Does it include chemical protective clothing or other safety equipment?

Are there instructions for cleaning equipment before and after sampling?

- Are instructions for equipment calibration and/or use included?
- Are instructions for cleaning or handling sample containers included?

Have instructions for each type of sample collection been prepared?

- Are numbers of samples and sample sizes designated for each type?
- Are any special sampling times or conditions needed?
- Are numbers, types, and sizes of all QC samples included?
- Are numbers, types, and sizes of exploratory and supplementary samples included?
- Are instructions for compositing samples needed?
- Are instructions for field preparations or measurements included?

Have instructions for completing sample labels been included?

- Do they include maximum holding times of samples?

Have instructions for packaging, transporting, and storing been included?

Have instructions for chain-of-custody procedures been included?

Have safety plans been included?

Table C3
Examples of Nonmeasurable Sources of Error(1)

- Biased sampling
- Sampling the wrong area
- Sampling the wrong matrix
- Switching samples prior to labelling
- Mislabelling sample containers
- Incorrectly preserving the sample
- Incorrectly aliquoting or weighing samples
- Incorrectly diluting or concentrating samples
- Incorrectly documenting any procedure
- Not recognising matrix-specific interferences
- Using the wrong method for analysis

APPENDIX D
SAMPLE CONTAINERS, VOLUMES AND HOLDING TIMES

Table D1
Sample Containers and Preservation for Constituents in Water

Parameter	Container	Minimum Volume (mL)	Preservative (if required)	Recommended Maximum Storage Time
Boron	P	100	Acidification permitted but not required	28 days
Metals – general	P	500	Add HNO_3 to pH<2	6 months
Chromium (VI)	P	250	4°C	24 hours
Phenols	Brown glass	500	4°C(1)	Extract within 7 days; analyse extract within 40 days of sample collection
Dioxins and Furans(2)	G	2000	4°C	Extract within 30 days; analyse extract within 45 days of sample collection

Abbreviations: P = Polyethylene
G = Glass

Notes: (1) In the presence of residual chlorine preserve with 0.08% $Na_2S_2O_3$
(2) Store samples in the dark.

Table D2
Sample Containers and Preservation for Constituents in Soil and other Solid Matrices[3]

Parameter	Container	Minimum Weight[2]	Preservative (if required)	Recommended Maximum Storage Time
Boron	P	100 g	–	28 days
Metals – general	P or G	250 g	–	6 months
Chromium (VI)	P or G	100 g	4°C	48 hours
Phenols	G[1]	250 g	4°C	Extract within 14 days; analyse extract within 40 days of sample collection
Dioxins and Furans[4]	G[1]	250 g	4°C	Extract within 30 days; analyse extract within 45 days of sample collection

Abbreviations: P = Polyethylene
G = Glass

Notes:
(1) Teflon or solvent washed aluminium lined cap.
(2) Where a sample is to be analysed for several components, 250g is usually sufficient.
(3) Typical only, depends on the requirements of the specific laboratory.
(4) Store samples in the dark.

APPENDIX E
REQUIREMENTS FOR PRESERVATION AND FILTRATION OF GROUNDWATER SAMPLES

The primary objective of any groundwater or surface water sampling programme is to obtain sample(s) that are representative of conditions within the aquifer or surface water body. Secondly, it is important to deliver the sample(s) to the laboratory in unchanged condition (as far as practical). Some relevant considerations are as follows:

- Whilst it is the objective of a groundwater monitoring programme to recover relatively "clear" samples, occasionally this may not be possible from a particular monitoring bore. Many monitoring bores that initially produce turbid samples can produce relatively "clear" samples following an extended period of development. However, the practicality of obtaining a "clear" sample is dependent on the nature of the aquifer. For some aquifers, particularly low/yielding systems, it is not possible within the time constraints of most site investigations, to obtain "clear" samples. In most cases turbidity within the sample is derived from fine particulate in the aquifer material or overlying soil disturbed in the process of drilling or bore construction. If a turbid sample is analysed without first removing the particulate material, (e.g. by filtration), then potentially, an artificially high result may be obtained. For most parameters, however, a very turbid sample (e.g. 200 mg/L TSS) is required to give a significant increase in the reported contaminant concentration.
- Sampling and/or filtration of a sample can induce changes in the water chemistry of the sample. In particular aeration of groundwater samples, which can occur during sampling, can lead to the precipitation of iron oxides. The iron oxide precipitate has the potential to adsorb other heavy metals, stripping metallic contaminants from solution. This can result in artificially low metallic concentrations if the sample is filtered before analysis. The formation of iron oxide precipitates is relatively slow under slightly acid conditions (e.g. pH 5-6) although it can occur within a short period of time (3 to 4 minutes) at a pH of 7 to 8. The preservation of samples by acidification avoids the formation of iron oxide precipitates and also provides conditions that limit the adsorption of metals on particulates that may be present in the sample.
- If a groundwater sample is preserved without filtration, and the sample is relatively turbid, then the resulting acid conditions can lead to the dissolution/desorption of contaminants associated with the particles. Such dissolution/desorption may give rise to an artificially high sample concentration. **If a sample is filtered prior to preservation (i.e. in the field) it must be completed quickly so as to avoid the formation of iron oxide precipitates.**
- Filtering of groundwater samples in the field can be a slow and difficult process, potentially allowing iron oxides to precipitate, depending on the nature of the sample and the equipment available. Further, field filtering of groundwater samples may introduce a source of cross-contamination, although the potential for this is reduced by the use of disposable filtration equipment.

- Metal contaminants in water samples may adsorb onto the wall of glass sample containers, potentially reducing the measured concentration. Acid preservation of water samples avoids such adsorption, and acid pre-rinsing of water sample containers may also reduce the potential adsorption of metal species.

An important consideration in the discussion of filtration practices for water samples is the use of the water resource. In most cases, groundwater or surface water is used unfiltered (with the exception of filtered potable water supplies) and therefore it is important that sampling practices reflect contaminant concentrations in the water body prior to treatment. With groundwater it is considered that if particulates are present in a sample at significant concentrations, the particulates are most often an artefact of sampling rather than being typical of conditions within the aquifer. **On this basis it is important to remove particulate from turbid groundwater samples.**

Based on the considerations outlined above the recommended sampling, filtration and preservation practices for water samples are presented in Table E1.

It is noted that an alternative to filtration and preservation of groundwater samples in the field is to recover the sample without aeration and then to forward the sample to the laboratory without preservation. The sample may then be filtered under laboratory conditions or the clear supernatant may be decanted after the sample has been allowed to stand for a period of time. A sample may be recovered without aeration by placing the outlet tube from an appropriate sampling pump into the sample container (without preservation) and allowing the groundwater to overflow the container for several minutes (several sample container volume changes) before capping the sample container. The sample may then be forwarded to the laboratory for analysis. Sampling without aeration avoids the formation of iron oxides.

It is to be stressed that whilst occasionally it is necessary to filter or otherwise remove particulates from groundwater samples, the objective is to construct and develop groundwater monitoring bores in such a way that "clear" samples are produced.

The above discussion refers principally to groundwater and to metal contaminants given the requirement for preservation of samples to be analysed for heavy metals. **Samples to be analysed for organic contaminants should not be filtered.** Similarly surface water samples should not be filtered given particulates present in surface water samples are likely to be representative of those in the surface water body, rather than being an artefact of sampling.

Table E1
Summary of Filtration and Preservation Requirements
for Groundwater Samples to be Analysed for Heavy Metals [2]

Sample Type	Filter		Preserve[1]	Comment
	Field	Laboratory		
Surface Water	No	No	Yes	Possibly slightly conservative approach
Groundwater				
– Clear Sample	No	No	Yes	Normal sampling
	or No	No	No	Sample without aeration
– Turbid Sample	Yes (quickly)	No	Yes	Normal sampling
	or No	Yes or decant clear supernatant	No	Sample without aeration

Notes: (1) Acidify with concentrated nitric acid to pH = 2
(2) Does not include hexavalent chromium. Samples to be analysed for hexavalent chromium should be filtered and preserved in the field (refer Section 4.7.6).

APPENDIX F
EXAMPLE TABLE OF CONTENTS FOR ASSESSMENT REPORT

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CHAPTER 3
FIELD SAMPLING PROCEDURES AND QUALITY ASSURANCE PLAN

TABLE OF CONTENTS

APPENDICES

3. FIELD SAMPLING PROCEDURES AND QUALITY ASSURANCE PLAN

3.1 INTRODUCTION

3.1.1 Scope

This chapter seeks to provide clear guidance on the level of detail and care that is required for the acquisition of samples during the environmental sampling programme at a timber treatment site. It is acknowledged that site-specific circumstances may require variation from the sampling protocols outlined here or even the adoption of different sampling techniques. In such cases it is important that variation in the application of these procedures be documented and carefully reviewed by appropriately qualified and experienced professionals.

The first part of the chapter involves a discussion of some aspects of quality assurance practice with regard to sampling, including the types and frequency of blank samples that may need to be acquired. Standard requirements on the keeping of field records and the cleaning of sampling apparatus in the field are also addressed.

Detailed sampling procedures for soil, groundwater, surface water, drain and sediment samples are provided in the second part of the chapter, accompanied by a discussion of the philosophy behind, and the information that is sought from, each sample type.

3.1.2 Chapter Summary

Guidance is given on the level of detail and care required to plan and carry out a programme of environmental sampling. QA/QC recommendations are provided on:

- sample blanks;
- field notes and record keeping;
- the cleaning of sampling apparatus and the use of equipment;
- procedures for obtaining samples of soil, surface water, and groundwater (including the installation and maintenance of groundwater bores).

3.1.3 Documentation

The following documentation should be prepared prior to initiating the field investigations:

- *Work Plan or Site Sampling Plan*
  Used to define the exact work requirements for a given site, including sample locations, depths, analytes, etc. Also used to document variations from the standard quality assurance procedures.
- *Health and Safety Plan*
  Used to inform workers of potential physical and chemical hazards, health and safety responsibilities, normal work precautions, monitoring requirements and action plans. An example table of contents for a Health and Safety Plan is included as Appendix A.

3.2 THE USE OF BLANK SAMPLES AND DUPLICATE SAMPLES AS QUALITY ASSURANCE AND QUALITY CONTROL MEASURES

The two data quality indicators (see Section 2.2.4) most often used to assess measurement quality objectives in field sampling are bias and precision.

Bias is defined as a systematic deviation (error) in data. Precision is defined as random variation in data. Bias can be assessed by using a variety of blank sample types. They are discussed in Section 3.2.1. Precision is typically estimated using the practice of duplicate sampling, discussed in Section 3.2.3.

3.2.1 Blank Samples Used to Estimate Sampling Bias

Various types of blank samples can be used to assess the following sources of bias:

- the possibility that extraneous material has been introduced to the samples;
- whether the site of interest is truly different from surrounding sites;
- whether the sample matrix affects the sampling and analytical process.

Field blanks are samples of analyte-free media similar to the sample matrix. They are transferred from one vessel to another or exposed to the sampling environment at the sampling site. They measure incidental or accidental sample contamination during the whole sampling and analytical process (sampling, transport, sample preparation and analysis).

Equipment blanks (or rinsate blanks) are samples of analyte-free media (usually high-purity distilled water collected in a suitable container) that have been used to rinse the sampling equipment. They document adequate decontamination of the sampling equipment after its use. These blanks are collected after equipment decontamination and prior to re-sampling.

Material blanks are samples of construction materials such as those used in groundwater wells. They document the potential contamination of samples from use of these materials.

Trip blanks (or transport blanks) are test samples of analyte-free media taken from the laboratory to the sampling site and returned to the laboratory unopened. They are used to measure cross-contamination from the container and preservative during transport, field handling and storage.

Background samples (or matrix blanks or field control samples) are samples of the media similar to the test sample matrix (dust, soil, surface water, etc.) and are taken near to the time and place where the analytes of interest may exist at background levels. The background sample measures the background presence of analytes of interest. Background samples assist in demonstrating whether the site of interest is contaminated or whether the elevated concentrations reported are naturally occurring.

Background samples can basically be taken from two different sorts of sites designated as “local control sites” and “area control sites”.

Local control sites are usually adjacent or very near to the test sample sites. The following principles apply to their use:

- Local control sites should be upwind or upstream of the sampling site.
- When possible, local control site samples should be taken first to avoid contamination from the sample site.
- Travel between local control sites and sampling areas should be minimised because of potential contamination caused by people, equipment and/or vehicles.

Area control sites are in the same area, e.g. city or district, as the sampling site, but are not adjacent to it. They are chosen where a suitable local control site cannot be found. All possible efforts should be made to make the sites identical except for the presence of the species of interest at the site under investigation. The principles applying to local control sites are also relevant for area control sites.

3.2.2 Number and Frequency of Blank Samples

It is prudent practice to acquire a range of the blank sample types described above.

Analysis costs are often the driving factor in determining the number of blank samples that are actually analysed from the pool of those collected. Where such costs are significant it may be possible to select an approach which minimises the number of blank samples that require analysis. For instance, if the field blanks show no sign of contamination, then any trip blanks can be discarded or stored as necessary. Similarly if the primary samples show analyte levels below the limit of detection or below levels considered significant, then there is a lesser requirement to run all blank types. This

approach is especially relevant for groundwater samples where there are likely to be a significant number of various blanks.

It is recommended that the following be collected:

- one field blank;
- one equipment blank;
- one trip blank;
- one duplicate sample (see 3.2.3).

per day or per 10 samples (whichever is more frequent) per collection apparatus.

Background samples for every matrix type should be acquired during the sampling exercise.

For groundwater samples the following additional blank samples are suggested:

- one standpipe material blank per batch of standpipe material;
- one filter pack (sand or gravel) material blank per batch of standpipe material;
- one drilling equipment blank per day;
- one sampling (e.g. pump, bailer, etc.) equipment blank per day or every 10 wells (whichever is the more frequent).

3.2.3 Duplicate Sampling to Estimate Precision

Duplicate samples are independent samples which are collected as close as possible to the same point in space and time. They are two separate samples taken from the same source, stored in separate containers and analysed independently. These duplicates are useful in documenting the precision of the sampling process.

3.3 RECORD KEEPING

A field log book will be maintained by each investigation work group. The log book will be used to record general progress, any deviation from the QA and Health and Safety Plans, changed conditions, any health and safety incidents and any other notable observations.

- **Sampling locations** will be located with reference to the site plan and by measuring distances from known features on the site plan. All sampling locations will be referenced by using a unique numbering system, for example, a location number and one of the following prefixes:

  HA Hand Auger
  BH Borehole
  TP Backhoe Test Pit
  GW Groundwater Monitoring Bore

A record of all sampling locations shall be kept. Test pits should be photographed with a measuring tape and the test pit number in the photo, where practical.

In addition, groundwater monitoring bores may need to be professionally surveyed and marked on a base map using an appropriate co-ordinate system, particularly where bore locations cannot be reasonably defined by reference to site features.

- **Sub-surface conditions** encountered at every borehole and auger hole will be logged on standard field log sheets. An example of the field log sheet is included in Appendix B.
- **All depths** will be **referenced** to the ground surface. All depths shall be recorded in metric units (metres). The elevation of each sample location, relative to an appropriate height datum, shall be determined by levelling by suitably experienced field personnel.

  A record of all **samples collected** shall be kept by the field supervisor. This record will incorporate the following information:
  - Job Number
  - Client/Job Name
  - Sampling Location Number
  - Sample Number (as defined in work plan. The Sampling Location Number and Sample Number may be the same number).
  - Sampling Depth (where appropriate)
  - Date
  - Initials of Sampling Personnel
- Each **sample** will be **labelled** with the following information, which correlates directly with the record of sampling to be kept by the field supervisor:
  - Job Number
  - Client/Job Name
  - Sampling Location Number
  - Sample Number (as defined in work plan)
  - Sampling Depth
  - Date
  - Duplicate (if the sample is a duplicate sample)
  - Triplicate (if the sample is a triplicate sample)
- **Chain-of-Custody** documentation shall be prepared by the field supervisor prior to delivery of the samples to the laboratory. A copy of a standard Chain-of-Custody form is included in Appendix B. A copy is to be retained by the field supervisor and a copy to be delivered to the laboratory with the samples.

Information to be recorded in the Chain-of-Custody will include:

- Job Number
- Project Name
- Date of Sample Collection
- Chemical Analysis Required
- Preservation requirements and maximum holding times
- Sample Numbers (as defined in work plan)
- Person delivering samples
- Person receiving samples

On submission of the samples to the laboratory, and the signing of relevant sections by the person relinquishing and the person receiving the samples, a copy of the Chain-of-Custody form will be sent to the assessor and the original Chain-of-Custody form will be returned with the certified results sheet.

The Chain-of-Custody documentation may also be used as the record of samples collected outlined above, if it is extended to include the appropriate information.

The field supervisor shall keep a record of any **change in conditions** encountered during field work, including unusual or unexpected sub-surface conditions, the presence of perched groundwater, odours or significant PID readings. This information should be recorded on the log sheets where relevant.

- Deviations from the documented health and safety, quality assurance and work plans should be noted by the field supervisor.

Additional specific record-keeping requirements are outlined in the following sections.

3.4 FIELD CLEANING PROCEDURES

An area will be established on-site where all sampling equipment may be cleaned without risk of contaminating areas to be sampled, or spreading contamination around or off the site. All field tools which are used for sampling and which come into direct contact with the material to be sampled must be cleaned and wrapped as described in this section.

The following **field cleaning procedure** will be utilised for cleaning field sampling equipment (e.g. hand augers, trowels, split barrel samplers, bailers, sampling pumps, etc.):

- Steam clean all field tools that cannot be readily washed using a detergent solution in a trough or similar (i.e. those prior to commencing the field sampling programme and prior to sampling at each location). Note: it is not practical or safe to steam clean small items of equipment using commonly available steam cleaning equipment.

- Wash in laboratory grade phosphate-free detergent.
- Rinse with tap water.
- Rinse with HPLC grade acetone where the sample is to be analysed for dioxins.
- Wipe over with a nanograde hexane soaked pad[1], where the sample is to be analysed for dioxins.
- Rinse with HPLC grade acetone, where the sample is to be analysed for dioxins, PCP (in oil) or PAHs.
- Rinse in high-purity analytical-grade deionised water.
- Sampling tools are to be stored in such a way as to prevent recontamination. Wrap in clean aluminium foil until the next use, where samples are to be analysed for dioxins.

If a drilling rig or backhoe is employed for soil sampling or groundwater bore construction, the drill string or backhoe bucket will be steam cleaned and the sampling equipment, e.g. split barrel sampler, will be subject to the above cleaning procedure. Wastes generated by equipment cleaning may be directed to the site waste treatment and disposal system, or drummed for off-site disposal as appropriate. Where tools such as crowbars and shovels do not come into contact with the material to be sampled, a less rigorous cleaning procedure, such as that used for a backhoe (i.e. steam cleaning), may be used.

Note: that where steam cleaning facilities are not available, suitable equipment may be hired or other rigorous cleaning procedures adopted. Steam cleaner and high pressure hot water washer are synonymous for the purposes of this document.

3.5 TYPICAL SOIL SAMPLING PROCEDURE

3.5.1 Outline of Field Investigation

The field investigations may involve the recovery of soil samples using a hand auger, backhoe, drill rig or other suitable equipment. This should be done in accordance with a documented field sampling plan, and a health and safety plan.

To penetrate the strata to be sampled, hand augering or other appropriate techniques will be used. Drilling or excavation using a backhoe may be necessary should difficult ground conditions be encountered, or excavations to depth be required.

[1] The pad must be of cotton, unbleached paper or cloth to eliminate a source of possible dioxin contamination.

Soil samples obtained may be analysed for a range of chemical parameters including:

- Pentachlorophenol
- Copper
- Chromium
- Arsenic
- Boron
- Dioxins and furans

It is noted that samples may be analysed for a range of other parameters as discussed in Chapter 2.

Many of these compounds, if present in soil samples, may be present in trace quantities which require very sensitive laboratory analytical procedures. Consequently it is important that soil sampling procedures are such that the quality of the samples obtained is assured.

Some samples may eventually be composited for analysis. Composite samples should be prepared from individually collected and documented sub-samples. It is recommended that compositing be performed in the analytical laboratory (refer Sections 2.6.2 and 4.5.3).

3.5.2 Hand Auger Sampling

The following procedures will be used when collecting shallow samples. As indicated in the sampling plan, where samples are collected from several positions within a given test location for later compositing by the laboratory, the same sampling tool and tray can be used, provided all loose dirt is removed from the tools.

The following is an indicative procedure for recovery of soil samples by hand augering.

Note: where the field cleaning procedure does not require wrapping of tools in aluminium foil, ignore such references in the following procedure.

Shallow Samples

- A clean area immediately adjacent to the sample location will be established, using a clean plastic sheet, on which all cleaned, and foil-wrapped equipment may be placed.
- Put on a clean pair of powder-free latex/PVC gloves.
- Unwrap a clean sampling trowel from the aluminium foil. Always rest the trowel on the foil.

- Remove grass etc. from the area to be sampled by hand or with the trowel.
- With the trowel remove soil to a depth of 100 mm from the sampling area and place directly in pre-cleaned glass sample jar.
- Depending on the analytical requirements, it may be appropriate to recover samples in more than one sample container, e.g. recovery of separate samples for PCP and dioxin analyses where these are to be completed by different laboratories.
- Label each sample jar as outlined in Section 3.3. Record the details of the sampling location and other pertinent data. Complete a chain-of-custody form for the samples.
- All samples to be analysed for organic constituents shall be stored at $<4^{o}C$ in a portable ice chest whilst in the field or in transit.
- If no further samples are to be taken at the location, then replace any surface soil removed from the hole.

Deep Samples

- Change to a clean pair of latex/PVC gloves.
- Unwrap a new sampling trowel from the aluminium foil. Always rest the tool on the foil, not on the plastic, whilst sampling.
- Unwrap a new sampling tray from the aluminium foil.
- Unwrap a pre-cleaned auger or a pre-cleaned shovel or crowbar from the aluminium foil. Always rest the equipment on the foil, not on the plastic, whilst sampling.
- The deeper samples will be recovered by hand auger, taking care to select material such that the possibility for cross-contamination is minimised. In order to minimise the likelihood of smearing or cross-contamination between sampling depths, the initial sample will be recovered using a sampling spoon or 75 mm diameter auger. The hole will then be advanced using the 75 mm diameter auger before, say, a 62 mm diameter auger is used to recover the second sample. All equipment is cleaned in accordance with Section 3.4 of this plan between each sample point.
- Label each sample jar as outlined in Section 3.3. Record the details of the sampling location and other pertinent data. Complete a chain-of-custody form for the samples.
- Backfill the hole. If the hand auger hole approaches the water table or passes through an aquitard the hole may be sealed (e.g. using bentonite pellets) to minimise contaminant migration.

It is noted that recovery of samples by hand auger is limited by practical considerations to a depth of approximately 2 m, depending on soil type. In addition, the risk of cross-contamination increases with sample depth when using a hand auger and therefore caution should be exercised when selecting this technique for sample recovery.

3.5.3 Boreholes

Boreholes may be drilled to sample soil and/or groundwater where hand auger techniques are not appropriate. The hollow auger drilling technique, with sample recovery using a split barrel sampler, is commonly employed in assessing unconsolidated formations. Alternative drilling techniques include cable tool, mud rotary, air rotary and air hammer.

Techniques that involve the use of drilling fluids, or the introduction of other substances that may result in contamination of the bore should be avoided where possible. If drilling techniques requiring the use of drilling fluids (e.g. water, mud, air) are used then the importance of bore development and stabilisation, prior to sampling, is increased. In addition, it is important to implement measures to reduce the potential for cross-contamination associated with the oil commonly present as a mist in compressed air supplies.

Drilling

- The drill string will be steam-cleaned prior to commencing each borehole.
- All sampling equipment should be cleaned in accordance with the procedures in Section 3.4 prior to commencing the borehole and prior to obtaining each sample.
- Typically, samples of sub-surface material will be recovered from the following depths (although samples may be recovered from other depths as required):
  - 0.5 metres
  - 1.0 metres
  - 2.0 metres or as listed in the sampling schedule
- Every member of the field staff who will come into direct contact with the soil being sampled **must** change to a clean pair of gloves for collecting each sample.
- Samples of sub-surface material will be recovered by driving a Split Barrel Sampler or other similar sampling device into undisturbed material.
- All boreholes will be sealed with cement grout or bentonite at the completion of drilling unless used to establish a groundwater monitoring well.

Sample Collection

- Samples will be recovered from the ground using the techniques specified in the previous sections.
- All equipment used for drilling, augering, digging or extracting samples will be cleaned using the cleaning procedure specified in Section 3.4.
- Field personnel will wear clean PVC/latex gloves whilst handling sampling equipment and carry out sampling.
- Every sample jar will be labelled in the manner outlined in Section 3.3.
- Each sample shall be recorded on the chain-of-custody documentation.
- All samples to be analysed for organic constituents shall be stored at $<4^{o}C$ in a portable ice chest whilst in the field or in transit.

3.5.4 Backhoe Testpits

A backhoe may be used to recover soil samples where ground conditions make the use of a hand auger impractical. The following precautions will apply:

- The backhoe bucket and boom will be steam cleaned prior to each test pit and at the end of each day's work, ensuring residual grease and oil are removed.
- The backhoe will be in good condition and free of oil or hydraulic fluid leaks.
- Following excavation to the target depth, all loose dirt will be removed from the backhoe bucket and a sample representative of the material at the target depth will be recovered using the backhoe. Field staff **must not** enter the test pit greater than say, 1.0 m deep under any circumstances, unless it has been made safe in accordance with relevant occupational health and safety regulations.
- Samples will be recovered at depths as specified in the sampling plan. Additional samples may be recovered at the discretion of the field engineer.
- A sample will be recovered from the backhoe bucket using a cleaned sample spoon or trowel, taking care to select material that has not contacted the sides of the bucket. The sample will be placed in a cleaned glass jar. In some circumstances samples may be recovered directly using a scoop, rather than from the backhoe bucket.
- Field personnel will wear clean PVC/latex gloves whilst handling sampling equipment and whilst carrying out sampling.
- Every sample jar will be labelled in the manner outlined in Section 3.3.
- Chain-of-custody documentation will be completed for each sample.
- All soil samples to be analysed for organic constituents shall be stored at $<4^{o}C$ in a portable ice chest whilst in the field or in transit.

3.6 TYPICAL GROUNDWATER SAMPLING PROCEDURE

3.6.1 Outline of Field Investigations

The field investigations are designed to obtain representative groundwater information of the site in order to:

- define the geologic profile and aquifer characteristics beneath the site;
- assess the current nature and level of soil and groundwater contamination;
- identify the principal sources of contamination;
- estimate the rate and direction of contaminant flow, on and off-site;
- evaluate remediation requirements for the site;
- identify likely zones of discharge.

The primary contaminants of concern have been outlined in Section 3.5.1.

The field investigations may involve the following:

- Installation of groundwater monitoring bores as indicated in the site-specific sampling plan, including one bore suitable for a pump test.
- Recovery of groundwater samples and measurement of the groundwater level and floating hydrocarbon (if present) in all groundwater monitoring bores.
- Rising head permeability testing at each of the groundwater bores.
- Recovery of soil samples from selected depths during drilling (refer soil sampling requirements).

3.6.2 Drilling

The borehole numbering system adopted will conform with that specified in the sampling plan.

Material handling and quality control measures will be directed towards clean drilling conditions and the elimination of down-hole contamination as a result of drilling operations.

Specific measures will include:

- The drilling rig to be used will be in sound working order and free of oil leaks.
- A cleaning pad will be established on the site where the drilling rig and other large equipment can be cleaned without risk of contamination to sampling locations. Power and water will need to be located nearby to enable use of a steam-cleaning unit.

- On arrival at the site the drilling rig will be decontaminated by steam-cleaning. This is to include all drilling equipment which will go into or be used near the borehole. The drilling rig and all drilling equipment will also be cleaned between boreholes.

Logs of the soil encountered will be prepared on standard borehole log sheets. The soil will be logged using the Unified Method of Classification and standard abbreviations will be used (refer attached information sheets). Record of Progress sheets will itemise all activities carried out, and detail of equipment placed into the hole, decontamination procedures and sampling episodes.

Particular note will be taken of the nature of possible soil contamination including an assessment of appearance and odour. Where contamination by volatile organic compounds is suspected, field screening of samples using an organic vapour analyser (e.g. PID) may be warranted. All samples will be ranked using a scale ranging from 0 to 3 taking account of appearance and odour. All information is to be recorded on log sheets.

All drill cuttings are to be placed in sealable containers or a covered waste disposal skip on-site for subsequent disposal.

- A range of drilling techniques may be used to install groundwater monitoring bores; however, preference should be given to techniques that do not rely on the introduction of drilling fluids (including air). Whilst hollow auger drilling techniques are frequently employed, the selection of a technique should be made on the basis of the expected ground conditions and the requirements for bore construction. On those sites covered by concrete paving, drilling will be preceded by concrete coring of a size to accommodate both drilling activities and subsequent borehole completion, including installation of borehead protectors.
- Accumulated drill cuttings will be removed from the borehead area as drilling progresses in order to prevent cuttings falling back into the borehole.
- Note that it is recommended that the background monitoring bore(s) be drilled first where possible.

3.6.3 Standpipe Installation

Records will be kept on the standard record sheets including all procedures adopted, materials used and the respective timing of the various stages of bore construction. Well completion reports may be used containing information on borehole configuration; piezometer configuration (e.g. screen location, casing length, diameter etc.); placement of screen filter pack and borehole seals; and bore development and completion details. All data will be recorded directly in the field and subject to physical measurement.

All materials placed in the hole will be free of any target contaminants listed in the project brief.

Prior to installation, standpipe materials will be subject to high pressure hot water wash, with phosphate-free detergent, followed by a rinse in potable quality water and final rinse with deionised water. Thereafter the standpipe materials will be handled only by field personnel wearing clean PVC/latex gloves.

Conventional solvent glues will not be used. Instead mechanical screw fittings will be used on all casing and screen joints.

The top of the screen generally will be placed between 1.0 m -1.5 m above the water table as logged during drilling or at the discretion of the field engineer/geologist, particularly where the depth to groundwater is less than 2.0 m. The intention is to identify the presence of any floating product and allow for fluctuations of the water table level. Following screen and casing installation, graded sand or gravel, sized to match the aquifer materials, will be placed around the screen and to a height of approximately 200 mm above the uppermost screen slots. The bentonite seal will be placed directly above the filter pack and will extend for a thickness of 1.0 m or more where possible.

The filter material will be pre-washed and screened to eliminate foreign material and should be appropriately graded to the aquifer material wherever possible. Sand or gravel will be brought on-site in bags and transferred directly from bag to hole when running the screen filter.

Holes will be back-filled above the bentonite seals to approximately 0.25 m below ground level, with final completion at the surface comprising a concrete collar seal and steel protective covers to provide security and prevent accidental damage. In most cases these covers will comprise cylindrical steel upstands fitted with lockable lids. Where vehicular traffic poses a problem, the installation will be finished flush with the ground surface using an appropriate protective cover. In this event, a sump will be provided around the top of the casing with sub-surface drainage installed to prevent build up of drainage water around the borehead. Generalised design drawings are included in Appendix C.

All loose material will be removed from the borehead working area prior to piezometer installation so as to avoid accidentally being dislodged into the open hole.

Final levels of both screen filter packs and bentonite seals will be verified by direct measurement using a slim probe lowered down the annular space between borehole wall and casing.

Monitoring bore basin and screens would typically be constructed from PVC pressure pipe of a nominal 50 mm diameter. Screen lengths will be determined on-site after drilling has established preferred screen zones. Typically, slot sizes will be nominal 0.5 mm width with two rows of slots per screen length and average spacing of 1 cm

between slots. Approximately 0.5 m of unslotted casing may be provided below each screen, to act as a sump for collection of any fines that may pass through the screens. Monitoring bores will be terminated with a fitted PVC end cap at the lower end and with a PVC cap at the surface.

The precise diameter, material and configuration of monitoring bores should be determined on a site-specific basis by a qualified professional. The above guidance provides an indication of a typical installation.

3.6.4 Bore Development and Aquifer Testing

Compressed air pumping, mechanical surging or other pumping will be used to achieve development depending on the aquifer characteristics, with gentle surging to promote removal of any residual fines. Development pumping will continue until water clears of residual sediment and yields stabilise. Adequate development will be verified on the basis of stabilisation of basic water chemistry parameters including electrical conductivity and temperature. Records of the above will be maintained.

The selection of an appropriate pumping system for bore development depends on the nature of the aquifer. However, care should be exercised to ensure the aquifer is not aerated. Some alternative pumping systems include compressed air with 'U' tube system to avoid aeration, Waterra pump, bladder pump, purge pump, Grundfoss pump or similar mechanical pumping systems. Pumping systems that avoid aeration of the samples are generally preferred.

On completion of development pumping, water levels will be in a depressed condition in the borehole. The groundwater recovery will be monitored by recording the rate of water level rise on cessation of pumping, and empirical analysis may be used to estimate permeability. Other tests may be necessary to characterise the aquifer, depending on site-specific conditions.

All items inserted into the bore will be decontaminated using high-pressure hot water used in conjunction with phosphate-free detergent, followed by final rise in potable-quality water and distilled water.

Effective construction and completion of the piezometers will be verified on the basis of recorded discharge and subsequent water level recovery data.

Data recording will include:

- Daily Record of Progress sheets, which will include details of all activities carried out, equipment installed, times and durations.
- Pumping Schedule, detailing pump operating periods and measurements or estimates of discharge volumes.

- Water Level Recovery Data, detailing time, elapsed period since pumping ceased and water level. Water levels prior to commencement of pumping will also be recorded.

3.6.5 Groundwater Sampling

Groundwater samples will be collected several days after the development pumping and recovery test phase. The borehole will be purged by removal of at least three bore volumes of water from each bore to remove any stagnant water or water which is not representative of the aquifer, before retaining any samples for analysis. During the purging process checks on temperature, pH and electrical conductivity will be carried out and pumping continued until these parameters stabilise. Records of temperature, pH and electrical conductivity measurements shall be maintained.

Samples will be collected in a stainless steel or teflon downhole bailer, or using an appropriate sampling pump (where disturbance of suspended solids must be minimised) which will be decontaminated between sampling sites by cleaning in accordance with the procedures specified in Section 3.4. Alternatively, a disposable bailer may be used for each sample, provided the bailer material is compatible with the suspected contaminants and is able to withstand the necessary solvent rinses when sampling for dioxins. Care will be taken when sampling to avoid any opportunity for excess aeration of the sample.

At the time of sampling, all samples collected will be transferred to storage at 4°C. Transfer to the analytical laboratory will be completed with 48 hours of sample collection.

Note: analysis for hexavalent chromium must be undertaken within 24 to 48 hours, so faster delivery to the laboratory is required for these samples.

Additional requirements are as follows:

(i) If a bailer is used, the bailer should be lowered gently to avoid disturbance of any sediment that may still be in the bore and to avoid damage to the bailer or the rope. Samples should be recovered from beside the slotted section of the standpipe.

(ii) Prior to commencement of sampling, a clean piece of plastic shall be placed on the ground beside the well. All equipment shall be placed on this sheet when not in use and all cleaning shall be carried out on the plastic sheet. As the bailer is removed from the well, care shall be taken to place the rope on the plastic sheet.

(iii) Water samples will be placed in screw-capped containers which will be supplied by the laboratory. Bottles supplied shall be polythene for metals and inorganics and glass for organics.

(iv) Water samples to be analysed for heavy metals may require filtration on-site to remove particles that could affect the concentration of metals (refer Appendix E

of Chapter 2). Filtering should take place before the water sample is added to the container with the preservative. Care must be exercised to minimise aeration of the sample during filtration. Alternatively, if relatively clear and low-turbidity samples can be collected, then the sample may be recovered without filtration and preservation, provided the sample is recovered without aeration (e.g. place outlet of pump directly into the base of the sample container and fill, allowing to overflow for several volume changes before sealing).

(v) A sample collection record form shall be completed for each sample collected.

3.6.6 Water Level Determination

Following well development, the standing water level shall be measured. Sufficient time will be allowed for stabilisation of water levels following development or other disturbance of the bore. The time required for stabilisation depends on the aquifer characteristics, and may range from minutes to days.

A cleaned dipper will be lowered down the well to ascertain the water level. The depth to the top of floating non-aqueous phase liquid (NAPL) will be determined using either a mechanical or electrical measuring device (e.g. interface probe). The depth to top of groundwater will be measured with a cleaned electrical dipper. The difference between the two is the thickness of floating NAPL. This thickness will be verified by bailing with a transparent bailer.

The cleaning procedure for these instruments shall be to wash copiously with tap water and then rinse with deionised water. If oil or grease is picked up on the bailer then additional washing with phosphate-free detergent will be required. The bailer may be rinsed with acetone to assist in removal of oil or grease, followed by rigorous rinsing with potable, then deionised water. Alternatively, a disposable bailer may be used.

Water levels will be referenced to ground surface and recorded to the nearest centimetre.

3.7 TYPICAL SURFACE WATER AND DRAIN SAMPLING PROCEDURE

3.7.1 Outline of Field Investigations

The field investigations are designed to obtain representative samples of water and/or sediment from site discharges, the appropriate receiving waters after mixing, and from various drains across the site. The primary contaminants of concern with regard to such investigations were outlined in Section 3.5.1.

It is noted that samples may be analysed for a range of other parameters as discussed in Chapter 2.

The field investigations may involve:

- Recovery of grab samples from selected locations in the receiving water body.
- Recovery of grab samples from selected locations within the site drainage system or at the point of discharge from the site.
- Recovery of sediment samples from selected locations within surface water bodies (including drains) in the vicinity of the site.

3.7.2 Stream Sampling

Samples will be recovered from the stream at locations designated in the sampling plan. All equipment used in stream sampling will be cleaned in accordance with the procedures outlined in Section 3.4 prior to the recovery of each sample.

Stream samples will be recovered from below the stream surface in order to prevent accidental sampling of surface slicks. A suitable sampling device, able to recover samples from a designated depth and prevent ingress of surface water, will be employed. Such devices are readily available. If possible, the sample will be taken directly into the sample container prepared by the laboratory.

Sampling should commence at the location furthest downstream, working back upstream in turn.

Care will be taken when sampling to avoid any opportunity for excess aeration of the sample.

All samples to be analysed for organic constituents will be stored at $<4^{o}C$ in a portable ice chest whilst in the field or in transit. Transfer to the analytical laboratory will be completed as soon as practical. Maximum recommended sample holding times are set out in Appendix D of Chapter 2.

Additional requirements are as follows:

(i) The sampling equipment should be lowered gently to avoid disturbance of any sediment.

(ii) Prior to commencement of sampling a clean piece of plastic shall be placed on the ground beside the sampling location. All equipment shall be placed on this sheet when not in use and all cleaning shall be carried out on the plastic sheet.

(iii) Water samples will be placed in screw-capped containers which will be prepared by the laboratory. Polythene bottles should be used for samples to be analysed for metals and inorganic constituents, and glass bottles should be used for samples to be analysed for organic compounds.

(iv) Only those samples which do not have preservatives in the bottles shall be filled to overflowing; those bottles with preservatives should be filled to maximum capacity but not to overflowing.

(v) Sample containers shall be placed in clean polyethylene bags to minimise the potential for cross-contamination.

3.7.3 Drain Sampling

Water samples will be recovered from various drains across the site, as designated in the sampling plan. All equipment used in the sampling of drains will be cleaned in accordance with the procedures outlined in Section 3.4, prior to the recovery of each sample.

Water samples may be recovered from drainage system manholes across the site using a stainless steel sampling container or glass jar. Sampling of the drains is likely to require field personnel to enter manholes in order to recover the samples. The following precautions will be adopted when entering manholes:

- The manhole cover will be removed using appropriate lifting equipment, and allowed to vent for a period of time.
- The atmosphere within the manhole will be monitored for explosive gases (using an explosimeter), oxygen deficiency and other volatile organics (using a photoionisation detector).
- One person will remain on the surface, at the manhole opening, as an observer.

During sampling the temperature and electrical conductivity of each sample will be recorded.

Care will be taken when sampling to avoid any opportunity for excess aeration of the sample.

All samples to be analysed for organic constituents will be stored at $<4^{o}C$ in a portable ice chest. Transfer to the analytical laboratory will be completed as soon as practical.

Additional requirements are as follows:

(i) The sampling equipment should be lowered gently to avoid disturbance of any sediment.

(ii) Prior to commencement of sampling a clean piece of plastic shall be placed on the ground beside the sampling location. All equipment shall be placed on this sheet when not in use and all cleaning shall be carried out on the plastic sheet.

(iii) Water samples will be placed in screw-capped containers which will be prepared by the laboratory. Bottles supplied shall be polythene for metals and inorganics and glass for organics.

(iv) Containers shall be filled to over-flowing except the metals container which shall have preservatives already added.

(v) A sample collection record form shall be completed for each sample collected.

3.7.4 Sediment Sampling

Sediment samples will be recovered from selected locations within streams, drains and other surface water bodies in the vicinity of the site, as designated in the sampling plan. Samples will usually be recovered from locations where sediment, associated with run-off from the site, is likely to collect, i.e. areas of lower flow velocity adjacent to, or downstream from, the site.

All equipment to be used in the recovery of sediment samples should be cleaned prior to the recovery of each sample, in accordance with the procedures outlined in Section 3.4. Sediment samples may be recovered using an appropriate scoop or other sampling tool in the case of shallow water bodies, or using purpose-designed sediment core sampling equipment for recovery of samples from deeper water bodies and where a vertical profile of the sediment is required.

Sediment samples shall be placed in clean glass sample jars, as for soil samples, or, where samples are recovered using core sampling equipment the sample may be retained in the coring equipment (e.g. plastic or aluminium tube), sealed and transferred to the laboratory for analysis. All samples to be analysed for organic constituents should be stored at <4°C, and transfer of the samples to the laboratory shall be completed as soon as possible. The maximum recommended sampling times are outlined in Appendix D of Chapter 2.

3.8 TYPICAL BUILDING DUST SAMPLING PROCEDURE

3.8.1 Outline of Field Investigations

The field investigations are designed to obtain representative samples of dust from buildings across the site, and on adjacent land, with the objective of assessing the risk to site workers and residents. The sampling should be directed to characterising the concentration of various contaminants in dust within those buildings regularly occupied, either by site workers or residents e.g. main work areas, tea rooms and residences. Two types of dust samples may be recovered, as follows:

- dust samples recovered from surfaces within the living areas of residences, or main work areas and surfaces of timber processing buildings;
- dust samples recovered from surfaces where dusts may accumulate without disturbance, e.g. ledges, the roof space of residences.

The primary contaminants of concern with regard to such investigations are as outlined in Section 3.5.1. Where PCP has been used on-site, particular attention should be focussed on characterising the dioxin concentrations in the dust. Samples may also be analysed for a range of other parameters as discussed in Chapter 2.

The field investigations involve the recovery of samples of dust from selected locations, as defined in the site-specific sampling plan. Frequently, several samples will be recovered from a single building and composited for analysis.

3.8.2 Dust Sampling

Building dust samples will be recovered at locations designated in the sampling plan. All equipment used in dust sampling will be cleaned in accordance with the procedures outlined in Section 3.4, prior to the recovery of each sample.

Due to the nature of dust sampling, the exact sampling methods must be determined on a site-specific basis, however, some examples are presented as follows:

- where dust is to be recovered from ledges or other locations where dust may accumulate without disturbance, samples may be recovered by scraping or scooping the dust into a screw-capped glass sample jar, using a stainless steel implement;
- where dust is to be collected from living areas of residences or similarly disturbed work areas, a vacuum sampling device may be used. The collected dust may be transferred to a screw-capped glass sample jar.

Dust samples from more than one location may need to be composited in order to obtain sufficient sample for analysis.

A rigorous sampling protocol should be developed by the site assessor, in accordance with the QA/QC framework outlined in Chapter 2, prior to beginning sampling.

Each sample should be labelled in accordance with the procedures outlined in Section 3.3. Details of each sampling location and other pertinent observations should be recorded, and a chain-of-custody form should be completed for all samples.

All samples to be analysed for organic constituents will be stored at $<4^{o}C$ in a portable ice chest in the field and in transit to the laboratory. Maximum recommended sample holding times are set out in Appendix D of Chapter 2.

3.9 DISPOSAL OF WASTES

A range of wastes may be generated as part of any sampling programme. Examples of such wastes include:

- washwater and solid residues from cleaning procedures;
- waste foil, cloth pads, plastic sheeting, etc. from cleaning and wrapping tools;
- excess spoil from sampling locations; and
- groundwater from bore development and purging.

Each of these wastes may be contaminated and should be packaged and disposed of in accordance with the relevant health and safety, dangerous good and landfill disposal regulations.

Contaminated wastewaters may be disposed of via the site wastewater treatment system, if available, subject to the necessary approvals. Planning for a field sampling programme should include planning for the disposal of waste materials.

APPENDIX A
SITE SPECIFIC HEALTH AND SAFETY PLAN FOR INVESTIGATION OF SUBSURFACE CONTAMINATION AT TIMBER PROCESSING SITES

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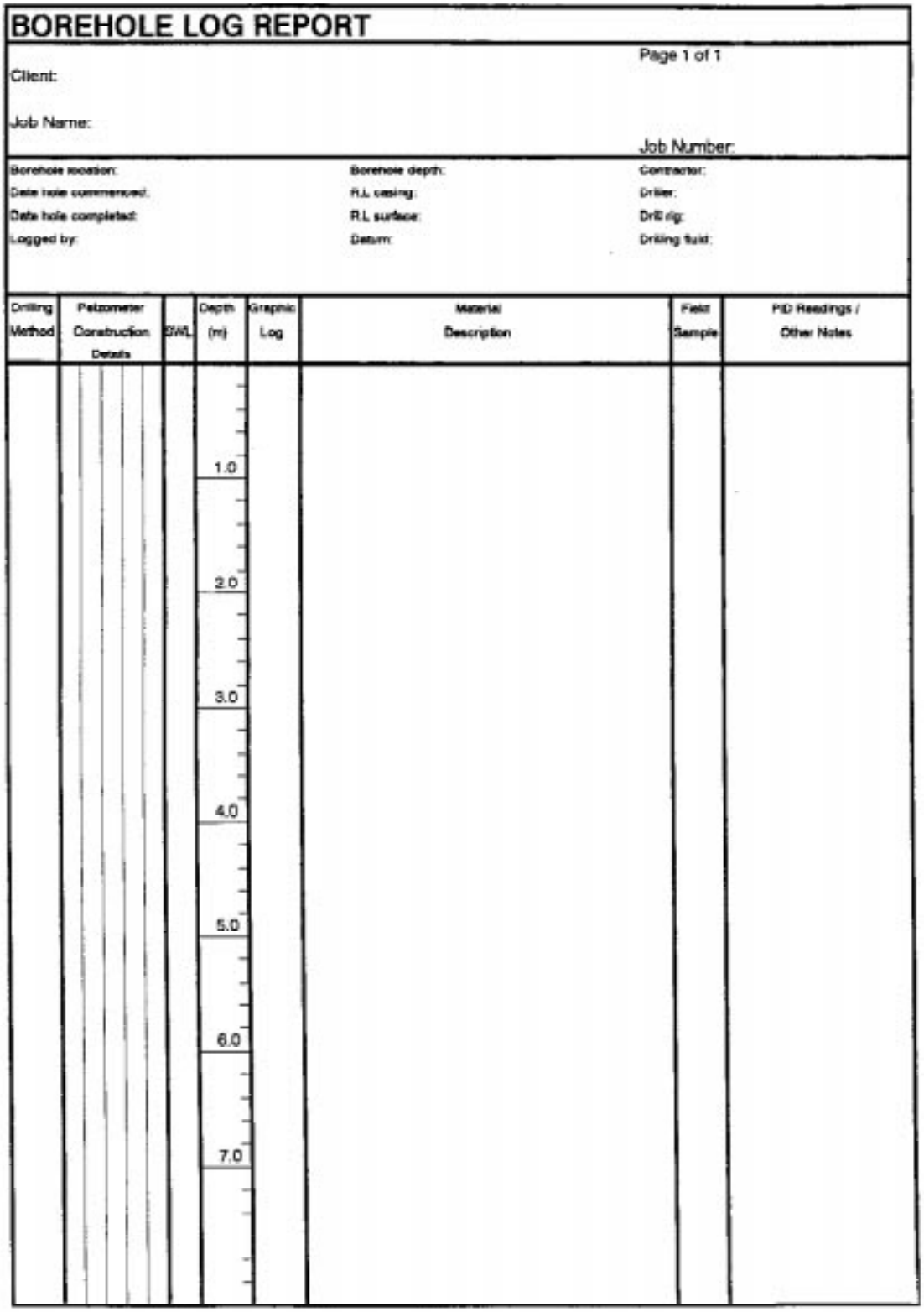
EXAMPLE FIELD RECORDS



BOREHOLE LOG REPORT

Page 1 of 1

Client:

Job Name:

Job Number:

Borehole location:	Borehole depth:	Contractor:
Date hole commenced:	R.L casing:	Driller:
Date hole completed:	R.L surface:	Drill rig:
Logged by:	Datum:	Drilling fluid:

Drilling Method	Piezometer Construction Details	SWL	Depth (m)	Graphic Log	Material Description	Field Sample	PID Readings / Other Notes
			1.0				
			2.0				
			3.0				
			4.0				
			5.0				
			6.0				
			7.0				

TESTPIT LOG REPORT

Client:

Job Name:

Job Number:

Testpit location:
Date pit commenced:
Date pit completed:
Logged by:
Checked by:

Contractor:
Excavator:
Bucket:
Testpit depth:

Depth (m)	SWL	Graphic Log	Material Description	Field Sample	Field Rank (0 – 3)	PID Readings / Other Notes
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1.5						
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2.5						
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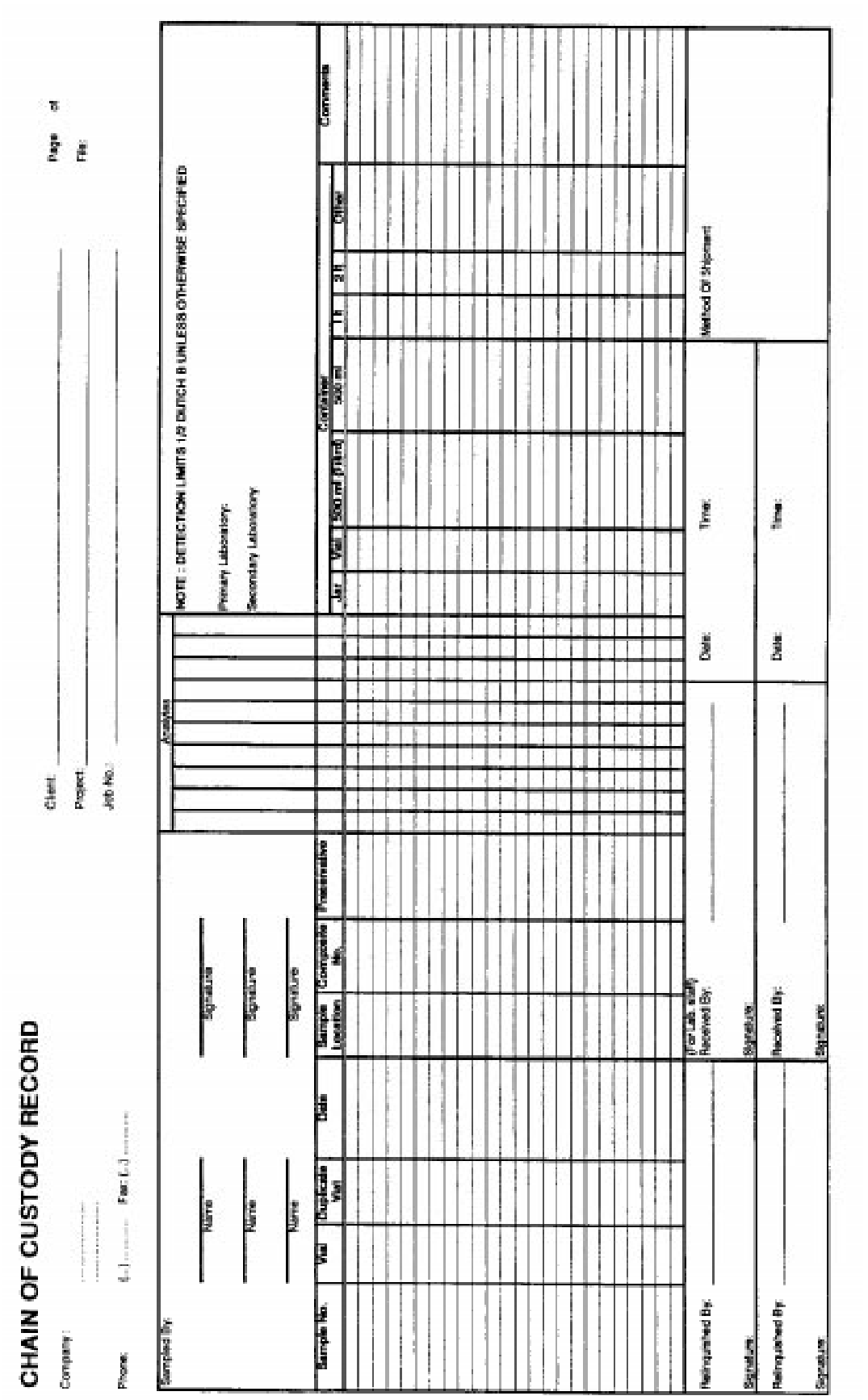
4.0						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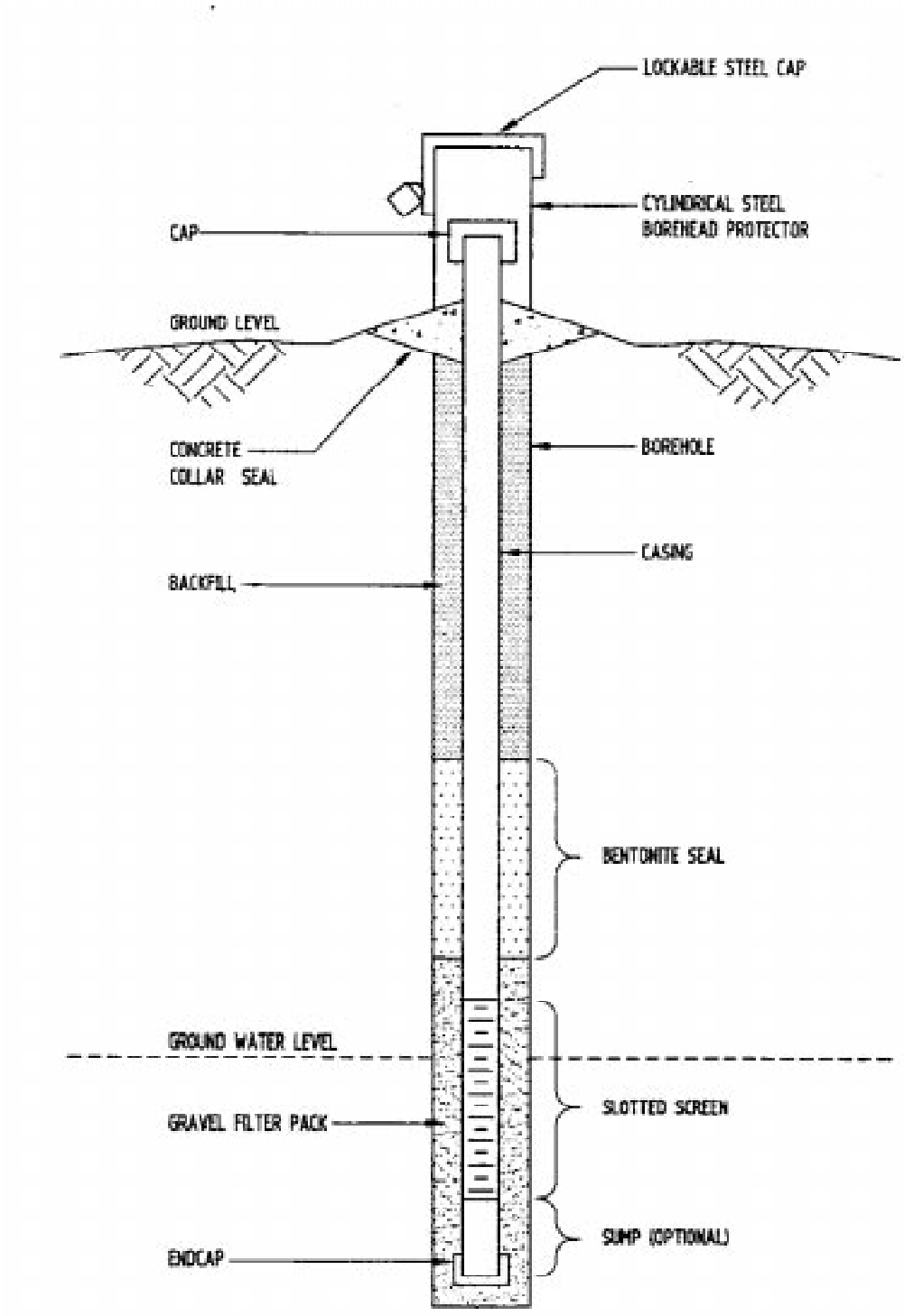
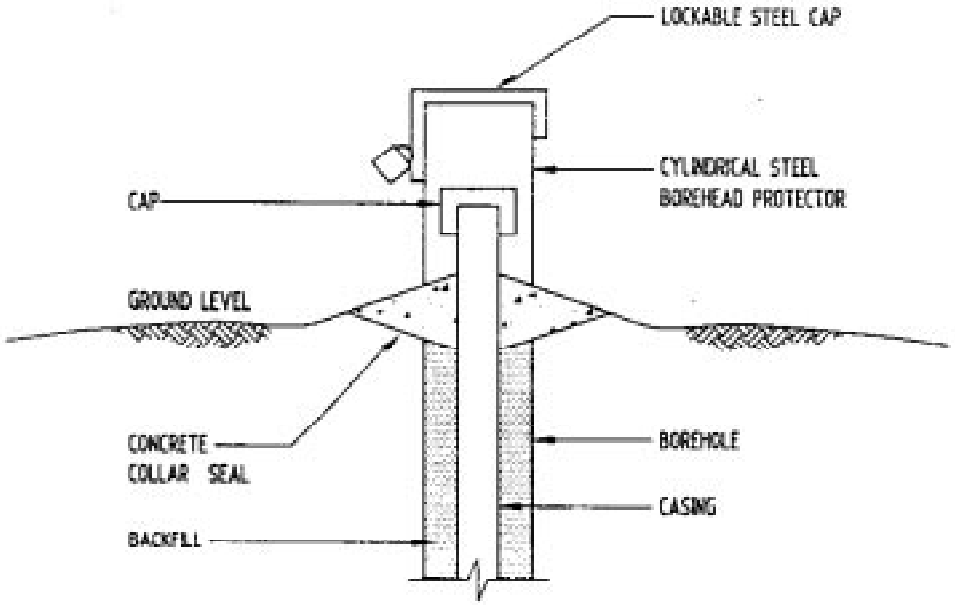
Phone:
Client:
Project:
Job No.:
Page of
File:
Sampled By:
Name
Signature
Name
Signature
Name
Signature
Sample No.
Vial
Duplicate Vial
Date
Sample Location
Composite No.
Preservative
Analytes
NOTE : DETECTION LIMITS 1/2 DUTCH B UNLESS OTHERWISE SPECIFIED
Primary Laboratory:
Secondary Laboratory
Container
Jar
Vial
500 ml (Pint)
500 ml
1 lt
2 lt
Other
Comments
(For Lab staff)
Received By:
Signature
Date:
Time:
Method Of Shipment
Relinquished By:
Signature
Received By:
Signature
Date:
Time:
Relinquished By
Signature

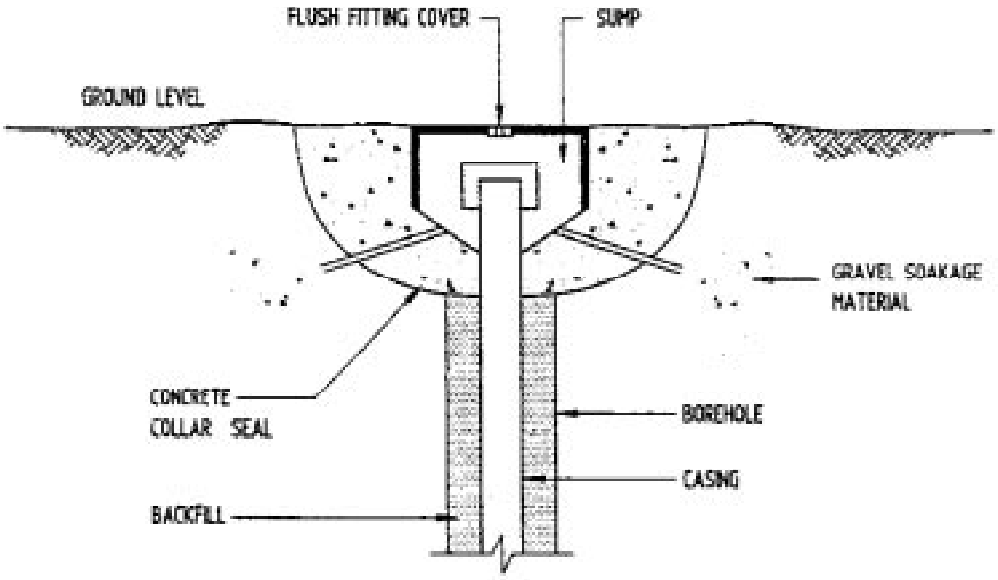
APPENDIX C
GENERALISED MONITORING BORE DESIGN



GENERALISED CONSTRUCTION DETAILS FOR GROUNDWATER MONITORING BORE



(a) ABOVE GROUND



(a) BELOW GROUND

GENERALISED BOREHEAD CONSTRUCTION DETAILS FOR GROUNDWATER MONITORING BORE

CHAPTER 4
LABORATORY ANALYSES

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APPENDICES

4. LABORATORY ANALYSES

4.1 INTRODUCTION

4.1.1 Scope

This chapter provides information and guidance on method selection and quality control strategies for the analysis of the principal contaminants of concern from New Zealand timber treatment sites:

- copper, chromium, arsenic, boron
- pentachlorophenol and related chlorophenols
- polychlorinated dibenzodioxins and dibenzofurans (PCDDs and PCDFs).

A range of other organic contaminants could also be involved at particular sites, including organochlorine insecticides, phenols, cresols, polycyclic aromatic hydrocarbons, volatile aromatic hydrocarbons and petroleum hydrocarbons. Methodologies for these classes of contaminants will not be specifically discussed but are available in the compendiums of methods from the US-EPA and other agencies.

Soil or sediment samples will additionally require determination of dry matter content. Other parameters such as pH, cation exchange capacity, organic carbon content and particle size fractions may also require determination. Water samples may additionally require determination of pH, total dissolved solids, colour, turbidity and anion/cation balance. These tests are reasonably standardised in laboratories routinely undertaking analyses of soils or waters for agricultural purposes.

The intention in writing this chapter was, wherever possible, to adopt a non-prescriptive approach and give individual analysts maximum freedom in selecting appropriate analytical methods and analytical instrumentation. To achieve this goal, and in the first part of the chapter, emphasis has been placed on a description of strategies which ensure that analytical results are of known quality.

Guidance for the specific selection of analytical methods for both inorganic and organic contaminants is provided in the second part of the chapter. For organic contaminants a number of reference methods have been suggested as performance benchmarks against which new methods can be validated. For inorganic contaminants, methods are prescribed for the analysis of soluble boron and Cr(VI) in soils and sediments and also for the digestion of all other species. However, analysts may use their discretion in selecting an instrument for the determination stage.

4.1.2 Chapter Summary

The following aspects relating to analysis and sample handling are considered:

- Criteria for selection of an analytical method and an analytical laboratory.
- Definition of laboratory analysis quality control terms.
- Quality control strategies for sample handling and analysis, including:
  - sample storage;
  - instrumental calibration standard preparation and care;
  - analytical quality control steps, including laboratory reagent blanks, replicate analyses, reference control samples, and sample fortification with surrogate compounds and internal standards.
- Requirements for method performance in terms of detection limits.
- Recommended methods for inorganic contaminants (copper, chromium, arsenic and boron) in soils, sediments and waters.
- Recommended methods and other available methods for organic contaminants (pentachlorophenol and related chlorophenols and PCDDs and PCDFs) in soils, sediments and waters. (A brief summary and critique of methods for pentachlorophenol analysis is provided in Appendix A.)
- Recommendations for the style of reporting data.

4.2 SELECTION OF AN ANALYTICAL METHOD

There are many method options for the analysis of metals and organic contaminants in soil and water, options often differing in scope, specificity, sensitivity, rigour and complexity. With respect to timber treatment chemicals the rigour of extraction must be consistent with the techniques upon which the human health and environmental protection criteria are based. For soils and sediments (except where criteria are expressed in terms of leachable and soluble) this is usually “total” contaminant content. This indicates that for metals analysis, the enduring soil science techniques are to be preferred over more recently espoused mild-extractions which aspire to measure bio-available concentrations. For organic contaminants, this initial value judgement on method rigour is less contentious in that the aspiration of all method options is to measure total content. However, most standard extraction protocols, e.g. US-EPA SW-846 (Method 3540: Soxhlet extraction of soil/sediment) have not been rigorously tested for completeness of extraction of weathered field residues. Method acceptance criteria may need to be largely based on recovery of spikes and results of inter-laboratory reference samples.

When there are multiple methods available, the principal considerations used to select the most suitable one for the situation at hand include the following:

- availability of instrumentation,
- confidence level needed,

- detection limits,
- potential interferences,
- applicability of the method for the matrix,
- complexity and cost.

The priorities of the above will vary depending on each specific situation.

Certainly one of the first considerations must be availability of instrumentation. If, for example, the method selected requires a mass spectrometer for analysis and the laboratory does not have that instrument, then clearly either another method or another laboratory must be selected.

Another early consideration involves the matrix for which the method has been designed. Some methods are designed for aqueous matrices and others for solid matrices (soils or sediments). Aqueous matrix methods usually are subdivided into drinking water, raw source water for drinking water, and industrial waste waters. Both surface waters and groundwaters are sources for drinking water, so all methods that mention raw source waters should be applicable for either of these water types. Most methods differ in their application for various matrices only in sample preparation. Once a sample has been prepared correctly according to matrix requirements, the instrumental analytical protocols from most other related methods should be able to be used after proper verification of precision and bias.

The selectivity of some methods is better than others. This will affect the degree of confidence in the identification of specific analytes as well as the possibility of false positive detections. Note that there is an important difference between detection and identification. Detection involves determining whether a signal produced by using a specific method is from the sample instead of being an artefact from instrumental noise, background contamination, or other types of interferences. A signal that meets detection criteria and that has the characteristics of the analyte of interest (e.g. a peak in a gas chromatogram at the correct retention time for that analyte) is often assumed to also identify that analyte. This is not necessarily true. Multiple identification characteristics are required for an identification to be valid. In the example above, repeating the analyses using a different GC column, so that a second and different retention time of the analyte can be compared to a standard of it, is one way to verify an identification. An alternative would be to check for the presence of characteristic ions and their ratios to one another using mass spectrometric detection.

Sensitivity can be an important consideration when concentration levels of the analytes of interest are likely to be very low. Sensitivity will vary among methods for most of the analytes. Instrument selection (e.g. ICP versus direct aspiration atomic absorption or electrothermal atomic absorption instruments) is important for metals. In the case of detectors for organic compound analyses, sensitivity and selectivity characteristics must be weighed against one another as both affect the method detection limits.

An important principle for method selection is the degree of generality. Preferred methods are those that share sample preparation and initial digestion or extraction steps with other methods being used on the samples. Similarly, methods that are suitable for both soils and sediment or surface and groundwaters will enhance productivity. A high degree of universality at the determination step can be useful e.g., ICP-MS or X-ray fluorescence for inorganics and high resolution gas chromatography with mass-spectrometric detection for organics. However, simpler determinative steps may be more cost-effective where a limited number of analytes are being covered.

Methods should be chosen which are internationally recognised and have been subjected to extensive validation and inter-laboratory study. In some cases existing methods may be unsuitable or modifications may be required for reasons such as improving cost-effectiveness on particular analytes. It is then necessary that the laboratory carry out detailed validation studies on the method as developed and applied. This validation data should be available for inspection by clients and auditing agencies along with other QC data produced during analysis of client samples. Full reporting of analytical data, on samples to which appropriate soil or water quality guidelines apply, is also important to allow clients to make correct judgements on the environmental significance of the concentrations observed. Where analyte concentrations are close to or below the method detection limit, clients will need to assess the reliability of the data. For example, data is only of quantitative significance when the concentrations reported are about four times higher than the MDL.

4.3 SELECTION OF AN ANALYTICAL LABORATORY

It is essential that a laboratory participating in studies have suitable equipment and staff experienced in the particular analyses required. The laboratory should have a comprehensive QA/QC programme in place which can prevent, detect and correct problems in the measurement processes so analytical data produced can be demonstrated to be of acceptable quality. Normally this will require that the laboratory meets ISO 9002 and ISO Guide 25 quality management standards.

A number of organisations are able to accredit and audit the more general ISO 9000 series standards. However, at present in New Zealand, TELARC is the organisation with the most experience in the ISO Guide 25 laboratory standards which include both a general audit of the quality management systems and a detailed assessment of particular analytical methods the laboratory chooses to register. It should be noted that not all the methods used in a laboratory need to be registered for TELARC accreditation. In many cases only the more routine and high-volume tests are covered. It is then incumbent on the client to ensure that an unregistered, perhaps more complex and specialised, method offered by the laboratory can be carried out to the required quality standards. This involves judgements on the experience of the analysts, the suitability of the method and its validation, and the QC analyses offered with that method.

4.4 DEFINITIONS

A range of terms is used as part of laboratory quality control procedures and as measures of data quality and method performance. It is desirable that a common set of terms is agreed upon and understood by all those involved in conducting site investigations or making decisions based on derived data. The following definitions largely follow those adopted by the US-EPA.

Internal standard: A pure analyte(s) added to a solution in known amount(s) and used to measure the relative responses of other method analytes and surrogates that are components of the same solution. The internal standard must be an analyte that is not a sample component. In practice internal standards are added prior to the final instrumental determining stage.

Surrogate analyte: A pure analyte(s), which is extremely unlikely to be found in any sample, and which is added to a sample aliquot in known amount(s) before extraction and is measured with the same procedures used to measure other sample components. Where mass spectrometric detection is employed, internal standards or surrogate standards may be isotopically labelled analogues of one or more of the analytes.

Laboratory duplicates: Two sample aliquots taken in the analytical laboratory and analysed separately with identical procedures. Analyses of duplicates give a measure of the precision associated with laboratory procedures, but not with sample collection, preservation, or storage procedures.

Field duplicates: Two separate samples collected at the same time and placed under identical circumstances and treated exactly the same throughout field and laboratory procedures. These give a measure of the precision associated with sample collection, preservation, and storage, as well as with laboratory procedures.

Laboratory reagent blank (LRB): An aliquot of reagent water or quartz sand that is treated exactly as a sample including exposure to all glassware, equipment, solvents, reagents, internal standards, and surrogates that are used with other samples. The LRB is used to determine if method analytes or other interferences are present in the laboratory environment, the reagents, or the apparatus.

Field control sample (FCS): A sample of field matrix which contains levels of the analytes of interest which are low compared to those expected in test samples. The FCS should otherwise be as similar as possible to the test samples. Aliquots of FCS, alone and fortified with analytes, carried through the complete method provide essential data on interferences, analyte recoveries and detection levels for a method as being applied in a given laboratory at a given time.

Laboratory performance check solution (LPC): A solution of method analytes, surrogate compounds, and internal standards used to evaluate the performance of the instrument system with respect to a defined set of method criteria.

Laboratory fortified blank (LFB): An aliquot of reagent water to which known quantities of the method analytes are added in the laboratory. The LFB is analysed exactly like a sample, and its purpose is to determine whether the methodology is in control, and whether the laboratory is capable of making accurate and precise measurements at the required method detection limit.

Laboratory fortified sample matrix (LFM): A portion of an environmental sample, usually a field control sample, to which known quantities of the method analytes are added in the laboratory and which is then analysed exactly like a sample. Its purpose is to determine whether the sample matrix contributes bias to the analytical results, i.e. whether the matrix causes interferences or reduced recoveries of the analytes. The background concentrations of the analytes in the sample matrix alone must be determined in a separate aliquot and used to correct the measured values in the LFM.

Stock standard solution: A concentrated solution containing a single certified standard that is a method analyte, or a concentrated solution of a single analyte prepared in the laboratory with an assayed reference compound. Stock standard solutions are used to prepare primary dilution standards.

Primary dilution standard solution: A solution of one or more analytes prepared in the laboratory from stock standard solutions and diluted as needed to prepare calibration solutions and other needed analyte solutions.

Calibration standard (CAL): A solution prepared from the primary dilution standard solution of the analytes and stock standard solutions of the internal standard(s) and surrogate analyte(s). The CAL solutions are used to calibrate the instrument response with respect to analyte concentration.

Quality control sample (QCS): A sample matrix containing method analytes, portions of which are regularly analysed to check that a method is in control. A QCS can be a fortified sample matrix (either laboratory or external). A thoroughly homogenised field sample with analytes present as weathered residues can also be used as a QCS. The QCS may be locally prepared from a bulk sample containing analytes in relevant concentration ranges (laboratory reference material) or from external sources where the QCS may have been carefully validated by a inter-laboratory collaborative study. These certified reference materials are available from US-NBS, European BCR and other national agencies but are expensive and may not contain priority timber treatment analytes in the relevant concentration ranges.

Accuracy: Closeness of a result or the mean of a set of results to the true value. Accuracy is assessed by means of laboratory fortified matrix samples or external QC samples.

Precision: A measurement of the agreement of a set of replicate results amongst themselves without assumption of any prior information as to the true result. Laboratory precision is assessed by means of analysis of duplicate/replicate sub-samples.

Method Detection Level or Limit (MDL): The lowest concentration at which individual measurements for a specific analyte are statistically different from a laboratory blank with a specified confidence level for a given method and representative matrix.

For a 95% confidence interval MDL = 3 S_B/M
where M = Slope of calibration line for analyte
S_B = Standard deviation of the noise level or the background signal (usually from a field control sample).

Reliable detection level (RDL): Lowest recommended concentration of analyte for making qualitative decisions based on individual measurements for a given method and representative matrix. Recommended to be 2 x MDL (CCME, 1993, Keith, 1991 a, b, c).

Reliable quantitation level (RQL): Lowest recommended concentration of analyte for making quantitative decision based on individual measurements for a given method and representative matrix. Recommended to be 4 x MDL (CCME, 1993, Keith, 1991 a, b, c).

4.5 LABORATORY QUALITY CONTROL

4.5.1 Importance of QA/QC

The basic features of laboratory quality assurance have been summarised in Chapter 2. These precepts are fundamental to providing a physical and managerial environment in which analytical systems of controlled quality may function. However, the mere use of approved methods in a facility operating a quality assurance programme does not guarantee adequate results. A wide range of unanticipated effects can cause inaccuracies. It is largely the QC components of each method as it is applied to test samples that allow the detection of these errors and that provide key validation of the data as reported.

A crucial factor for hazardous waste sites and their environs can be the concentration range of the analytes. Cross-contamination from dust, glassware or instruments will make it extremely difficult to obtain quality data from broader environmental samples in a laboratory that is also analysing highly contaminated site samples where analyte concentrations may be 3-5 orders of magnitude higher. Conversely, adequate sub-sampling and wide-ranging instrument calibration is more important for soil samples which are likely to be more hetereogeneous, and exhibit wider variations in contaminant concentration, than other environmental samples such as water or sediment. QC procedures must be designed to cope with these extremes.

The following sections are not comprehensive and fuller information is available in preambles to documented environmental analytical methodologies (SWP-846, US-EPA 1986; CCME 1993). However, it is useful to highlight areas which are likely to be crucial to obtaining quality analytical data on samples from contaminated timber treatment sites or surroundings, whether by fully documented and approved methods or by newer, perhaps in-house, methods.

4.5.2 Sample Storage

Tables D1 and D2 in Chapter 2, Appendix D summarise storage conditions suitable for samples received for analysis of priority timber treatment contaminants. Sample containers for organic analytes should be glass with Teflon-faced cap seals. Waters should be acidified to pH 2 on receipt to minimise microbial growth using HNO_3 for inorganics and HCl for organics.

4.5.3 Sample Preparation and Sub-sampling

As the distribution of contaminants in soil or sediment is often heterogeneous, samples must be adequately mixed before taking portions for analysis to ensure sub-sampling errors are small with respect to other errors in the analytical procedure. There is a trend towards smaller test portions in order to minimise costs of reagents and waste disposal and to simplify glassware manipulations. Some attention to detail is obviously required if an analysis of a 0.5 g portion is to adequately represent a 1 kg field sample. It is recommended that all sample preparation, including mixing, be undertaken by the laboratory, minimising the potential for cross-contamination (refer to Chapter 3 for details of the sampling procedures).

Where analytical portions are 10 g or greater, field-moist samples can be used after removal of stones and other large particles and thorough mixing of the sample. Superficial water can be decanted from sediment samples prior to mixing. For volatile analytes field-moist sub-samples should be tested without drying.

For high-clay, clumpy samples or where smaller test portions are to be taken, it is necessary to air dry appropriate sub-samples and carry out grinding and sieving prior to taking analytical portions. The following procedure is recommended:

1. Obtain a representative sub-sample of the laboratory sample by riffler or cone-and-quarter technique. Take at least 50% of the sample or 200 g, whichever is smaller.
2. Remove the largest stones and vegetation. Do not discard (see Step 4).
3. Air dry the sample (30^oC to 35^oC, <65% RH, 16 hours or longer if required).
4. Grind (mortar and pestle) and sieve so less than 5% w/w retained on 2 mm sieve (store this retained fraction together with the larger particles (from Step 2) for possible future examination).

5. Homogenise the fraction <2 mm diameter. If small analytical portions (<10 g) are to be taken, grind at least 10 g of this to pass smaller sieves as shown in Table 4.1.

6. Store the ground and sieved sub-sample in suitable glass air-tight container. In some cases, for example PCDDs, PCDFs, and chlorinated phenols, it may be necessary to protect the sample from decomposition by sunlight by storing in the dark.

Table 4.1
Recommended Sieve Sizes

Mass of Sample Required for a Single Analysis	Sieve Size Recommended (mm)
Less than 1 g	0.15
Less than 2 g	0.5
2 to 9 g	1.0
10 g or greater	2.0

The preparation of samples for composite analysis should only entail the use of documented site samples as sub-samples for which the appropriate drying, grinding and sieving steps have been carried out. Each sub-sample from which the composite is compiled should contribute equally in mass. Prior to analysis the material should be thoroughly mixed to ensure the sample is homogeneous. The composite sample should not be composed of more than four sub-samples. This restriction ensures that the contaminants appearing in a single sub-sample will not be diluted below the method detection limit.

Extreme care should be taken to avoid cross-contamination during the sample preparation process and to minimise spread of dust in the laboratory. Equipment and containers used must be thoroughly cleaned before each sample to prevent cross-contamination. Cleaning procedures will vary according to the analytes being determined. Generally detergent washing, followed by deionised distilled water rinsing and oven drying will suffice. For trace metal analysis it may be necessary to incorporate soaking in dilute acid before distilled water rinsing. Solvent rinsing followed by air drying will normally be required prior to homogenising samples for organics analysis. Frequent laboratory reagent blank analyses will be required to check for contamination.

WARNING: **Grinding of soils to fine dimensions may produce airborne particles which present a health hazard. Preparation should be performed in a fume hood, and appropriate respiratory protection should be worn.**

4.5.4 Calibration Standards

Inter-laboratory check sample programmes have consistently shown that the most common source of major bias in analytical data is inaccurate concentrations, or even identifications, of analytes in calibration standards. Consequently laboratory QA/QC systems and quality audits must put great emphasis on this area. It is essential that detailed procedures are in place and followed by the analysts to manage and document the traceability and validity of reference materials and derived solution standards used in analytical methods. Documentation should include:

- A suitable coding system for uniquely identifying all primary and derived standards.
- Records of receipt for all primary reference compounds or certified standards including source, purity and expiry date.
- Records of preparation for all stock standard solutions including dates of preparation and expiry, weight of reference material, final volume and solvent of dilution, signature of check by laboratory manager or person responsible for quality assurance policy in the laboratory.
- Record of preparation for all primary dilution and calibration (working) standard solutions including aliquot volume(s) or weight(s) of stock standard(s), final volume and solvent of dilution, expiry date, signature of check by laboratory manager.
- Records of confirmation of identity and concentrations of analytes in standard solutions including GC-MS, comparisons of concentrations with those of previous standards and comparisons of concentrations with those of standard solutions exchanged with other laboratories.

For inorganic solution stock standards, solutions prepared in acid solution (stored at $<4^{o}C$) and chlorophenol or PCDD/PCDF solution standards prepared in organic solvents (stored at $-18^{o}C$), the expiry dates can be long (at least 12 months) provided careful checks are made on the volumes. Running records of total weights before and after removing aliquots can be used to check for solvent losses during storage. These will be minimised by use of high-boiling solvents, e.g. ethyl acetate, toluene. Fresh calibration standard solutions should be prepared often and instrument responses compared to those for previous sets.

4.5.5 Recommended QC Procedures

It is recommended that the QC steps described in Chapter 1, “Quality Control” of “Test Methods for Evaluating Solid Water”, USEPA Publication SW-846, be adopted for all soil analyses and are also applicable to most water analyses.

In particular, it is expected that analysts would implement the following QC steps with each analytical batch, or with each 20 samples, whichever is the smaller:

1. ***Laboratory Reagent Blank:*** at least one determination of a blank to establish the contribution to the analytical signal by reagents, glassware etc. The blank should be subtracted from the gross analytical signal for each analysis before calculation of sample analyte concentration.

2. ***Replicate Analysis:*** duplicate analysis of at least one sample from the batch. The variation between replicate analyses should be recorded for each batch to provide an estimate of the precision of the method.

3. ***Quality Control Sample:*** analysis of at least one control sample, which comprises either a standard reference material, a laboratory reference material or a control matrix fortified with analytes representative of the analyte class. Recovery check portions should be fortified at concentrations which are easily quantified but within the range of concentrations expected for real samples.

4. ***Surrogate analytes:*** surrogates should be added to all analyses for determinations where it is appropriate (e.g. chromatographic analysis of organics). Surrogate spikes are known additions **to each sample, blank and matrix spike or reference sample analysis**, of compounds which are similar to the analytes of interest in terms of:

   (a) extraction,

   (b) recovery through clean-up procedures, and

   (c) response to chromatographic or other determinations, but which

   (d) are not expected to be found in real samples,

   (e) will not interfere with quantification of any analyte of interest, and

   (f) may be separately and independently quantified by virtue of, for example, chromatographic separation or production of different mass ions in a GC/MS system.

   Surrogates are added to the analysis portion **before extraction** to provide a means of checking, for every analysis, that no gross errors have occurred at any stage of the procedure leading to significant analyte losses.

   In the case of organic analyses the surrogate analytes may be ^{13}C, deuterated, alkylated or halogenated analogues, or structural isomers of analyte compounds.

5. ***Internal Standards:*** use of internal standards is highly recommended for chromatographic analysis of organics. Internal standards are added, **after all extraction, clean-up and concentration steps**, to each final extract solution. The addition is a constant amount of one or more compounds with similar qualities to 4(d), 4(e) and 4(f) above.

   Internal standards are used to check the consistency of the analytical step (e.g. injection volumes, instrument sensitivity and retention times for chromatographic systems) and provide a reference against which results may be adjusted in case of variation. The instrument is usually calibrated using the

ratio of peak height or area for analytes compared with that for the internal standard(s). Surrogates are treated as analytes for quantification.

Internal and surrogate standards are most useful for trace analyses where analyte losses during extraction or chromatography and small final volumes can give rise to considerable errors. They are of lesser utility for samples with very high concentrations of analytes as the responses of small quantities of added standards are likely to be swamped or to be lost in dilution of final extracts.

In addition to the above within-batch QC samples, it is also strongly recommended that the laboratory participate in inter-laboratory sample exchange and collaborative study programmes and periodically analyse certified reference materials. These QC activities provide invaluable experience and external reference to validate the analytical methodology and give confidence in the data produced.

It is also recommended that a field control sample spiked with analytes in the mid-range of anticipated sample concentrations be analysed for every matrix type from a site assessment study. Such samples provide information on the potential of the matrix to cause positive or negative bias. For soil and sediment samples the spike should be applied to fresh material which has already been dried, ground and sieved. An unspiked duplicate sample must also be analysed to establish the naturally occurring analyte concentrations.

4.6 METHOD PERFORMANCE SPECIFICATIONS

Method specifications must be such as to allow assessment of compliance with the various health and environmental guidelines. In general, guidelines for protection of aquatic ecosystems are more stringent than those for other beneficial uses and may well require implementation of laboratory practices not required for the majority of tasks. For this reason – and because ecosystem guidelines are not formally proposed in this document – specifications are presented in two categories. Laboratories unable to meet the requirements for testing compliance with ecosystem guidelines are not then excluded from participation in site assessments that do not involve aquatic ecosystems. Furthermore, current methods have difficulty meeting the MDLs required to determine PCP and dioxins at aquatic ecosystem guideline levels, particularly as reliable quantitation can only be achieved at concentrations more than four times the MDL (refer Sections 4.4 and 4.8).

The method detection limits in Table 4.2 for aquatic ecosystems are illustrative and are presented only to guide analytical work. Where protection of aquatic ecosystems is not a significant consideration, the illustrative method detection limits presented in Table 4.3 may be more appropriate.

Table 4.2
Illustrative Sediment and Water Method Detection Limits
(for aquatic ecosystem protection)

Contaminant	Method Detection Limit for Sediments (mg/kg)	Method Detection Limit (mg/L)
Cu	4	0.001
Cr (total)	6.5	0.002
Cr (VI)		0.002
As	1.5	0.010
B (total)		
B (soluble)	2.5	2
PCP		0.01 µg/L
Dioxins		0.002 ng/L

Table 4.3
Required Method Detection Limits for Soils and Waters
for Beneficial Uses other than Ecosystem Protection

Contaminant	Min. Soil Guideline (mg/kg)(1)	Method Detection Limit (mg/kg)	Min. Water Guideline (mg/L)(2)	Method Detection Limit (mg/L)
Cu	30	7.5	0.2	0.05
Cr (total)	600	150	0.1	0.025
Cr (VI)	10	2.5	0.05	0.015
As	10	2.5	0.012	0.003
B (total)	25	6.3	0.4	0.1
B (soluble)	3	0.75	(0.4)	(0.1)
PCP	0.1	0.025	0.003	0.00075
Dioxins (T.E.)			0.015 ng/L	0.0038 ng/L

Notes: (1) Refer to Chapter 5
(2) Refer to Chapter 6

4.7 METHODS FOR INORGANIC CONTAMINANTS

The methods nominated in this section seek to standardise the extraction rigour of a sample work-up. The prescription does not, in general, extend to dictating which instrumental technique is used in the final analysis step. So within the constraints of accuracy and specificity and the need to comply with the sensitivity requirements given in Table 4.3, there is sufficient flexibility for competent laboratories to apply techniques already in use. No method is described in full unless it is unpublished or not readily accessible.

4.7.1 Cu, Cr (total), As, B in Soils and Sediments

Sample is subjected to a mild acid digestion following the procedure set out in USEPA Method 200.2. The resultant solution is amenable to instrumental analysis using a number of modern techniques. If boron is analysed colorimetrically, the method must accommodate the presence of nitric acid in the digest. The Azomethine-H method is commonly used without problem, but use of Curcumin without adequate dilution of the test solution may attract interference from nitrate.

4.7.2 Soluble B in Soils and Sediments

There are several variants of this empirical soil test. Although all similar in concept, the operational differences are of significance in that the results obtained are very much a function of the extraction procedure. The method described in the Handbook on Reference Methods for Soil Analysis (Soil and Plant Analysis Council, 1992) is recommended, the essential details being consistent with procedures already in use in the major soil laboratories in New Zealand. The basis of the method is as follows:

- Mix 20 g of air-dry, 2 mm soil with 40 mL of water containing 0.5 mL of 10% $CaCl_2$. Bring to boil and gently reflux for 10 minutes.
- Without cooling, filter or centrifuge a suitable volume of suspension.
- Analyse the clarified solution for B. Express results on sample dry-weight basis.

4.7.3 Cr(VI) in Soils and Sediments

A suitable method is that described by Page (1982), and requires a fresh, moist sample.

- Shake 3 g of moist sample with 25 mL of 0.1 M KH_2PO_4 for 5 minutes.
- Centrifuge or filter.
- Add 1 mL of S-diphenylcarbazide reagent to 8 mL of extract, mix and measure colour developed after 20 minutes.
- Express results on dry-weight basis.

NB. The diphenylcarbazide reagent differs from that in Standard methods 3500-Cr D and is as follows:

> "Dissolve 0.4 g of S-diphenylcarbazide in 100 mL of ethanol, and mix this solution with 120 mL of 85% phosphoric acid diluted to 400 mL with water."

4.7.4 Suitability for landfill disposal

The USEPA TCLP test as outlined in Appendix B of Chapter 7 is applied.

4.7.5 Cu, Cr, As, B in Water

Elements Cu, Cr(total), As, B are to be determined using any suitable method following sample preparation according to USEPA Method 200.2. Any colorimetric finish for boron must be able to accommodate the presence of nitric acid in the digest (refer to Appendix E of Chapter 2 for a discussion of sample field filtration and preservation requirements). The Azomethine-H method is suggested as suitable.

4.7.6 Cr(VI) in Waters

Cr(VI) may be determined by the diphenylcarbazide colorimetric method. A suitable procedure is APHA Standard Methods 3500-Cr D.

The analysis should be performed on a sample filtered through a 0.45 μm membrane filter and acid-preserved at the time of sampling (refer Appendix E of Chapter 2). When separate analyses for total chromium and Cr(VI) are required, separate samples should be taken. A practical expedient is to analyse for Cr(VI) only if the result for Cr(total) indicates exceedance of the Cr(VI) guideline, although care is required to ensure maximum sample holding times (refer Appendix D, Chapter 2) for Cr(VI) are not exceeded.

4.8 REFERENCE AND SCREENING METHODS FOR ORGANIC CONTAMINANT ANALYSIS

Reference methods for analysis of pentachlorophenol (PCP) and polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDDs and PCDFs) in soils, sediments and waters are presented in Table 4.4. In some cases the complete method is sub-divided into extraction, clean-up and analysis (determination) stages, which are referenced separately themselves. These methods have been validated as being capable of producing quantitative analytical data in the concentration region of 4 times the quoted detection limits and above. The reference methods selected have been chosen on the grounds of their validation and inter-laboratory exchange history as well as international recognition. It should be emphasised that, provided suitable validation studies are conducted, other methods may prove to be more suitable, particularly for

screening. Appendix A summarises some of the characteristics of the various reference and screening methods for PCP.

Screening methods should be validated, typically by analysing an appropriate number of samples using a reference method and conducting a statistically based comparison.

Analyses for the highly toxic PCDDs and PCDFs are complex due to the very low levels that may be significant, the wide range of congeners (chlorination level) and positional isomers. The high selectivity, specificity and sensitivity provided by capillary GC-MS (selected ion mode) is essential to obtain reliable detection limits. Isotopically labelled internal and surrogate standards are extensively used for in-run QC. Due to the specialised nature of the analyses, the expensive equipment required and the expense/toxicity of standard materials, PCDD and PCDF analysis remains the province of very few laboratories.

The MDLs shown in Table 4.4 should be regarded as indicative values only. Individual laboratories will have to establish their own detection limits for analytes as specified in Section 4.4. For PCP analysis in soils and sediments and PCDD and PCDF analysis in water a five-fold improvement of the specified reference method detection limit is required to comply with the method performance standards suggested in Section 4.6. Several of the specified US-EPA methods for PCP were developed principally for neutral or weakly acidic contaminants and are unlikely to be able to produce reliable data for PCP at low detection limits. The validation data for the highly acidic PCP often showed poor precision and high MDLs. However, the methods do have value for analysis of a broad range of contaminants.

Table 4.5 shows a number of screening methods for the analytes in question. Screening methods are often considerably faster and cheaper to perform than the corresponding reference methods. In many cases these methods may also offer improved quality characteristics (MDL, precision). In many cases these methods only lack the full validation and interlaboratory study required of reference methods. However, screening methods may also provide analytical data which is qualified in some way. The qualification might be a possible positive bias and relatively low precision such as may occur with some of the immunoassay techniques for PCP. Also a method may have a lack of scope, such as with the DSIR OCDD screening analysis. This method can be used to provide quantitative data for hepta-chlorinated and octa-chlorinated dioxin and furan isomers substituted in the 2, 3, 7 and 8 positions, but is not suitable for the isomer-specific determination of tetra-chlorinated to hexa-chlorinated congeners.

Appendix A summarises some of the characteristics of the various reference and screening methods for PCP.

Analyses performed using screening methods should still be conducted according to the QA/QC requirements recommended in Section 4.5.5. Screening methods used under more relaxed circumstances, such as in the field, have some use for selection of contaminated areas for further sampling.

Table 4.4
Reference Methods for the Analysis of Pentachlorophenol and Polychlorinated Dibenzo-*p*-dioxins and Dibenzofurans

Analyte and Matrix	Determination	Extraction	Clean-up	Method Detection Limit
Pentachlorophenol				
Soil/sediment	USEPA 8270	USEPA 3540 or 3550	USEPA 3650 or 8040	0.5-3 mg/kg
	USEPA 8040	USEPA 3540 or 3550	USEPA 8040	0.5 mg/kg
Water (contaminated)	USEPA 8270	USEPA 3510 or 3520	USEPA 3650 or 8040	0.6-50 µg/l
	USEPA 8040	USEPA 3510 or 3520	USEPA 8040	0.6 µg/l
Water (drinking or 0.08 µg/1 ecosystem protective)	USEPA 515.1 USEPA 1653	USEPA 515.1 USEPA 1653	USEPA 515.1 USEPA 1653	0.08 µg/l 0.28 µg/l
Polychlorinated dibenzodioxins and dibenzofurans				
Soil/sediment (contaminated)	USEPA 8280	USEPA 8280	USEPA 8280	2 µg/kg
Water (contaminated)	USEPA 8280	USEPA 8280	USEPA 8280	10 ng/l
Soil sediment (trace)	USEPA 8290	USEPA 8290	USEPA 8290	1 ng/kg
Water (trace)	USEPA 8290	USEPA 8290	USEPA 8290	0.010 ng/1

Table 4.5
Screening Methods for the Analysis of Pentachlorophenol and Polychlorinated Dibenzo-*p*-dioxins and Dibenzofurans

Analyte	Matrix	Reference
Chromatographic Methods		
Pentachlorophenol	Soil or water	Stark (1969)
Pentachlorophenol	Soil or sediment	Lee et al (1987)
Pentachlorophenol	Water	Abrahamsson and Xie (1983) Lee et al (1984)
PCDDs and PCDFs	Soil	DSIR/ESR (1992)
Immunoassay Methods		
Soil or Water		Test packs : Millipore Corp Ohmicron Corp Ensy Corp

4.9 DATA REPORTING

Laboratories have a responsibility to provide reports that are complete, accurate and unambiguous so that clients can draw clear conclusions from the data without making any assumptions. Laboratories must also maintain full records of samples, methodology and experimental data so that auditing can be carried out at any time to verify the reported results. Particular attention must be given to the QC records in reports and filing.

Reports must contain the following information:

- Sample I.D. (client and laboratory) and description.
- Date of receipt and conditions of storage.
- Date extraction commenced.
- Details of sample preparation and fraction of sample taken for analysis.
- Citation and summary of analytical procedure – may be just the title for a validated regulatory method. Any modifications to the protocol should be noted.
- Date of reporting and signature of laboratory manager or other authorised signatory.

It is recommended that results of analyses should be reported using the following conventions. Those for concentrations in the region of the detection limit follow recent trends in North America (Canadian NWQL, ASTM, ACS) which leave any censoring of data to the client but provide guidance on the quality of the data.

- No results are to be reported for analyses that were outside the calibration range of the instrument. Dilutions must be made to bring extracts/digests into the linear range. For analyses using derivatisation, smaller aliquots of extract must be taken through the procedure.
- Concentrations of analytes in soils or sediments should be presented on an oven dry ($105^{o}C$) basis with moisture contents of the field samples presented separately if requested.
- Analyte concentrations should be corrected for the blank but not for recovery.
- Use SI units e.g. mg/kg, mg/L rather than ppm or ppb.
- No observed signal for the analyte – report as ND (not detected) at quoted Method Detection Limit (MDL).
- Analyte signal detectable but concentration less than the MDL – report concentration but flag as <MDL and in a region of high uncertainty. Terms such as “Trace” should be avoided.
- Analyte concentrations greater than MDL – report unflagged.
- Separate results should be presented for each field replicate.
- The MDL and analyte recovery (% from spikes) should be given based on actual QC samples run with the client samples and should not be estimates from previous method validation experiments. MDLs should be based on environmental control samples rather than laboratory blanks. If suitable control samples are not available then MDLs should be set on a conservative basis after a careful study of signals from field samples and blank samples.
- Results for laboratory replicates should be averaged and marked in the report with the number of measurements e.g. 0.31 (3). Sets of laboratory replicate data should be summarised in the form of confidence intervals to show within-laboratory precision.
- The mean and standard deviation of the recoveries for the surrogate analyte(s) across all samples should be reported.
- Results for all QC analyses (reagent blanks, field control samples, fortified laboratory matrix, QC samples) run with client samples should be reported with ranges, means and confidence intervals where appropriate.

4.10 REFERENCES

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APPENDIX A
BRIEF DESCRIPTION AND CRITIQUE OF REFERENCE AND PROPOSED METHODS FOR PENTACHLOROPHENOL ANALYSIS

The principal target phenol from timber treatment sites is pentachlorophenol (PCP). However, a variety of lower-chlorinated phenols are also generally present, either from impurities in the PCP formulations or microbial degradation of PCP, with 2, 3, 4, 6-tetrachlophenol predominating. It is desirable to use methods that analyse for a wide range of chlorophenols so a more complete inventory of site contamination and extent of PCP degradation can be obtained.

The following US-EPA methods are relevant to chlorophenol analysis.

SW-846 Method 8040A, rev. 1, 1990 (Phenols by gas chromatography).

This concentrates on the determination steps but indicates that chlorophenols can be recovered from waters by liquid-liquid partition (Method 3510 Separating funnel or Method 3520 Continuous liquid-liquid) or from solid waste by solvent extraction (Method 3540B Soxhlet or Method 3550B Sonication). Clean-up is by acid-base partitioning (Method 3650A) and, for low levels in soil, gel permeation chromatography (Method 3640A).

Reliable quantitation levels by packed column GC with flame ionisation detector (FID) are about 5 mg/L in waters and 0.5 mg/kg in soils for some di- and tri-chlorophenols and about a factor of 10 higher for PCP.

The specificity of packed column GC-FID is low and interferences from other acidic compounds may be expected. Also acidic phenols are liable to tailing and other adsorption effects in the GC, effects which can be variable and influenced by co-extractives and therefore lead to poor quantitation.

The method also provides for a derivatisation step to form pentafluorobenzyl-ethers of the phenols which have more reliable GC performance and give high responses to the electron capture detector (ECD). However, a time-consuming silica gel chromatographic clean-up is required to remove interferences including derivatised co-extractives. The method has been validated for a range of phenolics including cresols but a number of the relevant lower chlorophenols have not been formally included.

SW-846 Method 8270B (Capillary GC-MS for Semi-volatile Organics in Solid or Liquid Water).

This is a screen also based on solvent extracts prepared using the 3500 series protocols (see above). However, the low resolution mass spectrometric detection (full scan mode) covers a wider range of contaminants with higher selectivity than ECD or FID. The high resolution capillary column separation also improves selectivity and inertness in the analytical system. However, erratic performance of PCP at low levels will be a problem with crude extracts of

soils or waste waters. The method has not been validated for many of the chlorophenol isomers.

EPA-600 Method 525.1, rev. 2.2, 1991 (Determination of Organic Compounds in Drinking Water using Liquid/Solid Extraction and Capillary GC-MS).

This screen is similar to SW846 8270B in the determination of a wide range of contaminants using capillary GC-MS except that reversed phase adsorbents (column or disk) are used to concentrate contaminants from water samples. However, poor GC performance of the acidic PCP resulted in a method detection limit of 48 µg/L versus 0.03-0.5 µg/L for most neutral contaminants tested. No lower chlorophenols have been validated through the method.

EPA-600 Method 515.1, rev. 4.0, 1988 (Determination of Chlorinated Acids in Water by GC-ECD).

Solvent extraction of the water sample is followed by derivatisation with diazomethane to form the methyl esters/ethers. Chlorophenol methyl ethers are detected by capillary GC-ECD along with methyl esters of chlorinated and other electron capturing acids. A Florisil Chromatographic clean-up is used to reduce interferences but no data is provided on performance of the method on soil extracts. The method has been validated for PCP with a method detection limit of 0.08 mg/L but no data is provided for other chlorophenols.

EPA Method 1653 (Chlorinated phenolics in wastewater by *in-situ* acetylation and GC-MS).

Chlorophenolics in water are extracted with hexane after in situ acetylation with acetic anhydride at pH 9-11.5. After volume reduction to 0.5 mL the compounds are separated and determined by high resolution gas chromatography with mass spectrometry. Detection limits are 0.5-1 µg/L. This method covers a wide range of chlorophenols and apart from the GC-MS detection is simple and direct. Detection limits can be improved by selected ion monitoring (SIM).

None of these USEPA regulatory methods has the simplicity to recommend it as the first choice for screening of chlorophenols in soil or water. Method 515.1 seems likely to perform well for lower chlorophenols in water as well as PCP and it is probable that solvent extracts of soils could also be analysed after acid/base partition, methylation and Florisil clean-up. However, the procedure is relatively complex and diazomethane is a carcinogenic reagent. The soil extraction procedures using neutral conditions are likely to be relatively ineffective at removing weathered residues of chlorophenols due to ionic binding. Several of the methods use GC-MS detection which raises the cost of the assays and the equipment is not available in some laboratories. GC with electron capture detection is the preferred technique for screening of the tri- through to pentachlorophenols. Some of the following alternative approaches seem more promising for development into fully validated methods for screening of chlorophenols in soil or water.

Analysis of Pentachlorophenol Residues in Soil, Water and Fish (Stark, 1969).

Soils are extracted with 0.1 M potassium hydroxide solution and an aliquot is buffered to pH 6.5-7.0 using boric acid and mineral acid before partitioning with toluene. Acidified water samples are partitioned with toluene directly. The toluene extracts are treated with diazomethane and PCP methyl ether determined by GC-ECD. Use of the PCP-trimethyl silyl (TMS) derivative was recommended for confirmation. Detection limits for PCP in soil of less than 1 mg/kg were reported.

This method has been adopted by the Victorian Environmental Protection Agency (ANZECC, 1993) with minor modifications including a toluene wash prior to acidification to remove base-neutral co-extractives and use of the TMS derivative rather than methyl ether for primary GC quantitation.

Graysons Laboratories, Auckland have also made available an in-house method which is related to the above methods. An aqueous base soil extract is rinsed with dichloromethane, acidified to pH 2 and chlorophenols extracted with dichloromethane. Chlorophenols are determined as TMS derivatives using HRGC-ECD (split injection) with detection limits in soil of 0.1 mg/kg.

Although a base extraction of soils can be expected to be efficient for chlorophenol residues, substantial quantities of humic acids will be solubilised. These are likely to give rise to problems of gels and emulsions at the acid partitioning step, overcome to some extent by the buffering in the Stark procedure. The TMS derivatisation is not attractive as the reagent and derivatives are unstable to hydrolysis and silicone polymer impurities can be formed. These effects are liable to lead to interferences and contamination of the sensitive ECD. The extracts can also be derivatised using diazomethane to form the more suitable methyl ether derivatives.

Direct Determination of Trace Amounts of Chlorophenols in Fresh Water, Waste Water and Sea Water (Abrahamsson and Xie, 1983).

This remarkably simple method uses extractive acetylation to selectively transfer chlorophenols as their acetates from water (pH adjusted to 9) plus acetic anhydride into hexane. A range of chlorophenols (di- to penta-) were determined with minimal interferences and high recoveries at levels of below 0.1 mg/L using capillary GC-ECD.

Analysis of 15 Chlorophenols in Natural Waters by In Situ Acetylation (Lee et al., 1984).

This method is very similar to the above, using 0.5% potassium carbonate as the alkaline buffer. All 16 possible di- to penta-chlorophenols were extracted and analysed as their acetates by GC-ECD. Method detection limits in lake water were below 0.1 μg/L. This method forms the basis for EPA Method 1653.

Determination of Pentachlorophenol and 19 other Chlorinated Phenols in Sediments (Lee et al., 1987).

This method extends the acetylation procedure in the previous paper to the analysis of solvent extracts of sediments. Sediments were acidified to pH 1 and Soxhlet extracted with acetone/hexane. Partitioning/clean-up into 2% aq. potassium carbonate solution was followed by extractive acetylation and clean-up by silica gel chromatography. Method detection limits for chlorophenols were below 1 mg/kg using capillary GC-MS in selected ion mode (mono- to penta-) or using capillary GC-ECD (tri- to penta-). A dibromo-phenol was recommended as a surrogate analyte or internal standard. The method was not reliable for phenol itself or chloro methyl phenols and so it is unlikely to be applicable to cresols.

CHAPTER 5
SOIL ACCEPTANCE CRITERIA

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APPENDICES

5. SOIL ACCEPTANCE CRITERIA

5.1 INTRODUCTION

5.1.1 Purpose

This chapter describes the development of soil acceptance criteria to guide the assessment and management of land contaminated by timber treatment chemicals.

5.1.2 Goals of the Soil Acceptance Criteria

The criteria have been developed using established health and environmental risk assessment procedures. They represent acceptable residual contaminant concentrations in soils for the following generalised uses of a site:

- Agricultural
- Residential
- Industrial Use – unpaved
- Industrial Use – paved
- Maintenance Access

The soil acceptance criteria are based on the protection of the health of people on the site, protection of plant life, and contaminant uptake by plants and animals. Site amenity considerations have also been included, particularly as part of the review of criteria applicable for residential land use.

The soil acceptance criteria do not take account of the need to protect groundwater or surface rainfall runoff, which may result in more stringent criteria. This aspect is addressed separately in Section 5.10.7.

5.1.3 Chapter Summary

This chapter develops guidelines for acceptable levels of residual contaminants in soils which have been contaminated by timber treatment chemicals. These guidelines have been developed on the basis of potential generalised future uses of such sites, which are: agriculture, residential use and industrial use. Industrial use is further classified in terms of the likely exposure to the soil substrate, so that separate guidelines are proposed for paved and unpaved sites and for maintenance access.

The criteria have been developed by taking into account the goals of protection of the health of site users, protection of public health and protection of plant life on the site. Considerations which are not explicitly dealt with include the protection of groundwater quality and the protection of the health of on-site ecosystems.

Section 5.4 explains the way in which the criteria were developed. The protection of human health is achieved by adopting a risk-based approach which aims to ensure that the levels of residual contaminants which are suggested in the guidelines do not pose an unacceptable risk to human health. This is done by using available toxicological information for each contaminant to set a maximum intake level, calculating the likely exposure of site users to soils containing that contaminant, and then setting the maximum permissible level of contaminant in the soil so that the target intake is not exceeded.

Section 5.4 also deals with risk characterisation. It describes the different approaches taken for carcinogens and non-carcinogens and the approach taken to multiple contaminants.

Section 5.5 discusses the health hazards posed by arsenic, boron, chromium, copper, pentachlorophenol and dioxins and furans and the internationally accepted acceptable intakes of these substances. It also deals with mixtures of contaminants and with background exposures.

Section 5.6 deals with ecotoxicity and the protection of plant life (phytotoxicity).

Section 5.7 deals with exposure and risk assessment. It identifies the different receptor groups in the population and the different exposure routes which may be significant for each group and then develops mathematical models which allow the calculation of exposure by the ingestion of contaminated soil, the inhalation of contaminated dust, dermal absorption of soil contaminants and the ingestion of produce grown in contaminated soils.

Section 5.8 combines the toxicological data from Section 5.5 and the exposure models developed in Section 5.7 to derive maximum allowable levels of contaminants in soils on sites which are used for agriculture, residential use and industrial use (including both paved and unpaved sites and maintenance access). The uncertainty associated with the proposed values is discussed.

Section 5.9 summarises the soil acceptance criteria and discusses the interim criteria for dioxins and arsenic.

Section 5.10 discusses the application of the criteria. It deals with site assessment and management issues, particularly in circumstances in which it may be appropriate to use criteria different from those recommended in the guidelines.

5.2 ISSUES ADDRESSED IN THE DEVELOPMENT OF THE SOIL ACCEPTANCE CRITERIA

5.2.1 Considerations in the Development of the Soil Acceptance Criteria

As outlined above, soil acceptance criteria have been developed for a range of land uses, giving consideration to the:

- Protection of the health of site users and those engaged in maintenance works at the sites, including consideration of exposure via the following routes:
  - ingestion of contaminated soil;
  - dermal absorption of contaminants from soil;
  - inhalation of contaminated particulates;
  - consumption of home-grown produce;
- Protection of public health by consideration of the uptake and accumulation of contaminants in plant life and livestock; and
- Protection of plant life and livestock health.

5.2.2 Issues Requiring Separate Consideration

Considerations not specifically addressed in the development of these soil acceptance criteria include:

- Protection of groundwater quality (refer Section 5.10.7);
- Protection of terrestrial ecosystems (refer Section 5.6).
- Generation of dust and its accumulation within buildings (refer Section 5.10.6);

5.3 DEFINITION OF LAND USES

5.3.1 Agricultural

In these guidelines, agricultural use 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, e.g. ingestion of contaminated soil.

5.3.2 Residential

This scenario 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. Regional councils may select the most appropriate value on a site-specific basis.

5.3.3 Commercial/Industrial

The commercial/industrial land use is designed to reflect 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.

5.4 DEVELOPMENT OF SOIL CRITERIA PROTECTIVE OF HUMAN HEALTH

5.4.1 Risk-based Approach

A risk-based approach has been adopted for the development of soil acceptance criteria protective of human health. The general framework for the development of health-based soil acceptance criteria is presented in the ANZECC Guidelines (ANZECC, 1992). The approach is consistent with the requirements of the RM Act that discharges to the environment have “no significant adverse effect” on human health, the environment and a range of other considerations. Risk assessment is a tool which can be used to determine whether an adverse impact is expected as a result of a given discharge, and to define contaminant concentrations in environmental media at which no adverse impact is expected.

The use of risk assessment in the derivation of soil acceptance criteria is consistent with current international practice. It has been used by the United States Environmental Protection Agency (USEPA), the Australian and New Zealand Environment and Conservation Council (ANZECC), the National Health and Medical Research Council (Australia, NHMRC) and the World Health Organisation (WHO). A risk-based approach was used as part of the derivation of the Drinking Water Standards for New Zealand (DWSNZ, MoH, 1995).

5.4.2 Health Risk Assessment Framework

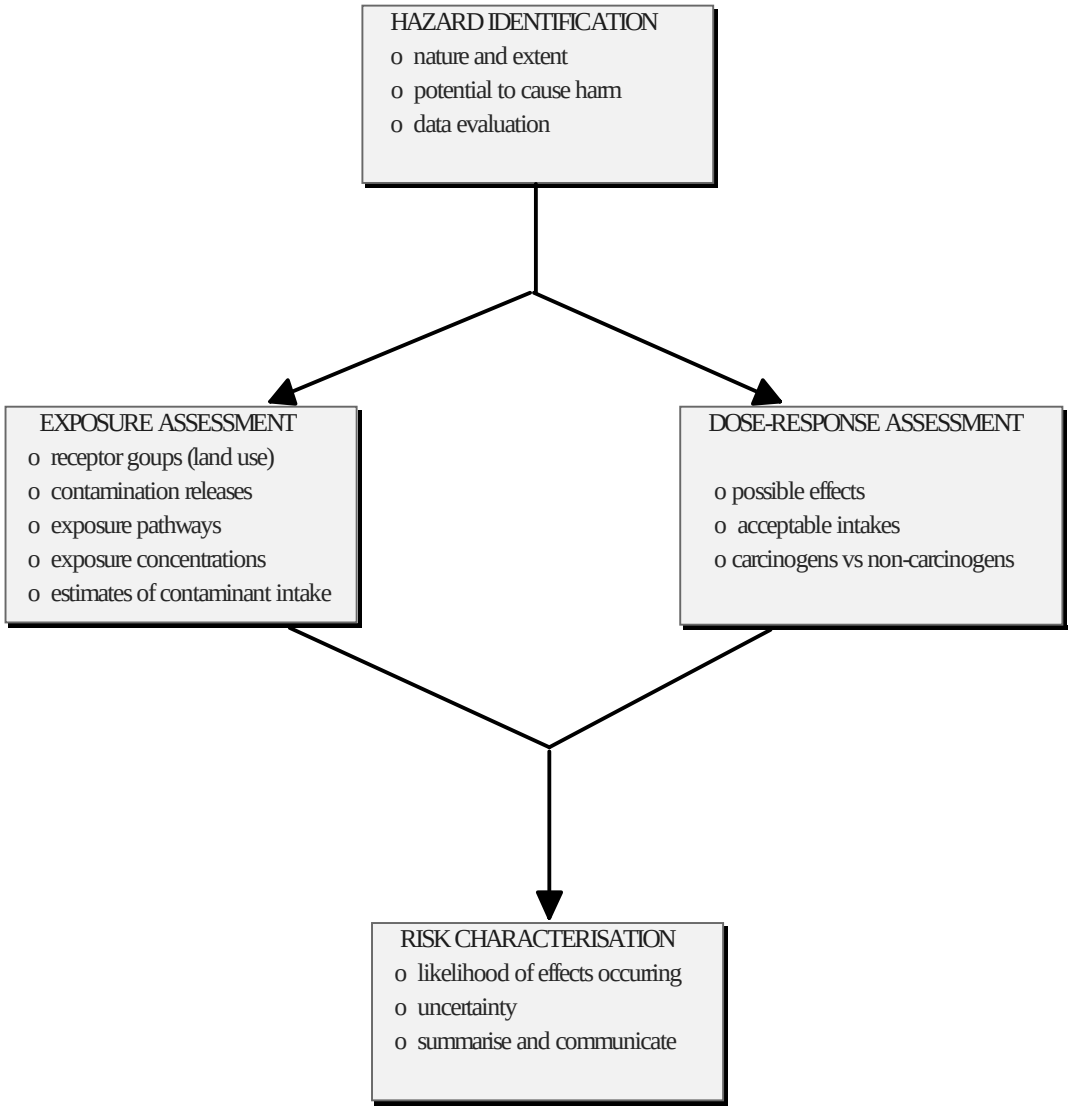
The following process has been developed for estimating soil acceptance criteria (refer Figure 5.1):

- preliminary assessment of the chemicals of concern based on reported levels at the site and safety data;
- identification of exposure paths for humans;
- estimation of the likely human exposure to each chemical of concern for significant exposure routes;
- estimation of the effects of human exposure from available animal, occupational health and epidemiology data;
- prediction of the allowable maximum concentration of chemicals in the soil which did not pose an unacceptable risk to human health.

The soil acceptance criteria are designed to be conservative[1] i.e. to be protective of human health at most sites.

[1] A conservative assessment overestimates the risk to human health and the environment, resulting in acceptance criteria that may be lower than would be necessary on the basis of an accurate assessment of the actual risk. A conservative approach is taken to ensure the protection of public health and the environment where uncertainty in the input parameters prevents a more accurate assessment of the risk.

Figure 5.1
Health Risk Assessment Framework



5.4.3 Risk Characterisation

The development of risk-based soil acceptance criteria involves working through the conventional risk assessment process in reverse, starting with target risk levels and using information about the toxicity of the contaminants and likely human exposure to determine soil concentrations which are consistent with the nominated risk targets.

A number of dose-response factors, such as reference dose (RfD) and slope factor (SF), have been developed by the United States Environmental Protection Agency (USEPA) and similar organisations to quantify the health risks associated with human exposure to various contaminants. The RfD and SF have been developed on the basis of available human and animal studies. The existing dose-response data are generally limited and are extrapolated to determine exposure levels that are consistent with a very low risk (typically 10^{-4} to 10^{-7}) to determine acceptance criteria. The available dose-response factors are summarised in Table 5.1. Published dose-response factors are conservative, and incorporate safety factors to account for the inherent uncertainties in such estimates.

Chemical contaminants may be divided into two broad groups according to their effects on human health, namely carcinogens and non-carcinogens. However, the division between carcinogens and non-carcinogens is becoming less distinct as the understanding of the range of mechanisms of carcinogenesis improves and as greater attention is focused on other effects such as reproductive toxicity.

The assessment of carcinogens is subject to considerable debate internationally. In particular, much of the debate has centred on whether carcinogens exhibit a threshold concentration below which no increase in cancer incidence may be expected. For these guidelines an approach has been taken which is consistent with that adopted in other New Zealand guidance and the WHO (1993) "Guidelines for Drinking Water Quality". It can be summarised as follows:

- Genotoxic carcinogens (i.e. carcinogens which act by damaging genetic material) are assumed to exhibit no threshold for carcinogenesis;
- Non-genotoxic carcinogens are assumed to exhibit a threshold dose below which no carcinogenesis may be expected.

A Technical Working Party on Cancer Risk Assessment, established by the Australian National Health and Medical Research Council, has developed a new approach for cancer risk assessment in relation to contaminated land. The draft discussion document, "Cancer Risk Assessment for Soil Contaminants", was still undergoing review when these guidelines were finalised and considerable further development and pilot testing would be required before the method could be applied.

(a) Non-threshold Carcinogens

For a non-threshold carcinogen the incremental lifetime risk of cancer is defined as follows (USEPA, 1989a):

Risk = CDI x SF (5.1)

Where: CDI = Chronic daily intake (refer Section 5.7.2)
SF = Slope factor (sometimes called Cancer potency factor)

The slope factor is an upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime, and can be used to derive the probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a potential carcinogen.

To allow comparison with published acceptable daily intakes (ADIs), a risk-specific dose (RSD) may be determined. The RSD is the chronic daily intake (CDI) which has been determined for a given risk using equation 5.1.

Any decision regarding the level of risk deemed to be acceptable requires consultation with the wider community. For the purpose of deriving generic soil acceptance criteria an incremental lifetime risk of cancer of 1 in 100,000 per substance has been adopted. The acceptable level of cancer risk has been the subject of public consultation as part of the development of these guidelines and the "Drinking Water Standards for New Zealand."

Key considerations in the selection of this level of risk are:

- The "Drinking Water Standards for New Zealand" (MoH, 1995) and the "Guidelines for Drinking Water Quality" (WHO, 1993) adopt an acceptable incremental lifetime risk of cancer associated with exposure to each substance in drinking water of 1 in 100,000.
- [A notable exception to this is arsenic which has been associated with an increased incidence of skin cancer in populations exposed through drinking water. Arsenic-related skin cancers are readily operable and not necessarily fatal, therefore the guideline value has been nominated to correspond with a risk of fatality from arsenic-related skin cancer in the order of 1 in 100,000.]
- The "Australian Drinking Water Guidelines" (NHMRC/ARMCANZ, 1996) nominate guideline values for genotoxic carcinogens based on consideration of: (i) a non-threshold dose-response relationship and an incremental lifetime risk of cancer of 1 in 1,000,000; (ii) a threshold dose-response model incorporating an additional factor of safety; and (iii) detection limit.
- The USEPA has indicated that risk values in the range 10^{-6} to 10^{-4} may be acceptable, depending on factors which include the size and nature of the exposed population, with 10^{-6} as a nominal threshold of concern for exposures to the general public.

The USEPA (1990b) indicates a maximum total risk level of 10^{-4} may be acceptable in some occupational contexts.

- The Dutch "National Environmental Policy Plan, Premises for Risk Management" (Directorate General for Environmental Protection at the Ministry of Housing, Physical Planning and Environment, 1989) nominates a maximum permissible individual risk of mortality per substance for existing substances of 10^{-6}/year (lifetime risk of 7 in 100,000). The maximum permissible individual risk of mortality for all existing substances combined is defined as 10^{-5}/year (lifetime risk of 7 in 10,000). The acceptable level of risk

for new substances is set at 10^{-8}/year and 10^{-7}/year for individual substances and for all substances combined respectively.

- Occupational exposure standards, such as those developed by the American Conference of Governmental Industrial Hygienists, generally correspond to an incremental lifetime risk of cancer (calculated using USEPA procedures) greater than 10^{-5} (Paustenbach D.J., 1990). However, such exposure standards generally do not provide for sensitive sub-populations, or provide full protection for all workers.

In general, the cancer risk associated with exposure to multiple contaminants, or exposure via multiple routes, is assumed to be additive. The cancer risk associated with each substance is, initially, determined separately, and the overall impact of all exposures is considered subsequently. (Potential antagonistic and synergistic effects have not been addressed in the derivation of soil criteria.)

The model of carcinogenicity underlying the USEPA approach assumes that the dose and consequent risk associated with exposure to carcinogens are cumulative over a lifetime. On this basis a weighted lifetime average CDI is estimated for use in equation 5.1.

(b) Non-carcinogens and Threshold Carcinogens

For non-carcinogenic species (and non-genotoxic carcinogens) a chronic hazard quotient is defined as follows (USEPA, 1989a):

$$HQ = \frac{CDI}{RfDc} \qquad (5.2)$$

Where:
HQ = Hazard quotient
CDI = Chronic daily intake (refer Section 5.7.2)
RfDc = Chronic reference dose

The chronic reference dose is an estimate of a daily exposure level for the human population, including sensitive sub-populations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Chronic RfDs are specifically developed to be protective for long-term exposure to a compound. They may have uncertainties spanning an order of magnitude.

The RfD is similar to the acceptable daily intake (ADI) or provisional tolerable weekly intake (PTWI) nominated by the WHO.

An HQ of 1 is appropriate to protect sensitive population groups (i.e. the chronic reference dose as developed by the USEPA allows for sensitive sub-populations).

Where more than one species has the same health effect (site and mechanism) or where exposure to a species may occur by more than one route, the HQ for each combination is summed to give a hazard index, HI. In the absence of further information, exposure to each substance is considered separately. Where information is available regarding

possible synergistic effects associated with simultaneous exposure to multiple contaminants, the risk associated with such exposure should be considered.

The toxicological model underlying the USEPA assessment approach for non-carcinogenic health effects assumes that effects and doses are not necessarily cumulative over a lifetime. The USEPA RfDs for chronic health effects have been developed for an exposure duration of several years. On this basis a year-average chronic daily intake is used to estimate the HQ in equation 5.2.

Children may be exposed to soil contaminants over a period of years at a rate significantly higher than that for adults (on a weight-standardised basis). If the exposures of children and adults are combined for the assessment of non-carcinogenic health effects over, say, the 30-year exposure duration for a residential scenario, then the year-averaged CDI for children would be underestimated, as would the likelihood of adverse health effects. Consequently, the assessment of non-carcinogenic health effects for residential and agricultural land uses is based on a year-average CDI for children only. This does not include consideration of those substances for which there is toxicological evidence to support special consideration of early childhood exposures.

Table 5.1
Summary of Dose-response Factors

Contaminant	Carcinogenic Category[11]	Parameter	Source				Value Adopted
			IRIS (1996)	DWSNZ (1995)	WHO DWG (1993)	WHO Other	
Arsenic	A	Oral RfD Inhal RfD Oral SF Inhal SF	3 x 10^{-4} mg/kg/d 1.5 $(mg/kg/d)^{-1}$ 4.3 x $10^{-3}/(\mu g/m^3)$[1]	2 x 10^{-3} mg/kg/d[5] 6 x $10^{-5}(\mu g/L)$[2] 0.0075 $(\mu g/m^3)$[3]	 6 x $10^{-5}(\mu g/L)$[2] 	2 x 10^{-3} mg/kg/d[5] 0.0075 $(\mu g/m^3)$[3]	2 x 10^{-3} mg/kg/day 0.15 $(mg/kg/d)^{-1}$ 15 $(mg/kg/d)^{-1}$
Boron	D	Oral RfD Inhal RfD Oral SF Inhal SF	9 x 10^{-2} mg/kg/d RfC = 2 x 10^{-2} mg/m^3[10]				9 x 10^{-2} mg/kg/d
Chromium (III)	D	Oral RfD Inhal RfD Oral SF Inhal SF	1.0 mg/kg/d			0.0003 mg/kg/d[7]	1.0 mg/kg/d 0.0003 mg/kg/d
Chromium (VI)	A	Oral RfD Inhal RfD Oral SF Inhal SF	5 x 10^{-3} mg/kg/d 1.2 x $10^{-2}/(\mu g/m^3)$[1][9]			4 x $10^{-2}/(\mu g/m^3)$[8], [4]	5 x 10^{-3} mg/kg/d 41 $(mg/kg/d)^{-1}$
Copper	D	Oral RfD Inhal RfD Oral SF Inhal SF		0.5 mg/kg/d 0.5 mg/kg/d			0.5 mg/kg/d 0.5 mg/kg/d
Penta-chlorophenol	B2	Oral RfD Inhal RfD Oral SF Inhal SF	0.03 mg/kg/d 0.12 $(mg/kg/d)^{-1}$	0.003 mg/kg/d		0.03 mg/kg/d(6) 0.003 mg/kg/d[6]	0.12 $(mg/kg/d)^{-1}$ 0.12 $(mg/kg/d)^{-1}$
Dioxins and Furans	B2	Oral RfD Inhal RfD Oral SF Inhal SF					10 pg/kg/d (TE)[12,13]

(1) Inhalation unit risk, corresponds to an inhalation slope factor of 1.5 $(mg/kg/d)^{-1}$
(2) Corresponds to a slope factor of 2.1 $(mg/kg/d)^{-1}$ or 0.15 $(mg/kg/d)^{-1}$ after allowance for 7% skin cancer fatality
(3) Corresponds to an inhalation slope factor of 10.5 $(mg/kg/d)^{-1}$
(4) Corresponds to an inhalation slope fraction of 140 $(mg/kg/d)^{-1}$
(5) WHO (1989) "The Evaluation of Certain Food Additives and Contaminants"
(6) WHO (1987a) "Environmental Health Criteria 71, Pentachlorophenol"
(7) WHO (1988) "Environmental Health Criteria 61, Chromium"
(8) WHO (1987b) "Air Quality Guidelines for Europe"
(9) Corresponds to an inhalation slope factor of 41 $(mg/kg/d)^{-1}$
(10) Reference Concentration, USEPA (1992b) Health Effects Summary Tables
(11) USEPA carcinogenic category (USEPA, 1992a)
(12) NTG (1992)
(13) Provisional

5.5 ASSESSMENT OF HEALTH EFFECTS FOR TIMBER TREATMENT CHEMICALS

5.5.1 Arsenic

Inorganic arsenic can cause a range of adverse chronic carcinogenic or non-carcinogenic health effects in humans. Long-term exposure to inorganic arsenic has been associated with an increased incidence of lung cancer in populations exposed primarily via inhalation, and an increased incidence of skin cancer as a result of arsenic contamination of drinking water supplies. The IARC classifies arsenic as a Group 1 human carcinogen (i.e. confirmed human carcinogen, USEPA Class A). In addition, chronic arsenic exposure may result in other non-carcinogenic effects on major human body organs.

There is a higher degree of uncertainty associated with the USEPA carcinogenicity assessment for arsenic than with those for other carcinogens (USEPA, 1992a), and the estimates based on the published slope factors may be overly conservative by at least one order of magnitude.

There has been considerable debate regarding the carcinogenicity of arsenic via the oral route and the form of the dose-response relationship. The failure to detect significant increases in cancer incidence at lower doses has been attributed to either a lack of statistical power in the epidemiological investigations or the existence of a practical threshold for effects (USEPA, 1987 in Langley, 1991). However, the evidence to support a threshold approach is inadequate (Chen, 1992, ASTDR, 1989).

The USEPA (1996) has nominated an oral RfD for the assessment of non-carcinogenic effects associated with exposure to arsenic, and an oral slope factor and a unit risk for carcinogenic effects resulting from arsenic in drinking water (refer Table 5.1) as well as an inhalation unit risk for carcinogenic effects.

The Joint FAO/WHO Expert Committee on Food Additives developed a provisional tolerable weekly intake (PTWI) for oral exposure to inorganic arsenic of 0.015 mg/kg (0.002 mg/kg/day), and noted that there is a very small margin between the PTWI and the likely background exposures (WHO, 1989). The Committee recognised the necessity of recommending tolerable intakes that took account of the natural occurrence of high arsenic levels in water supplies in some parts of the world.

The WHO PTWI was adopted by Langley (1991) as part of the Health Risk Assessment Workshop arranged by the South Australia Health Commission. The Health Investigation levels for arsenic developed at this workshop for residential land use were adopted in the ANZECC Guidelines.

The WHO "Guidelines for Drinking Water Quality" (WHO, 1993) provide an estimate of cancer risk associated with exposure to arsenic in drinking water. The principal studies used to develop the estimates of cancer risk are based on observations of a Taiwanese population (USEPA, 1992a, WHO, 1996). The cancer risk estimates derived from these studies are likely to overestimate the actual cancer risk, partly because other possible causes of skin cancers in the population were not quantified and partly due to dose-dependent variations in metabolism (WHO, 1996). Only 1-14% of the arsenic induced skin cancers were fatal. Consequently, for the proposed WHO drinking water guideline of 10 µg/L the incremental lifetime risk of cancer was estimated to be 6×10^{-4}, but the lifetime risk of fatal skin cancer was estimated to be in the range of 6×10^{-6} to 8.4×10^{-5} (WHO, 1992).

The above risk-based guideline agrees well with a drinking water guideline derived from the WHO PTWI (WHO, 1989), assuming 20% of the total arsenic exposure is due to arsenic in drinking water. The WHO and USEPA cancer risk estimates have been based on studies of exposure to arsenic in drinking water, and it is noted that exposure to arsenic in contaminated soils may result in different rates of increased skin cancer, in part due to reduced absorption of the arsenic from the soil, and other factors.

The DWSNZ established a maximum acceptable value for arsenic in drinking water of 0.01 mg/L based on an approach similar to that adopted in the WHO "Guidelines for Drinking Water Quality" (WHO, 1993).

The "Air Quality Guidelines for Europe" (WHO, 1987) nominate a unit risk for inhalation exposure to arsenic.

Ferguson (1995) reviewed published information and concluded that arsenic should be regarded as a genotoxic carcinogen and should be assessed using a non-threshold dose-response model. On this basis, dose-response factors consistent with those adopted in the derivation of the DWSNZ have been used in considering the cancer risk associated with exposure via the oral route. For the derivation of soil acceptance criteria a proportion of arsenic-related skin cancers that are fatal of 7% has been adopted, together with an incremental lifetime risk of fatality from cancer of 1 in 100,000. The USEPA inhalation unit risk has been adopted for the assessment of cancer risk via the inhalation route.

5.5.2 Boron

Boron has not been classified by the USEPA or WHO with regard to carcinogenicity. A range of animal studies indicate significant non-carcinogenic health effects for humans are likely to result from chronic exposure, albeit at exposures above those normally experienced by humans under most conditions. Amongst other effects, boron is known to adversely affect the male reproductive system in rodents and dogs. The USEPA reference dose was developed by application of an uncertainty factor of 100 to the highest no observed adverse effect level (NOAEL) in a lifetime study of dogs exposed to boron (USEPA, 1992a).

5.5.3 Chromium

Chromium is found in two oxidation states which show significantly different health effects. In many studies there has been significant difficulty in separating the effects associated with these different forms.

Trivalent Chromium – Cr(III)

Cr(III) is a sensitising agent which can cause skin reactions, asthma and irritation of mucous membranes. Cr(III) has not been evaluated by the USEPA with regard to carcinogenicity.

Residents living near ferro-alloy plants reported no adverse health effects at ambient air concentrations of 1 $\mu g/m^3$ (WHO, 1988). A Cr(III) air concentration of 1 $\mu g/m^3$ corresponds to an adult intake of 0.0003 mg/kg/d based on an inhalation rate of 20 m^3/d.

An inhalation reference (RfD) dose for Cr(III) of 0.0051 mg/kg/d (USEPA, 1984) was based on occupational exposure standards proposed by the American Conference of Governmental Industrial Hygienists (ACGIH). Since 1988, the occupational exposure standard (TLV) proposed by the ACGIH has been reduced by a factor of 10, suggesting the RfD should also be reduced to 0.0005 mg/kg/d.

On the basis of the WHO report, an inhalation RfD of 0.0003 mg/kg/d has been adopted for the purposes of this study.

A USEPA working group is currently working on a revised RfD for CR(III) by the inhalation route. An inhalation RfD of 5.7×10^{-7} mg/kg/d has been proposed (USEPA, 1991c); however, it is subject to formal review and ratification by the relevant USEPA work groups. An RfD or reference concentration (RfC) was deleted from the Health Effects Assessment Summary Tables (USEPA, 1992b), pending resolution of the Agency's position.

The inhalation RfD adopted in these guidelines should be reviewed following adoption of a revised RfD by the USEPA.

Hexavalent Chromium – Cr(VI)

Chronic occupational exposure to chromic acid above 1 $\mu g/m^3$ has been associated with adverse health effects (WHO, 1988). Cr(VI) has been associated with increased incidence of lung cancer and has been classified by the USEPA as a Group A human carcinogen (IARC Group I) by the inhalation route. In addition, occupational exposure to Cr(VI) has been associated with bronchitis, chronic lung congestion, skin irritation and sensitising and irritation and ulceration of the mucous membranes.

Both the WHO and USEPA have published unit risk estimates for exposure to Cr(VI) by inhalation, as shown in Table 5.1. The WHO unit risk estimate is approximately three times higher than the USEPA value. The USEPA unit risk factor has been converted to the slope factor format for this study.

There is some evidence to suggest that skin irritation effects associated with long-term exposure to Cr(VI) may exhibit a threshold of approximately 1,000 mg/kg. On this basis, even allowing for sensitive sub-populations, a Cr(VI) concentration of 100 to 200 mg/kg may not pose a risk to health with regard to skin irritation (Soong, 1993).

5.5.4 Copper

Copper is not classified as a carcinogen by the USEPA (Group D) given the lack of adequate data. Human exposure to copper compounds may result in a range of adverse health effects; however, such effects are associated with higher-level, acute exposures.

The maximum acceptable value (MAV) nominated for copper in the DWSNZ is based on the provisional tolerable daily intake of 0.5 mg/kg/day nominated by the Joint FAO/WHO Expert Committee on Food Additives in 1982. This value is regarded as uncertain due to the age of the information on which it is based. It has, however, been used in deriving the soil acceptance criteria. Only 10% of the tolerable intake has been assigned to contaminated soil exposure, due to the relatively high intake of copper from other sources.

5.5.5 Pentachlorophenol (PCP)

Chronic exposure of humans to PCP has been associated with a range of non-carcinogenic health effects. Such effects include irritation of the skin, mucous membranes and respiratory tract, signs of chloracne, neurasthenia, depression, headaches, porphyria and changes in kidney and liver function. Recent animal studies have provided evidence that technical-grade PCP is also carcinogenic; however, human studies for high-exposure groups, such as timber treatment workers, have not provided consistent evidence of increased cancer rates.

There appears to be a consensus regarding the oral NOAEL (rat) which is the basis for setting the various ADIs or RfDs for the non-carcinogenic effects of PCP. The WHO (1987a) employs an additional factor of 10 to account both for PCP's greater toxicity by inhalation (a factor which was not considered by the USEPA (1987a, 1992a)), and for the additional uncertainty associated with inter-species extrapolation (the human steady-state body burden is approximately 10 to 20 times that of rats).

In 1991, the USEPA (1992a) reclassified technical-grade PCP as a Group B2 carcinogen by both oral and inhalation exposure routes and assigned a slope factor of 0.12 $(mg/kg.bw/d)^{-1}$ for oral exposure. Ferguson (1995) reviewed the available information regarding the genotoxicity of PCP and concluded that it should be regarded as genotoxic. On this basis the USEPA slope factor, incorporating an

assumed non-threshold dose-response model, has been used to derive soil acceptance criteria.

While the USEPA has nominated dose-response factors for both carcinogenic and non-carcinogenic effects, criteria based on the carcinogenic effects are limiting, and acceptance criteria have been prepared on this basis.

Commercial or technical-grade PCP and NaPCP formulations may contain PCDD and PCDF impurities produced during manufacture. In some selected NaPCP formulations, the concentration of PCDD/PCDF impurities ranged from 0.194 to 1.85 µg/g (mean 0.99 µg/g) (Bingham, 1990). Such PCDD/PCDF impurities may be associated with PCP in the soil environment.

Some of the adverse health effects associated with exposure to technical-grade PCP may be caused by PCDD/PCDF impurities; however, the information required to confirm this is not readily available. In addition, PCP was used as the technical-grade with the associated impurities, and the impurities are frequently found in the soil environment with PCP. On this basis it is considered appropriate to develop soil acceptance criteria for PCP which are based on toxicological information for the technical-grade product and which account, in part, for the action of impurities. Note, however, the comments in the following section on the fate of dioxins in the environment.

5.5.6 Dioxins and Furans

2, 3, 7, 8-Tetrachlorodibenzodioxin (TCDD) is regarded as a probable human carcinogen by the USEPA and by other groups such as the WHO. As part of the Pentachlorophenol Risk Assessment Pilot Study (NTG, 1992) a maximum allowable daily intake (analogous to RfD) of 10 pg/kg.bw/d (TE) was adopted, following a review of the available literature, for the purposes of developing soil acceptance criteria. The NATO Toxic Equivalence Factors (TEF) have been used to express the concentration of other chlorinated dioxins and furans in terms of a 2, 3, 7, 8-TCDD toxic equivalent (TE) concentration.

The toxicology of PCDD/PCDF is the subject of ongoing research, and the nominated maximum allowable daily intake may require revision as further information becomes available. The toxicology of PCDD/PCDF will be reviewed in detail as part of the Organochlorines Programme.

The development of soil acceptance criteria based on toxicological information for technical-grade PCP makes some allowance for the effects of impurities in PCP formulations, such as PCDD/PCDF. However, PCP and PCDD/PCDF exhibit different environmental fate and transport properties such that soils may contain PCDD/PCDF concentrations out of proportion with the PCDD/PCDF content of the original formulations (e.g. as a result of leaching of PCP from surface soils). Thus it is necessary to consider soil criteria for both PCP and PCDD/PCDF.

5.5.7 Assessment of Chemical Mixtures

Published toxicological information generally relates to exposure to single chemicals, whereas in practice exposure to multiple contaminants is likely to occur, consistent with the composition of the formulations used. As outlined previously, it is common practice to sum cancer risks and HQs where non-carcinogenic effects are known to occur at the same site (in the body) and by similar toxicological mechanisms. Synergistic or antagonistic interaction between chemicals may occur; however, information in this regard is limited.

At timber treatment sites, contaminants are likely to occur in the following combinations:

- Copper, chromium and arsenic

  Exposure to copper, chromium and arsenic via the oral route may result in a range of health effects; however, the most sensitive effect (and therefore the effect used to determine the dose-response factor) differs between each contaminant. Published information regarding possible interactions between copper, chromium and arsenic was not identified.

  In practice, at most timber treatment sites arsenic is responsible for most of the estimated risk to human health.

- Pentachlorophenol and dioxins and furans

  Information about possible synergistic effects between PCP and dioxins and furans (or other impurities) is limited. Some of the health effects associated with exposure to technical-grade PCP may be associated with dioxins and furans or other impurities. The dose-response factors nominated for PCP have been, at least in part, based on toxicological studies using technical-grade PCP, and therefore some consideration of interactive effects has been included.

Due to the limited information available, the recommended soil acceptance criteria have been based on the assumption that each chemical acts independently. However, given the comments above regarding the significance of arsenic in CCA-contaminated soil and the use of technical-grade PCP, the criteria are considered appropriate for direct application. Separate consideration may be given to interactions between contaminants where there is reason to believe these may be significant.

5.5.8 Correction for Background Exposures

Significant exposure to many substances may be experienced from sources other than contaminated land: e.g. food, drinking water and air. For those substances for which an acceptable intake or RfD has been set, such acceptable intakes include exposure from other sources. In order to use the acceptable intake to calculate a health-based criterion for a substance in soil it is important to first make allowance for exposure from other sources. That fraction of the acceptable intake that is not accounted for by other sources may then be allocated to exposure to contaminated soil and an acceptance criterion determined on this basis. For some substances the contribution

from other sources, or background exposure, is negligible. The portion of the acceptable intake assigned to contaminated soil exposures is detailed in Table 5.2.

It should be noted that exposures from sources other than contaminated land are indicative only. They are based on published information from New Zealand, Australia, the USA and Canada and will vary with location, eating practices etc. Where background exposures significantly different from those listed in Table 5.2 may be experienced, the health-based acceptance criteria should be modified to account for this. In most cases diet (including drinking water) is the primary source of exposure to contaminants other than contaminated soil.

Table 5.2
Correction of Dose-response Factors for Typical Background Exposures

Contaminant	Exposure Route	Acceptable Intake[7] (mg/kg.bw/d)	Intake from Sources Other than Soil Contamination (mg/kg.bw/d)	Acceptable Intake Assigned to Contaminated Soil (mg/kg.bw/d)
Arsenic	Oral Inhalation	6×10^{-5} [8] 2×10^{-7}	Approx. 1×10^{-3}[2] ND	6×10^{-5} [1] 2×10^{-7}[1]
Boron	Oral	9×10^{-2}	4.5×10^{-2} [9]	4.5×10^{-2}
Cr(III)	Oral Inhalation	1.0 3×10^{-4}	2×10^{-3}[3] 2×10^{-6}[3]	1.0 3×10^{-4}
Cr (VI)	Oral Inhalation	5×10^{-3} 2×10^{-7}	ND ND	5×10^{-3} 2×10^{-7}[1]
Copper	Oral	0.5	4.5[4]	5×10^{-2}
Pentachloro-phenol	Oral Inhalation	8.3×10^{-5} 8.3×10^{-5}	1×10^{-5}[5]	8.3×10^{-5} [1] 8.3×10^{-5} [1]
Dioxins and Furans	Oral	10 pg/kg/d	1.5 pg/kg/d[6]	8 pg/kg/d

Notes: (1) The acceptable intake listed is based on an incremental lifetime risk of cancer of 1 in 100,000. The cancer risk level adopted as a threshold of concern is expressed in terms of an incremental risk associated with each exposure, and is therefore in addition to cancer risk associated with exposure to other sources (e.g. food).

(2) Langley, 1991; ATDSR, 1992

(3) WHO, 1988; Soong, 1993

(4) 10% of acceptable intake assigned to contaminated soil exposure, consistent with the derivation of the DWSNZ. 90% assigned to other exposures.

(5) Unpublished report, Environment Canada, 1992

(6) Hannah, 1992

(7) Acceptable intake corresponds to the reference dose for non-carcinogens and the dose corresponding to an incremental lifetime risk of cancer of 1 in 100,000 for carcinogens.

(8) Adjusted to reflect the assumption that only 7% of the arsenic-related skin cancers are fatal.

(9) 50% of acceptable intake is assigned to soil exposures. The current DWSNZ value corresponds to 10% of the acceptable intake and is under review.

5.6 PROTECTION OF PLANT LIFE AND OTHER ECOLOGICAL RECEPTORS

5.6.1 Ecotoxicity

The primary focus in assessing the possible ecological effects associated with a current or former timber treatment facility is on the protection of the off-site ecosystems, particularly through consideration of the impact of groundwater discharges and surface runoff on surface water quality and the associated aquatic ecosystems. Limited consideration is given to the protection of the on-site environment consistent with use of the site. For example, in the context of residential and agricultural use, consideration is given to the protection of plant life.

This focus reflects the fact that the establishment of a timber processing facility is likely to have a major impact on the local ecosystem, irrespective of any soil or water contamination issues. If the site were to be redeveloped for a more sensitive use, it is unlikely that protection of on-site ecosystems would be a major requirement, given past impacts (although some protection consistent with the proposed use might be required).

Assessing the impact of combinations of timber treatment chemicals on natural ecosystems is dependent on such a range of factors that it is best undertaken on a site-specific basis. However, various organisations have in recent years developed methodologies to assess potential ecosystem impacts associated with contaminated sites, and for the establishment of soil quality guidelines that are protective of on-site ecosystems. Such generic guidelines can be useful in screening sites for significant ecological impact.

The Dutch authorities have developed a methodology for determining the maximum tolerable risk (MTR) value for various contaminants in soil and water, for the protection of ecosystems. The Dutch approach is based on experimental no-observable adverse effect levels (NOAEL) and on the requirement to provide full protection to 95% of the species in the target ecosystem. There are two concerns, in particular, that have arisen out of this approach:

- The significance of protecting 95% of species has not been determined, although it is understood further work is under way in the USA to examine the impact of protecting 85% or 90% of species.
- Some of the maximum tolerable risk values determined for heavy metals indicate a significant environmental risk exists at concentrations below the natural background soil concentrations.

Thus, although the Dutch approach to ecotoxicological assessment is well-documented, the practical application of the resulting maximum tolerable risk values is uncertain.

Some published criteria for the protection of aquatic ecosystems have been included in Chapter 6 – Acceptance Criteria for Surface Water and Groundwater. These guidelines may be used as a screening tool to assist in assessing the impact on ecosystems associated with nearby surface water bodies. However, where a possible impact is identified, a detailed site-specific evaluation should occur, taking into account the sensitivity of the affected ecosystem.

5.6.2 Phytotoxicity

Many contaminants are toxic to plant life, although the available data regarding phytotoxicity is limited for many contaminants, particularly organic compounds. Some heavy metals are necessary at trace levels for healthy plant growth; however, they become toxic above a threshold level and may, for example, reduce crop yields. Threshold concentrations at which a number of heavy metals may begin to exhibit phytotoxic effects are presented in Table 5.3. Note that this information is not based on tests of New Zealand species.

The soil concentration at which a heavy metal becomes toxic to plant life depends on a range of factors, including plant type, soil type, soil pH and the form in which the contaminant is present. Phytotoxicity is the major concern with some heavy metals, particularly copper and Cr(III), in residential use (Alloway, 1990, p. 275) and is a significant concern with boron in residential soils.

In order to obtain a better estimate of the impact of ground contamination on plant life, it is necessary to assess the bioavailability of the contaminant. A number of tests are available to estimate the bioavailability of heavy metals (e.g. EDTA digestion) and data is available relating the results of such tests to the onset of phytotoxic effects (Charman, 1991).

In many soils, Cr(III) is not readily available for plant uptake and consequently potential phytotoxic effects are reduced. Thus, toxicity of Cr(III) to plants is unlikely except in extremely acid soils and this species is therefore often regarded as non-toxic; however, Cr(VI) is more available for plant uptake and is more toxic to plants (Alloway, 1990, p. 143).

Information on the phytotoxicity of PCP was collated as part of the development of the Dutch intervention values (Denneman, 1990). Reported no observed effect concentrations (NOEC) for individual species ranged from 0.32 mg/kg to 32 mg/kg whereas EC_{50} values ranged from 3.2 mg/kg to 100 mg/kg. Based on the information collated as part of the development of the Dutch intervention values, impacts on plant life appear to be among the more sensitive ecological effects associated with PCP. This information is useful in assessing the possibility of adverse effects on plant life,

but is limited in that it is not specific to New Zealand species. The Dutch intervention value for PCP is 5 mg/kg, which may be regarded as a reasonable threshold for possible phytotoxic effects, given the range of plant data summarised above.

Available information relating specifically to the effects of various contaminants on plant life may augmented by guideline values developed on the basis of ecological protection which would consider impacts on plant life, soil microflora and microfauna, invertebrates and a range of other relevant ecological receptors.

Environment Canada has proposed ecologically based soil acceptance criteria for arsenic as follows: agricultural/residential/parkland use, 19 mg/kg; commercial/ industrial use, 26 mg/kg (Environment Canada, 1997). The criteria for copper are: 63 mg/kg for agricultural/residential/parkland use; 100 mg/kg for commercial/ industrial use. CCME nominated an interim environmental quality objective for soil of 2 mg/kg for boron, but the basis for this is not stated (CCME, 1992).

The Dutch intervention values incorporate consideration of both human health and ecological considerations. In derivation of the intervention values an ecotoxicological intervention value is derived and combined with the health-based intervention value. The estimated ecotoxicological intervention values are summarised as (Swartjes, 1993, 1994): arsenic, 40 mg/kg; chromium, 230 mg/kg; copper, 190 mg/kg; and boron, 7 mg/kg.

Table 5.3
Summary of Phytotoxicity
(Alloway, 1990, CCME, 1991, Charman, 1991)
(All values expressed as total metal concentrations unless otherwise specified)

Contaminant	Approximate Threshold for Plant Toxicity (mg/kg)	
	Acid Sandy Soils	Neutral Clay Soils
Arsenic	10 (1) 20 (3)	25 (3)
Boron	3 (water soluble) (2)(7)	3 (water soluble) (2)(7)
Cr(VI)	25 (1)(4)	
Cr(III)	600 (1)	
Copper	130 (1)(2)	500-1000 (5)

Notes:
(1) Alberta Environment (1990) "Alberta Tier 1 Criteria for Contaminated Soil Assessment and Remediation" (DRAFT) (as cited in CCME, 1991).
(2) UK Interdepartmental Committee on the Redevelopment of Contaminated Land (ICRCL).
(3) Ontario Ministry of the Environment (1989) (as cited in CCME, 1991).
(4) UK mine spoil soil trigger for crop growth.
(5) Estimated on the basis of copper bioavailability in neutral clay soils.

(6) Thresholds for plant toxicity generally based on species exhibiting low tolerance.

(7) Analysis for soluble boron using a hot water extraction or similar.

5.7 HUMAN EXPOSURE ASSESSMENT

5.7.1 Site Uses, Receptor Groups and Exposure Routes

(a) General

The potential future site uses considered in the development of acceptance criteria and the associated exposure routes are summarised in Table 5.4.

Table 5.4
Receptor Groups and Exposure Routes

Site Use	Receptor Group	Exposure Routes
Agricultural/ Horticultural	Child residents Adult residents/workers Sub-surface maintenance workers	Ingestion of soil Inhalation of particulates Dermal absorption (PCP only) Ingestion of produce
Residential	Child residents Adult residents Sub-surface maintenance workers	Ingestion of soil Inhalation of particulates Dermal absorption (PCP only) Ingestion of produce
Industrial-Unpaved	Adult workers Sub-surface maintenance workers	Ingestion of soil Inhalation of particulates Dermal absorption (PCP only)
Industrial-Paved	Adult workers Sub-surface maintenance workers	Ingestion of soil Inhalation of particulates Dermal absorption (PCP only)

The contaminants of concern are of sufficiently low volatility that inhalation of vapours may be discounted. Further, copper, chromium, arsenic and boron are generally not readily absorbed dermally when present in a soil matrix, and consequently the dermal absorption route is only considered for pentachlorophenol.

(b) Sensitive Receptors

Subsurface maintenance works may expose workers to contaminants at levels higher than may be expected for routine site use, although generally the frequency of exposure would be less than for other site users. Sub-surface maintenance workers have been considered under each land use; however, criteria based on protection of this receptor group have only been found to be limiting for the paved industrial land use.

Young children are generally the most sensitive receptor group in residential use, due to their relatively high incidental soil ingestion rate and low body weight (increasing the weight-standardised exposure rate).

Some children have been observed to ingest large quantities of soil and other non-food material. Such behaviour is referred to as pica and is estimated to affect 1% of children between 0 and 5 years of age. Soil ingestion rates for children exhibiting pica range from 1 to 10 g/day, compared with a typical child soil ingestion rate of 0 to 100 mg/day. Children with pica thus represent a particularly sensitive sub-population. They have not been included as part of the derivation of soil acceptance criteria because:

- the probability that a child exhibiting pica would be resident on a former timber treatment site is relatively low;
- provision is made for the notification of contaminated land though the Land Information Memorandum (LIM) system, allowing site owners to take appropriate precautions if children exhibiting pica are resident at the site;
- soil criteria have been developed on the basis of the "reasonable maximum exposure" (refer Section 5.7.2) rather than the absolute worst case exposure scenario. Consideration of children with pica may be regarded as an absolute worst case scenario;
- the proposed approach is consistent with current practice internationally and the approach set out in the ANZECC Guidelines.

5.7.2 Exposure Estimation

(a) General Approach

Health-based soil acceptance criteria have been derived on the basis of an estimate of the reasonable maximum exposure (RME) likely to be experienced by site users or workers. The RME combines upper bound and average exposure factors to give an exposure scenario that is both protective and reasonable, and that is not the absolute worst case but represents a reasonable maximum exposure (USEPA, 1991b). The use of RME therefore seeks to provide full protection to almost all people, should they come in contact with contamination at a particular site. Due to the conservatism inherent in the risk assessment process, which is seeking to provide a high level of protection to the majority of the population, even highly exposed individuals are unlikely to be exposed to an appreciable risk of an adverse effect.

The approach to the assessment of health risk outlined in the ANZECC Guidelines is consistent with the use of RME, as is subsequent guidance, such as the Health-Based Soil Investigation Levels (NEHF, 1996). The use of RME to derive preliminary remediation goals for the protection of human health has been common practice in the USA (USEPA, 1989a).

The approach used in these guidelines for exposure assessment and development of the health-based-based acceptance criteria is based on the procedures developed by the USEPA (1989a, 1991d). In general, assumptions employed in the risk assessment are also based on recommendations by the USEPA (1989a, 1990b, 1991b, 1991d).

The estimated exposure (or intake) is normalised for time and body weight and is calculated as:

$$\text{Intake} = \frac{\text{Concentration x Contact Rate x Exposure Frequency x Exposure Duration}}{\text{Body Weight x Averaging Time}}$$

How this is applied to particular exposure routes is described in the following sections.

(b) Ingestion of Contaminated Soil

The chronic daily intake (CDI) may be determined from the following expression:

$$CDI = \frac{C \times CF \times IR_{adj} \times EF \times MF}{AT} \quad (5.3)$$

where:

- C = concentration of species in the soil
- CF = conversion factor = 10^{-6} kg/mg
- IR_{adj} = age-adjusted ingestion rate

$$= \frac{\Sigma\, ED_i \times IR_i}{BWi} \quad (5.4)$$

where:

- ED_i = exposure duration for receptor group ‘i’ (yr)
- IR_i = ingestion rate for receptor group ‘i’ (mg/d)
- BW_i = body weight for receptor group ‘i’ (kg)
- EF = exposure frequency
- MF = matrix factor, accounts for reduced bioavailability of contaminant due to binding to the soil matrix. In the absence of necessary information, MF is usually taken as 1.0 (USEPA, 1989a).
- AT = averaging time = (ED x 365) days for non-carcinogens by convention or (70 years x 365) days for carcinogens, representing lifetime exposure, by convention (USEPA, 1989a)

The CDI (mg/kg/day) is determined by using equation 5.3 is a weighted average, taking account of variation in body weight and ingestion rate with age.

(c) Inhalation of Contaminated Dusts

The chronic daily intake (CDI) by inhalation may be determined from the following expression:

$$CDI = \frac{C \times IH_{adj} \times CF \times EF \times MF \times R \times PEF}{AT} \quad (5.5)$$

where: C = concentration of species in the soil

IH_{adj} = age-adjusted inhalation rate

$$= \frac{\Sigma\, EDi \times IHi}{BWi} \quad (5.6)$$

where: ED_i = exposure duration for receptor group 'i' (yr)

IH_i = inhalation rate for receptor group 'i' (m^3/d)

BW_i = body weight for receptor group 'i' (kg)

CF = conversion factor
= 10^{-6} kg/mg

EF = exposure frequency

MF = matrix factor

R = proportion of particulates retained in lungs
= 0.75 (Hawley, 1985)

PEF = particle emission factor (m^3/kg)
(relates soil contaminant concentration to air contaminant concentration)

$$1/PEF = TSP \times FC \times P_{RP} \times CF \quad (5.7)$$

where: TSP = concentration of total suspended particulates (mg/m^3)

FC = fraction of suspended particulates from a contaminated source

P_{RP} = respirable fraction, i.e. proportion of suspended particulates <10 μm diameter

AT = averaging time
= (ED x 365) days for non-carcinogens by convention or (70 years x 365) days for carcinogens, a lifetime by convention

Refer to Appendix C for estimation of the PEF, and details of assumptions relating to FC and TSP.

The CDI determined using equation 5.5 is a weighted average, taking account of variation in body weight and inhalation rate with age.

(d) Dermal Absorption from Contaminated Soil

The chronic daily intake (CDI) for dermal absorption from contaminated soil may be determined by using the following expression:

$$CDI = \frac{C \times AR_{adj} \times AH \times AF \times EF}{AT} \quad (5.8)$$

where: C = concentration of species in the soil (mg/kg)

AR_{adj} = age-adjusted area of exposed skin

$$= \frac{\Sigma\, ARi \times EDi}{BW_i} \quad (5.9)$$

where: AR_i = area of exposed skin for receptor group 'i' (cm^2)
ED_i = exposure duration for receptor group 'i' (yr)
BW_i = body weight for receptor group 'i' (kg)

AH = soil adherence ($mg/cm^2/day$)

AF = absorption factor (unitless)

Assuming that 11% of an organic in a solvent base is absorbed by an adult over a 24 hr period and that a 15% matrix factor may be applied when the organic is applied in a soil matrix, for a 12 hr period, AF(PCP) = 0.008; and for an 8 hr period, AF(PCP) = 0.006. The above approach is based on information provided by Kimborough and others, as presented in GRI (1988). The above factors are typical values only. Actual contaminant absorption is dependent on a range of factors, including the concentration of the contaminant in the soil.

EF = exposure frequency

AT = averaging time
= (ED x 365) days for non-carcinogens by convention or (70 years x 365) days for carcinogens, a lifetime, by convention

The CDI determined in equation 5.8 is a weighted average, taking into account variation in body weight, skin area and exposure patterns with age.

(e) Ingestion of Produce

The chronic daily intake (CDI) for ingestion of produce may be estimated using the following expression:

$$CDI = \frac{CP \times IP_{adj} \times EF \times Pg}{AT} \quad (5.10)$$

where: CP = concentration of contaminant in produce (mg/kg) (refer Appendix A for determination of the contaminant uptake by plants)

IP_{adj} = age-adjusted ingestion rate for produce

$$= \frac{\Sigma\ IP_i \times ED}{BW_i} \quad (5.11)$$

where: IP_i = produce ingestion rate for receptor group 'i' (kg/d)
ED_i = exposure duration rate for receptor group 'i' (yr)
BW_i = body weight rate for receptor group 'i'(kg)
EF = exposure frequency (d/yr)
Pg = proportion of produce grown on-site
AT = averaging time
= (ED x 365) days for non-carcinogens by convention or (70 yr x 365) days for carcinogens by convention

The CDI estimated in equation 5.10 is a weighted average taking into account variation in body weight and produce consumption with age.

5.8 DEVELOPMENT OF SOIL ACCEPTANCE CRITERIA FOR SITE USES

5.8.1 Agricultural

(a) Protection of Human Health

Soil guideline values for the protection of human health are based on reasonable maximum case exposure assumptions (refer Table 5.5 and Table 5.6). The major exposure assumptions are based on published typical average and upper bound values and may be summarised as:

- exposure duration = 30 yr, assuming exposure from 0 to 30 yr of age

  The exposure duration is based on the reasonable maximum time spent on the one site in a rural context based on USEPA (1989a,b).

- exposure frequency = 350 d/yr (USEPA, 1989b)

  Studies have shown that a child is likely to spend less than 200 days/year playing outside. However, Hawley (1985) estimated that 80% of indoor dirt is derived from local soil, meaning a child may be exposed whenever on-site, not just outdoors.

- body weight: child (1-6 yr) = 15 kg (USEPA, 1991a)
  adult (7-31 yr) = 70 kg (USEPA, 1989b)
- soil ingestion rate: child (1-6 yr) = 00 mg/d (ANZECC, 1992)
  adult (7-31 yr) = 25 mg/d
- inhalation rate: child (1-6 yr) = 3.8 m^3/d (Langley, 1993)
  adult (7-31 yr) = 20 m^3/d (USEPA, 1989a)
- skin surface area: child (1-6 yr) = 2625[1] cm^2 (Langley, 1993)
  adult (7-31 yr) = 4700 cm^2

[1] Corresponds to value for 1 to 11 year old children.

- soil adherence: 1 mg/cm^2 allowing for soil contact typical of farming activities (USEPA, 1989a)
- ingestion of produce: child (1-6 yr) = 0.13 kg/d (Langley, 1993)
  adult (7-31 yr) = 0.45 kg/d

  These produce consumption rates are based on the median consumption rates from the 1985 Australian National Dietary Survey. For adults, the consumption patterns may be subdivided as follows:
  - Leafy vegetables: 31%
  - Root vegetables: 29%
  - Fruit: 40%
- proportion of produce grown on-site = 100%
- ambient total suspended particulate concentration = 0.020 mg/m^3, based on information provided in EPAV (1990), for a rural environment.

A number of assumptions relating to the types of agricultural use are inherent in these calculations, and higher exposures to contamination via dermal and inhalation routes may arise where intensive soil preparation and/or crop production is practised.

The contribution of sources other than contaminated soil to the overall intake of a given substance is discussed in Section 5.5.8. An allowance has been included for typical intake from sources such as diet, drinking water and ambient air. Where specific information is available about background exposure to arsenic, copper, chromium, boron or pentachlorophenol at a given site, the proposed guideline values may be corrected to account for this. Where significantly elevated background exposures are likely, e.g. elevated arsenic concentrations in some areas of New Zealand, then acceptance criteria lower than the proposed guidelines may be appropriate on a site-specific basis.

(b) Protection of Plants and Livestock

The impact of ground contamination on plant life and livestock may involve the following factors:

- protection of health of residents who may consume home-grown produce;
- protection of the health of consumers of livestock products;
- protection of plant life (phytotoxicity);
- protection of livestock health.

The issues concerning the protection of the health of consumers of produce[2] and phytotoxicity are considered in Section (a) above and Section 5.6.2 respectively.

[2] Section (a) considers the health risk associated with consumption of home-grown produce. Consumers of produce grown for commercial purposes will also be protected.

Uptake of Contaminants by Plants

Contaminant residues in produce depend on a range of factors affecting the uptake of contaminants by plants, as discussed in Appendix A. A preliminary estimate of the relationship between contaminant concentrations in the soil and those in produce can be made, based on published data and correlations developed on behalf of the Dutch and Canadian authorities (ECETOC, 1990, AERIS, 1992). The uptake of contaminants by plants is highly variable, and depends on soil type and chemistry, soil organic matter content, the age of contamination and other factors.

Table 5.5
Preliminary Health Risk Based Acceptance Criteria
Agricultural Site Use

Site Use:	Agricultural	Exposure Frequency:	350 d/yr	Body Weight (1-6 yrs):	15 kg	Produce Ing. (1-6 yrs, kg):	0.13
Receptor:	Children resident onsite for up to 30 yrs and adult workers	Averaging Time (carc.):	70 yrs	Body Weight (7-31 yrs):	70 kg	Produce Ing. (7-30 yrs, kg):	0.45
		(non-carc.):	ED yrs	Exposure Dur. (1-6 yrs):	6 yrs	Proportion of produce from contaminated source:	1
		Age adjusted	49 mg.yr/kg.d	Exposure Dur. (7-31 yrs):	24 yrs		
Target Risk:	1.00E-05	Ingestion Factor:		Ingestion Rate (1-6 yrs):	100 mg/d	Skin Area (1-6 yrs) (sq. cm):	2625
Target HI:	1	Age adj. inhalation rate:	8.4 cu.m.yr/kg.d	Ingestion Rate (7-31 yrs):	25 mg/d	Skin Area (7-30 yrs) (sq. cm):	4700
		Particle Emission Factor:	250000000 cu.m/kg	Inhalation Rate (1-6 yrs):	3.8 cu.m/d	Soil Adherance (mg/sq.cm):	1
		Age adjusted dermal exposure factor:	2661	Inhalation Rate (7-31 yr):	20 cu.m/d	Skin Absorption Factor (PCP):	0.006
				Particulate Retention:	0.75		

					Acceptable CDI						Preliminary Health-based Soil Acceptance Criteria (mg/kg)							
Contaminant	SF (1/(mg/kg/d))		RfD (mg/kg/d)		Carcinogenic			Non-carcinogenic			Carcinogenic				Non-carcinogenic			
	Oral	Inhalation	Oral	Inhalation	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Produce	Oral	Inhalation	Dermal	Produce
Arsenic	0.147	15			6.8027E-5	6.667E-07					1.0E+02	1.9E+03		4.4E+00				
Boron			4.50E-02					4.50E-02							7.0E+03			3.0E+00
Chromium (III)			1.00E+00	3.00E-04				1.00E+00	3.00E-04						1.6E+05	4.1E+05		1.1E+04
Chromium (VI)		41	5.00E-03			2.439E-07		5.00E-03				7.1E+02			7.8E+02			4.4E+00
Copper			5.00E-02	5.00E-02				5.00E-02	5.00E-02						7.8E+03	6.9E+07		3.9E+01
PCP	0.12	0.12			8.3333E-05	8.3333E-05	8.3333E-05				1.3E+02	2.4E+05	3.8E+02	6.9E-01				

Note: Preliminary health-based soil acceptance criteria for carcinogens based on average exposure over 30 years.
Preliminary health-based soil acceptance criteria for non-carcinogens based on exposure over the first six years for a child.

Table 5.6
Preliminary Health Risk Based Acceptance Criteria
Agricultural Site Use
Estimation of Target Soil Conceptions: Produce Based

Contaminant	Target Produce Concentration		Soil Conc/ Root Conc.	Soil Conc/ Stem Conc	Soil Conc/ Fruit Conc.	Target Soil Concentration	
	Carcinogenic	Non-carcinogenic				Carcinogenic	Non-carcinogenic
Arsenic	2.4E-02		100	500[2]	NA[1]	4.4E+00	
Boron		5.4E+00	0.3	1.5	NA		3.0E+00
Chromium (III)		1.2E+02	50	250	NA		1.1E+04
Chromium (VI)		6.0E-01	4	20	NA		4.4E+00
Copper		6.0E+00	3.5	17.5	NA		3.9E+01
PCP	2.9E-02		12.8	63.9	NA	6.9E-01	

Proportion of total fruit and vegetable consumption:

Root: 0.29
Stem: 0.31
Fruit: 0.40

Note: (1) NA, assuming negligible translocation of contaminants to fruit.
(2) Assuming stem/leaf concentration 20% of root concentration.

Research into the uptake of arsenic by plants has been undertaken in both New Zealand and Australia. Research into the relationship between arsenic accumulation in orchard soils and concentrations in plant matter indicated that, in most cases, concentrations of arsenic in edible plant portions did not exceed recognised health-based food standards (Merry, 1983, 1986). Further, in most cases the onset of plant toxicity was noted before the accumulation of significant concentrations of arsenic in plant matter. The uptake of arsenic by plants depends on the extent to which it is adsorbed on soil constituents such as iron oxides, clay particulates, and organic matter, which in turn is highly dependent on the soil type and contamination history (Merry, 1993).

Research completed by Landcare Research in New Zealand has shown that arsenic concentrations of potential significance for human health can accumulate, particularly in root crops (Yeates, 1993). Significant uptake of copper and chromium was also noted, particularly for root crops; however, the acceptable concentrations for these species in produce are higher than for arsenic. The research focused on pasture contaminated by contaminants. Other work investigating the uptake of contaminants by plants is presented in Appendix A.

Because the uptake of contaminants by plants is highly dependent on soil conditions and crop species each site should be evaluated independently. However, the criteria presented in Tables 5.5 and 5.6 are useful for screening purposes.

Uptake of Contaminants by Livestock

Lipophilic compounds such as PCP can accumulate within the fatty tissue of livestock that may be grazed on contaminated land. No MRL has been established for PCP in New Zealand; however, soil criteria protective of public health may be developed by considering the uptake of contaminants by livestock and the patterns of consumption of livestock products (e.g. milk and meat) by the general public.

Soil criteria for PCP, based on the accumulation of PCP in livestock and later consumption by the public, corresponding to an incremental lifetime risk of cancer of 1 in 100,000 are presented in Table 5.7. Refer to Appendix B for details of the derivation of the acceptance criteria.

The estimates presented in Table 5.7 should be regarded as indicative only, as the exact concentrations of contaminants in livestock products will depend on site-specific conditions.

The uptake and accumulation of heavy metals by livestock is expected to be limited, and correlations relating heavy metal concentrations in soil to those in livestock have not been identified. Consequently no maximum allowable metal concentrations in agricultural soils, based on uptake by livestock, have been derived.

Table 5.7
Acceptance Criteria for PCP in Agricultural Soils
Used for the Production of Meat and Milk

Produce	Estimated Human Exposure (mg/d) for Soil Concentration of 1 mg/kg	Soil Concentration (mg/kg) Corresponding to a Cancer Risk of 1 in 100,000
Meat	3.7×10^{-4}	
Milk	1.8×10^{-4}	
Combined	5.5×10^{-4}	11

Protection of Stock Health

Preliminary soil acceptance criteria for the protection of stock health are derived in Appendix D. The criteria are based on extrapolation from guideline values for stock watering (that are protective of stock health) presented in Chapter 6 and hence should be regarded as indicative.

The preliminary soil acceptance criteria derived on this basis are presented in Table 5.8. They indicate that protection of livestock health is not likely to be a limiting consideration for the agricultural land use.

Table 5.8
Preliminary Soil Acceptance Criteria Protective of Livestock Health

Contaminant	Preliminary Soil Acceptance Criteria (mg/kg)(1)
As	38
Cr	76
Cu	38-380
B	380
PCP	11

Note: (1) Because preliminary soil acceptance criteria are based on extrapolation from stockwater limits they may be regarded as available soil concentrations. In some cases higher total contaminant concentrations may have no adverse effect on livestock health.

5.8.2 Residential

(a) Protection of Human Health

Soil guideline values have been developed on the basis of reasonable maximum exposure assumptions. They are:

- exposure duration = 30 yr, assuming exposure from 0 to 30 years of age

  The exposure duration is based on the reasonable maximum time spent on the one site in a rural residential context, based on USEPA (1989a,b).

- exposure frequency = 350 days/year (USEPA, 1989b)

  Studies have shown that a child is likely to spend less than 200 days/year playing outside; however, Hawley (1985) estimated that 80% of indoor dirt is derived from local soil, meaning a child may be exposed whenever on-site, not just outdoors.

- body weight: child (1-6 yr) = 15 kg (USEPA, 1989b)
  adult (7-31 yr) = 70 kg

- soil ingestion rate: child (1-6 yr) = 100 mg/d (ANZECC, 1992)
  adult (7-31 yr) = 25 mg/d

- inhalation rate: child (1-6 yr) = 3.8 m^3/d (Langley, 1993)
  adult (7-31 yr) = 20 m^3/d (USEPA, 1989a)

- exposed skin surface area: child (1-6 yr) = 2625 cm^2 (Langley, 1993)
  adult (7-31 yr) = 4700 cm^2

- soil adherence: 0.5 mg/cm^2 (USEPA, 1989a)

- produce ingestion rate: child (1-6 yr) = 0.13 kg/d (Langley, 1993)
  adult (7-31 yr) = 0.45 kg/d
  (refer also to the additional information presented under section 5.8.1 above)

- proportion of produce grown on-site:
  High: 50% (eg. rural residential)
  Average: 10% (eg. urban residential)

- ambient total suspended particulates concentration = 0.026 mg/m^3, based on information provided in EPAV (1990).

The contribution of sources other than contaminated soil to the overall intake of a given substance is discussed in Section 5.5.8. An allowance has been included for typical intake from sources such as diet, drinking water and ambient air. Where specific information is available about background exposure to arsenic, copper, chromium, boron or pentachlorophenol at a given site, the proposed guideline values may be corrected to account for this. Where significantly elevated background exposures are likely, e.g. elevated arsenic concentrations in some areas of New Zealand, then acceptance criteria lower than the proposed guidelines may be appropriate on a site-specific basis.

The health-based soil acceptance values for residential land use are presented in Tables 5.9-5.12.

(b) Protection of Plant Life

Soil contaminant concentrations at which some adverse impacts on plant life may be expected are summarised in Table 5.3. Although the contaminant concentrations corresponding to the onset of plant toxicity are dependent on a range of factors, the values presented in Table 5.3 represent relatively conservative limits for the nominated soil types.

5.8.3 Industrial – Unpaved

Human health is the primary on-site concern where an ongoing industrial use is proposed. Where off-site transport of contaminants via soil movement, groundwater or surface water is likely, off-site environmental or health impacts may be controlling. The human-health-based preliminary remediation goals have been based on reasonable maximum exposure assumptions. These are:

- exposure duration = 20 yr (reasonable maximum time in one job)
- soil ingestion rate = 25 mg/day (for workers not directly involved in excavation) (ANZECC, 1992)
- inhalation rate = 9.6 m^3/d (based on 8 hour working day) (Langley, 1993)
- skin surface area = 4700 cm^2 (Langley, 1993)
- soil adherence = 1.0 mg/cm^2 (USEPA, 1989a)
- ambient total suspended particulate concentration = 0.14 mg/m^3 (GRI, 1988) for a typical industrial site without construction activities.

The health-based soil guideline values for an unpaved industrial site use are presented in Table 5.13.

Table 5.9
Preliminary Health Risk Based Acceptance Criteria Residential Site Use (High Home-grown Produce)

Site Use:	Residential	Exposure Frequency:	350 d/yr	Body Weight (1-6 yrs):	15 kg	Produce Ing. (1-6 yrs, kg):	0.13
Receptor:	Children resident onsite for up to 30 yrs	Averaging Time (carc.):	70 yrs	Body Weight (7-31 yrs):	70 kg	Produce Ing. (7-30 yrs, kg):	0.45
		(non-carc.):	ED yrs	Exposure Dur. (1-6 yrs):	6 yrs	Proportion of produce from contaminated source:	0.5
		Age adjusted	49 mg.yr/kg.d	Exposure Dur. (7-31 yrs):	24 yrs		
Target Risk:	0.00001	Ingestion Factor:		Ingestion Rate (1-6 yrs):	100 mg/d	Skin Area (1-6 yrs) (sq. cm):	2625
Target HI:	1	Age adj. inhalation rate:	8.4 cu.m.yr/kg.d	Ingestion Rate (7-31 yrs):	25 mg/d	Skin Area (7-30 yrs) (sq. cm):	4700
		Particle Emission Factor:	190000000 cu.m/kg	Inhalation Rate (1-6 yrs):	3.8 cu.m/d	Soil Adherance (mg/sq.cm):	0.5
		Age adjusted dermal exposure factor:	2661	Inhalation Rate (7-31 yr):	20 cu.m/d	Skin Absorption Factor (PCP):	0.006
				Particulate Retention:	0.75		

					Acceptable CDI						Preliminary Health-based Soil Acceptance Criteria (mg/kg)							
Contaminant	SF (1/(mg/kg/d))		RfD (mg/kg/d)		Carcinogenic			Non-carcinogenic			Carcinogenic				Non-carcinogenic			
	Oral	Inhalation	Oral	Inhalation	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Produce	Oral	Inhalation	Dermal	Produce
Arsenic	0.147	15			6.8E-05	6.7E-07					1.0E+02	1.5E+03		8.9E+00				
Boron			4.5E-02					4.5E-02							7.0E+03			6.0E+00
Chromium (III)			1.0E+00	3.0E-04				1.0E+00	3.0E-04						1.6E+05	3.1E+05		2.2E+04
Chromium (VI)		41	5.0E-03			2.4E-07		5.0E-03				5.4E+02			7.8E+02			8.9E+00
Copper			5.0E-02	5.0E-02				5.0E-02	5.0E-02						7.8E+03	5.2E+07		7.8E+01
PCP	0.12	0.12			8.3E-05	8.3E-05	8.3E-05				1.3E+02	1.8E+05	7.6E+02	1.4E+00				

Note: Preliminary health-based soil acceptance criteria for carcinogens based on average exposure over 30 years.
Preliminary health-based soil acceptance criteria for non-carcinogens based on exposure over the first six years for a child.

Table 5.10
Preliminary Health Risk Based Acceptance Criteria
Residential Site Use (High Home-grown Produce)
Estimation of Target Soil Concentrations:- Produce Based

Contaminant	Target Produce Concentration (mg/kg)		Soil Conc/ Root Conc.	Soil Conc/ Stem Conc	Soil Conc/ Fruit Conc.	Target Soil Concentration	
	Carcinogenic	Non-carcinogenic				Carcinogenic	Non-carcinogenic
Arsenic	4.8E-02		100	500[2]	NA[1]	8.9E+00	
Boron		1.1E+01	0.3	1.5	NA		6.0E+00
Chromium (III)		2.4E+02	50	250	NA		2.2E+04
Chromium (VI)		1.2E+00	4	20	NA		8.9E+00
Copper		1.2E+01	3.5	17.5	NA		7.8E+01
PCP	5.9E-02		12.7	63.5	NA	1.4E+00	

Proportion of total vegetable consumption:
Root: 0.29
Stem: 0.31
Fruit: 0.40

Note: (1) NA, assuming negligible translocation of contaminants to fruit.
(2) Assuming stem/leaf concentration 20% of root concentration.

Table 5.11
Preliminary Health Risk Based Acceptance Criteria
Residential Site Use (Typical Home-grown Produce)

Site Use:	Residential	Exposure Frequency:	350 d/yr	Body Weight (1-6 yrs):	15 kg	Produce Ing. (1-6 yrs, kg):	0.13
Receptor:	Children resident onsite for up to 30 yrs	Averaging Time (carc.):	70 yrs	Body Weight (7-31 yrs):	70 kg	Produce Ing. (7-30 yrs, kg):	0.45
		(non-carc.):	ED yrs	Exposure Dur. (1-6 yrs):	6 yrs	Proportion of produce from	0.1
		Age adjusted	49 mg.yr/kg.d	Exposure Dur. (7-31 yrs):	24 yrs	contaminated source:	
Target Risk:	0.00001	Ingestion Factor:		Ingestion Rate (1-6 yrs):	100 mg/d	Skin Area (1-6 yrs) (sq. cm):	2625
Target HI:	1	Age adj. inhalation rate:	8.4 cu.m.yr/kg.d	Ingestion Rate (7-31 yrs):	25 mg/d	Skin Area (7-30 yrs) (sq. cm):	4700
		Particle Emission Factor:	190000000 cu.m/kg	Inhalation Rate (1-6 yrs):	3.8 cu.m/d	Soil Adherance (mg/sq.cm):	0.5
		Age adjusted dermal	2661	Inhalation Rate (7-31 yr):	20 cu.m/d	Skin Absorption Factor (PCP):	0.006
		exposure factor:		Particulate Retention:	0.75		

					Acceptable CDI						Preliminary Health-based Soil Acceptance Criteria (mg/kg)							
Contaminant	SF (1/(mg/kg/d))		RfD (mg/kg/d)		Carcinogenic			Non-carcinogenic			Carcinogenic				Non-carcinogenic			
	Oral	Inhalation	Oral	Inhalation	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Produce	Oral	Inhalation	Dermal	Produce
Arsenic	0.15	15			6.8E-05	6.7E-07					1.0E+02	1.5E+03		4.4E+01				
Boron			4.5E-02					4.5E-02							7.0E+03			3.0E+01
Chromium (III)			1.0E+00	3.0E-04				1.0E+00	3.0E-04						1.6E+05	3.1E+05		1.1E+05
Chromium (VI)		41	5.0E-03			2.4E-07		5.0E-03				5.4E+02			7.8E+02			4.4E+01
Copper			5.0E-02	5.0E-02				5.0E-02	5.0E-02						7.8E+03	5.2E+07		3.9E+02
PCP	0.12	0.12			8.3E-05	8.3E-05	8.3E-05				1.3E+02	1.8E+05	7.6E+02	6.9E+00				

Note: Preliminary health-based soil acceptance criteria for carcinogens based on average exposure over 30 years.
Preliminary health-based soil acceptance criteria for non-carcinogens based on exposure over the first six years for a child.

Table 5.12
Preliminary Health Risk Based Acceptance Criteria
Residential Site Use (Typical Home-grown Produce)
Estimation of Target Soil Concentrations:- Produce Based

Contaminant	Target Produce Concentration (mg/kg)		Soil Conc/ Root Conc.	Soil Conc/ Stem Conc	Soil Conc/ Fruit Conc.	Target Soil Concentration	
	Carcinogenic	Non-carcinogenic				Carcinogenic	Non-carcinogenic
Arsenic	2.4E-01		100	500[2]	NA[1]	4.4E+01	
Boron		5.4E+01	0.3	1.5	NA		3.0E+01
Chromium (III)		1.2E+03	50	250	NA		1.1E+05
Chromium (VI)		6.0E+00	4	20	NA		4.4E+01
Copper		6.0E+01	3.5	17.5	NA		3.9E+02
PCP	2.9E-01		12.7	63.5	NA	6.9E+00	

Proportion of total vegetable consumption:

Root: 0.29
Stem: 0.31
Fruit: 0.40

Note: (1) NA, assuming negligible translocation of contaminants to fruit.
(2) Assuming stem/leaf concentration 20% of root concentration.

Table 5.13
Preliminary Health Risk Based Acceptance Criteria
Unpaved Industrial Site Use

Site Use:	Unpaved Industry	Exposure Frequency:	240 d/yr	Ingestion Rate:	25 mg/d
Receptor:	Adult Workers	Averaging Time (carc.):	70 yrs	Inhalation Rate:	9.6 cu.m/d
		(non-carc.):	ED yrs	Particle Emission Factor:	29000000 cu.m/kg
Target Risk:	0.00001	Exposure Duration:	20 yrs	Skin Area:	4700 sq.cm
Target HI:	1	Body Weight:	70 kg	Soil Adherance (mg/sq.cm):	1 mg/sq.cm
				Skin Absorption Factor:	0.006
				Particulate Retention:	0.75

Contaminant	SF (1/(mg/kg/d))		RfD (mg/kg/d)		Acceptable CDI						Preliminary Health-based Soil Acceptance Criteria (mg/kg)					
					Carcinogenic			Non-carcinogenic			Carcinogenic			Non-carcinogenic		
	Oral	Inhalation	Oral	Inhalation	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal
Arsenic	0.15	15			6.8E-05	6.7E-07					1.0E+03	1.0E+03				
Boron			4.5E-02					4.5E-02						1.9E+05		
Chromium (III)			1.0E+00	3.0E-04				1.0E+00	3.0E-04					NL	1.3E+05	
Chromium (VI)		41	5.0E-03			2.4E-07		5.0E-03				3.7E+02		2.1E+04		
Copper			5.0E-02	5.0E-02				5.0E-02	5.0E-02					2.1E+05	NL	
PCP	0.12	0.12			8.3E-05	8.3E-05	8.3E-05				1.2E+03	1.3E+05	1.1E+03			

Note: Preliminary health-based soil acceptance criteria for carcinogens based on average exposure over 20 years.
NL – denotes not limiting.

5.8.4 Industrial – Paved

The protection of human health is the primary on-site concern where an ongoing industrial site use is proposed. Where contaminated areas are fully paved and the integrity of the paving is maintained, exposure to non-volatile soil contaminants should be eliminated. The effectiveness of pavement as a barrier to the exposure of workers to ground contamination is, however, highly dependent on the integrity and design of the pavement and on the nature of the underlying soils. Spreading and other transport of contaminated soil from areas where contaminated soil is unpaved or from areas of failed pavement may mean that protection against worker exposure is compromised.

Similarly, the acceptable contaminant concentrations in soil on a paved industrial site may be controlled by exposures associated with ongoing maintenance of subsurface services or works. Such exposures may be effectively mitigated by the use of an appropriate site management plan requiring, for example, the use of protective clothing and equipment whenever the integrity of the pavement is compromised by subsurface works, and the diligent clean-up of soil and repair of the damaged areas. An estimate of maintenance worker exposure is presented in Section 5.8.5 and typical issues that should be addressed in a management plan are outlined in Appendix F.

The assessment of worker exposure for paved and partially paved sites is likely to be dictated by site-specific factors and each site should therefore be assessed with reference to:

- extent of paving,
- condition of pavements, and
- site management plans.

In this document, a range of acceptance criteria has been assigned for each contaminant. The lower values represent a situation where a nominal five-fold protection factor applies and contaminated soil areas are exposed either as a result of failure of pavements and/or non-paving of contaminated soils and contribute some 20% of the dust which workers in the area would ingest. If a management plan which can ensure appropriate protection of workers and the ongoing integrity of the paving or capping is in place, then contaminant concentrations would not be limited. This approach and the underlying assumptions should be critically reviewed on a site-specific basis. A summary of proposed acceptance criteria for industrial use is presented in Table 5.14.

Table 5.14
Proposed Acceptance Criteria for Industrial Use
(mg/kg, dry basis)

Contaminant	Unpaved Industrial	Maintenance Worker Protection (refer 5.8.5)	Paved Industrial	
			No Management Plan[2]	Management Plan[3]
Arsenic	500	650	650	2500 – NL[4]
Boron	NA[1]	NA	NA	NA
Cr(III)	NA	NA	NA	NA
Cr(VI)	360	510	510	1800 – NL
Copper	NA	NA	NA	NA
Pentachlorophenol	570	1000	1000	2800 – NL
Dioxins and Furans	0.018[5]	0.021[6]	0.021[6]	0.09 – NL

Notes: (1) NA denotes an estimated health-based criterion >10,000 mg/kg.

(2) Based on requirement for protection of maintenance workers.

(3) Ranging from a nominal five-fold protection factor applied to criteria developed for an unpaved industrial area, to a not limited condition for situations where the management plan ensures appropriate protection of workers and the integrity of the paving or capping.

(4) NL denotes not limited.

(5) NTG (1992).

(6) NTG (1992) modified to reflect slightly different exposure factors adopted in the development of these guidelines.

5.8.5 Maintenance

For each of the above site uses, human exposure to ground contamination may be associated with subsurface maintenance works, e.g. 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.

In order to develop reasonable but protective soil guideline values goals for adult workers involved in subsurface maintenance, the following exposure factors have been assumed:

- exposure duration = 20 yr, 90% upper bound for time spent in one job (USEPA, 1989b).
- soil ingestion rate = 100 mg/d (for workers directly involved in excavation (GRI, 1988).

- exposure frequency = 50 d/yr.
- inhalation rate = 9.6 m^3/d (Langley, 1993)
- skin soil adherence = 1.5 mg/cm^2 (USEPA, 1989a)
- ambient total suspended particulate concentration = 0.28 mg/m^3 (GRI, 1988).

It has been assumed that the maintenance workers described above are involved in routine servicing of the timber processing sites and consequently subsurface work at timber processing facilities may be part of the workers' main activity. The exposure frequency should be reviewed based on normal practice, which may vary from site to site, and the proposed acceptance criteria adjusted accordingly.

The above assessment also assumes that maintenance workers wear normal work clothes. The use of appropriate personal protective equipment may allow work within areas with contaminant concentrations in excess of the proposed criteria.

The health-based soil acceptance criteria for the protection of workers involved in subsurface maintenance works are presented in Table 5.15.

5.8.6 Uncertainties in Risk Assessment

The development of health-based acceptable soil concentrations entails a large number of assumptions to characterise possible future site use and the fate of contaminants in the soil environment. However, the objective of health risk assessment is not to characterise the actual health risks likely to be experienced by site users, but to estimate either typical or reasonable maximum exposure conditions on which to base future decision-making. The USEPA risk assessment procedures focus on the calculation of a reasonable maximum exposure (RME). The aim of the RME is to combine the various exposure factors "so that the result represents an exposure scenario that is both protective and reasonable; not the worst possible case" (USEPA, 1991b).

Parameters used in the assessment of exposure to soil contamination may be estimated from published typical data and local knowledge, and professional judgement is also required. Specific issues associated with the assessment of uncertainty are:

- The base data used to determine human response to chemical exposure are derived from limited human and animal response studies, which identify wide-ranging effects. Extrapolation of the available response data is required to:
  - account for the differences between animals and humans;
  - determine likely responses at low doses based on information about responses at high doses.

  Such extrapolation is conducted by applying a safety factor, typically 100, or by assuming that response is linearly related to dose and extrapolating from a limited population base (at high doses) to determine exposure levels consistent with very low risk levels, typically 10^{-4} to 10^{-6}.

Table 5.15
Preliminary Health Risk Based Acceptance Criteria
Subsurface Maintenance Works

Site Use:	All site uses	Exposure Frequency:	50 d/yr	Ingestion Rate:	100 mg/d
Receptor:	Workers involved in subsurface maintenance	Averaging Time (carc.):	70 yrs	Inhalation Rate:	10 cu.m/d
		(non-carc.):	ED yrs	Particle Emission Factor:	9000000 cu.m/kg
Target Risk:	0.00001	Exposure Duration:	20 yrs	Skin Area:	4700 sq.cm
Target HI:	1	Body Weight:	70 kg	Soil Adherance (mg/sq.cm):	1.5 mg/sq.cm
				Skin Absorption Factor:	0.006
				Particulate Retention:	0.75

Contaminant	SF (1/(mg/kg/d))		RfD (mg/kg/d)		Acceptable CDI						Preliminary Health-based Soil Acceptance Criteria (mg/kg)					
					Carcinogenic			Non-carcinogenic			Carcinogenic			Non-carcinogenic		
	Oral	Inhalation	Oral	Inhalation	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal
Arsenic	0.15	15			6.8E-05	6.7E-07					1.2E+03	1.4E+03				
Boron			4.5E-02					4.5E-02						2.3E+05		
Chromium (III)			1.0E+00	3.0E-04				1.0E+00	3.0E-04					NL	1.8E+05	
Chromium (VI)		41	5.0E-03			2.4E-07		5.0E-03				5.2E+02		2.6E+04		
Copper			5.0E-02	5.0E-02				5.0E-02	5.0E-02					2.6E+05	NL	
PCP	0.12	0.12			8.3E-05	8.3E-05	8.3E-05				1.5E+03	1.8E+05	3.5E+03			

Note: Preliminary health-based soil acceptance criteria for carcinogens based on average exposure over 20 years.
NL – denotes not limiting.

Published parameters, such as reference doses (RfD) and slope factors (SF), are used to relate exposure to human health risk, and hence incorporate the uncertainty resulting from extrapolation, as outlined above. However, the extrapolation method is such that the values published are reasonably conservative and protective of human health.

- The acceptance criteria based on the protection of plant life have been drawn from published information, and are appropriate for reasonable worst-case conditions of acidic sandy soils. Although there is some uncertainty associated with such values, the estimates are conservative and represent a reasonable worst case. Higher values may be present at some sites without an observable adverse effect on plant life due to the effect of site-specific considerations such as soil type.
- The estimated ratio of the contaminant concentration in produce or animal products to that in soil is typical of the upper part of the range of the values reported in the literature. The uptake of contaminants is expected to reflect, in part, the bioavailability of the contaminants in the soil environment and is thus highly site-specific and should be assessed on this basis. Lower uptake ratios may be expected at some sites, depending on specific factors.
- The guidelines have been developed on the basis of conservative estimates of typical soil ingestion rates which provide for full protection of the majority of the population. However, the amount of soil ingested is highly dependent on the behaviour of the individual. The childhood soil ingestion rate adopted for this study, 100 mg/day (ANZECC, 1992), is a conservative estimate of soil ingestion for a typical child. However, up to 1% of children ingest significant quantities of soil, paint chips, plaster etc., say, in the order of 1 to 10 g/day. Children who exhibit such a tendency are said to suffer from the condition pica. As discussed in Section 5.7.1, the guideline values may not be fully protective of children with pica; however, such behaviour is limited in incidence and duration.
- The bioavailability of each contaminant in the soil matrix is assumed to be 100%. In practice the bioavailability of contaminants in the soil environment is likely to reflect the source of the contaminant, the form in which it is present in the soil and the soil type and characteristics.

A summary of the uncertainties associated with the parameters used in exposure assessment is presented in Table 5.16 for a residential site use, by way of example. It is far more likely that the recommended acceptance criteria would give rise to human exposures that are below a threshold of concern with regard to human health than exposures above such a threshold of concern.

Because the estimated criteria are most likely to be conservative, possibly by up to one order of magnitude, it is not proposed to add an additional factor of safety to account for the uncertainty. This has been addressed in the selection of conservative values for exposure factors in the development of the soil guideline values.

Table 5.16
Summary of Uncertainties
Example Scenario – Residential

Parameter	Value	Estimated Range	Percentage Variation[(1)]
Exposure Duration	30 yr	9(av.) – 70 yr	+130%, -70%
Exposure Frequency	350 d/y	200-365 d/yr	+4%, -42%
Body Weight (adult)	70 kg	60-100 kg	+43%, -14%
Ingestion:			
Ingestion rate (mg/d)			
– adult	25 mg/d	5-100 mg/d	+300%, -80%
– child	100 mg/d	5-200 mg/d	+100%, -95%
Matrix factors	1.0	ND[(2)]	+0%, -95%
Dermal:			
Soil adherence	0.5 mg/cm^3	<0.5-1.5mg/cm^2	+200%
Skin surface area			
– adult	4700 cm^2	3000-11000cm^2	+134%, -36%
– child	2625 cm^2	1500-5000cm^2	+90%, -43%
Absorption Factor (PCP)	0.006	ND	
Inhalation:			
Particulate air concentration:	0.026mg/m^3	0.01-0.06mg/m^3	+130%, -60%
Particulate retention:	0.75	ND	
Respirable fraction:	0.46	0.4-0.6	+33%, -13%
Fraction from cont. soil:	0.2	0.05-0.5	+150%, -75%
Inhalation rate:			
– adult	20 m^3/d	15-30m^3/d	+50%, -25%
– child	3.8 m^3/d		
Produce			
Produce ingestion rate:			
– adult	0.45 kg/d	ND	
– child	0.13kg/d	ND	
Ratio soil conc/root conc (for arsenic)	100	10->500	>+400%, -90%
Proportion of produce grown	50%		100%, -96%
on-site	10%	2-100%	+900%, -80%

Notes: (1) Percentage variation of the estimated range compared to the value selected for each parameter.

(2) ND indicates no data.

5.9 RECOMMENDED SOIL ACCEPTANCE CRITERIA

Table 5.17 summarises the soil acceptance criteria developed in the preceding sections and the interim soil acceptance criteria nominated for arsenic and dioxins.

5.9.1 Derived Soil Acceptance Criteria

The process for selection of the derived soil acceptance criteria is:

(a) combine the health-based soil guideline values derived for each exposure route to give a single health-based acceptance criterion for each combination of site use and contaminant;

(b) for each contaminant select the lowest of the criteria developed for the protection of (as appropriate);
   – health of site users and workers;
   – health of the consumers of produce and livestock products;
   – plant life and livestock health.

(c) where the criterion selected above is less than the typical background concentration, adopt the background concentration as the criterion (estimated criterion noted in brackets in Table 5.17).

In some cases the values for the threshold of phytotoxicity are for a reasonable worst case, with the actual value being highly dependent on site-specific conditions. Higher contaminant concentrations may be acceptable on a site-specific basis depending on soil type and other site-specific conditions.

In the selection of the proposed acceptance criteria a lesser weighting has been assigned to resident ingestion of produce because of the higher degree of uncertainty associated with this exposure route. The actual levels of contaminants in produce grown on contaminated soil depend on factors such as site conditions and contamination history, and in general the health risk associated with the uptake of contaminants by produce for human consumption should be evaluated on a site-specific basis.

Issues which are not included in the derivation of the soil acceptance criteria (refer Table 5.17) and which require separate consideration on a site-specific basis include:

- leaching of contaminants from soil and the impact of soil contamination on groundwater quality (refer section 5.10.7);
- contamination of surface run-off and impact on nearby surface water bodies;
- protection of off-site ecosystems (refer section 5.6);
- protection of the on-site ecosystem, other than the protection of plant life for agricultural and domestic purposes (refer section 5.6);
- assessment of dust within buildings (refer section 5.10.6).

Table 5.17
Summary of Soil Criteria (all values in mg/kg dry basis)

Item	Arsenic	Boron	Chromium III	Chromium VI	Copper	PCP
Agricultural						
Health:						
– Oral	100	7,000	160,000	780	7,800	130
– Inhalation	1,900	–	410,000	710	NL	240,000
– Dermal	–	–	–	–	–	380
– Produce	4.4	3.0	11,000	4.4	39	0.7
– Combination	4.2	3	10,000	4.3	39	0.7
Livestock:						
– Human health	–	–	–	–	–	11
– Livestock health[5]	(38)	(380)	(76)	(76)	(38-380)	(11)
Protection of plant life	10-20	3 (Sol)	600	25	130	N/A[3]
Derived	4.2	3 (Sol), 3	600	4	40	0.7
Adopted (including interim values)	30[2]	3 (Sol)	600	4	40	0.7
Residential						
Health:						
– Oral	100	7,000	160,000	780	7,800	130
– Inhalation	1,500	–	310,000	540	NL	180,000
– Dermal	–	–	–	–	–	760
– Produce (50% home grown)	8.9	6	22,000	8.9	78	1.4
– Produce (10% home grown)	44	30	110,000	44	390	6.9
– Combination (50% produce)	8.1	6	18,000	8.7	77	1.4
– Combination (10% produce)	30	30	54,000	39	370	6.5
Protection of plant life	10-20	3 (Sol)	600	25	130	N/A
Derived:						
– 50%	8.1	3 (Sol), 6	600	9	80	1.4
– 10%	30	3 (Sol), 30	600	25	130	7
Adopted:						
– 50%	30[2]	3 (Sol)	600	9	80	1.4
– 10%	30	3 (Sol)	600	25	130	7
Industrial Unpaved						
Health – Workers:						
– Oral	1,000	170,000	NL[4]	21,000	210,000	1,200
– Inhalation	1,000	–	130,000	370	NL	130,000
– Dermal	–	–	–	–	–	1,100
– Combination	500	170,000	130,000	360	210,000	570
Health – Maintenance:						
– Oral	1,200	230,000	N/A	26,000	260,000	1,500
– Inhalation	1,400	–	180,000	520	N/A	180,000
– Dermal	–	–	–	–	–	3,500
– Combination	650	230,000	180,000	510	260,000	1,000
Adopted	500	NL	NL	360	NL	570
Paved Industrial[1]	650	NL	NL	510	NL	1000
Background[6]	2-30	9->30	<4-219		0.1-107	

Notes:
(1) Refer Table 5.14, Paved – No Management Plan
(2) Interim values, refer Section 5.9.2
(3) N/A denotes criteria not available
(4) NL indicates consideration not limited
(5) Preliminary criteria only based on extrapolation from stockwater limits
(6) Spier (1997)

5.9.2 Interim Soil Acceptance Criteria

(a) Dioxins

Soil acceptance criteria have not been specifically developed for dioxins and furans (PCDDs and PCDFs; "dioxins") as part of these draft guidelines. However, site-specific acceptance criteria have been developed for dioxins and furans at the Waipa Processing Complex, as part of the "Pentachlorophenol Risk Assessment Pilot Study" (NTG, 1992). These criteria may be used as a general indication of acceptable concentrations of dioxins in soil at other sites. The acceptance criteria for dioxins developed as part of the PCP Risk Assessment Pilot Study are:

- Agricultural: 0.01µg/kg (TE)
- Residential: 1.5 µg/kg (TE)
- Industrial – Unpaved: 18 µg/kg (TE)
- Industrial – Paved, with Management Plan: 90 µg/kg (TE) to not limited
- Maintenance: 21 µg/kg (TE)

Acceptance criteria for dioxins will be reviewed as part of the Organochlorines Programme being undertaken by the Ministry for the Environment. In the interim, care should be exercised when applying the above values to other sites, given they were developed for specific scenarios at a particular site.

(b) Arsenic

Background

Calculated acceptance criteria for arsenic based on the protection of human health are summarised in Table 5.18 (refer also Tables 5.5, 5.6 and 5.9-5.14).

Table 5.18
Calculated Health-Based Acceptance Criteria for Arsenic

Land use	Calculated Criterion (mg/kg)[1]	
	Including Produce Consumption	Excluding Produce Consumption
Agricultural (100% produce)	4.2	100
Residential (50% produce)	8.1	100
Residential (10% produce)	30	100
Commercial/Industrial	–	500

Note: (1) In practice a lower limit in the range 10 to 30 mg/kg may be nominated based on typical background concentrations in soil.

There are particular uncertainties associated with the derivation of arsenic criteria which resulted in a decision to establish interim acceptance criteria for arsenic to enable further research to be carried out.

The factors leading to this decision include:

- It was noted that the calculated criteria for arsenic for agricultural and residential land uses were low and possibly impractical, given their relationship to background concentrations. These criteria are a direct function of key assumptions such as the uptake of contaminants by plants, the consumption of produce, the dose-response relationship and the acceptable cancer risk.
- Additional information regarding the background concentrations of arsenic in New Zealand soils was sought to confirm whether significantly elevated arsenic concentrations occur in some parts of New Zealand.
- Use of interim soil guideline values would facilitate release of the guidelines. The interim guidelines would reflect international directions and would be supported by a research programme designed to address some of the key points of uncertainty.
- Bioavailability was noted as one of the key areas of uncertainty in the derivation of soil guideline values for arsenic. A research programme designed to address this issue should be developed to support the release of the interim guidelines, and to enable a final set of soil acceptance criteria for arsenic to be developed. In particular, consideration should be given to determining whether naturally occurring or anthropogenic arsenic exhibits differing bioavailability and whether this impacts on the health significance of naturally occurring arsenic.

Interim Soil Acceptance Criteria for Arsenic

On this basis interim soil acceptance criteria for arsenic of 30 mg/kg for agricultural and residential uses and soil acceptance criteria of 500 mg/kg for a commercial/ industrial use are proposed. The basis for the selection of these values is as follows:

- The commercial/industrial soil guideline value is based directly on health risk considerations, corresponding to an incremental lifetime risk of mortality from arsenic-related skin cancer of 1 in 100,000 (assuming a 7% mortality rate). No consideration of the protection of plant life has been included in nominating the soil guideline value for commercial/industrial use.
- Background concentrations of arsenic in New Zealand soils typically range from 2 to 30 mg/kg.
- The proposed guidelines lie well within the range of guideline values nominated internationally. The lower international values reflect lower target risk levels (e.g. the USA and Canada adopt an incremental lifetime risk of cancer of 1 in 1,000,000). Refer to Table 5.1 for comparison with other guideline values.

- The proposed guideline value for agricultural and residential use (i.e. 30 mg/kg) corresponds to an incremental lifetime risk of mortality from cancer of 1 to 14 in 100,000[1] for agricultural use; 0.8 to 11 per 100,000 for residential use assuming 50% of produce consumed is home-grown; and 0.5 to 7 in 100,000 for residential use assuming 10% of produce consumed is home-grown.
- The above estimates of cancer risk assume that 100% of the arsenic present in soil is bioavailable. The bioavailability of arsenic in soil is uncertain but is likely to depend on the soil type and conditions and the form in which the arsenic is present. Further work is required in this area to confirm the bioavailability of arsenic arising from timber treatment activities and any differences compared to arsenic from other sources. The recommendation of an interim value assumes research is being done to resolve these issues. The interim soil guideline values for arsenic may be revised as a result of the outcome of this work.
- Arsenic may also result in adverse effects on a range of ecosystems. In the first instance consideration may be given to the effect of arsenic on plant life (phytotoxicity). This depends on the bioavailability of the arsenic, as outlined above. However, published thresholds for the onset of adverse effects on plant life are generally in the order of 30 mg/kg for acid, sandy soils, ranging upward for finer-grained soils. The Dutch intervention value of 55 mg/kg is based on the protection of 50% of species in an ecosystem, including consideration of plant life.

Further information regarding overseas soil acceptance criteria, background concentrations of arsenic in New Zealand soils and the phytotoxicity of arsenic in soil is presented in Appendix G.

5.10 APPLICATION OF THE SOIL ACCEPTANCE CRITERIA

5.10.1 General

The recommended soil acceptance criteria provide general guidance for each of the nominated land uses. In practice, the impact of soil contamination is influenced by a wide range of factors which should be considered on a site-specific basis. These include:

- the off-site impact of the soil contamination, particularly by contamination of groundwater;
- the extent and distribution of soil contamination;

[1] Range of risks refers to the range of in the mortality rate associated with arsenic-related skin cancers. Between 1 and 14 % of arsenic-related skin cancers have been found to be fatal (WHO, 1993).

- the bioavailability and mobility of soil contaminants as affected by the composition and structure of the soil and the form in which the contaminants are present;
- the extent of pavement or other ground cover which limits the exposure of workers, residents etc. to soil contamination; and
- site management and works practices.

Where continuing industrial use of a site is proposed, the need to avoid contamination of groundwater or surface receiving waters may determine the acceptable soil contaminant concentrations, rather than the criteria listed in Table 5.17 (refer 5.8.3).

5.10.2 Averaging Contaminant Concentrations

The recommended soil acceptance criteria have been based on the assumption of a largely unpaved, uniformly contaminated site. In practice, a typical timber treatment site shows limited areas of relatively higher-level contamination with more widespread areas of lower-level contamination. This may require more detailed assessment at any particular site.

Where chronic human exposure to ground contamination is the primary concern, it is reasonable to compare average contaminant concentrations, rather than the maximum measured concentration, with the proposed acceptance criteria. The area across which contaminant concentrations are averaged should be selected on the basis of the typical area in which a person may spend most of their time. For residential land use, the averaging area may be selected as the area of a typical backyard.

However, the approach of comparing an average contaminant concentration with the acceptance criteria may be inappropriate in some circumstances, for example acute toxicity concerns or where the particular criterion is based on the protection of plant life. Further, criteria set on the basis of phytotoxicity may be relaxed in some situations depending on the soil type and chemistry and the tolerance of various plant species. Such variation must be evaluated on a site-specific basis; refer to Section 5.6.2 for details.

Further comments about the averaging of contaminant concentrations are presented in Chapter 2.

5.10.3 Paving and Other Ground Cover

Reduction of the exposure of site users to contaminated soil by the engineered containment of areas of soil contamination is an acceptable remediation/management strategy in some circumstances. Containment may involve the use of an impermeable surface cap of bitumen or concrete paving. A well-defined management plan should accompany such a containment strategy. Very high levels of contamination may be acceptable with such a containment strategy, provided the contamination does not

produce other off-site impacts e.g. via groundwater or surface rainfall runoff (refer section 5.8.3). Appropriate containment strategies may be used to mitigate such off-site effects. In addition, mechanisms must be put in place to ensure the long-term integrity of any management plan and containment works, taking into account property transfer and other possible scenarios. The implementation of a containment strategy is most practical in the context of ongoing commercial/industrial use, or possibly high-density residential use.

The protection offered by pavement is, however, highly dependent on the integrity of the pavement. A relatively new area of paving of high integrity is likely to effectively eliminate exposure of site personnel to ground contamination except where the pavement is disturbed and the integrity of the pavement compromised. Where an area of pavement is relatively old and broken, with say 30 to 50% of the area seriously affected, so that soil can spread from areas of pavement failure to areas remaining intact, the pavement may provide negligible protection for site personnel. The integrity of an area of pavement and its likely effectiveness in reducing exposure must be evaluated on a site-specific basis.

Redevelopment of former timber treatment sites for agricultural or residential use is likely to involve the importation of some clean fill and topsoil. Most timber treatment facilities are either paved or covered with hard stand areas which would not be suitable for redevelopment without significant preparatory work. The placement of such clean fill provides a barrier to the exposure of site users to soil contamination, and therefore higher contaminant concentrations may be allowable below such fill. The placement of clean fill over contaminated areas may also be an acceptable management strategy in the case of ongoing industrial use.

Considerations relevant to the appropriate use of such a clean fill cover include:

- the root zone of most home vegetables and the depth of digging as part of gardening activities does not generally extend beyond 0.5 m;
- normal maintenance activities at a residential site (e.g. maintenance of underground services) may bring contaminated soil to the surface and spread it around;
- contaminant concentrations below a cover layer should not pose a short-term health risk to people who may disturb the contamination;
- contaminant concentrations below a cover layer should not significantly impact on the health of trees and other plants where the root zone may extend beyond 0.5 m;
- ongoing management, and re-establishment of the cover layer after significant works or redevelopment, are required;
- the use of a barrier may not be appropriate in a flood plain zone or other areas subject to significant erosion;
- the existence of contamination below any barrier should be notified via the LIM system administrated by the TLAs.

On this basis, a protection factor of 5 is suggested as appropriate (consistent with that assigned to paving in the context of industrial use) to the placement of 500 mm of clean fill or topsoil, provided that an appropriate management plan is in place and the existence of contamination beneath the barrier is recorded on a LIM. The owner, occupier, and purchaser of the site as well as the TLA will be made aware of the contamination so that this can be taken into account during any maintenance or construction activities. As indicated previously, separate consideration should be given to possible impacts on groundwater quality.

5.10.4 Site Work and Management Practices

Where site work or management practices result in reduced worker exposure the acceptable contaminant concentrations may be increased. For example, timber workers may spend only 2 hours per day within a contaminated portion of a treated timber storage yard, and the acceptable contaminant concentrations in that area in the soil may therefore be increased by a factor of 4 [(8 hr/day)/(2 hr/day)] provided work practices do not change and exposure does not occur elsewhere on the site. Other management or work practices such as the use of protective equipment and clothing can also reduce exposure. If such an approach is adopted, it is essential that the acceptability of contamination is subject to ongoing review.

Where workers are exposed to contaminants from sources other than contaminated land (e.g. drinking water or other occupational exposure), the acceptable contaminant concentrations in the soil should be reduced by a proportionate amount.

5.10.5 Background Concentrations and Bioavailability

The acceptance criteria nominated for some contaminants, particuarly arsenic, are similar to background concentrations in some areas. This can pose problems in assessing the significance of site contamination.

The uptake of contaminants by plants is dependent on the bioavailability of the contaminant in the soil environment. Unfortunately, however, quick, reliable and generally applicable techniques for the assessment of bioavailability are not yet available.

In general, the bioavailability of arsenic and heavy metals in soil is higher in the case of anthropomorphic contamination, as compared to the natural occurrence of these substances. In addition, aging of contamination frequently reduces the bioavailability of arsenic and heavy metals.

However, the information required to rigorously demonstrate the differences in the bioavailability of naturally occurring arsenic and arsenic from a contaminated source in a general context is not available. Therefore the following approach is proposed for the assessment of arsenic contamination at former timber treatment facilities:

- Measure arsenic concentrations on-site;
- If the measured concentrations are lower than the acceptance criterion then no further action is required.
  - If the measured concentrations are higher than the acceptance criterion compare the measured concentrations with background concentrations. This may require a detailed assessment of the typical background concentrations in the vicinity of the site.
- If the measured concentrations are similar to the background concentrations then no further action is required.
  - If the measured concentrations are higher than the background concentrations determine the difference between the measured concentration and the background concentrations and compare this with the relevant criterion. This gives a measure of added contamination.
- If the added contamination is less than the criterion then consideration may be given to providing some evidence that the natural and the added contamination exhibit different bioavailabilities. Alternatively, depending on the quantity of soil involved and nature of the site, it may be appropriate to manage or remediate as if all of the measured arsenic is bioavailable.
  - If the added contamination is higher that the criterion, manage or remediate as required.

5.10.6 Assessment of Dusts Within Buildings

Investigations to date report elevated contaminant concentrations in dust samples recovered from former timber treatment facilities and from houses on and near former timber treatment facilities (CRC, 1995). In general, such dust should be treated as contaminated unless it is demonstrated that it is not. The following approach is proposed for the assessment and management of contaminated building dusts associated with timber treatment activities:

- Ongoing timber processing use:
  Assessment and management of contaminated dusts in accordance with OSH requirements, including compliance with occupational exposure limits.
- Redevelopment of former timber processing facilities for non-residential use (for example, as a storage facility for agricultural use):
  As part of the decommissioning a management plan should be implemented that includes the careful removal of free dusts (sawdust and other dusts) that may be contaminated. Dust removal should be undertaken in such a way as to minimise the quantity of free dust remaining to which site users may be exposed.

- Residential dwellings:
  Where contaminated dusts are detected within the living space of residential dwellings on or near to former timber treatment facilities, the measured contaminant concentrations should be compared with the preliminary criteria nominated in Appendix E, and managed accordingly.

5.10.7 Protection of Groundwater Quality

Where groundwater contamination is a potential concern, a groundwater monitoring programme should be initiated to further characterise conditions on-site and to quantify contaminant concentrations. Groundwater and surface water acceptance criteria, based on a range of uses, have been proposed in Chapter 6 and these criteria should be used to assist in the assessment of groundwater contamination.

Although such direct measurement of groundwater contamination is usually essential, several models are available to assist in estimating the contaminant concentration in groundwater associated with a given contaminant concentration in soil in the unsaturated zone. These models are necessarily simplified, and usually do not accurately represent conditions on-site. Despite these limitations, the models are useful in gaining some understanding of the relationship between groundwater quality and soil contamination.

Contaminant concentrations in the groundwater will depend on a wide range of factors including:

- contaminant concentrations in the soil,
- soil type, permeability, organic carbon content,
- infiltration rate,
- groundwater flow, including gradient and aquifer permeability,
- groundwater mixing,
- depth to groundwater, and
- the type and form of the contaminant.

When considering soil acceptance criteria, it is likely that concern regarding groundwater quality will be secondary where an agricultural or residential land use is proposed, because the criteria for these usages should also provide for protection of groundwater. However, experience has shown that groundwater quality may be the controlling factor where continued industrial use of the site is proposed, and where there is potential for groundwater contamination to occur, it must be considered in detail on a site-specific basis.

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APPENDIX A
UPTAKE BY PLANTS

OVERVIEW

The uptake of contaminants by plants, particularly edible fruits and vegetables, and consumption of such produce, is a potential human exposure route for residential and agricultural site users. Limited information is available regarding the uptake of heavy metals by plants; however, some typical published data is presented in the following sections.

Where available, measured contaminant concentrations in the edible portions of plants provide the most reliable estimate of plant uptake, particularly if information is available on a site-specific basis. Frequently published information on plant uptake is limited and correlations based on the accumulated data for a range of chemicals may be of use. Plant uptake is heavily influenced by site-specific factors and therefore the generic soil acceptance criteria have been based on conservative estimates of plant uptake.

In the following sections published data is presented alongside predictions made using published correlations. A typical value has then been selected for use in derivation of the acceptance criteria.

ESTIMATING PLANT UPTAKE

Procedures for the estimation of contaminant uptake by plants are presented in ECETOC (1990), AERIS (1991) and Travis (1988).

The estimation methods are based on uptake from soil via the roots and on foliar deposition.

2.1 Inorganics

Plant Uptake – Roots

The contaminant concentration in plant material as a result of root uptake may be estimated as follows (ECETOC, 1990):

$$Cp_{root} = Cs \times BCF_{root} \quad \textbf{(A1)}$$

where Cp_{root} = concentration of contaminant in plant due to root uptake (mg/kg)
Cs = concentration of contaminant in soil (mg/kg)
BCF_{root} = bioconcentration factor

$$\ln(Kd) = 3.02 - 0.85\ln(BCF) \quad \textbf{(A2)}$$

where Kd = distribution co-efficient (mL/g)

Predicted and measured values for BCF are presented in Table A1.

Table A1
Plant Uptake Parameters – Root

Contaminant	Kd	BCF_{Root}	
		Estimate	Range (1)
Arsenic	200	0.07	0.01 – 0.1
Boron			1.0 – 10.0
Chromium (III)	5000	0.015	0.01 – 0.1
Chromium (VI)	70	0.24	
Copper	60	0.28	0.1 – 1.0

Note: (1) Range of BCF experimentally determined (ECETOC (1990)).

Plant Uptake – Foliar Deposition

The estimated contaminant concentrations in plant material due to foliar deposition are outlined as follows (ECOTOC, 1990):

$$Cp_{FD} = Cs \times BCF_{FD} \quad \textbf{(A3)}$$

$$BCF_{FD} = (fin/Yv \times fei) \times (1 - (1 - e^{-fei.te})/fei \times te)) \times DRo \times frs \quad \textbf{(A4)}$$

where
- fin = initial fraction of interception = 0.4
- Yv = vegetative productivity = 0.033 d^{-1}
- fei = weathering constant = 0.033 d^{-1}
- te = crop growth period = 120 d
- DRo = deposition rate = 230 mg/m^2
- frs = fraction of soil in dust = 0.5

The predicted and measured values for BCF and the adopted soil to root concentration ratios for the inorganic contaminants of concern are presented in Table A2.

Table A2
Summary of Plant Uptake Parameters

Contaminant	BCF_{root}	BCF_{FD}	Ratio Soil Concentration/ Root Concentration		Adopted Ratio of Soil Concentration to Root Concentration[2]
			Predicted	Measured	
Arsenic	0.07 (0.01-0.1)	0.004	13.5	1.7-1600 [2]	100 [3]
Boron	3.0[1]	0.004	0.3	0.1-1 [4]	0.3
Chromium (III)	0.015	0.004	52	11-88 [2]	50
(VI)	0.24	0.004	4.1	-	4
Copper	0.28	0.004	3.5	1-10 [4]	3.5

Notes: (1) Assumed value within typical range
(2) Refer Table A3
(3) Refer Section 3
(4) Refer Table A1

The above estimates of the average ratio of soil concentration to root concentration reflect contaminant concentrations in root tissues under normal soil conditions (reasonable soil adsorption capacity and aged contamination). For the purposes of estimating soil acceptance criteria, contaminant concentrations in leafy vegetables are assumed to be 20% of those in root vegetables (estimated above). Based on comparison of BCF_{root} and BCF_{FD} in Table A2, this is expected to be a conservative assumption. Uptake and translocation to fruit (by fruit trees) is assumed to be negligible.

Further, contaminants taken up by plants often accumulate near the skin of root crops, which is often removed prior to consumption, and therefore exposure may be less than that estimated on the basis of the roof uptake estimates. The uptake of contaminants by plants is a key area of uncertainty in the development of soil acceptance criteria. Therefore, a site specific assessment of contamination may include work to refine the estimates of plant uptake.

2.2 Organics

An empirical relationship describing the uptake of organics by plants has been derived by Travis (1988), based on the octanol-water partition coefficient (K_{ow}).

The bioconcentration factor (Bv) is the measure of a chemical's potential to accumulate in vegetation, and is defined as the ratio of the concentration in aboveground parts (mg of compound/kg of dry plant) to the concentration in soil (mg of compound/kg of dry soil). Travis (1988) nominates the following relationship:

$$\log B_v = 1.588 - 0.578 \log K_{ow} \quad \textbf{(A5)}$$

where: B_v = Bioconcentration Factor for Vegetation
K_{ow} = Octanol Water Partition Coefficient.

The bioconcentration factor (B_v) for an organic in vegetation is inversely proportional to the square root of the octanol-water partition coefficient (K_{ow}).
B_v is based on the dry weight of vegetation as estimated in equation A5, however the fresh weight concentration may be estimated assuming a moisture content of 80% (i.e. divide Bv by a factor of 5).

UPTAKE OF ARSENIC

A summary of typical plant and soil arsenic concentrations is presented in Table A3.

Considerable research into the uptake of arsenic by plants has been undertaken in Australia and overseas. Limited test work specific to the Tweed Valley dip sites has been undertaken by NSW Agriculture (Tyler, 1993). In addition research completed by CSIRO investigating the relationship between arsenic accumulation in orchard soils and concentrations in plant matter indicated that in most cases, concentrations of arsenic in edible plant portions did not exceed recognised health based food standards (Merry et.al. 1983, Merry et.al. 1986). Further, in most cases the onset of plant toxicity was noted prior to the accumulation of significant concentrations of arsenic in plant matter. The uptake of arsenic by plants depends on the extent to which it is adsorbed on to iron oxides, clay particulates, organic matter, etc., which is highly dependent on the soil type and contamination history (Merry, 1993).

Research completed by Landcare Research has shown that arsenic concentrations of potential significance with regard to human health can accumulate, particularly, in root crops (Yeates, 1993).

Based on the information presented in Table A3, a soil/root uptake factor of 100 has been selected for the development of arsenic soil acceptance criteria. The selection of a factor of 100 reflects:

- a typical uptake ratio, consistent with the objective of determining the reasonable maximum exposure;
- the wide variation in published information;
- the preference for information related to food crops, rather than pasture, as the basis for developing soil acceptance criteria;
- preparation of root vegetables will frequently remove the skin where much of the arsenic absorbed through the roots may be expected to concentrate.

Table A3
Summary of Typical Soil and Plant Metal Concentrations

Contaminant	Soil Concentration (mg/kg)	Plant Concentration (mg/kg)		Ratio Soil/Plant		Source
Chromium	Background 36.1 to 61 62000	ND to 0.19 2.5 to 4.1 700-5400		14 11-88		WHO
		Herbage	**Roots**	**Herbage**	**Roots**	
	47.3 148 382 739	2.4 2.8 5.2 7.9	8.7 23.2 62.3 39.8	19.7 52.9 73.5 93.5	5.4 6.4 6.1 18.6	Yeates (Pasture)
Copper	19.3 109 425 835 44 174	10.5 14.6 18.4 23.9 3.6 13.5	22.9 44.9 136 162	1.8 7.5 23.1 34.9 12.2 12.9	0.8 2.4 3.1 5.2	
Arsenic	35 to 108	0.8 to 2.1		43 to 51		WHO
		Herbage	**Roots**	**Herbage**	**Roots**	
	12 161 469 790 4.9 64	1 3.5 5.7 11.1 0.2 2.45	7.2 28.8 54.5 66.2	12.0 46.0 82.3 71.2 24.5 26.1	1.7 (2) 5.6 (2) 8.6 (2) 11.91 (2)	Yeates (Pasture)
	Sandy Soil					
	6		0.09 0.96		67	Taylor (Carrots)
	23		0.15 0.21		109	
	120		1.5 1.2		80	
	640		1.3 1.3		492	

Contaminant	Soil Concentration (mg/kg)	Plant Concentration (mg/kg)		Ratio Soil/Plant		Source
Arsenic (cont'd)	Clay Soil	**Herbage**	**Roots**	**Herbage**	**Roots**	
	10		0.05 0.07		142	
	32		0.02 0.02		1600	
	78		0.29 0.31		251	
	350		1.1 0.80		318	
				~ 300 (1)		HESP
	322	0.17		1890		lettuce
	322		0.04		8050	onion
	322		0.034		9470	beetroot
	322		0.042		7670	carrot
	322	0.008		40200		pea
	322	0.008		40200		bean
	95	0.3		317		silverbeet
	120	0.4		300		silverbeet
	117		0.96		122	radish
	117	0.92		127		radish
	117	0.6		195		silverbeet

Notes: (1) Average of root and stem based on BCF (stem) = 0.006 and BCF (root) = 0.003.
(2) Relates to fibrous root material, not edible portion.
(3) Shell, 1994

UPTAKE OF PENTACHLOROPHENOL

Unpublished information from Canada (Environment Canada, 1993) indicates that soil acceptance criteria for PCP may be based on a plant (total) concentration: soil concentration ratio of 0.056 compared to a value adopted for the purposes of these guidelines of 0.043 (= 1/23).

Published information regarding the uptake, metabolism and elimination of PCP is summarised as follows (Environment Canada, 1993):

- PCP is readily metabolised by plants so that while PCP products may be detected, little intact PCP is found in plants;
- 0.1% of applied PCP (1 mg/kg) was taken up by carrots in one growing season. Most was either recovered from the soil (57.6%) or lost to the atmosphere (42%);
- A greenhouse study determining the effects of sludge on plant uptake using tall fescue, lettuce and carrots, found minimal intact PCP in the fescue and lettuce and no intact PCP was detected in the carrots (PCP application up to 5 mg/kg soil);

- The fate of 100 mg/kg PCP was investigated in a soil-crested wheat grass system. After 155 days 21% of the PCP was associated with plant roots and 15% with the shoots; and
- Uptake of PCP in soybean and spinach was studied in pot experiments (sterilised) with a loamy sand soil treated to give 10 mg PCP/kg soil. Soybean shoots and stems were measured to contain 5 and 0.1 mg/kg (fresh weight) respectively. Spinach plants contained 9 and 20 mg/kg (fresh weight) in the shoots and roots respectively.

Environment Canada (1993) note bioaccumulation in plants is low (<0.01 for fescue, lettuce and carrot), although specific bioaccumulation studies for PCP are limited.

As indicated in Section 2.2 published correlations such as that by Travis (1988) may be of use in predicting plant uptake. Such correlations generally relate uptake to K_{ow} or K_{oc}. PCP may be present in soil as both pentachlorophenol and pentachlorophenate, each of which exhibits differing K_{oc} and K_{ow} values. Therefore, K_{ow} for PCP in soil depends on the proportion present in an unionised form. At the pHs of most concern in soil, very little PCP is present in an unionised form (0.41 at pH = 4.6, 0.01 at pH = 6.8 (USEPA, 1994)) reducing the tendency for PCP to sorb on soil particles. Environment Canada (1993) summarised published K_{ow} values for a range of pH values. At neutral pH, log K_{ow} for PCP has been measured as 3.3 (compared to log K_{ow} = 5.05 for unionised PCP).

Based on log K_{ow} = 3.3, and equation A5, B_v (fresh weight) may be estimated to be approximately 0.09 (B_v (dry weight) = 0.4).

In contrast, preliminary modelling using HESP suggests an estimate for B_v (fresh weight) at low PCP concentrations, in the order of 2. Environment Canada (1993) estimated B_v to be equal to 0.056.

On this basis an estimate of B_v of 0.09 has been used in the derivation of soil guideline values given:

- Uptake experiments indicate plant uptake is variable but can occur;
- Bioconcentration may be limited by degradation of PCP in the soil and metabolism in the plants; and
- Estimates of B_v range from <0.01 to 2 (adopted value approximates the geometric mean).

Note: B_v estimated using the Travis (1988) correlation relates to the overall produce concentration the adopted root and stem uptake ratios as used in Table 5.6, Table 5.10 and Table 5.12 which have been selected based on:

- an overall uptake ratio (B_v) of 0.09;
- an assumption that stem concentrations are 20% of root concentrations;
- bioconcentration in fruit is negligible; and
- root and stem crops contribute 29% and 31% respectively of total produce consumption.

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APPENDIX B
UPTAKE BY CATTLE

The uptake of contaminants by cattle and other livestock is a potential cause for concern with regard to an agricultural land use (due to the potential exposure of consumers of livestock products to PCP residues). Normally the intake of contaminants via cattle grazed on contaminated soil is relatively low, however in the case of some contaminants, particularly some chlorinated organics, have the potential to accumulate as a residue in livestock. ECETOC (1990) have published correlations useful for estimating the residual organic contaminant concentrations in cattle. Note that similar procedures for the uptake of inorganic contaminants by livestock are less readily available, however this is not expected to be of concern for the inorganic contaminants addressed by these guidelines. It is assumed that ingestion of contaminated soil is the dominant uptake route for cattle (however consideration could also be given to uptake via the consumption of vegetation).

INTAKE VIA INGESTION OF SOIL

DIc = Cs x AIDc x (toc/124) x fac x N **(B1)**

where:
- DIc = direct ingestion of contaminant through soil ingestion
- Cs = soil concentration
  say, for example, 100 mg/kg, assumed
- AIDc = amount of soil ingested by cattle
  = 0.72 kg/d (Shell, 1994)
- toc = time spent outside per day
  = 24 hr/d
- fac = absorbed fraction
  = 1.0, i.e. assume 100% of PCP intake is absorbed.
- N = fraction of days on which this occurs

therefore DIc = 72 mg/day

Total daily intake by cattle = 72 mg/d

Concentration in product = Total intake x Kp **(B2)**

where Kp = contaminant partition co-efficient for product ‘p’

For organics

$\log K_{meat} = -6.88 + 0.832 \log K_{ow}$ **(B3)**
$\log K_{milk} = -6.786 + 0.731 \log K_{ow}$ **(B4)**
$\log K_{fat} = -3.457 + 0.500 \log K_{ow}$ **(B5)**

Note $\log K_{ow}$ (PCP) = 5.05 (AERIS 1991)

The estimated concentration of PCP in livestock products and resulting incremental lifetime risk of cancer associated with a concentration in soil of 1 mg/kg are presented in Table B1. By ratio, the PCP concentration associated with an incremental lifetime risk of cancer of 1 in 100,000 is estimated to be 11 mg/kg.

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Table B1
Preliminary PCP Soil Acceptance Criteria Based on Protection of Human Health: Exposure Via Consumption of Livestock

Oral slope	0.12	mg/kd/d[1]	Base soil conc:	1 mg/kg
Tarket risk	1.0E-05		Cattle ingestion rate:	0.72 kg/d
			Human body weight:	70 kg

Product Consumption (g/d)			
Product	**Male**	**Female**	**Average**
Meat	308	183	246
Milk	356	247	302

Cattle product	**Partition coefficient**	**Total intake for cattle (mg/d)**	**Concentration (mg/kg, fresh weight)**	**Average consumption rate (g/d)**	**Estimated human intake (mg/d)**	**Body weight (kg)**	**CDI (mg/kg/d)**	**Dose response factors $(mg/kg/d)^{-1}$**	**Estimated cancer risk for 1 mg/kg PCP in soil**
Meat	2.10E-03	7.20E–01	1.51E–03	246	3.7E–04	70	5.30E–06	0.12	6.36E–07
Milk	8.10E-04	7.20E–01	5.83E–04	302	1.8E–04	70	2.51E–06	0.12	3.01E–07
Combined					5.47E–04	70	7.81E–06	0.12	9.38E–07

Notes: (1) Equates to target risk

Acceptable soil concentration:[1] **10.66 mg/kg**

APPENDIX C
DETERMINATION OF PARTICLE EMISSION FACTOR

The particle Emission Factor is determined as follows (refer 5.7.2(c)):

$$(1/PEF) = TSP \times CF \times P_{RP} \times FC \qquad \textbf{(C1)}$$

where:
- TSP = concentration of total suspended particulates (mg/m^3)
- P_{RP} = proportion of particulates respirable (i.e. < 10μm)
- CF = conversion factor
  - = 10^{-6} kg/mg
- FC = fraction of TSP from a contaminated source.

Refer EPAV (1991) "Discussion Paper on Particulates, Publication No. 297" Environment Protection Authority, Victoria for typical air monitoring data for Victoria.

For Macarthur Street, Melbourne:

PM_{10} = 3.8-71 $\mu g/m^3$ (Annual av ~ 26 $\mu g/m^3$)

where:
- PM_{10} = concentration of particulates less than 10μm diameter
- PM_{10}/TSP = 0.57 (6 month average (1987))
- PM_{10}/TSP = 0.46 (12 month average (1988))

Latrobe Valley:

Concentration of respirable particulates in urban environment in Latrobe Valley is 35% greater than that in a rural environment within the Latrobe Valley.

- PM_{10} (urban) = 3.7 – 48 $\mu g/m^3$, (20 $\mu g/m^3$ annual average)
- Based on PM_{10} (country urban) = 20 $\mu g/m^3$ average
  - PM_{10} = TSP x P_{RP}
  - TSP x P_{RP} = 0.02 mg/m^3

Based on EPAV data for Rural/Provincial Urban environment 10-20% of PM_{10} is derived from soil and similar sources.

New Zealand:

Typical suspended particulate monitoring data for New Zealand are as follows:

- Typical background total suspended particulate (TSP) concentrations in the range 10 to 20 $\mu g/m^3$.
- Typical urban area TSP concentrations in the range 20 to 50 $\mu g/m^3$.
- Typical urban/industrial area TSP concentrations in the range 30 to 60 $\mu g/m^3$.
- TSP immediately adjacent to industrial sources up to 200 $\mu g/m^3$.
- Mean proportion of TSP that is less than 15 mm diameter is in the range 50 to 70%.

Winter time TSP concentrations in Christchurch can reach 100 μg/m^3, with up to 100% of the TSP less than 15 μm diameter.

The above information indicates TSP and PM_{10} concentrations in New Zealand are typically marginally less than the measured values for Victoria. The information selected for calculation of the PEF represents a conservative estimate of TSP and PM_{10}.

For an agricultural site use:

PM_{10} = 20 μg/m^3

Therefore TSP X P_{RP} = 20 μg/m^3

If FC = 0.2
CF = 10^{-6} kg/mg

Then PEF = $1/(0.02 \times 0.2 \times 10^{-6})$
= 2.5×10^8 m^3/kg

For residential site use:

PM_{10} = 26 μg/m^3

Therefore TSP X P_{RP} 26 μg/m^3

If FC = 0.2
CF = 10^{-6} kg/mg

Then PEF = $1/0.026 \times 0.2 \times 10^{-6}$
= 1.9×10^8 m^3/kg

For an industrial setting:

TSP = 0.142 μg/m^3 (GRI, 1988)
Assume PM_{10}/TSP = 0.5 and PI = 0.5
Assume 50% of TSP is resuspended local soil, i.e. FC = 0.5
PEF (Industrial) = 2.9×10^7 m^3/kg

For subsurface maintenance:

PEF (subsurface maintenance) = 0.9×10^7 m^3/kg
where TSP = 0.280 μg/m^3 for construction works (GRI, 1988)
P_{RP} = 0.5
FC = 0.8

Note: using particulate emission models, USEPA estimates PEF $\tilde{}$ 4.6×10^9 m^3/kg for a residential context. The PEF determined using the typical data is conservative. The PEF may increase where a large proportion of ground is covered effectively preventing emission of particulates, an increase in PEF would result in a decrease in the estimated exposure via inhalation of contaminants.

APPENDIX D
PROTECTION OF LIVESTOCK HEALTH

The potential for uptake of contaminants by livestock grazing on contaminated soil has been discussed in Appendix B. A further concern with regard to redevelopment or return of a site to agricultural use is the protection of the health of the animals grazing that site. Some livestock, particularly sheep, have been reported to be sensitive to some trace elements, most notably copper. In order to make a preliminary assessment of the approximate trace element concentrations required to protect animal health, published guidelines for stockwater quality have been extrapolated to apply to soil quality. The approach used to extrapolate from stockwater guidelines is outlined below:

$$\text{Preliminary soil acceptance criteria (available, mg/kg)} = \frac{\text{Water Consumption (L/d)} \times \text{Stockwater Guideline mg/L}}{\text{Soil Ingestion Rate (kg/d)}}$$

The preliminary soil acceptance criteria have been based on available soil and water consumption information for cattle (ECETOC, 1990):

where: Water Consumption = 55 L/d
Soil Ingestion Rate = 0.72 kg/d

The above soil quality guidelines apply to the available concentration of each parameter, which may be significantly less than the total concentration for many heavy metals. Typically, the available fraction for heavy metals is in the range 10% to 50% (although lower bioavailability is observed in some cases).

When an allowance is made for the fact that the preliminary soil acceptance criteria presented in Table D1 are expressed in terms of the available concentration (i.e. availability comparable with contaminants in drinking water); comparison with the acceptance criteria in Table 7.1 indicates protection of livestock health is not likely to be the controlling consideration for agricultural landuse.

Table D1
Summary of Soil Quality Guidelines to Protect Livestock Health

Parameter	Stockwater Guideline (mg/L)	Soil Guideline (available, mg/kg)
As	0.5	38
Cr	1	76
Cu	0.5-5(1)	38-380
B	5	380
PCP	0.15	11

Note: (1) Based on range of values for different species, the most stringent being based on sheep, which are particularly sensitive to copper.

APPENDIX E
ASSESSMENT OF DUST CONTAMINATION

1. OVERALL APPROACH

The following discussion outlines an approach for the assessment and management of contaminated dust within occupied building spaces. Preliminary acceptance criteria are presented to assist in the assessment of dust within residential dwellings. The overall approach to the assessment of dust contamination is discussed in Section 5.11.6.

The acceptance criteria for dust may be applied to fine material recovered from within living spaces including floors, benchtops and other surfaces, collected using a wipe test or vacuum sampling technique (as appropriate given the surface). The criteria are not intended for direct application to the assessment of contaminated dust within roof spaces where human contact may be limited.

General issues related to the assessment of soil contamination, such as averaging of contaminant concentrations may also apply to the assessment of dust contamination.

2. INDOOR DUST ACCEPTANCE CRITERIA FOR RESIDENTIAL DWELLINGS

Exposure to contaminants associated with dust inside the home (or other building) can occur via a range of exposure routes similar to those assumed for exposure to contaminants in soil. The exception to this being that exposure via the consumption of produce is not relevant in the case of indoor dust. While the exposure routes for indoor dust may be similar to those for outdoor soil, the rate of exposure is expected to be lower, reflecting the limited quantity of dust within most homes.

The development of acceptance criteria for contaminant concentrations in indoor dust has not been practiced as widely as the derivation of soil acceptance criteria and therefore information regarding exposure factors is limited. The derivation of acceptance criteria for indoor dust has been based on the assumptions and procedure outlined in J.K. Hawley, Assessment of Health Risk from Exposure to Contaminated Soil, Risk Analysis, Vol. 5, No. 4, 1985.

Hawley (1985) establishes three age groups for consideration:

- 2.5 year old child
- 6 year old child
- adult.

The main assumptions relating to indoor contamination are as follows:

- Level of suspended particulate matter in outdoor air = 26 $\mu g/m^3$ (same as for residential use in the derivation of soil criteria)
- The concentration of suspended particulate matter indoors equals 75% of that in outdoor air, i.e. 18 $\mu g/m^3$.
- The average dust covering of soil surfaces is 560 mg/m^2.

Exposure factors for a 2.5 year old child are summarised as follows:

- Rate of ingestion of indoor dust is assumed to be 50 mg/d (compared to 100 mg/d for soil).
- The quantity of soil from which dermal absorption can occur is assumed to be 28 mg/d, based on an exposed area of 0.05 m^2 (one half of the surface area of the child's hands, feet and forearms) and the dust loading of 560 mg/m^2.
- Dermal absorption factor of 0.009, based on 6% absorption rate of pure chemical for 12 hour exposure, and a 15% soil matrix effect (PCP only).
- Inhalation rate of 3.8 m^3/day (in accordance with derivation of soil criteria).
- 75% of inhaled particulate matter is retained.

Exposure factors for a 6 year old child are summarised as follows:

- Soil adherence is assumed to equal 0.056 mg dust/cm^2 of skin, giving a total quantity of dust for dermal absorption of 22 mg (area of both hands).
- For 12 hr exposure a 6% absorption rate of pure chemical, and a 15% soil matrix effect were assumed giving an overall dermal absorption factor of 0.009 (PCP only).
- The rate of ingestion of indoor dust is assumed to be 3 mg/d which equates to the quantity of soil adhering to the inside of the hands (based on estimates by Hawley).
- Inhalation rate for 6 year old children is assumed to be 3.8 m^3/day as used for the 1-6 year age group in the derivation of soil criteria.
- 75% of inhaled particulate matter retained.

Exposure factors adopted for adults are summarised as follows:

- Dust collecting areas such as attics are not included in this assessment. Initial investigation showed that total exposure due to this scenario is about 10% of the normal living conditions scenario (Hawley, 1985).
- Adults are assumed to ingest dust at an equivalent to that adhering to 10 cm^2 of skin. At a soil adherence of 0.056 mg/cm^2 the soil ingestion rate equals 0.56 mg/d.
- Dermal contact is assumed to occur from the hands (910 cm^2), with a soil adherence of 0.056 mg /cm^2.

- For 12 hr exposure a 6% absorption rate of pure chemical, and a 15% soil matrix effect were assumed, giving an overall dermal absorption factor is 0.009 (PCP only).
- Adult inhalation rate is 20 m^3 for 24 hrs (as assumed for derivation of soil acceptance criteria).
- 75% of inhaled particulate matter is retained.

To remain consistent with the derivation of criteria for soil, two groups are considered; children ages 1–6, and adults. The two child age groups are therefore combined to give an average intake or contact rate for each pathway for the one child group. Assumptions regarding adult exposure remain unchanged. A summary of the dust contact rate used in estimating the indoor dust acceptance criteria is given in Table E1 (taking into account exposure frequency and absorption rates).

Table E1
Summary of Exposure to Dust (mg/day)

Receptor Group	Exposure Route		
	Oral	**Inhalation**	**Dermal(1)**
Adults	0.56	0.15	0.46
Children	27	0.043	0.23

Note: (1) Incorporates allowance for absorbed fraction only (i.e. 0.009 x quantity of soil)

The contact rates with indoor dust estimated for children are uncertain, reflecting the uncertainty in the availability of indoor dust for consumption and the relevance of standard assumptions regarding soil ingestion. Children that have a tendency to consume soil and dust may be expected to consume more outdoor soil than indoor dust (due to the limited availability of indoor dust). On this basis the assumed oral contact rate for indoor dust of 27 mg/day which is generally consistent with the assumed soil ingestion rate of 100 mg/day.

Other input parameters of relevance in the derivation of dust criteria include:

- Target Cancer Risk = 1×10^{-5}
- Target Hazard Index = 1
- Body Weight – Child = 15 kg
- Body Weight – Adult = 70 kg
- Exposure Duration – Child = 6 years
- Exposure duration – Adult = 24 years
- Averaging Time (carcinogenic) = 70 years
- Averaging Time (non-carcinogenic) = exposure duration.

Acceptance criteria for non-carcinogenic chemicals are based on a year averaged estimate of exposure, calculated for the most critical exposure, i.e. the child's first 6 years. In contrast, acceptance criteria for carcinogenic chemicals are estimated based on a combined exposure of the child's 6 years followed by the adult's 24 years of exposure.

The preliminary dust acceptance criteria are presented in Table E2 and summarised in Table E3.

Table E2
Preliminary Health Risk Based Acceptance Criteria
Residential Site Use
Estimation of Target Dust Concentrations

Site Use:	Residential	Averaging Time (carc):	70 yrs	Ave Oral Intake (1-6 yrs):	26.5 mg dust/day	Produce Ing. (1-6 yrs, kg):	0.13
Receptor:	Child resident onsite for up to 30 yrs	(non-carc.):	ED yrs	Ave Oral Intake (7-31 yrs):	0.56 mg dust/day	Produce Ing. (7-30 yrs, kg):	0.45
		Exposure Dur. (1-6 yrs):	6 yrs	Ave Inhale Intake (1-6 yrs):	0.043 mg dust/day	Proportion of produce from contaminated source:	0.5
		Exposure Dur. (7-31 yrs):	24 yrs	Ave Inhale Intake (7-31 yrs):	0.15 mg dust/day		
Target Risk:	1.00E–05	Body Weight (1-6 yrs):	15 kg	Ave Dermal Contact (1-6 yrs):	0.23 mg dust/day	Skin Area (1-6 yrs) (sq. cm):	2625
Target HI:	1	Body Weight (7-31 yrs):	70 kg	Ave Dermal Contact (7-31 yrs):	0.46 mg dust/day	Skin Area (7-30 yrs) (sq. cm):	4700

					Acceptable CDI						Preliminary Dust Acceptance Criteria (mg/kg)							
Contaminant	SF (1/(mg/kg/d))		RfD (mg/kg/d)		Carcinogenic			Non-carcinogenic			Carcinogenic				Non-carcinogenic			
	Oral	Inhalation	Oral	Inhalation	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Oral	Inhalation	Dermal	Combined	Oral	Inhalation	Dermal	Combined
Arsenic	1.5E-01	1.5E+01			6.8E-05	6.7E-07					4.4E+02	6.8E+02		2.7E+02				
Boron			4.5E-02					4.5E-02							2.5E+04			2.5E+04
Chromium (III)			1.0E+00	3.0E-04				1.0E+00	3.0E-04						5.7E+05	1.0E+05		8.8E+04
Chromium (VI)		4.1E+01	5.0E-03			2.4E-07		5.0E-03				2.5E+02		2.5E+02	2.8E+03			2.8E+03
Copper			5.0E-02	5.0E-02				5.0E-02	5.0E-02						2.8E+04	1.7E+07		2.8E+04
PCP	1.2E-01	1.2E-01			8.3E-05	8.3E-05	8.3E-05				5.4E+02	8.5E+04	2.4E+04	5.3E+02				

Note: Preliminary remediation goal for carcinogens based on entire 30 years, for non-carcinogens based on most critical six years.
Intake rates are averaged based on yearly exposure.

Table E3
Summary of Preliminary Acceptance Criteria for the Assessment of Dust in Residential Dwellings

Contaminant	Preliminary Acceptance Criteria for Dust (mg/kg)
Arsenic	270
Boron	NA
Chromium (III)	NA
Chromium (VI)	250
Copper	NA
Pentachlorophenol	530

Note: (1) NA denotes predicted criterion greater than 10,000 mg/kg.

APPENDIX F
MANAGEMENT PLAN ISSUES

This section deals with approaches to assist in reducing risk rather than reducing contamination.

1. List the contaminants present on-site and, where appropriate, the first aid procedures in the event of acute exposure.

2. Develop a comprehensive health and safety plan for all activities likely to occur within contaminated areas.

3. Draw up a map of the site showing those points at which contaminant concentration is known and/or isometric boundaries extrapolated from the results of site sampling.

4. Categorise the site into areas where different conditions apply in respect of land use, access, activities allowed, worker protection, environmental monitoring, etc.

5. Specify the conditions applying in each area category. For example:

   Category 1: Unprotected access for all activities.
   No environmental monitoring required.

   Category 2: Unprotected access for above ground activities.
   Subsurface workers require dust masks of NZSA rating, and dedicated overalls.

   Category 1: All maintenance works restricted to informed and appropriately equipped staff.
   No eating or smoking in the area.
   Dust control measures required during subsurface operations.
   Regular monitoring for off-site migration of contaminants desirable.

6. List the names and contact details of any agency or organisation which could/should be contacted during ongoing management of the site. For example:

   - Regional Council staff
   - Area Health Board
   - Occupational Safety and Health
   - waste management contractor/s.

7. Establish contingency procedures for emergencies, fire, flooding, spillage, etc.

APPENDIX G
SUPPORTING INFORMATION FOR INTERIM SOIL ACCEPTANCE CRITERIA FOR ARSENIC

1. OVERSEAS GUIDELINE VALUES

A summary of overseas guideline values nominated for arsenic is presented in Table G1. The guideline values nominated for residential use based on a brief review of international literature range from 0.4 mg/kg to 100 mg/kg.

Table G1
Summary of Overseas Soil Guideline Values

Country	Agricultural		Residential Use		Commercial/Industrial Use	
	Value (mg/kg)	**Limiting Consideration**	**Value (mg/kg)**	**Limiting Consideration**	**Value (mg/kg)**	**Limiting Consideration**
New Zealand (Interim)	30	Refer above	30	Refer above	500	Refer above
Australia(1)	NS		100 20	Human health Environmental	500	Human health
Canada(2)						
Current Interim	20	Not specific	30	Not-specific	50	Not-specific
Proposed	12	Human health	12	Human health	12	Human health
USEPA(3)	NS		0.4	Human health		
United Kingdom	10	Not specific	10	Not-specific		
The Netherlands(4)						
Target	29		29			
Intervention	55		55			
(Health based Intervention)	680		680			

Notes: (1) The Health Investigation Levels (HILs) published by the NEHF include values for residential and commercial/industrial use. The residential values assume negligible consumption of home grown produce. HILs for scenarios including plant uptake are yet to be nominated. The arsenic HIL is based on the WHO PTWI. The Environmental Investigation Levels (EILs) reflect consideration of background concentrations and the level at which adverse effects may become apparent.

(a) The Canadian Interim Soil Quality objectives were published in 1991 and were based on a so called "mosaic approach" which made reference to other published guidelines. The remediation criteria for agricultural, residential/parkland and commercial/industrial use were 20 mg/kg, 30 mg/kg and 50 mg/kg respectively. The final draft of the risk-based limits which will replace the Interim Soil Quality Objectives nominates/recommends a value of 12 mg/kg for all uses based on:

- 1 in 1,000,000 incremental lifetime risk of cancer (giving soil concentration of 2.1 mg/kg)
- background soil concentration of 10 mg/kg, noting that where soil concentrations differ markedly from 10 mg/kg site specific evaluation may be warranted.
- consideration of soil ingestion in adults only (20 mg/day soil ingestion rate).

(b) The USEPA "Technical Background Document for Soil Screening Guidance" nominates soil screening levels for a limited number of exposure routes only. In the case of arsenic attention is focused on ingestion of soil and inhalation of particulates. Plant uptake and dermal exposure is not considered. A relatively low SSL for soil ingestion is nominated (0.4 mg/kg) reflecting the assumption of 1 in 1,000,000 cancer incidence (not death from cancer) and soil ingestion rates of 100 mg/kg for adults and 200 mg/kg for children.

(c) The Netherlands nominate Target and Intervention values for arsenic which are a combination of background conditions and consideration of impact on human health and the terrestrial ecosystem. The Target and Intervention Values do not relate directly to any specific land use. In each case human health is not the limiting consideration. The health based Intervention Value reflects:

- residential exposure scenario;
- The WHO PTWI (assuming 100% is assigned to contaminated soil exposure);
- exposure averaged over the lifetime (child and adult exposures are combined);
- 10% of produce home-grown.

(c) NS denotes not specified.

2. BACKGROUND ARSENIC CONCENTRATIONS IN NEW ZEALAND

Based on the available information (refer Table G2), an average arsenic concentration in New Zealand soils is likely to be in the order of 6 or 7 mg/kg. However in considering the development of generic criteria for arsenic, the concentration that would not be exceeded by background arsenic in most cases is more useful[1]. The nominated criterion should not be less than a reasonable background concentration of arsenic. In this regard a value of, say, 30 mg/kg may be appropriate (Spier (1997) notes that typically background concentrations of arsenic in New Zealand soils range from 2 to 30 mg/kg).

While considerable variation in background arsenic concentrations may be expected from site to site, the information available regarding the arsenic concentrations in New Zealand soils does not suggest that soils of volcanic origin are significantly elevated in arsenic content (as has been previously thought). Areas of elevated arsenic would need to be assessed on site specific basis.

[1] The purpose of the background concentration is to provide a lower limit for a reasonable criterion. Use of the average background concentration may be problematic in that whenever the arsenic concentrations exceed the criterion (which may be as low as the average) additional work to determine the site specific background may be required.

Table G2
Summary of Background Arsenic Concentrations in new Zealand Rural Soils(4)

Soil Type	Sample Number	Min	Max	Mean
Yellow brown loam	95	0.57	36.67	**6.56**
Yellow brown pumice	28	1.00	30.67	**7.65**
Yellow brown earth	106	0.80	18.00	**3.24**
Yellow grey earth	84	0.73	7.33	**2.37**
Gley	28	0.57	12.67	**5.14**
Peat	16	0.57	58.00	**12.60**
Brown granular loam	15	1.67	12.87	**5.42**
Alluvial	26	1.33	9.00	**3.50**

Notes: (1) Soil Depth: 0-7.5cm;
(a) Yellow brown loam and yellow brown pumice soils are volcanic in origin;

(2) Data is for both native (86 sites) and agricultural soils (312 farm pasture sites) of North and South Island rural soils (Robert et al., 1996).

(3) Roberts A.H.C, Cameron K.C., Bolan N.S., Ellis H.K. and Hunt S. (1996) "Contaminants and the soil environment in New Zealand" in R. Naidu et al. Contaminants and the soil environment in the Australian-Pacific region. pp 579-628, Kluwer, Bodmin.

3. THRESHOLDS FOR PHYTOTOXICITY

The threshold for the onset of phytotoxicity (or adverse effects on plant life) depends heavily on the bioavailability of the arsenic (as does uptake of contaminants by plants) which in turn depends on soil type, the chemical nature of the arsenic released, the age of the contamination and a range of other factors. Consideration of the impact of contamination on the broader biological function of the soil is essential in setting criteria for the protection of plant life. Consideration of phytotoxicity addresses this issue in part.

A brief review of published literature and guideline values for the onset of plant effects is presented in Section 5.6.2.

In addition to the published information relating directly to the onset of phytotoxic effects, it is noted that the Dutch guidelines give consideration to the impact of the ecological function of the soil. In particular the Intervention Value for arsenic of 55 mg/kg has been nominated

for a Standard Soil[2] on the basis of protection for 50% of the species that may be expected in the soil.

Therefore, it is expected that arsenic concentrations in the order of 20 to 30 mg/kg may represent a reasonable threshold for the onset of adverse effects on plant life for acid sandy soils. Sheppard (1991)[3] noted there are relatively few reports of toxicity below 10 mg/kg, with a geometric mean of 40 mg/kg. It is noted that thresholds for the onset of phytotoxicity are five-fold higher in clays compared to sands.

In circumstances where the arsenic may exhibit lower bioavailability eg., clays, arsenic associated with minerals, then much higher arsenic concentrations may be associated with negligible impact on plant life.

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2 The Dutch Standard Soil is defined as containing 10% organic matter and 25% clay fraction. Where soils exhibit a higher clay or organic matter content the concentration corresponding to protection of 50% of species may be higher and a revised Intervention Value may be nominated.

3 Sheppard S.C. (1991) "Summary of Phytotoxic Levels of Soil Arsenic". Water, Air and Soil Pollution 64: 539-550, 1992.

CHAPTER 6
SURFACE WATER AND GROUNDWATER ACCEPTANCE CRITERIA

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APPENDICES

6.1 SURFACE WATER AND GROUNDWATER ACCEPTANCE CRITERIA

6.1 INTRODUCTION

6.1.1 Background

This chapter describes the development of acceptance criteria for potentially contaminated surface water and groundwater. The acceptable levels of contaminants in groundwater or surface water depend on a range of site-specific factors, including current and potential future uses.

When assessing contamination of groundwater or surface water associated with a particular timber treatment facility or contaminated site, it is necessary to critically review the potential uses of any groundwater or surface water, and to select acceptance criteria on the basis of these uses. Consideration may also be given to the designation of attenuation or mixing zones within which contaminant concentrations may exceed the nominated criteria, provided use is restricted.

As part of the development of acceptance criteria, the following potential uses of groundwater and surface water have been considered:

- potable water (human drinking);
- stock watering;
- irrigation
- primary contact recreation (bathing); and
- protection of aquatic ecosystems.

The guidelines have been developed following a review of published information from CCME, ANZECC, USEPA, WHO and other similar organisations. Where appropriate, health risk assessment procedures have been used in the development of acceptance criteria. The 'Australian Water Quality Guidelines for Fresh and Marine Waters' (ANZECC, 1992) are Australian guidelines and have no official status in New Zealand. (This is in contrast to the 'Australian and New Zealand Guidelines for the Assessment and Management of Contaminated Sites' (ANZECC, 1992), which have the status of government policy in New Zealand.)

The Ministry for the Environment is working towards the development of a framework for the protection of aquatic ecosystems. This will lead, in due course, to the adoption of the ANZECC Water Quality Guidelines, which are currently under review.

If site discharge limits are based on the criteria in this document, the criteria selected should be based on the use of the appropriate section of the receiving water, and then corrected to account for dilution, etc. It is more appropriate to apply the criteria directly to the appropriate receiving water as this recognises changes in the prevailing conditions and other external influences. However, this approach is frequently not possible and it is more practical to apply the criteria to the discharge, after allowing for typical dilution.

Guideline values for dioxins and furans (expressed as a 2, 3, 7, 8-TCDD toxic equivalent concentration) are included in Section 6.7 as interim values only; these values are those adopted for the 'Pentachlorophenol Risk Assessment Pilot Study' (NTG, 1992). Final acceptance criteria for dioxin and furan concentrations in groundwater and surface water will be determined as part of the Organochlorines Programme of the Ministry for the Environment.

6.1.2 Chapter Summary

This chapter presents acceptance criteria for the levels of timber treatment chemicals in water used for a range of purposes. The potential uses considered include: potable water, stock watering, irrigation and primary contact recreation (i.e. bathing). The protection of freshwater aquatic ecosystems is also discussed, but it is noted that further work on this topic is in process, and general guidelines are not proposed although some advice is offered.

For each of these potential uses, the guidelines propose maximum allowable levels of each of the contaminants of interest (i.e. arsenic, boron, chromium, copper and pentachlorophenol) and set out the reasons for adopting these levels, usually by reference to published criteria.

Table 6.3 presents the recommended water quality guidelines in summary form, allowing easy identification in terms of the water use or contaminant of interest to readers.

6.2 POTABLE USE

6.2.1 Introduction

This section presents guidelines for the concentration of contaminants in water intended for potable use (human drinking). These guidelines have been drawn from the 'Drinking Water Standards for New Zealand' (DWSNZ; MOH, 1995). The DWSNZ give consideration to:

- human health;
- aesthetic concerns; and
- protection of assets.

The DWSNZ are generally consistent with the WHO 'Guidelines for Drinking Water Quality' (WHO, 1993). The DWSNZ for the major contaminants of interest in timber treatment sites are summarised in Table 6.1.

6.2.2 Arsenic

Arsenic has been associated with a range of adverse, chronic, non-carcinogenic health effects, particularly associated with the central nervous system. Arsenic contamination of water supplies has also been linked to an increased incidence of skin cancer and arsenic is classified as a confirmed human carcinogen (Group A) by the USEPA (1992b). Both common forms of arsenic are readily bioavailable and toxic to humans. Based on an incremental lifetime risk of cancer of 10^{-5} and the USEPA IRIS database slope factors, the acceptable concentration of arsenic in drinking water can be estimated to be 2.4×10^{-4} mg/L. There is a higher degree of uncertainty associated with the USEPA carcinogenicity assessment for arsenic than with those for other carcinogens (USEPA, 1992), and this estimate may be overly conservative by at least one order of magnitude. More detailed consideration of the toxicology of arsenic is presented in Chapter 5.

Table 6.1
Summary of Drinking Water Standards for New Zealand (MOH, 1995)

Contaminant	Health Based MAV (1) (mg/L)	Aesthetic Guideline Values (mg/L)
Arsenic	0.01	
Boron	0.3	
Chromium	0.03	
Copper	2	1
Pentachlorophenol	0.01	

Note: (1) MAV denotes maximum acceptable value.

The DWSNZ and the WHO guidelines nominate a limit of 10 µg/L for arsenic, which corresponds to an estimated incremental lifetime risk of skin cancer of 6×10^{-4}. However, only 1-14% of the arsenic related skin cancers are fatal, and therefore the estimated risk of fatal cancer spans the nominal threshold risk of 1×10^{-5}.

6.2.3 Boron

Boron is rapidly absorbed by humans, both orally and dermally, but is also readily excreted. Boron exposure may result in a range of non-carcinogenic health effects. Based on health risk assessment procedures, an acceptable concentration of boron in drinking water is estimated to be in the range 0.3 to 3 mg/L, corresponding to 10% to

100% of the acceptable daily intake (ADI). The DWSNZ and the WHO 'Guidelines for Drinking Water Quality' nominate a guideline value of 0.3 mg/L, corresponding to an allocation of 10% of the total exposure to drinking water.[1]

6.2.4 Chromium

The environmental properties of Cr(VI) and Cr(III) differ markedly. Further, the human health effects associated with exposure to each form differ. Cr(VI) has been classified as a confirmed human carcinogen (Group A) by the USEPA via the inhalation route but is not classified as a carcinogen by the oral route. However, exposure to Cr(VI) via inhalation is of lesser relevance in the case of potable water supplies. Both Cr(III) and Cr(VI) cause irritation of the skin and mucous membranes and Cr(VI) has also been associated with sensitisation and ulceration of the skin and mucous membranes. Cr(VI) tends to be the dominant species in aerated or chlorinated water supplies (WHO, 1984), but is readily reduced to Cr(III) in the aquatic environment.

Most published guidelines for drinking water quality nominate limits for total chromium in the range 0.05 to 0.1 mg/L on the assumption that all chromium is present as Cr(VI). The DWSNZ and the WHO guidelines nominate a value of 0.05 mg/L for total chromium on the basis that all chromium is present as Cr(VI). Given the very low solubility of Cr(III) in most groundwaters and surface waters, there is some justification for this assumption.

Cr(III) and Cr(VI) can now be readily differentiated analytically and it is considered appropriate to set separate guidelines for the two species. Using health risk assessment procedures, the acceptable concentrations of chromium in water can be estimated to be 0.02 mg/L and 4 mg/L for Cr(VI) and Cr(III) respectively, based on allowing drinking water to contribute up to 10% of the acceptable intake.

6.2.5 Copper

Copper has been associated with a wide range of chronic and acute adverse health effects in humans, although these are generally only observed at relatively high doses. Quantitative dose-response information is limited for copper. Copper is not classified as a human carcinogen by the USEPA. Based on the available information, health risk assessment suggests that an acceptable concentration of copper in drinking water may be in the order of 0.2 mg/L. Tainting has been reported at copper concentrations in excess of 2.6 mg/L and staining of laundry has been reported at copper concentrations in excess of 1.0 mg/L for copper.

The DWSNZ and the WHO guidelines nominate a health-based MAV of 2.0 mg/L and an aesthetic guideline value of 1.0 mg/L for copper.

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[1] MoH is reviewing the current MAV for boron.

6.2.6 Pentachlorophenol

Chronic exposure to pentachlorophenol (PCP) may result in a range of adverse health effects in humans, including irritation of the skin and mucous membranes, chloracne, neuraesthenia, depression, headaches and changes in kidney and liver function. In addition, PCP has been classified as a probable human carcinogen (Group B2) by the USEPA, for exposure via the oral route. Prior to the recent reclassification of PCP as a carcinogen, drinking water quality criteria up to 60 mg/L were established for PCP on the basis of taste and odour considerations. The USEPA has indicated PCP concentrations in drinking water in the range 0.3 to 3 mg/L may be associated with an incremental lifetime risk of cancer in the range of 10^{-6} to 10^{-5}.

The DWSNZ nominate a provisional MAV of 10 µg/L for PCP based on an ADI of 3 µg/kg/d and allocation of 10% of the ADI to drinking water exposure.

6.3 STOCK WATERING

6.3.1 Introduction

This section sets out water quality guidelines for stock watering. The guidelines have been based principally on the Canadian Water Quality Guidelines (CCREM, 1991) and the Australian Water Quality Guidelines (ANZECC, 1992), which are based on published studies completed in the USA, Canada and Australia.

6.3.2 Arsenic

Arsenic is an essential element for livestock at low levels; however, at higher concentrations it is toxic. The toxicity of arsenic depends greatly on the chemical form in which it appears. Organic arsenic is sometimes used as a feed additive for pigs and poultry. Generally, published stock water quality guidelines for arsenic are in the range 0.2 to 0.5 mg/L, with concentrations up to 5 mg/L tolerated if arsenic intake from other sources, including feed, is low.

As arsenic associated with timber treatment is likely to be in inorganic form, a stock water quality guideline for arsenic of 0.5 mg/L has been nominated, which is consistent with the ANZECC (1992) water quality guidelines.

6.3.3 Boron

There is little information about the toxicity of boron, even at relatively high concentrations. For cattle, the consumption of water contaminated at 150 mg/L was associated with decreased hay consumption and weight loss, and the tolerance concentration for boron was estimated to be between 40 and 150 mg/L for cattle (Green, 1977). Previously published guidelines indicate that a boron concentration of

5 mg/L is acceptable, and on this basis a guideline value of 5 mg/L has been nominated, which is consistent with the ANZECC (1992) guidelines.

There is no evidence that boron concentrations at several times the guideline value would be associated with adverse effects (CCREM, 1991), and it is possible that further information about health effects at higher levels of exposure may indicate that higher values would be acceptable.

6.3.4 Chromium

For humans, Cr(VI) is of most concern, and, assuming domestic animals respond similarly, greatest concern should be focused on Cr(VI). Cr(III) has a very low solubility and is poorly absorbed by the gastrointestinal tract. By contrast, Cr(VI) is relatively soluble in water and is readily absorbed (Owen, 1990). Studies involving domestic animals indicated that a concentration of 5 to 6 mg/L of Cr(VI) did not cause tissue damage whereas a concentration of 10 mg/L resulted in chromium accumulation in the muscle.

Published guidelines for Australia, Canada and the USA each nominate an acceptable concentration of total chromium of 1.0 mg/L based on the effects of Cr(VI). A guideline value of 1.0 mg/L is nominated for Cr(VI). This guideline may also be used as the default for the assessment of total chromium concentrations: i.e. if the chromium is soluble it is likely to be Cr(VI). Given the lower toxicity of Cr(III), it is likely that higher concentrations of Cr(III) would be acceptable.

6.3.5 Copper

Copper is an essential trace element for animal growth. However, excess copper intake can lead to copper toxicosis. Excess copper intake by livestock is normally feed-related, but, because of the use of copper-based chemicals in agriculture, there is a need to set limits for stock water. Livestock species differ in their tolerance to copper, and the following limits have been adopted by the Canadian authorities, and are also proposed by ANZECC:

- Sheep – 0.5 mg/L
- Pigs and poultry – 1.0 mg/L
- Cattle – 5 mg/L

The above species-specific values have been adopted for the purposes of these guidelines.

6.3.6 Pentachlorophenol

None of the available published guideline for stock watering include reference to pentachlorophenol. Both the Canadian and ANZECC guidelines refer to drinking water guidelines for toxic organics, i.e. 10 µg/L for PCP. Higher values may be acceptable depending on the livestock use, the levels of risk that may be acceptable for livestock health, and the potential accumulation of PCP in livestock tissue and milk.

Appendix B presents the derivation of a preliminary guidelines value for stock watering based on consideration of both:

- protection of the health of consumers of livestock products; and
- protection of livestock health.

A preliminary stock watering guideline value for PCP of 0.15 mg/L is nominated, based on the protection of stock health (refer Appendix B). Where appropriate, the preliminary stock watering guideline for PCP may be modified to reflect site-specific information or additional information about the effects of PCP on livestock as it becomes available.

6.4 IRRIGATION

6.4.1 Introduction

This section sets out water quality guidelines for irrigation. They are based principally on the Canadian Water Quality Guidelines (CCREM, 1991) and the Australian Water Quality Guidelines (ANZECC, 1992), which are based on studies completed in the USA, Canada and Australia.

Note that the guidelines have generally been set to protect the most sensitive crop. Higher contaminant concentrations may be acceptable in irrigation water depending on site-specific considerations, such as the crop being grown, the irrigation regime and the capacity of the local soils to assimilate contaminants.

6.4.2 Arsenic

Generally, yield reduction and crop failure are the primary concerns associated with elevated arsenic concentrations in irrigation water and, consequently, soils. Arsenic does not generally accumulate in edible plant portions at levels potentially dangerous to consumers (ANZECC, 1992). Under some circumstances, arsenic uptake from contaminated soils may result in unacceptable contaminant concentrations in the root portions. Nutrient solutions containing 0.5 to 10 mg/L arsenic have been associated with toxic effects on crops. ANZECC and other authorities have adopted an irrigation water guideline concentration of 0.1 mg/L; however, arsenic concentrations up to 2.0 mg/L may be acceptable in neutral to alkaline soils, particularly clays, where

arsenic is adsorbed strongly on the soil particles. On this basis arsenic should not exceed a concentration of 0.1 mg/L in water applied to sandy and acidic soils, with concentrations up to 2 mg/L being acceptable in some circumstances.

6.4.3 Boron

The tolerance of various plant species to boron in the soil water varies considerably, from less than 0.5 mg/L for blackberries to 15 mg/L for asparagus. Boron in small quantities is necessary for the normal growth of plants but at higher concentrations is toxic. The allowable concentration of boron in irrigation water depends on the sorption capacity of the soil and, consequently, higher concentrations of boron in the irrigation water may be allowed where soils are alkaline. While a guideline suitable for all soils cannot be set, boron should not exceed a concentration of 0.5 mg/L for sensitive species, although concentrations up to 6 mg/L may be acceptable in some circumstances, as noted in the ANZECC guidelines.

6.4.4 Chromium

There is no evidence that chromium is essential to plant life. Yield reduction has been associated with chromium concentrations in the range 1 to 10 mg/L, for some plant species. Cr(III) and Cr(VI) are similarly bioavailable in nutrient solutions but in the soil environment, Cr(III) is readily adsorbed on soil particles and is generally less bioavailable.

The CCREM (1991) and ANZECC (1992) guidelines nominate an acceptable concentration of chromium in irrigation water of 0.1 mg/L. The US authorities (CCREM, 1991) indicate that chromium concentrations up to 0.1 mg/L are acceptable for continuous irrigation of acid, sandy soils but that concentrations up to 1.0 mg/L may be acceptable for up to 20 years on neutral to alkaline fine-textured soils. On this basis, an irrigation guideline value of 0.1 mg/L has been nominated for chromium (total).

6.4.5 Copper

Copper is necessary in soils at concentrations greater than 6 mg/kg in order to ensure healthy plant growth; however, at concentrations between 150 and 400 mg/kg copper is toxic to some plants. The toxicity of copper to plant life is pH-dependent, with higher concentrations being tolerated in fine-textured alkaline soils. The Canadian guideline values for copper in irrigation waters (CCREM, 1991) are as follows:

- continuous irrigation of all plant species on all soils – 0.2 mg/L
- low-sensitivity crops (e.g. cereals) – 1.0 mg/L
- neutral to alkaline soils for up to 20 years – 5.0 mg/L

The ANZECC (1992) guidelines nominate a value of 0.2 mg/L for all conditions; however, the Canadian guidelines are considered to represent a more flexible, site-specific approach. In the first instance a guideline value for copper of 0.2 mg/L may be used for assessment of irrigation water quality.

6.4.6 Pentachlorophenol

Published guidelines for pentachlorophenol in irrigation water were not identified. PCP is an effective biocide, with algae being sensitive at concentrations of 0.5 mg/L, and it is possible that some plant species will be sensitive at low concentrations. Sorption of PCP on soil particles may reduce the effective concentration to which plants are exposed. In dilute solution, PCP can be expected to degrade in the soil environment, and is unlikely to accumulate if irrigated at low concentration. In the absence of other information the potable use guideline of 10 µg/L may be used.

6.5 PRIMARY CONTACT RECREATION

Limited published information is available on acceptable concentrations of contaminants in water to be used for primary contact recreation, such as swimming. The ANZECC (1992) guidelines indicate that water containing chemicals which are either toxic or irritating to the skin or mucous membranes is unsuitable for primary contact recreation, and that the concentration of toxic substances should not exceed levels given for untreated drinking water.

Health risk assessment has been used to better quantify the potential adverse effects of bodily immersion in water containing contaminants, and the resulting health-risk-based acceptance criteria for recreational water are presented in Table 6.2. For details of the health risk assessment procedures, refer to Appendix A.

The guideline values presented in Table 6.2 have been developed on the context of regular swimming activities (1 hr/day, 150 day/year). Guideline values more typical of occasional recreational bathing are presented in Appendix A.

Table 6.2
Preliminary Health Risk Based Acceptance Criteria (refer Appendix A for details)
Primary Recreational Use of Surface Water

Receptor:	Children and adults resident onsite for up to 30 yrs.		Target Risk:	1E-05		
			Target HI:	1		
	Exposure Frequency:	150	d/yr	Body Weight (4-10 yr):	30	kg
	Averaging Time (carc.):	70	yrs	Body Weight (adult):	70	kg
	(non-carc.)	6	yrs	Exposure Dur. (4-10 yr):	6	yrs
		24	yrs adult	Exposure Dur. (adult):	24	yrs
	Ingestion Rate:	100	mL/event	Surface Area (4-10 yr):	8290	sq. cm (50th percentile)
	Event Duration (t):	1	hr/d (av.)	Surface Area (adult):	18000	sq. cm (50th percentile)

Contaminant	SF (1/(mg/kg/d))		RfD (mg/kg/d)		Absorption in GI tract (%)[2]	Acceptable CDI (mg/kg/d)				Permeability Constant (cm/h)[4]	Preliminary Health-based Soil Acceptance Criteria (mg/kg)					
						Carcinogenic		Non-carcinogenic			Carcinogenic			Non-carcinogenic		
	Oral	Dermal[3]	Oral	Dermal[3]		Oral	Dermal	Oral	Dermal		Oral	Dermal	Combined	Oral	Dermal	Combined
Metals:																
Arsenic	0.15	0.21			70	6.75E-05	4.7E-05			8.0E-04	2.1E-01	1.3E+00	1.8E-01			
Boron			4.5E-02	4.5E-03	10			4.5E-02	4.5E-03	8.0E-04				3.3E+01	5.0E+01	2.0E+01
Chromium (III)			1.0E+00	4.0E-03	0.4			1.0E+00	4.0E-03	8.0E-04				7.3E+02	4.4E+01	4.2E+01
Chromium (VI)			5.0E-03	5.0E-04	10			5.0E-03	5.0E-04	8.0E-04				3.7E+00	5.5E+00	2.2E+00
Copper			5.0E-02	5.0E-03	10			5.0E-02	5.0E-03	8.0E-04				3.7E+01	5.5E+01	2.2E+01
Organics:																
PCP	0.12	0.15			80	8.3E-05	6.7E-05			6.5E-01	2.6E-01	2.2E-03	2.2E-03			

Note: (1) Preliminary water quality criteria for carcinogens based on average exposure over 30 years and for non-carcinogens on the six years of child exposure.
(2) Owen (1990)
(3) Dermal SF and RfD based on absorbed dose rather than administered dose.
(4) USEPA (1992a)

6.6 PROTECTION OF AQUATIC ECOSYSTEMS

6.6.1 Introduction

The development of general guidelines for the protection of aquatic ecosystems is a difficult and uncertain practice. The acceptable level of a given contaminant in the aquatic environment can depend on a wide range of factors, including:

- the form of the contaminant discharged to the aquatic ecosystem;
- the water chemistry (e.g. pH, hardness, dissolved oxygen content, presence of complexing agents and/or organic matter, temperature);
- the frequency and intensity of discharges (e.g. shock loads vs continuous loads);
- the aquatic species present in that section of the aquatic environment;
- synergistic and antagonistic effects of other contaminants;
- the current state of the aquatic ecosystem and the level of protection to be afforded (i.e. other factors may lead to the degradation of the aquatic ecosystem, and applying guidelines set on the basis of protection of sensitive species in a pristine environment may be unwarranted); and
- the definition of mixing zones associated with discharges consented under the provisions of the RM Act and the requirements for protection of the ecosystem within the mixing zone[1].

Because of these factors, there is a need to assess the requirements for protection of aquatic ecosystems on a site-specific basis. This approach is consistent with the RM Act, which indicates that, where an aquatic ecosystem is to be protected, there should be an assessment of the sensitivity of the local ecosystem, an indication as to whether an adverse effect has occurred and a clear demonstration of where in the downstream water path the ecosystem has been examined.

Less stringent requirements for water quality may apply within the mixing zone: e.g. a requirement for no acute toxicity and a requirement that contaminant levels should not act as a barrier to the migration of aquatic life. Note that these guidelines relate to concentrations in the receiving water where the effect is likely to occur, external to a mixing zone where localised higher concentrations may be tolerated.

1 Depending on conditions associated with the granting of a consent for a discharge to the environment less stringent requirements for the protection of aquatic ecosystems may apply within the mixing zone (e.g. a requirement that fish can pass through the mixing zone without adverse effect).

The Ministry for the Environment has embarked on a process of developing guidelines for the protection of aquatic ecosystems. The framework document 'A Process for the Development of Guidelines for the Protection of Aquatic Life in New Zealand' (MfE, 1995) has been the first stage. This will be followed in due course by the adoption of the ANZECC Water Quality Guidelines when the current revision is completed.

In this study the consideration of aquatic ecosystems has been restricted to freshwater receiving waters, as such systems are likely to be of primary concern given the location of most processing facilities. For information on potential ecosystem impacts in marine environments refer to the Canadian and ANZECC water quality guidelines.

Notwithstanding the desirability of undertaking a site-specific evaluation of the impacts on aquatic ecosystems, a number of published guidelines are available and are useful as part of an initial review.

6.6.2 Arsenic

The toxicity of arsenic depends on its form in the aquatic environment, although As(III) and As(V) exhibit similar bioavailability and toxicity to aquatic organisms (ANZECC, 1992). Because of its chemical similarity to phosphate, arsenate is readily absorbed by phytoplankton and thus enters the food chain (Sanders, 1980). Studies indicate the following toxicological information regarding arsenic in the aquatic environment (CCREM, 1991):

- Invertebrates exhibited signs of acute toxicity associated with As(III) at 812 μg/L.
- Adult fish exhibited signs of acute toxicity associated with As(III) at 13,300 μg/L upward.
- The alga *Scenedesmus obliquis* exhibited signs of toxicity at 48 μg/L.

A guideline concentration of 50 μg/L for total As has been recommended by CCREM (1991), Hart (1982) and ANZECC (1992). The USEPA recommends a concentration of 190 μg/L (4-day average) for As(III), not be exceeded more than once every three years.

6.6.3 Boron

There are no readily available published guidelines for boron for the protection of aquatic ecosystems. It may be argued that given impacts on plant life are some of the most sensitive effects associated with boron that guideline values nominated for irrigation may be used as a first indication of the likelihood of adverse impact on aquatic species, particularly plants. The USEPA refers to the irrigation guideline of 750 μg/L for sensitive crops. The ANZECC (1992) guidelines nominate a limit of 500 μg/L for sensitive crops.

Hickey (1989) conducted aquatic toxicity testing for a range of cladocerans, including some New Zealand species. Reported EC_{50} values for boron ranged from 101 to 319 mg/L, while EC_{10} values ranged from 38 to 250 mg/L. The lowest NOAEL reported for boron was 10 mg/L. It is understood unpublished information indicates some other species, particularly aquatic plants, are more sensitive to boron by a factor in the order of 3.

Based on the information outlined above, a preliminary aquatic ecosystem protection criterion for boron in the order of 0.75 mg/L is expected to be protective of the aquatic environment.

6.6.4 Chromium

The form of chromium, i.e. Cr(VI) or Cr(III), affects both its fate and toxicity in the aquatic environment. Cr(III) is much less soluble than Cr(VI) but may be present in the aquatic environment in suspension, or in solution complexed with organic anions. The ratio of Cr(III)/Cr(VI) is dependent on the organic matter present and the concentration of dissolved oxygen. Chromate, because of its chemical similarity to sulphate, is taken up by phytoplankton and can bioaccumulate in higher aquatic organisms. Studies indicate Cr(VI) is much more toxic to aquatic organisms than is Cr(III). The CCREM (1991) sets a guideline of 20 µg/L (total) for the protection of fish, and 2 µg/L (total) for protection of all aquatic life.

The Canadian water quality guidelines assume, for simplicity, that all chromium is present as Cr(VI), and set a value of 2 µg/L for total chromium for the protection of aquatic ecosystems (at a hardness of 50 mg/L). The ANZECC (1992) guidelines nominate a value of 10 µg/L (total). The USEPA provides separate guidelines for Cr(III) and Cr(VI) of 120 µg/L and 11 µg/L respectively. Given the differences in toxicity between Cr(VI) and Cr(III), it is considered appropriate that separate guidelines be set for each form. Where chromium concentrations are expected to be low, it may be appropriate to analyse for total chromium; however, where the concentration of total chromium exceeds the guidelines based on Cr(VI), analysis for each form becomes important.

The toxicity of Cr(III) to freshwater aquatic life is hardness-dependent; the USEPA guideline value for Cr(III) may be determined using the following relation:

$$\text{Acceptable chromium concentration (mg/L)} = e^{[0.819 \ln (\text{hardness (mg/L)}) + 1.561]}$$

6.6.5 Copper

The toxicity of copper to freshwater aquatic life is also dependent on the water hardness: toxicity increases with decreasing hardness. High concentrations of chelating agents and suspended solids lead to the formation of complexes that are less bioavailable, thus reducing the toxicity of copper. In the aquatic environment more than 98% of copper tends to be bound to organic material. The tolerances of fish,

invertebrates and freshwater plants to copper appear to be similar CCREM (1991). In assessing the impact of copper on aquatic ecosystems, the following should be considered:

- At a hardness of 50 µg/L, acute toxicity data for freshwater species ranged from 17 mg/L for *Ptychocheilus* to 10,000 mg/L for *Acroneuria* (USEPA in ANZECC, 1992).
- At a hardness of 50 µg/L, chronic toxicity values for 15 freshwater species ranged upward from 4 mg/L (USEPA in ANZECC, 1992).
- Changes in fish behaviour have been demonstrated at values as low as 4 µg/L (CCREM, 1991).
- CCREM (1991) developed a guideline of 2 to 6 µg/L depending on hardness.
- Hart (1982) established a criterion of 5 µg/L for filterable copper in soft waters.
- ANZECC (1992) established a guideline of 2 to 5 µg/L for copper in fresh waters, depending on water hardness.
- The USEPA has developed ambient water quality criteria for the protection of freshwater aquatic ecosystems such that the acceptable concentration of copper for chronic exposure is given by:

  Acceptable copper concentration (mg/L) = $e^{(0.8545 \ln [\text{hardness (mg/L)}] - 1.465)}$

  At a hardness of 50 mg/L, this expression results in an acceptable copper concentration for chronic exposure of 6.5 µg/L.

6.6.6 Pentachlorophenol

Studies involving PCP-contaminated freshwater lakes and ponds have shown that, although photolysis can result in rapid degradation of PCP, the chemical can persist for several months in water and fish, and for years to decades in sediments (Rao, 1978; Minister of Supply and Services, Canada, 1989). Fish were found to accumulate PCP and PCP degradation products rapidly from water, with fish liver and gill tissue exhibiting the highest concentrations of PCP. In addition, the persistence of PCP and its degradation products in sediments was found to provide a source of ongoing contamination of the aquatic environment (Rao, 1978; Minister of Supply and Services, Canada, 1989).

PCP, like many other chlorinated organic compounds, is soluble in fatty substances (ANZECC, 1991) and tends, therefore, to accumulate in the fatty tissues of living organisms. This process is known as bioaccumulation and it can occur by two pathways: bioconcentration, in which minute quantities of a chemical can be taken up and progressively concentrated in the fatty tissues; and biomagnification, in which the concentration of a chemical can be magnified several-fold by the consumption of organisms already contaminated with the chemical. Rainbow trout, which feed by predation, have been found to bioaccumulate PCP at up to 600 times the levels found

in the surrounding water by feeding on organisms which are already contaminated (ANZECC, 1991).

Several factors affect the toxicity of PCP in freshwater (ANZECC, 1990). PCP is more toxic to aquatic organisms at low pH, where concentrations of dissolved oxygen are low, or at elevated temperatures. These effects can either be species or life-stage specific.

The toxic effects of PCP have been studied in a range of organisms. Sensitivity can vary greatly between species. The 96 hour LC_{50} values for fish, for example, can vary from 0.03 to 3 mg/L (ANZECC, 1990). The WHO monograph on PCP (WHO, 1987) provides additional toxicological information.

Canadian environmental authorities have developed various PCP limits for protection of receiving waters (Environment Canada, 1988). Generally these limits apply to the water column of receiving waters, and have the objective of protecting the most sensitive biological species within the relevant ecosystems. These limits have been formulated with respect to Canadian ecosystems, and there may be some differences in the sensitivity of New Zealand ecosystems.

The acceptable concentration of PCP in aquatic biota and the concentration at which chronic sublethal effects will be observed are not well-defined. The British Columbia Ministry of the Environment (Environment Canada, 1988) has proposed a maximum PCP concentration in fish muscle of 100 mg/kg (wet weight). However, this figure appears to be based on considerations of background concentrations and suitability for human consumption, and since there is uncertainty as to its relevance as an indicator of impact on aquatic ecosystems it is considerated inappropriate for use as a criterion until its basis has been clarified.

Studies have shown that PCP at 0.03 mg/L in the water column causes tainting of the flesh of fish and other aquatic organisms; however, toxicity to aquatic species is found to be the limiting consideration compared to tainting of fish flesh.

Acceptable concentrations of PCP in sediments are not usually specified; however, the BC Ministry of the Environment has proposed a maximum PCP concentration in bottom surface sediments of 10 mg/kg based on background concentration considerations (Environment Canada, 1988).

Given the pH dependence of PCP toxicity, the USEPA (1986) has developed ambient water quality criteria for the protection of freshwater aquatic ecosystems based on the following relationships:

Acceptable PCP concentration – Acute (µg/L): $e^{(1.005\ (pH) - 4.830)}$ (1 hr average)

Acceptable PCP concentration – Chronic (µg/L): $e^{(1.005(\ pH) - 5.290)}$ (4 d average)

At pH 7, the above expression results in a chronic (4 d average) value of 5.7 µg/L.

While the ANZECC (1992) and CCREM (1991) guidelines adopt a similar approach for the development of guideline values protective of aquatic ecosystems, different values are nominated, reflecting differences in the toxicity data on which the guideline is based. The ANZECC (1992) guidelines have adopted the objective of 0.05 mg/L, based on an acute toxicity value of 4.4 µg/L for larval carp and an application factor of 0.01 (default value for a chemical which is persistent or which requires additional caution because of a limited data set). The CCREM (1991) guidelines nominate a value of 0.5 µg/L based on the lowest **mean** toxicity concentration of 55 µg/L for coho salmon, and an application factor of 0.01.

For the purposes of Chapter 7 it is necessary to nominate a PCP aquatic ecosystem protection guideline value for interim use. The recommended guideline values for the protection of aquatic ecosystems are:

- Modified ecosystems: 0.5 µg/L
- Pristine ecosystems: 0.05 µg/L

The interim recommendations will be reviewed as part of the Organochlorines Programme.

6.7 SUMMARY OF PROPOSED CRITERIA

The guidelines proposed for each contaminant and each potential use of groundwater or surface water are summarised in Table 6.3. Detailed consideration of the basis of the proposed values is presented in Sections 6.2 – 6.6.

The proposed guidelines have been based on total contaminant concentrations in groundwater and surface water, with the exception of chromium, for which separate values have been proposed for Cr(III) and Cr(VI). Where contaminant concentrations are low, as a first check, it may be appropriate to compare the results of total chromium analyses with the guideline value for Cr(VI).

In some cases the guidelines consist of a range rather than a single value, indicating that the application of the guidelines is particularly dependent on site-specific considerations.

As discussed in Section 6.6, contaminant concentrations associated with an adverse impact on aquatic ecosystems can vary significantly depending on the water chemistry, and the nature of the contaminant and aquatic flora and fauna. On this basis, no specific guideline values have been proposed for ecosystem protection (other than the interim values for PCP for use in Chapter 7), and it is recommended that the potential impact of contamination on aquatic ecosystems be reviewed on a site-specific basis, with reference to published guidelines as may be appropriate. Such an approach is considered to be consistent with the provisions of the RM Act, which may be interpreted as requiring consideration of the sensitivity of the local ecosystem and of whether an adverse effect has occurred. Published criteria may be used as a screening

tool, and site-specific evaluation should follow where appropriate (Hannah and McFarlane, 1992).

The guidelines in this document have been framed in terms of protecting different uses of surface waters and groundwaters, rather than being based on various classes or types of water body.

In applying the water quality guidelines it is important that a critical review be undertaken of the possible beneficial uses of a given groundwater or surface water body, and consideration be given to site-specific factors in any assessment of surface water or groundwater contamination. For example, natural water quality (e.g. salinity) and availability may restrict the potential beneficial uses of a groundwater or surface water body, and consequently it may not be appropriate to protect such water bodies for all beneficial uses. Further, the extent of a mixing zone, in which localised higher concentrations may be accepted, should be considered.

For these reasons the proposed guidelines should be viewed as being flexible and indicative only, with some variation being expected on a site-specific basis.

Water quality guidelines for dioxins and furans have not been specifically developed as part of these guidelines. A review of published water quality guidelines was undertaken as part of the 'Pentachlorophenol Risk Assessment Pilot Study' (NTG, 1992). The criteria adopted for dioxins and furans for the purposes of the NTG study are as follows:

- Potable use: 0.015 ng/L (TE)
- Aquatic ecosystem protection: 0.01 ng/L (TE)

These values may be used as interim criteria, which provide a preliminary indication of maximum acceptable dioxin and furan concentrations in surface water and groundwater.

Table 6.3
Summary of Water Quality Guidelines
(all values expressed in mg/L unless otherwise specified)

USE	Contaminant						
	Arsenic	Boron	Chromium			Copper	PCP
			(III)	(VI)	Total		
Potable	0.01	0.3			0.05	1	0.01
Stock watering	0.5	5		1	1	0.5(1)	0.15
Irrigation	0.1(2)	0.5(3)			0.1	0.2(3)	0.01
Primary contact recreation	0.21	20	41.5	2.19		21.9	0.002
Aquatic ecosystem protection							
• CCREM (1991)	0.05	–		0.002	0.002	0.002-0.004	0.0005
• ANZECC (1992)	0.05	–		0.01	0.01	0.002-0.005	0.00005
• USEPA	0.19	–	0.12(4)	0.011		0.0065(4)	(4)
• Other		0.75					

Note: (1) Based on sheep – higher values may be tolerable for other livestock.
(2) Based on acid, sandy soils – higher values may be tolerable under other conditions.
(3) Based on sensitive crops – higher values may be acceptable depending on the crop.
(4) Refer expression relating guideline value to pH or hardness.

6.8 REFERENCES

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APPENDIX A
HEALTH RISK-BASED ACCEPTANCE CRITERIA FOR DRINKING WATER AND PRIMARY CONTACT RECREATION

A1. EXPOSURE ROUTES AND RECEPTOR GROUPS

A1.1 Drinking Water

For the purposes of this assessment the Maximum Acceptable Values (MAVs) nominated in the DWSNZ have been adopted as acceptance criteria for potable use of water. Therefore risk-based criteria for potable use have not been independently derived for any of the contaminants of concern; however, an approach similar to that presented in the following sections may be used if necessary.

In the case of drinking water exposure, the lifetime ingestion of water by adults is usually used as the basis for the derivation of criteria. While a child may be exposed to a weight-standardised rate approximately twice that of an adult, the duration of exposure for adults may be up to 10 times that for children. For some volatile contaminants, such as benzene, human exposure during showering and similar activities can be a significant exposure route. However, given the contaminants of concern in this assessment exhibit low volatility the primary exposure route for potable water supplies is considered to be ingestion of drinking water. The derivation of the DWSNZ reflects consideration of the ingestion of water only.

A1.2 Primary Contact Recreation

Primary contact recreational activities such as bathing necessarily involve intimate contact between those involved and the potentially contaminated water. Both children and adults are considered in this assessment. The intention is to quantify risks associated with regular swimming in surface water bodies. However, the criteria derived may also be applied to the assessment of groundwater used to fill swimming pools, except in the case where such pools are used for very regular training activities (>150 events/year, e.g. commercial swimming pools). The assessment therefore focuses on recreational bathing rather than regular training activities. Both incidental ingestion of water during bathing and dermal absorption have been considered in this assessment.

A2. RISK ASSESSMENT AND THE DEVELOPMENT OF RISK-BASED ACCEPTANCE CRITERIA

A2.1 General

A discussion of risk assessment principles and the process for the development of risk-based acceptance criteria is presented in Section 5.4. Such a process can be applied to the derivation of criteria for both soil and water.

The risk assessment process may be summarised in four steps as follows:

- Hazard Identification;
- Exposure Assessment;
- Toxicity Assessment; and
- Risk Characterisation.

In derivation of acceptance criteria, the above process is run in reverse, starting with definition of an acceptable level of risk.

A2.2 Toxicity Assessment

In order to relate estimates of exposure to the risk of adverse health effects in humans a range of dose response factors have been developed by various health and environmental agencies, based on a review of published toxicological and epidemiological information. The health effects associated with exposure to copper, chromium, arsenic, boron and pentachlorophenol, and the dose response factors nominated for each of these chemicals are discussed in Section 5.5. Also refer to Section 5.10.5 for details of the corrections to the adopted dose-response factors necessary to account for background exposure.

A2.3 Exposure Assessment

A2.3.1 Ingestion of Contaminated Water

The Chronic Daily Intake (CDI) may be determined by the following expression.

$$CDI = \frac{C \times IR_{adj} \times EF \times MF}{AT} \quad \text{(A1)}$$

where: C = concentration of contaminant in the water (mg/L)

IR_{Adj} = age adjusted ingestion rate (L/d)

$$= \frac{\Sigma\ ED_i \times IR_i}{Bwi} \quad \text{(A2)}$$

where: ED = exposure duration for age group 'i' (yr)
IR = ingestion rate for age group 'i' (mg/d)
BW = body weight for age group 'i' (kg)

EF = exposure frequency

AT = averaging time
= (ED x 365) days for non-carcinogens by convention or (70 years x 365) days for carcinogens, a lifetime, by convention

MF = matrix factor, accounts for reduced bioavailability of contaminant due to binding to the soil matrix. In the absence of necessary information, MF usually taken as 1.0.

A2.3.2 Dermal Absorption from Contaminated Water

The Chronic Daily Intake (CDI) for dermal absorption from contaminated soil may be determined from the following expression (based on USEPA, 1988):

$$CDI = \frac{t \times AV_{adj} \times C \times PC \times EF \times CF}{AT} \quad (A3)$$

where: t = duration of exposure (hours/event)

AV_{adj} = age-adjusted skin surface area

$$= \frac{\Sigma\ AV_i \times ED_i}{BW_i}$$

Where: AV_i = skin surface area for age group 'i' (cm^2)
ED = exposure duration for age group 'i' (yr)
BW = body weight for age group 'i' (kg)

C = contaminant concentration in water (mg/L)

PC = dermal permeability constant (cm/hr)

EF = exposure frequency (event/yr)

AT = averaging time (days)

CF = conversion factor
= 10^{-3} L/cm^3

Note, the USEPA (1992) has recently released further guidance regarding the estimation of dermal exposure. The above procedure is retained for the assessment of exposure to inorganics whereas some revision of the procedure has been adopted for organic contaminants. The revised approach may be described as follows:

$$CDI = \frac{DA_{event} \times ED \times EF \times AV}{BW \times AT} \quad (A4)$$

where: DA_{event} = Absorbed dose per event (mg/cm event)

DA_{event} is a function of the duration of each exposure event, the concentration of the contaminant in water and a number of contaminant specific parameters. Refer to USEPA (1992) for details.

Note, for the purposes of developing human health-based preliminary remediation goals for non-carcinogenic health effects, the most sensitive receptor, i.e. children, only is considered in the assessment of primary contact recreational exposure.

A3. DEVELOPMENT OF ACCEPTANCE CRITERIA FOR PRIMARY CONTACT RECREATIONAL USE

Health-based preliminary acceptance criteria for primary contact recreational use of water have been developed giving consideration to both the ingestion and dermal absorption exposure routes. The major exposure assumptions are summarised as follows:

• exposure duration	=	Child (4-10 yrs): 6 yrs Adult (10-30 yrs): 20 yrs	
• water ingestion rate	=	130 mL/event	(ANZECC, 1992)
• skin surface area	=	Child (4-10 yrs): 8290 cm^2 Adult (10-30 yrs): 18000 cm^2	(USEPA, 1989a)
• body weight	=	Child (4-10 yrs): 30 kg Adult (10-30 yrs): 70 kg	(USEPA, 1989b)
• exposure frequency	=	150 event/yr	(USEPA, 1992)
• event duration	=	1 hr/event	(USEPA, 1992)

A4. REFERENCES

ANZECC/AWRC (1992) 'Australian Water Quality Guidelines for Fresh and Marine Waters' Australian and New Zealand Environment and Conservation Council.

USEPA (1989a) 'Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part A)' EPA/540/1-89/002.

USEPA (1989b) 'Exposure Factors Handbook' EPA 600/8-89-043.

USEPA (1992) 'Dermal Exposure Assessment: Principles and Applications' EPA/600/8-91/011B.

APPENDIX B
DERIVATION OF STOCKWATERING GUIDELINE FOR PCP

B1. GENERAL

Two issues to be considered in deriving stockwater acceptance criteria are as follows:

- Protection of animal health
- Protection of human health (considering consumption of livestock products).

Information regarding the possible adverse health effects of PCP on livestock health is not readily available. Historically, in the absence of other information, the potable water quality guideline for PCP, which is protective of human health, has been used. In practice, a lower level of concern is attached to the protection of animal health, compared to the protection of human health. On this basis, while a potable guideline for PCP of 10 µg/L may be appropriate for the protection of human health, a higher value may be appropriate as a stockwater guideline.

B2. PROTECTION OF HUMAN HEALTH

Preliminary acceptance criteria for PCP for the protection of human health where water is used for stockwatering purposes have been derived based on:

- An incremental lifetime risk of cancer of 1 in 100,000;
- Typical livestock product consumption rates (Langley, 1996);
- Published correlations between intake of contaminants and concentrations in livestock products (ECETOC, 1990).

For the purposes of criteria derivation, cattle have been used as a representative species.

A correlation between the concentration of PCP in stockwater and the concentration of PCP in various cattle tissues/products based on published relationships (ECETOC, 1990).

$$I = C \times Q \qquad \text{(B1)}$$

where I = PCP intake (mg/d)

C = Concentration of PCP in stockwater (mg/L)

Q = Consumption of water by cattle
= 55 L/day

$$Cp = I \times Kp \qquad \text{(B2)}$$

where Cp = Concentration of PCP in the product (mg/kg)
I = Intake (mg/d)
Kp = Contaminant partition co-efficient for product ‘p’

For organics

$$\log K_{meat} = -6.88 + 0.832 \log K_{ow} \quad \text{(B3)}$$

$$\log K_{milk} = -6.786 + 0.731 \log K_{ow} \quad \text{(B4)}$$

$$\log K_{fat} = -3.457 + 0.500 \log K_{ow} \quad \text{(B5)}$$

Note: log Kow (PCP) = 5.05 (AERIS, 1991)

Average livestock product consumption rates (average of mean value for males and females) are summarised as follows (Langley, 1996):

- Meat: Male: 308 g/d
  Female: 183 g/d
  Average: 246 g/d
- Milk: Male: 356 g/d
  Female: 247 g/d
  Average: 302 g/d

Table B1 presents estimates of the intake of PCP by consumers of livestock products associated with exposure to 1 mg/L of PCP in stockwater. The acceptable concentration of PCP in stockwater can then be determined by ratio as required. On this basis a preliminary acceptance criterion for PCP in stockwater of 0.14 mg/L is nominated. The nominated criterion should be regarded as preliminary reflecting the uncertainty associated with modelling the uptake and accumulation of PCP in livestock.

B3. PROTECTION OF STOCK HEALTH

The derivation of water quality criteria protective of stock health presented below draws on that outlined in Appendix XV of the Canadian Water Quality Guidelines (CCME, 1993).

For the purpose of deriving acceptance criteria, cattle have been selected as a representative of livestock as it has been reported that the range of water consumption per unit body weight ratio is similar in both the larger (i.e. cattle) and the smaller animal (i.e. goat) (CCME,1993).

The acceptance criteria may be derived as follows:

$$\text{Acceptance Criterion} = \frac{\text{Acceptable Daily Intake (ADI)} \times \text{Body Weight}}{\text{Water Ingestion Rate}} \times \text{Proportion Assigned to Drinking Water} \quad \text{(B6)}$$

where:
Body Weight = 500 kg (Shell, 1994)
Water ingestion rate = 55 L/day (Shell, 1994)
Proportion assigned to drinking water = 0.5

Table B1
Preliminary PCP Groundwater Acceptance Criteria Based on Protection of Human Health: Exposure Via Consumption of Livestock

Oral slope	0.12	mg/kd/d[1]	Base water conc:	1 mg/L
Tarket risk	1.0E-05		Cattle ingestion rate:	55 kg/d
			Human body weight:	70 kg

Product Consumption (g/d)			
Product	**Male**	**Female**	**Average**
Meat	308	183	246
Milk	356	247	302

Cattle product	Partition coefficient	Total intake for cattle (mg/d)	Concentration (mg/kg, fresh weight)	Average consumption rate (g/d)	Estimated human intake (mg/d)	Body weight (kg)	CDI (mg/kg/d)	Dose response factors (mg/kg/d)[1]	Estimated cancer risk for 1 mg/kg PCP in soil
Meat	2.10E-03	5.50E+01	1.16E-01	246	2.8E-02	70	4.1E-04	0.12	4.86E-05
Milk	8.10E-04	5.50E+01	4.46E-02	302	1.3E-02	70	1.9E-04	0.12	2.30E-05
Combined					4.18E-02	70	6.0E-04	0.12	7.16E-05

Notes: (1) Equates to target risk

Acceptable groundwater concentration! **0.14 mg/L**

In selecting the dose response factors (e.g. ADI) for use in establishing the stockwater acceptance criteria, it has been assumed that:

- Cancer is not a relevant endpoint for the establishment of stockwater criteria protective of cattle. The basis for this assumption relates to:
  - the lower level of protection that may be assigned to livestock compared to humans;
  - the relatively low risk of cancer expected at exposure levels consistent with protection against other adverse health effects; and
  - at least in the case of beef cattle, the relatively short duration of exposure compared to the natural life span of cattle; and
- Full protection of all sensitive sub-populations is not required.

On this basis, the ADI for the contaminants of concern has been determined from experimental data for a non-carcinogenic endpoint, assuming a threshold dose response relationship. The preliminary acceptance criteria for PCP based on the protection of livestock health, and the basis for the derivation, are summarised in Table B1.

Toxicological information for PCP is summarised in Section 5.5. Reference Doses/ Acceptable Daily Intakes for PCP nominated by various agencies range from 0.03 mg/kg/day to 0.003 mg/kg/day. The DWSNZ nominate a NOAEL for PCP of 3 mg/kg/day and apply an Uncertainty Factor of 1000. On this basis an acceptable daily intake for the protection of stock health of 0.03 mg/kg/day has been used as the basis for the derivation of the stockwater acceptance criteria (i.e. Uncertainty Factor of 100 rather than 1000 reflecting the lower level of protection assigned to stock compared to humans). It may be argued that an acceptable intake of 0.3 mg/kg/day could be used for derivation of stock water criteria, given the Uncertainty Factor nominated in the DWSNZ includes some consideration of the potential carcinogenicity of technical grade PCP. However, in this case a conservative approach has been adopted for the derivation of preliminary criteria.

Based on the assumptions outlined above, a preliminary stockwater acceptance criterion for PCP, that is protective of stock health, of 0.015 mg/L has been nominated.

Considering both the protection of stock health and human health a preliminary acceptance criterion for PCP in stockwater of 0.15 mg/L is nominated.

B4. REFERENCES

Aeris (1991) ‘AERIS Model Technical Manual, V 3.0’ AERIS Software Inc. Ontario.

CCME (1993) ‘Canadian Water Quality Guidelines, Appendix XV’ Environment Canada, Ottawa.

ECETOC (1990) 'Technical Report No. 40, Hazard Assessment of Chemical Contaminants in Soil' European Chemical Industry Ecology and Toxicology Centre.

Langley A. and L. Sabordo (1996) 'Exposure Factors in Risk Assessment' Proceedings of Third National Workshop on the Health Risk Assessment and Management of Contaminated Sites, SA Health Commission.

Shell (1994) 'The Concepts of HESP, Reference Manual, Human Exposure to Soil Pollutants, V 2.10a' The Hague.

CHAPTER 7
DISPOSAL OF TIMBER TREATMENT WASTES TO LANDFILLS

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APPENDICES

7. DISPOSAL OF TIMBER TREATMENT WASTES TO LANDFILLS

7.1 INTRODUCTION

7.1.1 Context

Historical use of timber treatment chemicals, including PCP, has given rise to a range of wastes containing chemical residues requiring treatment and/or disposal.

Landfills have been the traditional means to dispose of the majority of wastes in New Zealand. Due to the lack of past controls on disposal, New Zealand now has a significant legacy of hundreds of landfills many of which are likely to pose a high or moderate risk to the environment (MFE, 1992).

Although landfill guidelines (CAE, 1992) should ensure that future landfills are better constructed and managed, present-day experience with managed co-disposal of hazardous wastes is limited. Many practices have not been based on good scientific understanding. Therefore the guidance provided in this chapter adopts a precautionary approach so as to be protective of the environment. The methodology used to derive disposal criteria is, however, consistent with that used to derive the soil and water criteria in earlier chapters of these guidelines.

While the co-disposal of some PCP wastes may be appropriate in some circumstances, there is still a risk of landfill failure and consequential release of contaminants to the environment. This is a factor to be weighed in favour of achieving the destruction, if at all practicable, of organochlorine constituents such as PCP and dioxins.

7.1.2 Hazardous Waste Policy

There is a general duty under the RM Act 1991 (s.17) that every person must avoid, remedy or mitigate any adverse effect of their activities on the environment. This duty would apply to the disposal of hazardous substances and wastes.

There is also a requirement under the Fourth Schedule of the RM Act for resource consent applicants to assess all methods of discharging contaminants. In addition regulatory authorities will only grant consents that are consistent with waste management policies and plans.

7.1.3 Aim and Objectives

The main aim of this chapter is to provide guidance to industry and regulatory authorities on the disposal of PCP-containing wastes to landfill. Interim guidance is provided on wastes containing dioxin constituents.

The objectives of this chapter are:

(a) to develop a risk assessment framework for the disposal to landfill of PCP and other hazardous constituents;

(b) to develop acceptance criteria and guidelines to ensure that what is co-disposed to landfill is in accordance with good practice and appropriate to the nature of the landfill.

The risk assessment methodology employed can be applied to the assessment of other contaminants, but no detailed guidelines have been developed for other constituents.

7.1.4 Principles and Assumptions

This chapter is based on the following principles and assumptions:

- that it is more appropriate to destroy higher-strength PCP and dioxin wastes than to have these landfilled;
- that technologies to destroy organochlorine-bearing wastes will become commercially available in New Zealand in the short to medium term;
- disposal to landfill is a last-resort action under the Government's waste management hierarchy;
- that there is a legitimate place for the carefully managed co-disposal of smaller quantities and/or lower concentrations of PCP wastes;
- that there will be some concentration below which a chemical constituent of waste is no longer hazardous for landfill disposal.

7.1.5 Summary of Landfill Disposal Strategy

Landfill Classification System

To facilitate the proper management of the disposal of timber treatment wastes to landfill, a classification system for landfill sites in New Zealand has been developed. In developing this system, reference is made to landfilling principles, such as the natural attenuation and engineered containment approaches, and to landfill development methods. The current New Zealand practice is discussed in the light of these principles.

A three-tier landfill classification systems has been developed, as follows:

Class 1: Represents the formation of small, specially developed and lined cells within a Class 2 site.

Class 2: A site that is suitable for co-disposal of wastes containing relatively low concentrations and quantities of hazardous constituents.

Class 3: An appropriately sited, engineered and operated landfill of older design receiving municipal waste only.

Criteria for classification of landfills are presented in Table 7.2.

Classification and Characterisation of Waste Materials for Landfill Disposal

In order to characterise wastes and determine their suitability for disposal to landfill it is necessary to assess:

- the waste constituent concentrations and the total amount of each constituent,
- the tendency of contaminants to leach out of the waste matrix.

(Other properties such as corrosivity and flammability may also need to be assessed.)

A waste manifest system should be employed to monitor and record the generation, transport and disposal of wastes. An example waste manifest is presented; however, where a regulatory authority operates a separate waste manifest system, this may suffice.

Acceptance Criteria for Disposal of PCP Wastes by Landfilling

Acceptance criteria have been developed on the premise that wastes, when properly disposed and contained within the landfill, do not pose a risk to human health or the environment. However:

- a failure of the landfill capping system may release solid waste constituents to the soil environment;
- leachate seepage or runoff may transport soluble waste constituents to surface water or groundwater.

Therefore, in accordance with a classification system (Class 1, 2 or 3):

- waste concentrations should not exceed specified threshold values for PCP (refer Table 7.4);
- wastes should be solid or in spadeable form, and should pass a TCLP leachate test for PCP (refer Table 7.5);
- wastes containing **less than** 10 μg/kg (TE) of dioxins based on full congener analysis, or less than 5 μg/kg (TE) based on the OCDD screening methodology, may be considered for disposal at a Class 1, 2 or 3 landfill, provided the requirements for landfilling PCP-contaminated waste are also met;
- wastes containing **more than** 10 μg/kg (TE) based on full congener analysis of dioxins, or more than 5 μg/kg (TE) based on the OCDD screening methodology, should not be disposed to landfill and should be stored pending destruction or remediation.

The above landfill acceptance criteria for PCP are summarised in Table 7.1.

Table 7.1
Thresholds for Disposal of PCP- and Dioxin-Containing Wastes to Landfill

Threshold	Landfill Classification		
	1	**2**	**3**
Pentachlorophenol:			
Quantity (g)	–	–	250(1)
Concentration (mg/kg)	700	210	28
Leachate Concentration:			
– Potable Resource (mg/L)	50	25	1
– Aquatic Ecosystems (mg/L)	2.5	1.25	0.05
Dioxins:			
Concentration (µg/kg TE)(2)	10(4)	10(4)	10(4)

Notes: (1) Subject to landfill specific risk assessment if threshold exceeded.

(2) Not to be disposed to landfill if threshold exceeded.

(3) A case may be established to modify the landfill disposal criteria following a site-specific assessment of the landfill in accordance with the principles set out in Section 7.5.

(4) Based on full congener analysis[1] or 5 µg/kg (TE) based on the OCDD screening methodology[2].

7.1.6 Chapter Outline

A brief overview of the present status of alternative waste management options is provided in Section 7.2. Thereafter this Chapter considers the landfill disposal issue, and develops guidelines for the disposal of waste materials to landfills. The risk assessment methodology adopted is intended to be generally applicable to a range of contaminants found in soil and other wastes generated by industry at large; however, this document focuses on pentachlorophenol (PCP).

Section 7.3 examines landfilling practices in New Zealand and provides a classification basis for suitability of the co-disposal of hazardous contaminants.

[1] The term full congener analysis refers to the isomer-specific determination of all polychlorinated dibenzodioxin or dibenzofuran compounds with chlorine substituents in the 2, 3, 7 and 8 positions.

[2] The OCDD screen shall include quantification of 2, 3, 7, 8 – substituted hepta- and octachlorinated dioxins and furans.

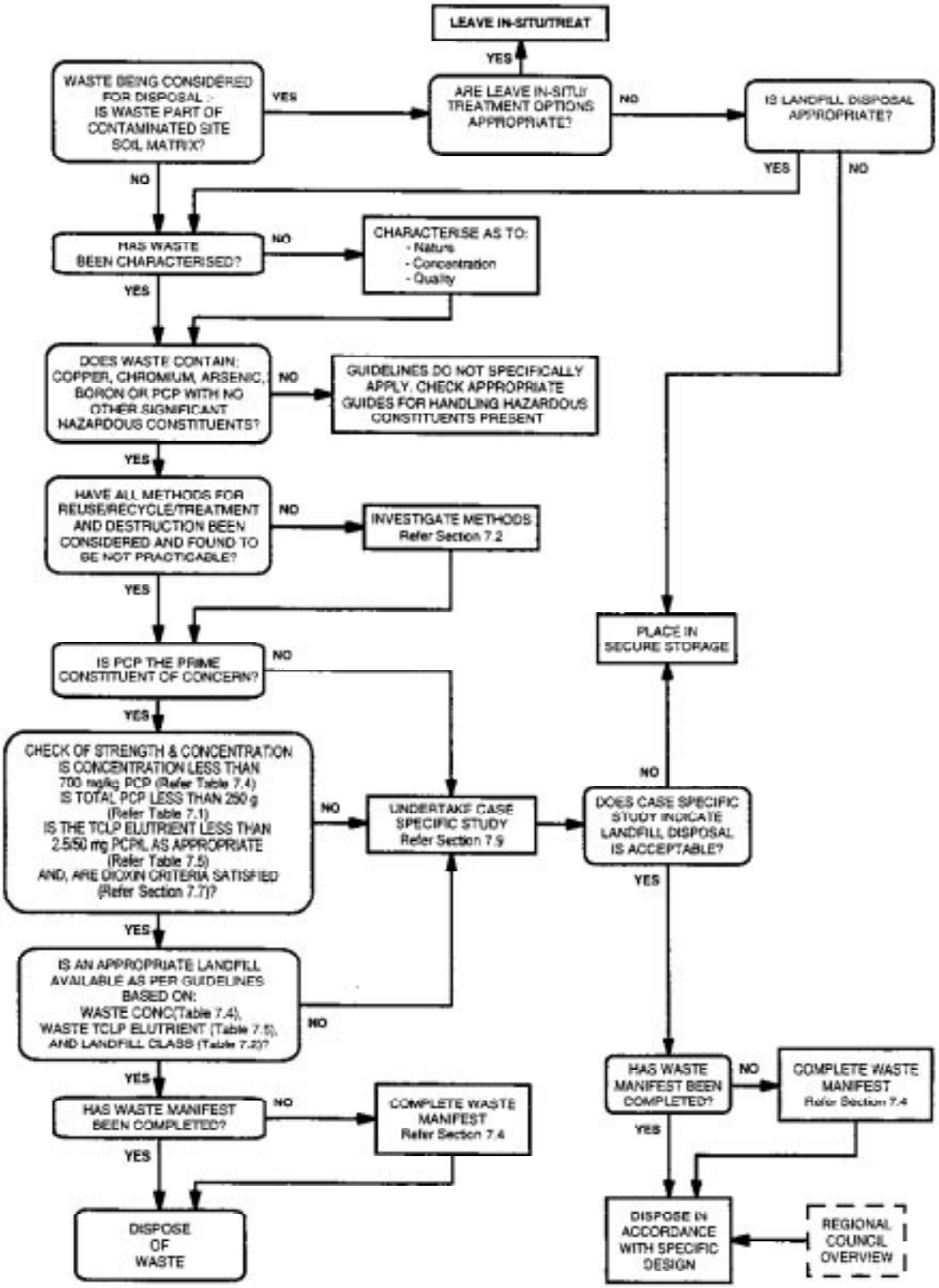
Section 7.4 reviews the use of leaching or elutriation tests, and discusses the development of a waste manifest system. Section 7.5 develops preliminary criteria for disposal of timber treatment wastes to different classes of landfill, based on:

- leachate;
- potential health and environmental effects; and
- waste disposal practices.

Section 7.6 discusses the application of the acceptance criteria methodology to other waste constituents, and Section 7.7 addresses the disposal of dioxin-contaminated wastes.

The procedure for landfilling PCP contaminated wastes outlined in this chapter is illustrated in Figure 7.1.

Figure 7.1
Logic Chart: Procedure for Landfilling PCP Contaminated Wastes



7.2 ALTERNATIVE MANAGEMENT OPTIONS TO LANDFILLING

7.2.1 Purpose

The purpose of this section is to provide a brief overview of alternative waste management methods.

7.2.2 Waste Management Hierarchy

As with other waste streams, the application of the "5 Rs" waste management hierarchy is promoted for timber treatment wastes. The 5 Rs are, in order of priority: reduce, reuse, recycle, recover, and residual management. An example of these principles being applied is the recovery of copper-chrome-arsenic from timber treatment sludges and sump residues; the extraction of reusable chemical being carried out by the chemical supplier or a waste management firm.

This chapter focuses on PCP wastes for which residual management is the only realistic option.

7.2.3 Overview of Treatment Requirements

Treatment methods must recognise the range of contaminants present and ensure that the treatment process, and the disposal of resulting residues, will not adversely affect human health or the environment.

7.2.4 Categories of Waste and Treatment Methods

Several categories of wastes can be distinguished on the basis of their treatment requirements:

- metal-containing wastes which cannot be recycled will ultimately require disposal in a landfill, perhaps requiring stabilisation or immobilisation;
- organic wastes such as PCP, which are potentially degradable by physical or biological processes;
- organic wastes containing dioxins which probably require a thermo-chemical treatment method.

These waste groups are discussed separately below.

7.2.5 Metal-Contaminated Wastes

Treatment and disposal methodologies for soil and sludges which are predominantly contaminated by metals such as copper, chromium and arsenic are well-established.

Specific considerations are as follows:

- Unfortunately, recycling and reuse of metals in soils and sludges is often impractical, because the chemical nature of the metals is altered and they cannot be readily converted back to the soluble forms required for use. However, the practicality of metals recovery should be considered with reference to specialist companies in this field, before proceeding with landfill disposal.
- Treatment to effect stabilisation or immobilisation of the metals may be required in order to permit safe disposal. Stabilisation should be in accordance with accepted procedures which use chemical additives to ensure the metal contaminants are in insoluble, non-available forms, and are incorporated in a stable matrix. A simple stabilisation procedure is outlined in Waste Management Guide: 02 Treatment and Disposal of Timber Preservative Wastes: Copper, Chromium and Arsenic (Dept of Health, 1986). More advanced immobilisation methods, including the use of cementitious reagents, may be necessary if the wastes are concentrated and there are leachate concerns.
- Check that non-metal contaminants (such as PCP) are not present in the waste at concentrations which would preclude its disposal in an appropriate landfill.

7.2.6 PCP-Contaminated Wastes

PCP-contaminated wastes include PCP/oil mixtures, PCP in soil, and PCP in timber (such as green chain tables). Potential biological and physical/chemical treatments include:

Bacterial Degradation

At moderate concentrations (<500 mg/kg) bacterial degradation is possible, although difficult in practice. Recent work has shown that the more advanced treatment methods utilising slurry reactors can be applicable, but further work is required to confirm the applicability of simple treatment methods (such as aerated piles).

At high concentrations (>500–2000 mg/kg), PCP appears resistant to bacterial degradation, as bacterial action is strongly inhibited. Concentrated wastes are probably best treated by physical or chemical means (in a similar way to dioxin-contaminated wastes), although there is potential to dilute the waste, reducing the PCP concentrations to levels where biological breakdown will occur. Test work would be required to establish whether this is practical, and such an approach would require the availability of a treatment system which is successfully operating at lower concentrations. Note that it would be important to not dilute wastes without having first confirmed that this would make treatment possible, as it could otherwise result in greatly increased quantities of wastes requiring treatment and disposal.

Fungal Degradation

Fungal treatment (e.g. by composting with an appropriate lignin-based material) has been shown in test work to degrade PCP at moderate concentrations (<500 mg/kg); however, the practical extension of this technique to a commercial scale has not been demonstrated consistently. Fungal degradation is not necessarily as complete as bacterial degradation, and stable intermediates (such as odorous anisoles) can form.

Soil Washing and Soil Classification

Soil washing relies on PCP being in a soluble form and able to be extracted from the soil. Initial test work suggests that soil washing may well be a useful technique to substantially reduce soil PCP concentrations at some sites, although further test work is required.

Soil classification seeks to separate the various soil particle-size fractions, with the objective of isolating the bulk of the contamination on a small portion of the total soil mass (usually the fine particle fraction). This technique is most applicable when the soil particle size is relatively coarse (e.g. sand). Test work with pumiceous soils has shown soil classification to be of little value.

7.2.7 Dioxin-Contaminated Wastes

Wastes containing dioxins at concentrations in excess of levels which will permit their safe disposal to an appropriate landfill will require treatment, or secure storage until treatment can be effected.

Dioxins appear resistant to degradation by biological means, and proven treatment methods at this time are restricted to physical/chemical techniques. A review of technologies has recently been undertaken on behalf of TIEC and MFE (CMPS&F, 1995, Envirochem, 1994) and these reports provide information relevant to dioxins.

Specific considerations are as follows:

- High-temperature incineration is a proven, widely established and generally preferred method, as it can achieve very high destruction efficiencies. However, at this time there are no incinerators in New Zealand or Australia licensed for this purpose.
- There are chemical treatment methods which test work has shown can be effective in breaking down dioxins. Of these, Base Catalysed Dechlorination (BCD) is currently in operation in Australia for treatment of recalcitrant PCB wastes, and the process is potentially applicable to wastes containing PCP and dioxins, although this requires confirmation. The application of chemical technology is relatively costly, and for economic reasons is likely to be applied only to smaller volumes of more concentrated wastes.

- Other treatment techniques are being considered in Australia; these include hydrogenation, molten metal and plasma arc systems. The demand for these systems in New Zealand is uncertain.
- Soil washing and extraction of dioxins, and soil classification to separate dioxins from the bulk of the soil, have not yet been tested in New Zealand.
- Organic compounds such as PCP are likely to be satisfactorily treated by a dioxin treatment method, although test work would be required to confirm that an appropriate level of treatment of the mixed waste is achieved. Metals will not destruct, and will remain as residuals following treatment. It should be confirmed that residual metals will be at acceptable levels for final disposal; supplementary treatment such as immobilisation may be required prior to disposal.

7.3 CLASSIFICATION OF LANDFILL SITES IN NEW ZEALAND

Given the general landfill principles and the practice in New Zealand with regard to the siting and operation of existing and likely future landfills (refer Appendix A), three landfill classifications have been defined for the purposes of these guidelines:

- Class 1 represents the formation of a small specially developed and lined cell within a Class 2 site. Any Class 2 landfill can undertake this, although those identified in Appendix C as Class 1 landfills are considered more likely to have the required handling expertise.
- Class 2 is a site that is suitable for co-disposal of wastes containing relatively low concentrations and quantities of potentially hazardous constituents within the framework of operating procedures common to such landfills in New Zealand.
- Class 3 is the standard that an appropriately sited, engineered and operated landfill of older design, receiving municipal refuse only, should currently be able to meet.

A comprehensive set of New Zealand-specific landfill engineering guidelines was produced by the University of Canterbury, in 1992 (CAE, 1992). Whilst few, if any, New Zealand sites would currently conform completely with the recommendations given in the CAE guidelines, they represent an appropriate standard for waste acceptance operation for sites designated Class 1 and 2, and for site engineering for sites designated Class 1.

It is noted that there are many sites in New Zealand currently used for waste disposal that do not conform to the standard of a Class 3 landfill. Such uncontrolled waste disposal sites are not considered suitable for the disposal of wastes contaminated with timber treatment chemicals.

A list of some of the criteria for distinguishing various landfill classes is given in Table 7.2. This should be considered to be indicative of desirable site characteristics for the various classes rather than rigidly specific. In some cases the particular features of a landfill may make certain criteria unnecessary for that particular landfill (e.g. impermeable geological features may obviate the need for engineered lining requirements). For many disposal cases, more detailed consideration in accordance with risk assessment principles should be undertaken before wastes with potentially hazardous constituents are accepted into a particular landfill. Section 7.9 provides information on factors which need to be considered in such an assessment.

Tables listing New Zealand sites which potentially conform to Classifications 1, 2 and 3 are given in Appendix C, although these landfills may not currently conform with every stated requirement of the respective landfill categories. Conformity with the classification criteria should be considered to be indicative only and subject to confirmation.

The three classes allow for a graduation in landfill quality commensurate with graduated levels of waste strength. This allows landfill disposal, within the waste strength and quantity constraints indicated in Tables 7.4 and 7.5, to be undertaken at least for very low-strength wastes in most areas of New Zealand. As the waste strength increases, the controls placed on the landfill in terms of management and engineering become more rigorous. **On no account should waste materials be diluted to allow them to be placed in a lower class of landfill site.**

Table 7.2
Landfill Classes

Class	Landfill Design and Operation Criteria
1.	• Meets Class 2 criteria; and • accepted hazardous wastes to be mixed with mature refuse if appropriate, and disposed in discrete cells with low permeability capping and lining material, • has leachate capture and either recirculation, treatment, or disposal to sewage treatment facility.
2.	• Meets Class 3 criteria; and • has an appropriately designed and operated leachate and groundwater quality surveillance programme which indicates insignificant levels of groundwater contamination and will be regularly monitored for potentially hazardous constituents following acceptance, • site applies cover on a daily basis and low-permeability intermediate and final cover, • site has adequate low-permeability/attenuating lining materials and appropriate subsoil conditions as evaluated by a detailed hydrogeological investigation, • site is further than 3 km from any significant point of water abstraction and use within the same hydrogeological catchment.

Class	Landfill Design and Operation Criteria
3.	• site is securely fenced and has personnel in attendance during all times of operation capable of assessing whether documentation with wastes is adequate. Additionally, personnel must be available who can decide how to evaluate specific wastes and determine the required disposal option, and who are fully instructed in the requirements for safe handling of the particular waste both for themselves and other landfill users. Where wastes are proposed to be accepted, appropriate testing (concentration and leachability of constituents) should be carried out, • site has at least a 4 m depth of well-compacted refuse available above the site base, • site has acceptable control of stormwater, and applies cover at least on a weekly basis, • site is further than 1 km from any significant point of water abstraction and use, • closure to include a low-permeability protective cap, • site to be further than 500 m from residential areas, • site to be located and engineered such that extreme meteorological events will not cause significant mobilisation of wastes by such processes as erosion, wave action, and stormwater runoff, • site to have in place appropriate operational, quality assurance, emergency response, and post-closure management plans.

7.4 CLASSIFICATION AND CHARACTERISATION OF WASTE MATERIALS FOR LANDFILL DISPOSAL

7.4.1 Outline of Leachate Tests

All landfills produce a liquor rich in organic and inorganic species called leachate. Leachate arises both from the penetration of rain and surface water into the landfill and from the physical and biochemical breakdown of wastes. Where wastes containing potentially hazardous constituents are co-disposed with municipal refuse and come in contact with leachate, the constituents of the waste may become solubilised unless the constituents are stabilised or immobilised by an appropriate treatment. Depending on the concentration of these constituents in the waste and the rate of dissolution, the leachate may become sufficiently contaminated that there is potential for the leachate to adversely affect the environment external to the landfill.

Leaching tests are normally laboratory-based procedures which investigate the extent to which contaminants leach from a sample. Virtually all leaching tests fall into two broad categories: dynamic tests, and extraction tests. With dynamic tests the fluid used to leach substances is systematically renewed during the test, whereas with extraction tests it is not.

Some leaching tests are designed to specifically model the leaching processes within a landfill. In the preparation of these guidelines a number of regulatory leaching test protocols were evaluated. It is recommended that the United States Environmental Protection Agency's Toxicity Characteristic Leaching Procedure (TCLP) be adopted as the standard leaching test procedure for New Zealand. The reasons for this choice are discussed in Appendix B.

Flow diagrams documenting the procedure for conducting the TCLP test are provided in Appendix B. Briefly, a waste sample of typically 100 g is ground or broken up to pass a 9.5 mm sieve. The waste is then agitated for 18 hours with leaching fluid chosen from two types, dependent on the alkalinity of the waste. After agitation, the filtered leaching fluid is analysed to determine the concentration of contaminant species of interest.

Interpretation of the significance of the concentration of any contaminant species in the TCLP extract requires careful consideration of both the use to which this information will be put and the limitations of the method.

In the United States the TCLP is used primarily for the assessment **with a high degree of confidence of whether a waste displays the characteristic of toxicity** and must be rated as such under Federal hazardous waste regulations. A waste is considered to possess this characteristic if the concentration of any one of 40 nominated contaminant species in the test extract exceeds specified regulatory levels. A further use of the method in the US is to assess the efficacy of defined treatment technologies in treating specified waste types. In this case, to be eligible for landfill deposition, nominated contaminant concentrations in the TCLP extract from treatment residues must not exceed limits specified in Table CCWE of Part 268.41 of Section 40 of the Code of Federal Regulations. Examples of the application of this are provided in Appendix B.

Interpretation of TCLP extract contaminant concentrations in the New Zealand context is more sharply focused on the way in which such information can be used to indicate the likelihood of adverse effects on the environment resulting from disposal of a waste in a landfill. The significance of the TCLP extract concentration for a particular waste is dependent on three factors:

- the limitation of the TCLP technique in providing information on the **rate** at which leaching occurs; i.e. the test can be interpreted as providing information on the average leaching rate over a period, but the test cannot predict the maximum concentration of a constituent in landfill leachate arising from the deposition of a specific waste.
- the levels of constituent attenuation in the landfill and leachate dilution in receiving water that can reasonably be expected before the constituent impacts on the environment;
- the requirement of the RM Act that discharges should not cause adverse effects at their point of impact; i.e. into water or air, or into or onto land. Depending on the point of discharge, an acceptable waste constituent concentration may vary from a water quality standard protective of sensitive aquatic life (if the

leachate were to enter surface water of designated value in a Regional Resource Management Plan), to levels based on the drinking water standards, or a wastewater treatment plant's ability to remove or assimilate the substance.

Of the three factors, the third is the only one which is defined (or has the potential to be defined) in regulation. Only estimates can be made of the effect that the other two factors have on the concentration of a waste substance at the point of impact. Such estimates require knowledge of typical landfill attenuation and receiving water dilution factors, and also some information about the rate at which leaching will occur over the waste's lifetime in a landfill.

In Section 7.5 appropriate receiving water acceptance criteria, and attenuation and dilution factors have been selected to derive appropriate TCLP limiting concentrations for various timber treatment chemicals.

7.4.2 TCLP Testing of PCP-Contaminated Wastes

(a) Bulk Water Sampling

The waste sampling protocol of the draft European Community Directive on Landfilling Wastes (91/C190/01) provides a good sampling methodology, and it is recommended that this protocol be adopted as the strategy for selecting from bulk waste for disposal. This sampling protocol is as follows:

(i) Definitions:

- *Homogeneous waste:* All types of waste which at the time of sampling are liquid or can be pumped and whose characteristics are the same throughout the whole mass, as well as those wastes whose homogeneity can be visually established.
- *Heterogeneous waste:* All other wastes.

(ii) Representative sample:

A sample is to be considered as representative if the small amount of material weighed out for the analyses has the same average composition as the large mass from which it is derived (*Reference:* General guidelines on sampling technology, ISO 5667-2-1988).

Number of samples and amounts to be taken:

- For wastes not delivered in containers
  - for homogeneous wastes: one sample of 1000 g per delivery;
  - for heterogeneous wastes: one representative sample of 1000 g per five tonnes of waste or part thereof. For large shipments (>20 tonnes) sufficient samples shall be taken to characterise the waste.

- For wastes delivered in containers

  The number of samples and amounts are listed in Table 7.3. These values are valid for containers with the same content, and for which the waste is considered homogeneous at the time of sampling. If the containers are emptied into a collecting tank, the cumulative sample can be taken from the tank.

Table 7.3
Sampling Requirements for Wastes Delivered in Containers

Weight per container	Weight and number of containers to be sampled for a laboratory sample
< 5 kg	Sufficient for a cumulative sample of at least 1 kg taken from at least $x^{(1)}$ containers
> 5 kg	Sufficient for a cumulative sample of between 1-2.5 kg taken from at least $x^{(1)}$ containers

Note: (1) Where $x = \sqrt{(n + 1)}$, where n equals the total number of containers.
Source: Draft EC Directive on Landfilling Wastes (91/C190/01))

The above requirements relate to the sampling of wastes on delivery. Alternatively, the detailed characterisation of wastes in-situ (e.g. during site assessment) may avoid the need for rigorous sampling at the point of disposal.

(b) Leaching Tests for Monolithic Wastes

The USEPA TCLP protocol requires that all wastes for testing pass a 9.5 mm sieve, and makes no provision for testing the leachability of waste which is monolithic in nature (i.e. waste which essentially comprises a single solid item as occurs when waste is encapsulated in concrete or is actually a functional object such as a telegraph pole).

Although monolithic waste testing was a component part of the TCLP test predecessor – the so-called EP Toxicity test – this option was not included within the TCLP test for reasons specified in Appendix B.

It should be borne in mind that there are practical constraints on the testing of monolithic waste materials in that the container in which leaching takes place would not typically have a neck of diameter greater than 100 mm, or a total volume of greater than 4 litres. In practice, all materials have to be sub-sampled and divided on an agreed basis if consistent and comparative extraction test results are to be achieved.

It is considered that the TCLP requirement that material should pass a 9.5 mm sieve is probably reasonable for achieving uniform and consistent results, and there is not a good basis for selecting an alternative sample size (such as 19 mm diameter).

It is recommended that monolithic waste material be subdivided for TCLP testing to enable it to conform with the normal requirements for TCLP testing.

In effect, monolithic wastes need to be reduced such that two dimensions are less than 9.5 mm, the sample fits into the extraction flask and all surfaces are equally accessible to the extractant.

It is noted that the disposal of wastes in a monolithic form to landfill (e.g. large pieces of treated timber or waste fixed in a cement matrix) should generally result in reduced leaching of contaminants from the waste, particularly compared to untreated contaminated soil or sawdust. The benefits of disposal of wastes in a monolithic form is acknowledged and such practices are to be encouraged where practical; however, such benefits are difficult to quantify.[3]

In addition, damage to the monolith during disposal and compaction of the landfill may allow leachate to contact a much larger waste surface area than initially anticipated, increasing leaching of contaminants. The TCLP test procedure nominated above may overestimate the leaching of contaminants from a monolithic waste. However, it does allow some consideration of the benefit of monolithic treatment given the waste must only pass a 9.5 mm sieve, rather than being ground to give a particle size, say, comparable with soil.

(c) Application of the TCLP Test to PCP Contaminated Wastes

The US EPA TCLP leachate test indicates that the extraction procedure should be undertaken using a buffered solution of acetic acid at pH 5 for low-alkalinity wastes. Such an extraction regime is expected to reasonably represent the worst-case conditions for mobility of contaminants, particularly some heavy metals, within a landfill. It is expected that conditions within a landfill will range between moderately acid (pH 5-6) to mildly alkaline over the life of the landfill.

Whereas acid conditions are the conditions under which heavy metals are most mobile, this is not necessarily the case for PCP. Under acid conditions PCP will tend to be present as pentachlorophenol, whereas under alkaline conditions PCP will tend to be present as the pentachlorophenate anion, which is more soluble in water. This suggests that PCP will be more mobile in a landfill under alkaline conditions rather than the acid conditions simulated by the TCLP test.

In practice, there are many factors that affect the mobility of PCP in a landfill, such as the presence of oils, and the adsorptive nature of the wastes within the landfill. Research by the US EPA has shown that leaching of organic materials is relatively insensitive to variations in pH, at least over the range of pH encountered in most landfills.

On this basis, it is considered that the TCLP tests should be used without modification to characterise the potential mobility of PCP within a landfill.

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3 Unusual waste fixation or encapsulation practices may need to be considered on a case by case basis.

7.4.3 The Waste Manifest System

The Hazardous Waste Management Handbook (MFE, 1994) offers definitions of hazardous properties. Under the handbook recommendations, wastes exhibiting hazardous characteristics should be analysed, and if qualifying as hazardous, a waste manifest should be prepared prior to disposal. Co-disposal should only be considered if appropriate and all other reasonable minimisation/treatment alternatives have been exhausted. A further definition, that of "problem wastes", includes chlorinated phenols.

The handbook discusses two manifest systems: a four-form system, and a six-form system. The six-form system by way of example includes copies for a regulatory agency (two copies) as well as the producer (two copies), the transporter (one copy), and the waste disposer (one copy). The handbook concludes: "due to the relatively high costs of monitoring a six form system, this may only be appropriate where other management methods are unsuccessful or environmental risks are especially high".

Current practice in New Zealand in the use of waste manifests for landfill acceptance includes Northern Disposal Systems, Wellington City Council, Hamilton City Council, and Hutt City Council. Additionally, the CAE report provides an example movement manifest as Appendix E to Part 3, page 267 (CAE, 1992).

The Transport Act 1962, and the Traffic Regulations 1976 provide regulatory controls covering the transport of hazardous substances on land. These substances are defined as those listed in Table 10 of NZS 5433:1988, with the proviso that minimum quantities apply for some classes of hazardous substance. Wastes containing any of these hazardous substances and the containers in which the substances or wastes have been held are required to be treated as hazardous substances under these regulations. The definition of hazardous wastes given under these regulations may be different from that given in the New Zealand Chemical Industries Council Waste Management Guidelines.

The Regulations specify requirements for documentation to accompany wastes, placarding and labelling of vehicles, segregation from other substances, and driver training.

A review of these separate documents shows some similarities and some differences. Aspects common to most, and which are recommended to be included in a manifest system for the timber treatment wastes covered in these guidelines are:

- multiple copies, for generator, transporter, and disposer;
- names, addresses, and contact phone/fax numbers for all three parties including after-hours numbers;
- a two-stage acceptance procedure, with the generator making an application along with documentation, followed by the acceptance, rejection, or request for further information from the disposer;
- a unique numerical identifier for each case;

- signed declarations by the generator, transporter, and disposal authority, verifying statements made, and knowledge of legal responsibilities;
- use of the New Zealand Chemical Industries Council modified OECD waste identification code;
- waste quantity;
- quantity of hazardous constituents;
- concentration of hazardous constituents;
- TCLP results;
- packaging method, packing group, UN number, and Hazchem classification;
- hazardous goods licence endorsement valid if waste classifies as hazardous under NZS 543:1988;
- location of waste;
- check sample taken for analysis, and results;
- verification of weight of load;
- proposed date of transport;
- generator's safe handling instructions;
- details of any pretreatment;
- physical state of waste;
- landfill class;
- check of procedure;
- disposal method;
- location of waste in landfill.

An example of an appropriate form is given in Figure 7.2. Note that it is the generator's responsibility to ensure that sampling and analysis conform to the guidelines or alternative acceptable methodologies.

Many districts and/or regions are likely to introduce manifest systems in the near future which may be used in preference to a manifest specific to timber treatment wastes, provided they contain the required information.

The manifest is intended to act as an application form from the generator to the disposer, an acceptance/rejection form from the disposer to the generator, a declaration form for the transporter, and a record of disposal methodology for the disposer.

The procedure is as follows:

1. Generator fills out Part 1 and sends all copies to proposed disposer.
2. Proposed disposer reviews information and asks generator for more data, or, accepts/rejects waste as indicated at the start of Part 2, returns forms to generator retaining one copy.
3. If waste is accepted, generator organises transport, and provides disposer with copy showing transport arrangements.
4. Manifest is carried by transporter who retains one copy.
5. Disposer completes Part 2, retains one copy and sends one copy back to generator.

Figure 7.2
Example of Appropriate Manifest Form for
Timber Treatment Waste Transport and Disposal

MANIFEST FOR THE DISPOSAL OF TIMBER TREATMENT WASTES

Manifest #:

PART 1: GENERATOR TO FILL OUT

1. Location Of Waste: .....
2. Proposed Disposal Location: .....
3. Hazardous Content(s) of Waste (Proper Shipping Name): ..... Physical State: .....
4. Pretreatment (Y/N?): ..... Method: .....
5. Samples Taken for Analysis (Y/N?): ..... Location: ..... Number: ..... Results Attached (Y/N) .....
6. TCLP Results: ..... Date Of Proposed Transport: ..... Vehicle Reg No(s): .....
7.

Waste I.D. Table (Ref NZCIC C.O.P.)	Table DA	Table DB	Table YA	Table YB	Table H	8. Quantity Waste (kg)
						9. Quantity Constituent(s) (g)
Generator's Storers Safety And Handling Instructions.						10A. Conc. Constituent(s) (mg/kg)
						10B. UN No.
						10C. Hazchem
Packaging Method:						10D. Packing Group

PART 2: DISPOSER TO FILL OUT

11. Material Accepted for Disposal (Y/N?): ..... Weight/Quantity Checked (Y/N?): .....
12. Check Sample Taken (Y/N): ..... Results: ..... Results Attached (Y/N) .....
13. Landfill Name: ..... Landfill Class: ..... Disposal Method: .....
14. Location In Landfill: Elevation: ..... m, Co-Ordinates: .....

	Name	Address	Work Phone	Home Phone	Fax	Signed	Date
Generator[1]							
Transporter[2]							
Disposer[3]							

1 - By signing the Generator declares that they are making disposal as per Section 7.1 of the Health and Environmental Guidelines for Selected Timber Treatment Chemicals and the waste is accurately described and is in a proper condition for transport in accordance with the New Zealand Transport Act.
2 - By signing the Transporter declares that the waste consignment described above has been received and has been checked as per the procedure in Figure 7.1, of the Health and Environmental Guidelines for Selected Timber Treatment Chemicals and that it is being transported in accordance with NZS 5433:1988.
3 - By signing the Disposer declares that the waste consignments described above has been received, that the procedure in Figure 7.1 of the Health and Environmental Guidelines for Selected Timber Treatment Chemicals has been followed for appropriate disposal.

COPY 1: - GENERATOR COPY 2: - TRANSPORTER COPY 3: - DISPOSAL FACILITY COPY 4: - GENERATOR

7.5 DEVELOPMENT OF ACCEPTANCE CRITERIA FOR DISPOSAL OF WASTES BY LANDFILLING

7.5.1 Current New Zealand Acceptance Criteria

As at mid-1994, a range of acceptance criteria for hazardous wastes were being utilised by New Zealand landfill operators. This variable practice could be broadly categorised as follows:

- "Hazardous" wastes are not accepted, with the USEPA Subtitle C definition used to define "hazardous" wastes; analyses are required for wastes which may approach the classification limit.
- Many wastes (including some designated as hazardous by Subtitle C definition) are accepted subject to specific consideration by the landfill operator; manifests and analyses are required for most wastes considered to be hazardous.
- "Hazardous" wastes are not accepted, with "hazardous" usually not defined, and few if any controls in place to determine and regulate compliance.

The consent requirements of the Resource Management Act are, however, creating improvements in management practices in many of these sites (including acceptance criteria), and resulting in the closure of many of the poorer ones.

The CAE report "Our Waste, Our Responsibility" (CAE, 1992) on the subject of co-disposal of hazardous wastes with municipal wastes, states: "co-disposal is only acceptable when the site is contained, and there is a leachate collection and treatment system in place". The CAE document offers the OECD categorisation system for guidance rather than strict adherence. The OECD system rates chlorophenols (classification number Y39) as "waste prohibited from landfills", if the waste is in an untreated form. However, this implies that even trace quantities of chlorophenols (such as would be acceptable in a residential situation) are prohibited in an untreated form, and this is considered to be an overly conservative requirement.

7.5.2 Scheduled Waste Management in Australia

The draft ANZECC National Model Regulation for the Management of Scheduled Waste (1992) was a key reference that assisted the development of landfill acceptance criteria for timber treatment wastes in New Zealand. The useful ideas brought forward from the draft Model Regulation were:

- waste constituent concentrations should not exceed specified threshold values which are based on environmental and health risk considerations;
- wastes should be solid or spadeable in form, and should pass a leachate test (which is also based on environmental and health risk considerations).

The threshold values referred to the concentration or quantity of a specific scheduled constituent of the waste (such as PCP); "quantity" meant the total quantity of hazardous constituent to be disposed of in a specific parcel of waste; and "concentration" meant the concentration of hazardous constituent in a specific parcel of waste.

7.5.3 Key Mechanisms

In this section acceptance criteria for landfilling wastes containing potentially hazardous constituents are developed. These criteria have been developed for landfill Classes 1 to 3 by considering the following alternative mechanisms by which waste constituents may be released to the environment:

- failure of the landfill capping system with release of solid waste constituents to the environment in the vicinity of the landfill; and
- leachate seepage or runoff, with transport of soluble waste constituents to surface or sub-surface receiving water.

The first release mechanism has been used to derive acceptance criteria in terms of total contaminant concentration of the incoming timber treatment waste, while the second release mechanism has been used to derive acceptance criteria in terms of the elutriable fraction of the incoming waste.

The acceptance criteria are based on assumptions about complex physico-chemical processes which are difficult to quantify accurately. Professional judgement has been applied consistent with prudent landfill management.

(a) Release of Solid Waste Constituents

Important factors which influence the significance of solid waste releases, for example by failure of containment in the very long term (e.g. greater than 50–100 years), are:

- The initial concentration of hazardous constituents of the waste and their concentration in the landfill overall (i.e. in the bulk landfill material as would be relevant if there were to be a large-scale release in the very long term); and
- The protection afforded by capping, or burying materials at depth, noting that:
  - material at depth is much less likely to be exposed and released than material at the surface; and
  - the greater the depth of cover, the greater the effective dilution if release should ever occur.

For the purposes of this document, these two dilution or attenuation factors have been termed the "solid waste mix ratio", and the "capping control" factors respectively.

The acceptance criteria for disposal of wastes to landfill are based on the assumption that the timber treatment wastes will comprise only a very small proportion of the wastes accepted at the landfill, and that the bulk of the wastes will comprise non-hazardous wastes (e.g. municipal refuse).

Specifically it has been assumed that the contaminated wastes comprise less than 1% by weight of the landfill waste on an ongoing pro rata basis within the zone of deposition of timber treatment wastes within the landfill (i.e. excluding the landfill base and cover materials). In practice, acceptance criteria for disposal will need to be set on the basis of anticipated and permitted total quantities of a particular waste type to be disposed of in the landfill, rather than attempting to set criteria on a load-by-load basis.

The assumption that timber treatment wastes will comprise only a very small proportion of the total waste volume reflects the situation which is expected to apply in New Zealand landfills.

In determining waste treatment and disposal alternatives, the following principles should apply:

- wastes should not be intentionally diluted to effect disposal;
- if practicable, concentrated wastes should be treated to reduce the concentration of hazardous constituents (e.g. soil washing);
- if the wastes are of sufficiently low concentration to be acceptable for disposal in a landfill, the wastes should be isolated and buried at depth in a defined and recorded location within the landfill, and not intentionally mixed with other waste materials.

Waste materials such as contaminated soil are generally non-homogeneous and the concentration of constituents can vary markedly through material of similar origin. With such materials it is the average concentration of constituents in individual containers (such as drums or skips) which should apply for the purposes of the guidelines below, rather than the maximum concentration of small sub-samples taken from the containers.

However, in such circumstances the significance of the maximum concentration should be assessed in relation to the average concentration. Wastes to be disposed of to Class 2 or 3 landfills should not report a maximum contaminant concentration in excess of the limit for disposal to a Class 1 landfill.

(b) Release of Soluble Waste Constituents

Soluble constituents of timber treatment wastes may also be released to the environment via leachate seepage and run-off, with subsequent impact on groundwater or surface waters down gradient of the landfill.

Important factors controlling the significance of such releases are:

- the dilution of soluble waste constituents within the leachate of the landfill; and
- further attenuation and dilution of the leachate constituents upon transportation and discharge to a surface or subsurface receiving water.

These factors have been termed the "leachate mix ratio", and the "receiving water dilution" factors respectively.

The solid and soluble waste release mechanisms described above represent separate but not independent processes by which adverse effects on the environment can occur, and in practice there will be some interaction between the two mechanisms.

For example, failure of the landfill capping systems can result in a release of waste to the surface environment, but can also result in increased rainfall infiltration with increased generation of leachate and possibly greater impact on the receiving water environment.

In fact the movement and concentration of waste constituents will vary with time and location, and clearly the assignment of dilution factors represents a simplification of the processes which determine the ultimate concentration of waste constituents in the environment. For example, there will be a significant variation in the concentration of soluble constituents as the leachate migrates through the wastes within the landfill, through the landfill base (including liner if present), within the aquifer, and ultimately within the receiving waters.

Biodegradation of some organic constituents can also be expected to occur and to reduce the concentration of substances (such as PCP); however, because biodegradation cannot be well-quantified and will depend on local conditions no allowance has been made for reductions in concentrations arising from biodegradation.

In most cases the assumptions and the derived factors are considered to be conservative, and a detailed risk assessment of a particular landfill and wastes may lead to site-specific criterion different from those derived below.

7.5.4 Derivation of Waste Acceptance Criteria

Waste acceptance criteria have been derived for each landfill class firstly by assuming that certain dilution and attenuation factors apply, and secondly with reference to the relevant acceptance criteria for the receiving environment (soil and waters) brought forward from Chapters 5 and 6.

The receiving environment acceptance criteria (subject to any revision of Chapters 5 and 6) are as follows:

Soil Agricultural	C_E(PCP)	=	0.7 mg/kg
Soil Human Health (Residential):	C_H(PCP)	=	1.4-7.0 mg/kg
Aquatic Ecosystems:	C_A(PCP)	=	0.5 µg/L
Drinking Water:	C_D(PCP)	=	10 µg/L

In consideration of environmental criteria developed in other countries (refer Chapter 6), a value of 0.5 µg/L PCP is adopted as the interim surface water criteria for the protection of modified aquatic ecosystems, pending Ministry for the Environment guidelines.

As previously outlined, the soil guideline values (Chapters 5 and 6) address the requirement to protect human health from contaminant residues in edible plants and livestock. Although soil guideline values protective of the terrestrial ecosystem have not been developed as part of these guidelines, reference can be made to the published information and international criteria.

The No Observed Adverse Effect Levels for PCP reported in the literature for plants and invertebrates are soil concentrations of 1 mg/kg and approximately 3 mg/kg respectively (Sheppard et al., 1992). The Canadian authorities have developed a methodology for the development of soil guideline values protective of the local ecosystem. Based on this protocol a soil guideline value for PCP, protective of the ecosystem, would be in the order of 0.2 mg/kg. The Dutch soil guidelines for the protection of the environment are as follows for PCP (Ministry of Housing, Spatial Planning and the Environment, 1994):

- Intervention Value: 5 mg/kg
  (corresponds to full protection of 50% of species)
- Maximum Tolerable Risk Value: 0.2 mg/kg[4]
  (i.e. full protection of 95% of species)
- Target Value: 0.002 mg/kg
  (corresponds to negligible risk level, which is set at 1% of the limit value).

In this context, the soil guideline value for PCP for agricultural use of 0.7 mg/kg adopted in Chapter 5 can be expected to be generally protective also of the terrestrial environment.

[4] Value inferred from Target Value

7.5.5 Release of Solid Wastes

(a) Solid Waste Mix Factor

As noted in Section 7.5.3, wastes containing a particular hazardous constituent form a very minor proportion of the total landfill mass (e.g. less that 1%). On this basis, a solid waste mix factor of 100:1 has been adopted for Class 1 landfills. It is assumed that in the event of a breakdown of the landfill waste placed at depth within the landfill would be effectively mixed with other refuse during the breakdown process and a dilution of at least 100:1 would be achieved upon release to the surface.

Ratios of 50:1 and 10:1 have been adopted for Class 2 and 3 landfills, respectively, reflecting the assumption that wastes may not be as well-covered and higher concentrations of waste may exist closer to the surface of the landfill, and lesser dilution may occur upon release.

Site-specific factors will depend on the landfill configuration; in-ground landfills are less likely to suffer breakdown and release than an above-ground landfill.

(b) Capping Control Factor

It is assumed that waste containing hazardous constituents will be covered by waste comprising either general municipal waste or capping material (not containing the constituents), and that the depth of this cover varies with the class of landfill. A capping control factor of 10:1 has been adopted for Class 1 landfills which include a relatively thick capping (for example, greater than 1 metre) and a relatively deep placement (for example, greater than 1 metre below the base of the landfill cap). The ratio reflects the protection afforded by the capping (i.e. the reduced risk of waste being released at the surface) and the additional dilution that will be effectively achieved through mixing with the inert waste or capping material and in the environment external to the landfill.

Capping control factors of 6:1 and 4:1 have been adopted for Class 2 and 3 landfills reflecting the varying thicknesses of landfill caps and depths of burial likely to be used in the various classes.

It is considered very unlikely that broad-scale breakdown of a landfill capping system would occur, and the factors assumed are therefore conservative, although rather arbitrary. In the very long term some degradation of organic waste constituents is also likely but no account is taken of this.

The long-term integrity of the landfill, and the success of the post-closure management plan, are key aspects in determining the degree of conservatism and the corresponding factors that should apply. In practice, if a long-term management plan can be applied and is assured, a lesser degree of conservatism and higher contaminant levels may be acceptable. It is anticipated that post-closure management plans addressing long-term monitoring and aftercare would be mandatory for Class 1, 2 and 3 landfills.

The adequacy of the assumptions relating to contaminant release may be reviewed in the context of a specific landfill and waste disposal scenario in order to establish site-specific waste acceptance criteria.

(c) Application of Environmental and Agricultural Criteria

Waste acceptance criteria derived for each of the three classes of landfills are shown in Table 7.4. These criteria have been developed using the solid waste mix ratio and the capping control factor (to estimate the attenuation and dilution of the waste that will occur), and from this the concentration of constituents in the waste which must not be exceeded if environmental and agricultural criteria are not to be exceeded.

The agricultural use criteria are controlling in determining the waste acceptance criteria, as they are generally the lower and represent the most stringent criteria for protection. Note that the agricultural use criteria are usually associated with broader-scale applications (e.g. cattle grazing), and would be relevant to general release of landfill contents over a broad area (not just a localised house-lot-size area). As such, the mix ratio and capping control factors should reflect broad-scale release rather than localised breakdown and lesser dilution upon release.

The protection of human health can be considered to apply for future use of the landfill area itself. Generally, use of former landfills is restricted to recreation or open space and more localised use such as residential is less likely. Recreation/open space usage reflects broad-scale use of the areas, and as such, averaging of concentrations over relatively large areas would be applicable (similar to that in setting environmental criteria, above). The agricultural land use guideline values developed in Chapter 5 are protective of human health associated with agricultural use, and are expected to also be protective of human health for other uses.

Table 7.4
Nominal Acceptance Criteria for Total Waste Constituent Concentrations

Landfill Class	Assumed Solid Waste Mix Factor	Assumed Capping Control Factor[2]	Estimated Resultant Threshold (mg PCP/kg waste)	Acceptance Criteria for PCP (mg/kg)
1	100	10	$1000C_E$	700
2	50	6	$300C_E$	210
3	10	4	$40C_E$	28

Notes: (1) C_E = Soil agricultural/environmental criteria – (0.7 mg/kg for PCP)
(2) Includes consideration of both a constructed clay cap and a layer of uncontaminated refuse beneath the cap.

7.5.6 Release of Soluble Waste Constituents

(a) Leachate Mix Factor

Leachate mix factors of 100:1, 50:1 and 10:1 have been adopted for landfill classes 1-3 respectively. These are based on the same concept as the solid mix ratios and have been adopted for similar reasons as those previously outlined, i.e. the factors account for mixing and dilution within the landfill, with the factors reflecting poorer management practices at lower-class landfills.

In practice the deposition of wastes in cells of low permeability will result in a reduced volume of leachate from the contaminated waste and greater effective dilution of leachate from waste materials. In the Class 3 case, the dilution of leachate within the landfill is assumed to be less, and allowance is made for the possibility of localised channelling of leachate through waste zones.

It is considered that the leachate mix factors adopted are conservative.

(b) Receiving Water Dilution Factor

For Class 1 and 2 landfills a groundwater and surface water dilution factor of 50:1 has been applied. In practice the siting of such landfills in an area of low permeability and/or the provision of lining systems and leachate collection will reduce the rate of leachate seepage from the base of the landfill, thereby increasing the potential for dilution within the aquifer. Consequently the 50:1 groundwater dilution factor may be a conservative assumption depending on the nature of the landfill and distance to the receptor point. The criterion of a 3 km (Classes 1 and 2) and a 1 km (Class 3) separation distance should afford considerable protection.

For Class 3 landfills a lower dilution factor in the receiving waters has been assumed due to the reduced distance between the landfill and the receptor point.

In some cases, sufficient monitoring information may exist regarding groundwater flow and quality to estimate a site-specific receiving water dilution factor.

The proposed approach assumes that leachate will ultimately discharge to a receiving water. However, in some landfills, leachate is collected and recirculated to the landfill, or discharged to a sewerage system. In these cases, the impact on the sewerage system may be a limiting factor. Consideration also needs to be given to the significance of leachate generation after landfill closure or beyond the life of the sewerage system.

(b) Application of Receiving Water Criteria

The above factors are applied to the surface water and groundwater acceptance criteria. The surface water criterion (C_A) applies to the protection of aquatic ecosystems, while the groundwater criterion (C_D) relates to the protection of drinking and stock water.

Nominal elutriation acceptance criteria derived for each of the three categories of landfills based on surface water (aquatic ecosystems) and groundwater (potable use) criteria are shown in Table 7.5. These may be conservative for landfills where leachate clearly will not discharge to surface waters (in which case the drinking water criteria may be applicable), and more relaxed acceptance criteria may be appropriate on a site-specific basis.

It is noted that the above receiving water dilution factors have been developed on the basis of discharge of leachate to surface water directly or discharge of leachate to groundwater. Whereas in the case of Class 3 landfills direct discharge of leachate to surface water is possible, in the case of Class 1 and 2 landfills it is assumed that any discharge to surface water would occur via groundwater. In such cases where there may be additional dilution of any leachate contaminated groundwater as it discharges to the surface water, a greater dilution factor may be appropriate.

The drinking water threshold concentrations have been applied to groundwater whereas the aquatic ecosystem threshold concentration has been applied to surface water. Consequently, where leachate does not discharge directly to surface water (i.e. Class 1 and 2 landfills) an additional dilution factor, assumed to be say, 5, may be applied. Where it can be shown that leachate from a Class 3 landfill does not discharge directly to surface water, a similar additional dilution factor may be applied, although this requires site-specific evaluation of the landfill. In such cases there may be additional dilution of any leachate-contaminated groundwater as it discharges to the surface water, and it may be appropriate to apply greater dilution factors.

Table 7.5
Nominal Acceptance Criteria for the Elutriable Fraction of Soluble Waste Constituents
(Criteria for PCP are shown in brackets)

Landfill Class	Assumed Leachate Mix Ratio[4]	Assumed Receiving Water Dilution Factor[5]	Estimated Resultant Threshold[3] (μg PCP/L elutrient)	
			Aquatic Ecosystem Protection[1]	Drinking Water Protection[2]
1	100	50	$5000C_A$[9] (2500)	$5000C_D$[9] (50000)
2	50	50	$2500C_A$[9] (1250)	$2500C_D$[9] (25000)
3	10	10	$100C_A$ (50)	$100C_D$ (1000)

Notes: (1) C_A = receiving water aquatic ecosystem protection (= 0.5 μg/L for PCP)

(2) C_D = receiving water drinking water protection (= 10 μg/L for PCP)

(3) Concentration of constituent in elutrient from TCLP test.

(4) Includes attenuation within the landfill.

(5) Includes attenuation by the landfill base lining, and attenuation prior to discharging to the receiving water (in the subsurface or surface environment).

(6) Proposed criteria assume aquatic ecosystem protection is relevant; if discharge to a surface receiving water does not occur then some other criteria, e.g. drinking water, may be more relevant.

(7) This is an example of the most conservative assumption. Assumptions used as part of a specific assessment must be developed on a site-specific basis.

(8) Lower values may be required if there is an aquifer in the vicinity of the landfill which requires protection; notwithstanding the requirement that there be no drinking water abstraction within 3 km.

(9) Leachate protection considerations could limit the acceptable value; refer to Section 7.5.6 (d).

(d) Leachate Quality Protection

Consideration should be given as to whether it is important to protect the quality of leachate generated from the landfill where wastes are proposed to be accepted. In most cases the receiving environment will be the determining consideration; however, if leachate is irrigated over the landfill or is discharged to a sewerage system, then there may be specific requirements to protect its quality.

An estimate of the landfill leachate quality that might result from the deposition of the wastes can be obtained from Table 7.5, by multiplying the leachate concentration (TCLP value) by the assumed leachate mix ratio (which reflects the assumed dilution of the waste leachate).

Usually landfill leachate will not conform with drinking water guidelines; typically contaminant concentrations 10 to 100 times the drinking water guidelines occur.[5] It is suggested that if leachate quality is to be protected, (i.e. the placement of waste should not affect leachate quality to a greater extent than normally occurs), then the TCLP concentration should not exceed 1000 times the drinking water quality value. This would be a more restrictive TCLP test criterion than that listed for drinking water protection in the receiving environment, and would be similar to the aquatic ecosystem protection value.

[5] CAE (1992) guidelines; Table 4.1 p. 318 as per NECAL Report Services 88/5.

7.5.7 Discharge to Sewer

In the case of landfills where leachate is discharged to a sewer, consideration should be given to: the anticipated dilution in sewage (with allowance for the possible presence of the substance in the sewage), reduction in levels by attenuation or removal in the treatment facility, and appropriate criteria for discharge of effluent from the sewage treatment plant and for the disposal of sludge.

7.5.8 Action in the Event of Excessive Discharge

The criteria outlined above have the specific objective of avoiding the future discharge of contaminants from landfills at levels adverse to the receiving environment. However, it is possible that unacceptable discharges may occur, especially from past uncontrolled waste disposal.

In the event of an unacceptable discharge (e.g. leachate problems from past uncontrolled waste disposal) the following control options could be considered:

- intercept leachate and return to the landfill, or treat and dispose;
- intercept groundwater and return, or treat and dispose;
- inject grout or install cut-off walls to contain leachate and/or groundwater;
- inject micro-organism cultures and nutrients, or modify the landfill and aquifer physical and chemical parameters to enable the existing biological and physiochemical processes to function more effectively;
- fix in situ;
- locate the problem waste materials within the landfill from the waste manifest; excavate, treat and dispose.

7.5.9 Acceptance of Wastes at Particular Landfills

In addition to the approach outlined above for establishing acceptance criteria for timber treatment wastes at a particular landfill, a quantity threshold has also been adopted as an additional precautionary measure.

The quantitative threshold for PCP is 250 grams. This value has been nominated in the ANZECC scheduled waste guidelines (ANZECC, 1992), and corresponds to approximately 10 tonnes (three truckloads) of waste material at the acceptance limit for a Class 3 landfill (28 mg/kg PCP).

It is recommended that landfill operators, when requested to accept waste containing potentially hazardous constituents, proceed on the following basis:

- where the quantity of waste to be disposed of is less than the threshold quantity, and the concentration and leachability of the waste conforms with the requirements for Class 3 landfills outlined in Tables 7.4 and 7.5, then the waste

load can be accepted for disposal without requiring a specific study or approval by the Regional Council;

- where the quantity of waste to be disposed of is greater than the threshold quantity, or the concentration and leachability of the waste does not conform with the requirements for the relevant class of landfill, acceptance criteria for the waste should be established for the landfill using a risk-based assessment approach, i.e. this should establish the maximum total concentration and leachability of the waste constituent (and that the waste will not pose an explosive, corrosive or ignitable hazard).

These acceptance criteria should be approved by the regional council prior to acceptance of waste.

7.5.10 Examples of the Application of the Criteria for PCP

Two hypothetical examples are provided to illustrate the development of landfill-specific criteria for:

Landfill A — landfill generally conforms with the Class 3 criteria, but the landfill is in a quarry with surrounding low-permeability materials, and there is no useable groundwater in the area of the landfill.

Landfill B — also conforms with Class 3 criteria; but the landfill has been built up on the ground not far from a creek where aquatic ecosystem protection is required. The quantity of wastes to be disposed of is very small compared with the quantity of normal refuse.

Differing solid waste mix ratios have been adopted for each landfill reflecting differing waste handling and placement practices. The selection of relevant mixing and dilution factors is based on professional judgement and is outlined as follows:

- Solid Waste Mix Ratio

  Waste is deposited in Landfill A in accordance with general practice for a Class 3 landfill and the total quantity of waste deposits is approaching 1% of the total landfill contents. On this basis the generic solid waste mix ratio of 10 has been adopted.

  In contrast Landfill B is a much larger operation and the quantity of PCP wastes to be deposited is very small in relation to the quantity of normal refuse received. Therefore a higher solid waste mix ratio (20) has been adopted.

- Capping Control Factor

  Generic value of 4 adopted for each landfill.

- Leachate Mix Ratio

  Leachate mix ratios of 10 and 20 have been adopted for Landfill A and Landfill B, respectively, reflecting similar considerations to those on which the selection of the solid waste mix ratios are based.

- Receiving Water Dilution Factor

  Landfill B is sited relatively near to a surface water body, and therefore the generic receiving water dilution factor (10), which reflects a relatively sensitive scenario, has been adopted.

  Landfill A is remote from sensitive surface water bodies, the landfill is surrounded by low permeability formations and there is no useable groundwater in the area. Consequently, leaching of contaminants from the landfill is not a limiting consideration and therefore no receiving water dilution factor has been nominated.

The various factors which are assumed to apply are shown in Table 7.6 and may be compared with those in Tables 7.4 and 7.5.

Table 7.6
Case Example – Disposal of PCP Contaminated Soil
at Two Different Class 3 Landfills

	Landfill A	**Landfill B**
Solid Waste Mix Ratio	10	20
Capping Control Factor	4	4
Allowable Total Waste PCP Concentration in Soil		
– environmental (0.7 mg/kg)	28 mg/kg	56 mg/kg
– human health (7.0 mg/kg)	Not limiting (280 mg/kg)	Not limiting (560 mg/kg)
Leachate Mix Ratio	10	20
Receiving Water Dilution Factor	Not applicable	10
Allowable Elutrient Concentration		
– aquatic ecosystem (0.5 μ/L)	Not applicable	0.1 mg/L
– drinking water (10 μ/L)	Not applicable	0.2 mg/L

It can be seen from this case example that quite different acceptance criteria can apply; with Landfill A the soil concentration criterion of 28 mg/kg can be expected to be limiting, whereas with Landfill B the TCLP elutrient concentration (0.1 mg/L) could limit the permissible soil PCP concentration.

This case example illustrates how factors other than human health can be significant in determining acceptable waste contamination landfill acceptance criteria.

7.6 APPLICATION OF THE METHODOLOGY TO OTHER WASTE CONSTITUENTS

In principle, the methodology of this chapter can be applied to derive landfill acceptance criteria for waste constituents other than PCP.

Important factors to be considered in assigning factors and deriving criteria for a particular landfill include the following:

- Attenuation in the subsurface environment will vary with time, distance, and geology; attenuation could be very high, for example, where the landfill is in an area of low-permeability materials.
- If discharge of leachate can occur only to groundwater (and not directly to surface receiving waters), the use of criteria based on the protection of potable use are likely to be more relevant than the protection of aquatic ecosystems.
- The total waste constituent concentration limits have been derived assuming that the constituents are bioavailable, e.g. from contaminated soil; the total waste concentration limits should not be applied to wastes that have been treated to contain or encapsulate waste constituents (e.g. in a concrete matrix, as the potential for environmental or human exposure is reduced by treatment of the wastes). The allowable elutrient concentrations apply irrespective of whether waste has been treated; however, the significance of the total waste constituent concentrations in a treated waste may need to be assessed on a case-by-case basis.
- Leachates from some New Zealand landfills already have elevated levels of contaminants. These levels must be taken into account when accepting further material, and may well act as controlling factors when determining acceptable concentrations and quantities.
- Some constituents require special handling and treatment e.g. mixing with some wastes may mobilise metal contaminants.

It is illustrative to consider the development of criteria for copper and arsenic, assuming the two hypothetical landfill case examples from Section 7.5.10:

Landfill A: where the landfill generally conforms with the Class 3 criteria, but the landfill is in a quarry with surrounding low-permeability materials, and discharge of leachate can only occur to groundwater, and direct discharge to surface waters cannot occur.

Landfill B: which also conforms with Class 3 criteria, but the landfill has been built up on the ground not far from a creek, where aquatic ecosystem protection is required. The quantity of wastes to be disposed of is very small compared with the quantity of normal refuse.

The selection of relevant mixing and dilution factors for Landfill A and Landfill B is based on professional judgement and is outlined as follows:

- **Solid Waste Mix Ratio**

  Contaminated soil is deposited in Landfill A in accordance with general practice for a Class 3 landfill and the total quantity of soil deposited is approaching 1% of the landfill contents. On this basis, the generic solid waste mix ratio (10) has been adopted.

  In contrast Landfill B is a much larger operation and the quantity of contaminated soil to be disposed is very small in relation to the quantity of normal refuse received. Therefore a higher solid waste mix ratio (20) has been adopted.

- **Capping Control Factors**

  Generic value of 4 adopted for each landfill.

- **Leachate Mix Ratio**

  Leachate mix ratios of 10 and 20 have been adopted for landfills A and B respectively, reflecting similar considerations to those on which the solid waste mix ratios are based.

- **Receiving Waste Dilution Factor**

  Landfill B reflects the conditions for which the generic receiving water dilution factor (10) was adopted, i.e. direct discharge of leachate to a surface water can occur.

  Direct discharge to surface water from Landfill A cannot occur, rather discharge occurs to groundwater which is not used in the immediate vicinity. Further, the low-permeability materials surrounding the landfill minimises the rate of contaminant leaching. Significant dilution occurs on discharge to the groundwater and within the aquifer prior to extraction and use and therefore a higher receiving water dilution factor (40) has been adopted. It is assumed that groundwater does not discharge to a surface water where ecosystem protection may be required.

The various factors which area assumed to apply are shown in Table 7.7, and may be compared with those in Tables 7.4 and 7.5.

Table 7.7
Case Example – Disposal of Contaminated Soil
at Two Different Class 3 Landfills

	Landfill A	**Landfill B**
Solid Waste Mix Ratio	10	20
Capping Control Factor	4	4
Allowable Total Copper Concentration in Soil		
– environmental (40 mg/kg)	1600 mg/kg	3200 mg/kg
– human health (370 mg/kg)	14800 mg/kg	29600 mg/kg
Allowable Total Arsenic Concentration in Soil		
– environmental (30 mg/kg)	1200	2400
– human health (30 mg/kg)	1200	2400
Leachate Mix Ratio	10	20
Receiving Water Dilution Factor	40	10
Allowable Elutrient Copper Concentration		
– aquatic ecosystem (0.002-0.005 mg/L)	Not applicable	0.4-1 mg/L
– drinking water (1 mg/L)	400 mg/L	200 mg/L
Allowable Elutrient Arsenic Concentration		
– aquatic ecosystems (0.05 mg/L)	Not applicable	10 mg/L
– drinking water (0.01 mg/L)	4 mg/L	2 mg/L

It can be seen from this case example that quite different acceptance criteria can apply; with Landfill A the soil copper concentration criterion of 1600 mg/kg can be expected to be limiting, whereas with Landfill B the TCLP elutrient copper concentration (0.4-1 mg/L) based on aquatic ecosystem protection will limit the permissible soil copper concentration. In contrast, for arsenic, the TCLP elutrient concentration based on drinking water considerations may be limiting for each landfill.

7.7 DIOXIN CONTAMINATED WASTES

This chapter has focused on the disposal of PCP-contaminated wastes to landfill. However, dioxins may be present in PCP-contaminated wastes as a secondary contaminant and in these situations dioxins may be a significant contaminant. In order to ensure the proper management of dioxin-contaminated wastes, guidance is provided regarding the maximum allowable concentration of dioxins in a waste accepted for landfill disposal.

Dioxins are persistent and bioaccumulative and some congeners are known to cause adverse effects at relatively low levels of exposure. On this basis, some argue that the landfilling of such wastes should be banned. However, the landfilling of wastes containing very low levels of dioxins may be the only practical option in many situations. It is assumed that the disposal of dioxin contaminated wastes to landfill should be highly restricted.

Studies undertaken by NECAL (1991) indicate that the concentration of dioxins in PCP formulations used in New Zealand in 1986 was approximately 1 mg/kg dioxin (TE).[6] If a similar ratio between the PCP and dioxin concentrations was to be maintained in contaminated soil, then soil containing 100 mg/kg of PCP would contain 0.1 μg/kg dioxins (TE). However, proportionally higher dioxin concentrations may be detected in some wastes, given:

- PCP will tend to be removed from contaminated soil by leaching, ultraviolet degradation and microbiological action, whereas dioxin will tend to persist; and
- Sludges from the bottom of PCP tanks etc. tend to contain higher concentrations of dioxin than in the bulk formulation.

It is noted that:

- The European Community permits the landfill co-disposal of dioxin-contaminated wastes containing up to 10 μg/kg TCDD;[7]
- The draft ANZECC (1992) "National Strategy for the Management of Scheduled Wastes" and the associated model regulation nominated an interim threshold concentration for the disposal of dioxin-contaminated wastes to landfill of 25 μg/kg (TE) (i.e. maximum allowable for disposal to a secure landfill) and an interim threshold for classification of a waste as a scheduled waste of 2.5 μg/kg (TE) (i.e. below this concentration there were no special requirements for disposal to landfill).

  Dioxins have subsequently been removed from Schedule A of the model regulation included in the draft strategy, on the basis that ANZECC considered that dioxins are not a significant component of wastes in Australia.
- On the other hand, the USEPA determines the acceptance of dioxin-containing wastes for landfilling on the basis of the TCLP test (CFR Part 40, Section 268.41; i.e. the concentration of TCDD, TCDF, PCDD, PCDF, HCDD and HCDF in the extract from the TCLP test, must not exceed 1 ppb for each congener).

It is common for landfill disposal limits to be developed with reference to the relevant residential soil acceptance criterion.

A residential soil acceptance criterion for dioxins has not yet been formally established in New Zealand. However, the soil acceptance criteria developed for dioxins as part of the PCP Risk Assessment Pilot Study (NTG, 1992) are brought forward as interim criteria as follows:

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6 Refer to Appendix D for details of the NATO Toxicity Equivalence Factors.

7 2, 3, 7, 8 Tetrachlorodibenzodioxin

- Agricultural: 0.01 μg/kg (TE)
- Residential: 1.5 μg/kg (TE)
- Industrial – Unpaved: 18 μg/kg (TE)

By way of comparison, the CCME (1991) "Interim Canadian Environmental Quality Criteria for Contaminated Sites" nominates the following values:

- Background: 0.01 μg/kg (TE)
- Agricultural: 0.01 μg/kg (TE)
- Residential: 1 μg/kg (TE)

The Victorian Environment Protection Authority (VicEPA) has nominated limits for the disposal of low-level contaminated soil to landfill. The landfills to which such low-level contaminated soil can be disposed may be described as good-quality municipal landfills, rather than secure landfills which may accept more highly contaminated wastes. Criteria for the disposal of low-level contaminated soil are based on a leachate test requirement (e.g. TCLP) and a total constituent concentration 10 times the residential use criterion. Where there is significant uncertainty regarding the possible impact of landfilling a given waste, the VicEPA have required that such waste be disposed to a secure landfill, while maintaining a total constituent limit 10 times the residential criterion.

From a consideration of the above, the following approach and criteria is adopted in these guidelines:

- PCP-contaminated wastes should be assessed for dioxin contamination, in-situ where possible, by judicious use of the OCDD screen as part of the site characterisation. All PCP-contaminated wastes designated for off-site disposal from sources which may contain significant dioxin concentrations should be assessed to determine the concentration of dioxins;[8]
- if the waste contains less than 10 μg/kg (TE) of dioxins based on full congener analysis, or less than 5 μg/kg (TE) based on the OCDD screening methodology,[9] the waste may be disposed of at a Class 1, 2 or 3 landfill, in accordance with the requirements for disposal to such a landfill, and in accordance with the requirements for landfilling PCP-contaminated wastes;

[8] Where PCP-contaminated wastes are not adequately assessed for dioxin content as part of the site characterisation process, additional OCDD screening analyses may be required prior to acceptance by the landfill.

[9] The OCDD screen shall include quantification of 2, 3, 7, 8 substituted hepta- and octachlorinated dioxins and furans; refer to Chapter 4, Section 4.8 for details of the OCDD screening method, and to Appendix D for details of the determination of toxic equivalent concentrations.

- if the waste contains more than 10 μg/kg (TE) of dioxins based on full congener analysis or more than 5 μg/kg (TE) based on the OCDD screening methodology,[10] the waste shall not be accepted for landfill disposal, and shall be stored pending treatment to reduce the dioxin concentrations.

These criteria will be reviewed as part of the development of Government policy on dioxins to be completed during 1999.

7.8 SUMMARY OF THE RECOMMENDED APPROACH FOR ACCEPTANCE OF TIMBER TREATMENT WASTES IN LANDFILLS

The acceptance of wastes containing timber treatment chemicals for landfill disposal should be on the following basis:

(a) Consideration of the waste; whether it is soil and can be safely left in situ in accordance with these guidelines.

(b) The waste should be characterised to determine its nature, the range of contaminants and their concentration and quantity. (The present guidelines cover PCP, dioxins, copper, chromium, arsenic and boron.) Where the PCP concentration in the waste exceeds 100 mg/kg, or where contaminated soil is from an area of potentially significant dioxin concentration, then the waste should be specifically tested to determine the dioxin concentration, and managed in accordance with the procedure outlined in Section 7.7. It is preferred that PCP wastes be assessed for dioxin as part of the site characterisation, rather than on a load-by-load basis at the time of disposal.

(c) A detailed consideration of alternative methods of dealing with the waste should then be undertaken (refer Section 7.2).

(d) If other options for dealing with the waste are not feasible or practicable and landfill disposal is necessary, then a waste manifest should be completed to define the waste quantity and composition for formal consideration of landfill disposal (refer Section 7.4).

(e) The availability of conforming landfill facilities should then be reviewed. The preference should be:
  – a landfill within the region (first priority);
  – the most secure landfill (second priority).

(f) If neither suitable landfills nor treatment options are available then the wastes should be placed in secure storage.

[10] The OCDD screen shall include quantification of 2, 3, 7, 8 substituted hepta- and octachlorinated dioxins and furans.

(g) The relevant regional council should be consulted to establish the consent requirements, and the receiving environment acceptance criteria of the preferred landfill.

If the quantity of the PCP is small (e.g. less than 250 g of PCP), and the concentration and leachability of the waste conforms with the acceptance criteria for Class 3 landfills (Table 7.4 and 7.5), the landfill operator may choose to accept the waste load without more detailed consideration.

(h) If the quantity of the PCP constituent is greater than 250 g PCP, more detailed consideration is required and acceptance criteria appropriate for the particular landfill should be determined with consideration to the various factors discussed in Section 7.5, including:

– concentration of PCP and other constituents;
– leachability of these constituents;
– is the total bulk of hazardous waste less than 1% of the other refuse being co-disposed in the waste disposal zone;

If the waste complies with the acceptance criteria and consent is obtained, then the landfill operator may accept the waste.

If the waste does not comply with the acceptance criteria, then consideration may be given to other landfills in surrounding regions, or treatment of the waste to achieve compliance. If compliance with the requirements for waste acceptance cannot be achieved, then the waste should be placed in secure storage.

These procedures are summarised in a logic diagram, Figure 7.1, presented as part of the introduction to this chapter.

7.9 COMMENTS ON THE APPLICATION OF THE GUIDELINES

The logic chart shown in Figure 7.1 prescribes a number of situations where the specific tables given in the guidelines will not be appropriate for use. This may be because:

- the proposed landfill does not conform to the prescribed categories;
- the wastes are at a greater concentration or amount than the maximum prescribed level for the proposed landfill site;
- the aquatic ecosystem and/or drinking water maximum allowable concentration values are not considered appropriate;
- constituents other than or as well as PCP are of concern; or
- the landfill has reached its allowable dilution ratio.

In such situations a rational approach utilising as much site-specific data as possible should be made. Where such data and information exist they should be used to demonstrate that the guideline values are able to be increased. (The tabulated values of the guidelines are deliberately conservative.)

A landfill evaluation should include detailed consideration of the following factors:

- hazardous constituent(s) and their nature;
- previous and potential future acceptance of similar or incompatible material in landfill;
- past current and future state of landfill operations;
- landfill engineering parameters;
- detailed risk analysis including identification of waste migration pathways to exposed populations and potential levels of exposure;
- monitoring data from site including leachate, groundwater and surface water quality.

For this information to be properly evaluated, technical expertise in the areas of hydrogeology, chemistry of contaminants and risk evaluation would be required.

The regional council may wish to view details of the landfill evaluation and possibly advise additional conditions or constraints as indicated on Figure 7.1.

7.10 REFERENCES

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APPENDIX A
LANDFILLING PRINCIPLES AND PRACTICE

1. LANDFILLING PRINCIPLES

1.1 Requirements

A landfill accepting hazardous waste materials – which may remain hazardous – must be designed and constructed on the basis that it will be the final repository for the waste materials. The landfill must control any release of the wastes into the surrounding environment that could adversely affect the beneficial uses of the land and the environment near the facility.

Conceptually, there are two basic approaches to the safe landfilling of waste: “natural attenuation” and “engineered containment”.

1.2 Natural Attenuation Approach

In the natural attenuation approach, landfills designed to receive wastes are located only in geologically stable areas where attenuation of the waste constituents in the environment can be achieved naturally. This approach allows the possibility of achieving a condition where less maintenance is required over the long term. The major disadvantage in the natural attenuation approach is that it is based largely upon predictions about the level of protection provided by the natural environment. Natural attenuation also requires the presence of suitable hydrological and geological conditions.

1.3 Engineered Containment Approach

In the engineered containment approach, reliance is placed on engineered facilities rather than on natural attenuation to protect the environment. Typically, landfills designed using this approach combine engineered liners (e.g. clay, flexible membrane), covers, leachate and/or gas collection and treatment systems to control the release of contaminants to the environment.

While engineered containment reduces the potential risk associated with the migration of waste constituents into the environment in the short term, questions remain concerning the integrity and functionality of such systems over the long term. The additional cost and responsibility associated with the maintenance and operation of such systems have to be accounted for. In practice, a specific landfill design may incorporate features from both basic design approaches to achieve the necessary level of environmental protection.

Regardless of the approach taken to landfill design, it is of paramount importance to have accurate information on the geological and hydrogeological characteristics of the site and the surrounding environment. Specific attention should be given to the development of a waste constituent transport model to assist in understanding and estimating the potential for off-site environmental effects.

1.4 Landfill Development Methods

There are three general methods for developing a landfill.

- **Cell Development Method**

  In the cell development method, discrete cells are constructed for depositing waste. Each cell is opened, filled and closed as a unit during a period of weeks or months. This may involve a relatively short timeframe compared to the life of the overall landfill site. Cells will tend to be relatively square in plan view.

- **Trench Development Method**

  In the trench development method the cell is extended such that the length to width ratio is substantially greater than 1:1. Unlike a discrete cell, the trench is developed continuously, with opening, placing and covering activities advancing in unison as the waste is placed in the trench. A trench is operated over a longer period of time than a cell, ranging from months to years. The length of the trench is constrained by the overall dimensions of the landfill site.

- **Area Development Method**

  In the area development method the landfill is developed over the full area available on the site. Similar to the trench method, the landfill operation is continuous with the opening, placing and covering of waste advancing steadily. The area is developed in this fashion over the full life of the landfill site.

  In descending order, the area, trench and cell development methods offer the most efficient utilisation of available space on the site, all other things being equal. Nevertheless, depending on project-specific circumstances, there may be justification to choose one method over another. For example: seasonal constraints due to harsh weather conditions, the need to provide a cover over the open working face, site configuration, and other reasons, may be sufficient to justify using the discrete cell method.

1.5 Siting

Sites suitable for the natural attenuation approach will usually consist of areas where no water resources are present that can be adversely affected, or where water resources are naturally protected by sound geological formations. The natural attenuation approach can also be appropriate where the types of wastes placed in the landfill can be controlled so that any leachate produced will be adequately attenuated by the environment.

Proponents for natural attenuation landfills must emphasise the need for a thorough hydrogeological assessment of the landfill site. This assessment must predict the long-term environmental effect of the landfill on the groundwater and surface water resources and address the need for any abatement measures to reduce adverse effects. The assessment must also investigate the applicability and nature of contingency measures and monitoring plans.

1.6 Leachate Control

All containment structures exposed over long periods of time to the weather and various chemicals absorb water. Waste exposed to this water produces leachate. A natural attenuation landfill in a humid climate that relies on clay deposits for environmental protection is expected to accumulate water over the long term. This water and the resulting leachate will eventually reach an equilibrium elevation above the base of the waste. Leachate that is produced in the landfill is often not recovered. Its movement and quality are controlled by the permeability and attenuation capacity of the natural environment at the site.

1.7 Monitoring

Monitoring is particularly important because it is through long-term monitoring that data is gathered to verify the hydrogeologic assessment and permit the final closure of the site.

1.8 Long-Term Care

The guarantee of long-term care, until the site can be finally decommissioned, is a particularly difficult and complex problem. In the case of natural attenuation landfills, long-term care must include the post-closure costs of facility insurance, site maintenance, contingency plans and monitoring.

One of the advantages of the natural attenuation approach is that activities such as leachate collection and treatment are not required, or are required for a limited period only and, therefore, the costs of long-term care are minimised.

1.9 Compensating Features

Where adequate natural conditions do not exist, compensating features are required.

The natural attenuation approach will accommodate the incorporation of sophisticated technology and “engineering” into landfill designs. This procedure reduces contaminant discharge to levels that can be accommodated by the natural attenuation capacity of the surrounding environment.

For example, it may be necessary to reduce the quantity and concentration of a particular leachate constituent by modifying the cover design. It may also be feasible to incorporate some type of accelerated leaching or in-place treatment system into the

design in order to reduce the strength of the leachate generated during the first few decades after site closure.

The natural attenuation approach will accommodate the use of flexible membrane liners and leachate collection facilities as added protection in a naturally protective environment, provided these facilities do not interfere with site monitoring.

2. NEW ZEALAND PRACTICE

Typical traditional New Zealand practice for refuse disposal site selection and method of operation has been to utilise gully, quarry or estuarine sites. Many of these sites have bases submerged in water, high inflow water budgets to the site, and little attenuation, and dilution is the only mechanism acting on the leachate passing from the site.

There is now general recognition of the inadequacies of these historical sites, and planning is well advanced for the closure and replacement of most of those which remain.

Newer sites are being chosen on more appropriate criteria, although until recently, many were still selected on a subjective basis rather than a rational analysis of contaminant transport processes and risk evaluation.

Operationally, the development of New Zealand refuse disposal sites almost exclusively involves the "area" method (using standard landfill nomenclature), although discrete trenches are often excavated in mature refuse to accommodate liquid or hazardous wastes.

Almost all sites accept liquid wastes, some in quantities which would represent more than 5% of the total water budget. Only one New Zealand site has been established as a total engineered containment site. This was designed to accommodate a specific waste, and was not operated on a continuous basis. A rigorous hydrogeological evaluation of most sites would classify them as attenuation rather than containment sites.

The application of cover is generally infrequent and inadequate from a point of view of minimising leachate formation. Working areas are often too large, and use of cover material is minimised due to cost, availability, and conservation of landfill space.

Modern landfill design stresses the view of the landfill as a "biological reactor", where adequate moisture content is a prerequisite for higher rates of decomposition. This view is correct; however, the theory is used in some quarters as an excuse to not provide adequate controls of infiltration to the site, with the use of small working areas and low permeability intermediate and final cover. Indications from water budgeting, modelling, field moisture content observations, and levels of leachate generation are that most areas of New Zealand by virtue of climatic conditions naturally support adequate moisture contents for acceptable degradation rates. In such areas, not

minimising infiltration will result in undesirable mobilisation of waste constituents and excessive levels of leachate formation. In areas where the field moisture content is low, the moisture content should be monitored and, if necessary, modified in a controlled manner such as by recirculating leachate rather than allowing infiltration.

Some sites which do provide daily and intermediate low-permeability cover do so in a continuous layer, thereby creating undesirable stratification within the site. Such cover should either be removed prior to the next day's operation or lift of the site, or at least have "windows" cut in it to permit percolation of moisture and gas. In many areas daily cover with permeable materials such as sand or sawdust is an acceptable solution.

Many newer sites collect leachate. The subsequent treatment/disposal of leachate is generally by recirculation within the landfill and/or discharge to a municipal sewer. In some cases, the level of sewage treatment and ultimate disposal is such that discharging leachate to a sewage treatment facility designed for treating municipal sewage is probably not environmentally acceptable. With the current retrofitting of leachate capture facilities to more remote landfill sites, on site treatment is likely to become more common.

Gas collection for power generation or direct energy use is being investigated at a limited number of larger sites, with two landfill gas power generation schemes already in operation. Some modern designs incorporate gas collection layers and vents. Most sites have no such facilities, although retrofitted leachate collection systems may also act as release mechanisms for gases heavier than air.

A review of relevant overseas landfill standards and designations as well as New Zealand operating practice has been undertaken to yield guideline criteria for this study.

APPENDIX B
LEACHING TESTS AND CO-DISPOSAL OF WASTES IN A SANITARY LANDFILL

1. INTRODUCTION

Under the ANZECC sponsored document "National Strategy for the Management of Scheduled Wastes", interim disposal of scheduled wastes may occur in a "controlled landfill" (this term is defined in the regulations) provided three requirements are met:

(i) It has been shown that such disposal will not cause any significant effect on the beneficial uses of the environment external to the landfill;

(ii) The concentrations of chemicals in the waste or residues are less than the threshold concentrations for landfill disposal listed in the schedule;

(iii) The waste or residues are in solid or spadeable form and meet the requirements of an elutriation test to be specified by the agency.

The requirements of the elutriation test are not specified in the document and it would appear unlikely that the ANZECC authorities will mandate particular test details in the future. It is understood that individual Australian States are already applying their own leaching criteria. There is therefore a need for New Zealand to assess a protocol for leaching tests and corresponding acceptable concentrations of scheduled waste species in test eluent which will enable the tenet of the Resource Management Act – that of sustainable management – to be kept.

This report is divided into five parts. In Section 2 an overview of the various stages of waste degradation in a landfill is given. A rationale for the necessity of buffered acetic acid leaching fluid is given and the optimum phase at which hazardous waste disposal should occur in a landfill is indicated. In Section 3 the various types of leaching test are classified and the information that can be derived from each particular test is specified.

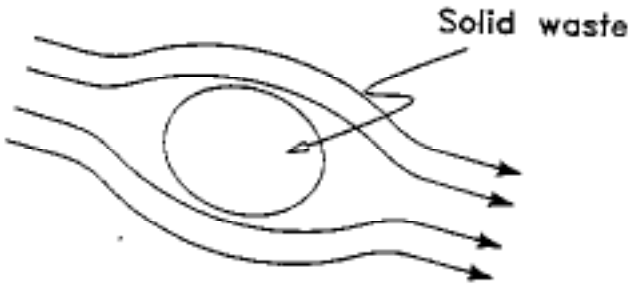
A summary of specific leaching test protocols currently in use by regulatory authorities is given in Section 4. In Section 5 a comparison of two prominent leaching test approaches is made and desirable and disadvantageous features of both are listed.

2. LEACHING PROCESSES RELATED TO WASTE DEGRADATION IN A LANDFILL

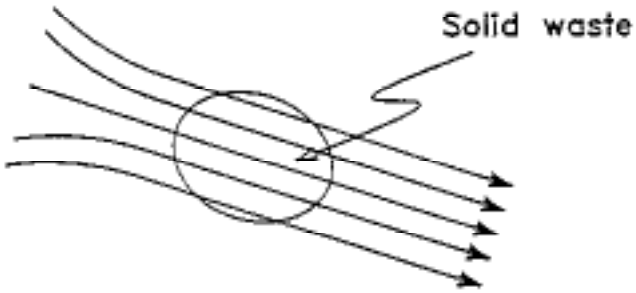
2.1 The Process of Waste Leaching (Environment Canada, 1990)

Leaching occurs when the contaminants in a waste come into contact with a leachant. Contact can occur by leachant flowing around the waste, or through the waste, or a complex combination of processes depending on the waste's porosity.

Once contaminants have been contacted by a leachant, leaching may ensue. Leaching encompasses the physical and chemical reactions that mobilize a contaminant, as well as the mechanisms of transport that carry the contaminant away from the waste.



(a) Water flowing around the waste



(b) Water flowing through the waste

In nonporous waste forms such as glasses and ceramics, where there are no voids within the waste, leaching is the result of interfacial exchanges at the outer surface by dissolution. Leachant renewal at the geometrical interface causes washing away of the surface of the waste form.

In porous wastes, which occur much more commonly, leaching is initiated at the pore scale, or the particle interface. These wastes consist of individual particles, which may or may not be consolidated, with voids between the particles.

If a porous waste is wet, there could be many different phases present: several solid phases, an aqueous phase, a nonaqueous liquid phase and a gas phase. Before contact with a leachant, the waste is normally at or approaching the state of chemical equilibrium, contaminants are associated with specific phases, and there is no net transfer between phases. Contacting the solid waste with leachant disrupts this chemical equilibrium, initiating the leaching process.

For wastes that are initially dry, wetting initiates leaching by mobilizing those constituents that are easily dissolved or desorbed. Immobile constituents become mobile as a result of complex combinations of physical reactions, such as dissolution, desorption, and subsequent transport through the liquid film of immobile water that surrounds a particle of waste, by molecular diffusion.

Leaching, therefore, is the result of chemical reactions at the scale of individual particles, which act to mobilize contaminants, coupled with transport processes dependent on the magnitude of leachant flow.

2.2 The Production and Character of Landfill Leachant – "Leachate"

All landfills produce a liquid phase of leaching liquor with a high organic and inorganic content commonly referred to as "leachate". Leachate arises from a number of sources but the principal are:

(i) Penetration of rain and sur face water into the landfill.

(ii) Liquid phase present in wastes deposited.

(iii) The physical and biochemical breakdown of wastes.

Although hazardous waste species will also be present in the leachate from a co-disposal landfill, the bulk leachate character will largely be derived from the products of breakdown of domestic wastes. To predict the likely gross composition of leachate in a landfill it is necessary to understand the various biochemical and microbiological stages that occur over a landfill's lifetime. Typically every landfill may exhibit the characteristics of five phases. The major microbiological/biochemical processes occurring in each phase are documented in Section 2.2.1. Figure 1 shows changes in selected chemical parameters with time and through the respective phases.

2.2.1 The Five Biochemical Stages of a Landfill's Lifetime

(Rushbrook, 1990, Pohland et al., 1985)

Phase I. Aerobic decomposition

Waste placement and moisture accumulation

Organic wastes decompose in the presence of oxygen. Putrescible (vegetable and food wastes) materials degrade most readily, followed by paper, wood, natural textiles and rubbers. This phase is characterised by rising carbon dioxide concentrations from aerobic respiration of micro-organisms and rising waste temperatures, derived from accelerating exothermic microbial decomposition processes. Also there are rising carboxylic acid concentrations in leachates formed as products of incomplete metabolic degradation by bacteria. This phase lasts only a few days or weeks in well-run controlled landfills. In poorly run landfills with a low density of waste emplacement and no compaction this phase can predominate. In these circumstances the landfill will be characterised by high temperatures, strong odours, and high carboxylic acid concentrations in leachates.

Phase II. Anaerobic, acetogenic decomposition

In most landfills oxygen is rapidly depleted and the environmental composition becomes more reducing. Anaerobic bacterial systems take over. This phase will last up to several months at well-run sites, and during this period carbon dioxide concentrations rise to over 70% by volume and carboxylic acid concentrations also continue to increase. This phase is propagated by acid-forming and acetogenic bacteria whose metabolic conversion of cellulose produces carboxylic acids (predominantly acetic acid), carbon dioxide and smaller quantities of hydrogen. Some landfills operate at this phase permanently. On the surface they may appear to be well-run, but the waste degradation achieved can produce excessive quantities of high BOD leachate containing carboxylic acids. This is principally due to the establishment of insufficiently "reduced" chemical conditions in the landfill to enable strictly anaerobic methane-producing bacteria to thrive and utilise the carboxylic acids produced by the acetogenic bacteria.

The continued existence of acetogenic decomposition in a landfill generally indicates a need to improve the covering of wastes to seal them from the atmosphere.

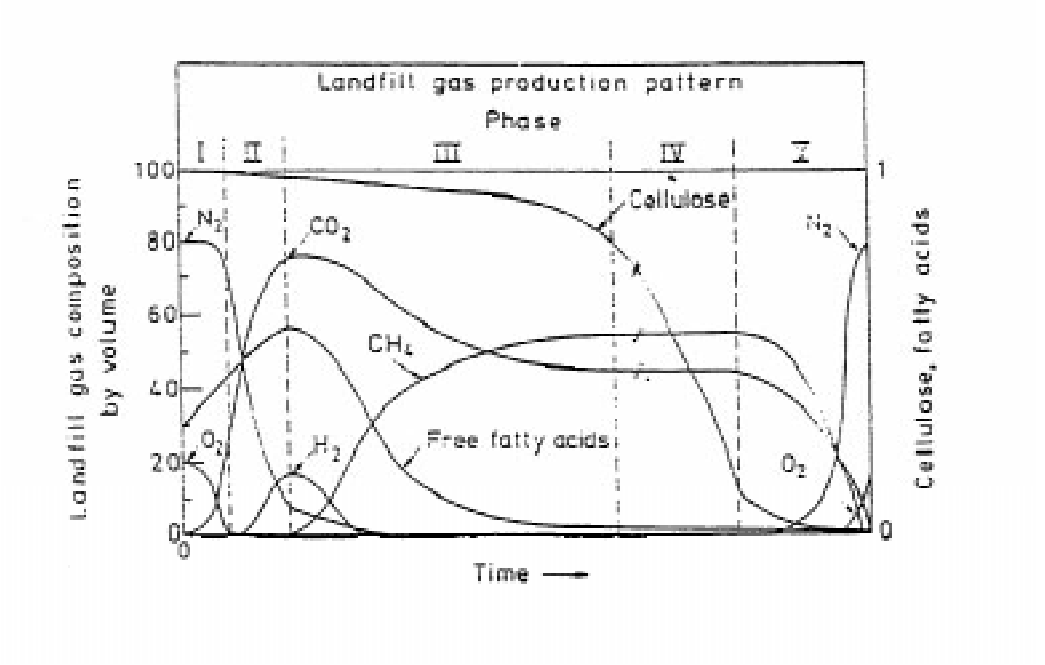
Phase III. Anaerobic, rising methanogenic decomposition

As oxygen depletion continues and the redox potential (Eh) of interstitial waters drops to below approximately – 200 mV, then the conditions become suitable for methanogenic activity to develop. Over the period of a few weeks methane concentrations begin to rise and carboxylic acids decline. This is due to the acetic acid in the leachates being utilised by the methanogens to produce methane, carbon dioxide and water. Landfill temperatures usually become stabilised in the mesophilic range (i.e. up to 40°C).

Phase IV. Anaerobic, stable methanogenic decomposition

This phase represents the most stable period in the decomposition of waste in controlled landfills. It is believed to persist for at least 15 to 20 years in temperate climatic areas and is characterised by methane and carbon dioxide concentrations of around 65 and 35% respectively. Lower carboxylic acid concentrations in leachates are observed and there is a gradual depletion of the available organic carbon substrate in the waste.

Figure 1
The Relative Composition of Selected Landfill Components Versus Time and Designated Phase



Phase V. Rising aerobic gaseous composition

No-one has yet studied waste decomposition in a landfill to completion. However, evidence from very old sites suggests that once the available organic carbon is used up the methanogenic microbial activity diminishes and methane and carbon dioxide concentrations gradually decline. At some future point, it has been argued, oxygen levels will begin to rise. Eventually the remaining waste would be regarded as biologically "inert" and atmospheric gaseous conditions may become re-established. This situation has not been demonstrated in the field.

In reality the five phases discussed above could be achieved simultaneously at one site because in most situations landfills are filled gradually. In addition the length in time of each phase will vary from site to site because of climatic, operational, management and waste type factors.

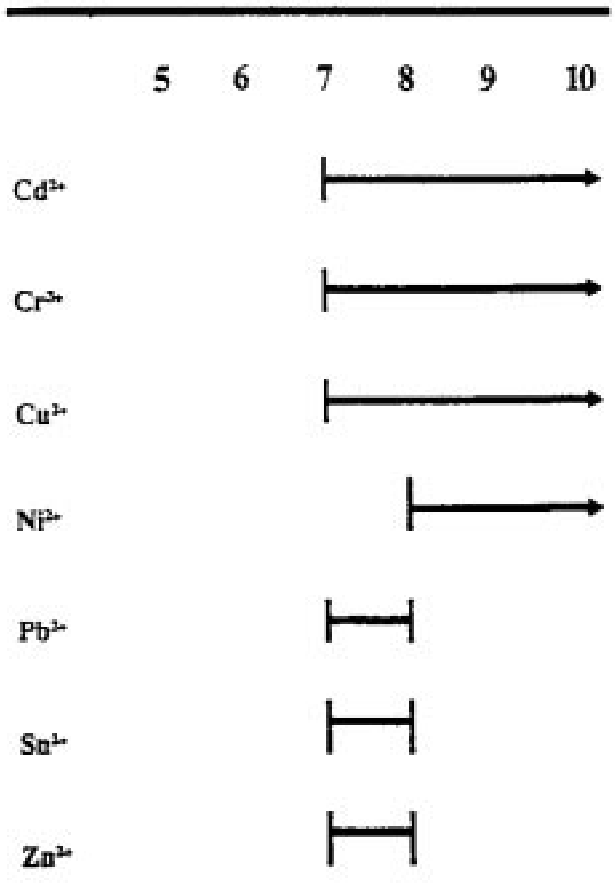
For co-disposal to be undertaken in a controlled and safe manner, it is important that stable anaerobic methanogenic conditions are established within the deposited municipal wastes. In practice this will require exhumation of an area of the landfill where this condition exists and admixing of the hazardous waste at pre-determined loading rates.

2.2.2 Relationship Between pH and Mobility of Heavy Metal Species – the Requirement for an Acid Leaching Medium

Figure 2 below indicates the pH range for effective precipitation (and consequent immobilisation) of various heavy metal species. A pH of between 8 and 9 is a useful compromise between achieving sufficient hydroxyl ion concentration to precipitate species such as metal hydroxides and avoiding excess hydroxyl ion concentration and production of soluble $[ML(OH)_4]^{-2}$ species.

As was discussed in the previous section, landfills may pass through developmental stages where large quantities of soluble organic acid species are produced. In recently emplaced wastes research has shown that up to 90% of the soluble organic carbon can be accounted for as short-chain volatile fatty acids: acetic, propionic and butyric acids being present in greatest concentrations (Bingham, 1987). Although these organic acids are considered relatively weak acids, when present in significant concentrations they may significantly reduce the pH of landfill leachate and cause mobilisation of fixed metal species present in the landfill. As a consequence the synthetic leaching media adopted in leaching tests typically involve acetic acid, the most prevalent acid species present in natural leachate, buffered to an acid pH – typically about 5.0.

Figure 2
pH Range for Precipitation of Metal Hydroxides (Bingham, 1987)



3. FUNDAMENTAL ASPECTS OF LEACHING TESTS (Environment Canada, 1990)

3.1.1 Experimental Variables in Leaching Tests

The following variables common to most leaching tests are discussed in Sections 3.1.1 to 3.1.7:

- sample preparation
- leachant composition
- method of contact
- liquid-to-solid ratio
- contact time
- temperature
- leachate separation

An attempt has been made to provide a rationale for the practices adopted.

3.1.2 Sample Preparation

Depending on the nature of the waste itself, the sample may require one of the following preparatory steps:

- liquid/solid separation
- particle size reduction
- surface washing

Liquid/solid separation may be necessary on wastes containing a free liquid phase. In some cases the liquid will be regarded as initial leachate and combined with the extract derived from the solid portion after the leaching test has taken place.

Particle size reduction is required for most extraction tests (an exception being those tests designed to cater for monolithic waste samples). The goal is to reduce the time required to reach steady-state conditions by increasing the surface area of contact between the waste and the leachant. Test requirements vary in their particle size prescription.

Surface washing may be performed prior to testing small monolithic samples in flow-around tests.

Leachant Composition

The release of contaminants from a waste in any leaching test may be strongly influenced by the leachant composition. Table 1 shows three types of leaching fluid commonly used and the advantages and disadvantages associated with each.

Table 1
Leachant Types and Advantages and Disadvantages
Associated With Their Use

Type of Leachant	Purpose	Advantages	Disadvantages
Pure water	Non-aggressive baseline medium without buffering capacity	Reliable simple standard. Waste establishes the chemical environment	May bear little resemblance to the actual leaching solution in the field
Site liquid (e.g. land-fill leachate)	Simulates site-specific leaching conditions	Best field-case model	Requires characterisation (to obtain leaching results by subtraction). Requires careful storage
Synthetic chemical solution	To examine metal speciation and organic compound binding	A compromise between distilled water and actual leachate	May be too aggressive and prove difficult to relate data to field conditions

3.1.3 Method of Contact

Agitation of the leachant-waste mixture allows steady-state conditions to be reached at a faster rate by maintaining maximum contaminant concentration gradients at the leachant-waste particle interface. Different methods can be used to agitate the waste, including:

- shaking (wrist action or reciprocation)
- stirring (magnetic or paddle type)
- tumbling in closed containers
- gas bubbling.

3.1.4 Liquid-to-Solid Ratio

The liquid-to-solid is the ratio of the amount of leachant in contact with the waste to the amount of waste being leached.

The liquid-to-solid ratio can be expressed as:

(i) Volume of leachant/mass of waste.

(ii) Mass of leachant/mass of waste.

(iii) Volume of leachant/surface area of waste (for monolithic waste).

Where contaminant species are highly soluble (e.g. sodium, potassium), their concentration in the test solution will be inversely proportional to the liquid-to-solid ratio.

However, if release of the contaminant species is limited by its solubility in the leachant, the final concentration is independent of the liquid-to-solid ratio and simply equals the maximum solubility concentration.

The liquid-to-solid ratio chosen should be low enough to avoid dilution of contaminants to less than analytical detection limits. However, it must also be sufficiently high to prevent species solubility from limiting the amount of contaminants that can be leached from the waste. In practice, values ranging from 1:1 to 100:1 are generally chosen.

3.1.5 Contact Time

The total amount of time that a leachant is in contact with a waste sample before the attainment of steady-state conditions will influence the amount of contaminant released. In extraction tests, the contact time is equivalent to the duration of the test; whereas in dynamic tests, it is a function of the flow rate, or the number of elutions, in addition to the test duration.

The contact time for extraction tests should allow steady-state conditions to be reached for the contaminants of interest. This is generally on the order of hours to days for samples that have undergone particle-size reduction. For monolithic samples, it can be on the order of weeks to months.

The contact time for dynamic tests should be sufficient to allow for observation of the processes of interest. Diffusion processes may be quantified within a few weeks, although several months may be required to study slow chemical reactions.

3.1.6 Temperature

Although temperature influences kinetic processes such as desorption and diffusion such that mechanisms relevant to leaching vary exponentially with (absolute) temperature, for convenience most leaching tests are performed at room temperature.

3.1.7 Separation of the Test Eluent from the Waste Sample

The eluent from a leaching test is normally separated from solid waste sample by filtration through a glass fibre or membrane filter of defined particle retention (typically 0.45 • m). The filtration process allows the definition of a convention for “soluble” species.

Classification of Leaching Tests

For the purposes of this discussion, leaching tests have been separated into two broad categories on the basis of whether or not the leachant is renewed:

(i) extraction tests (no leachant renewal)

(ii) dynamic tests (leachant renewal).

3.2 Extraction Tests (Environment Canada, 1990)

Extraction tests include all tests in which a specific amount of leachant is contacted with a specific amount of waste for a certain length of time, without leachant renewal. (This definition does not include analytical extractions or digestion procedures, which are used to measure the total contaminant concentration in a waste.) The leachate is analysed at the end of the test for species of interest.

The underlying assumption in this type of test is that a steady-state condition is achieved by the end of the extraction test (i.e. the concentrations of constituents in the leachate become constant by the end of the test). In this no-flow system, a steady-state condition occurs when there is no net transfer of components from the liquid to the solid, or vice versa.

Extraction tests can be further divided into four subcategories:

(i) agitated extraction tests

(ii) nonagitated extraction tests

(iii) sequential chemical extraction tests

(iv) concentration build-up tests.

3.2.1 Agitated Extraction Tests

Agitated extraction tests are performed to reach steady-state conditions as quickly as possible. They measure the chemical properties of a waste-leachant system, as opposed to physical rate-limiting mechanisms. Agitation ensures a homogeneous mixture and promotes contact between the waste and the leachant. Sample particle size reduction is often performed to increase the surface area of contact and to eliminate mass-transfer limitations. This reduces the duration of the test by reducing the time required to reach a steady-state condition in the leachate. This procedure may also have the effect of overestimating the short-term release of constituents.

3.2.2 Nonagitated Extraction Tests

A nonagitated extraction test is performed to study the physical mechanisms that are rate-limiting in leaching. The underlying assumption behind a nonagitated extraction test is that the physical integrity of the waste matrix affects the amount of contaminants that are leached during the test. Consequently this type of test is often applied to monolithic waste samples.

The disadvantage of running a nonagitated test is that a much longer contact period may be required to reach steady-state conditions than required in an agitated test. The advantage of this type of test is that rate-limiting mechanisms of leaching, due to the physical integrity of the waste form, are taken into account.

3.2.3 Sequential Chemical Extraction Tests

A sequential chemical extraction test is composed of a battery of agitated extraction tests. It involves performing sequential elution of aliquots of sample with different leachants, which are increasingly more aggressive, in terms of chemical attack, towards the waste. This method assumes that each successive leachant also extracts the sum of contaminants extracted by all preceding leachants. This test can also be conducted by subjecting the same aliquot of sample to each leachant. The amount extracted in each elution is associated with a certain chemical form or mineral phase in the waste matrix.

3.2.4 Concentration Build-up Tests

In a concentration build-up test, an extraction is achieved at a very low cumulative liquid-to-solid ratio. Aliquots of samples of waste are successively contacted with the same leachant. The contact of leachate with fresh waste can be considered to model an elemental volume of water flowing through a large body of waste, approaching saturation with respect to specific contaminants. The purpose of this test is not to collect kinetic information, but to characterize a leachate saturated with soluble waste constituents. In some cases, this may simulate the actual pore water composition of a waste.

3.3 Dynamic Tests (Environment Canada, 1990)

Dynamic tests include all tests in which the leachant is continuously or intermittently renewed to maintain a driving force for leaching. The intermittent tests may be conducted by alternating leaching periods with dry periods to study the effects of desiccation or unsaturated flow conditions.

Dynamic tests provide information about the kinetics of contaminant mobilization. Information is generated as a function of time, and attempts are often made to preserve the waste form's structural integrity. These two factors lend this category of leaching tests to the investigation of more complex mechanisms of leaching.

Dynamic tests can be further divided into four subcategories according to how the interface between the waste and the leachant is defined. Tests in which individual waste particles are used to define the interface are called serial batch tests. Tests in which a characteristic dimension of the waste, such as the external geometric surface area or the geometric surface area perpendicular to flow, is used to define the interface include:

(i) serial batch tests

(ii) flow-around tests

(iii) flow-through tests

(iv) soxhlet tests.

3.3.1 Serial Batch Tests

A serial batch test is conducted using a granular or crushed monolithic waste sample, which is mixed with leachant at a given liquid-to-solid ratio, for a specified period of time. The leachate is then separated from the solids, and replaced with fresh leachant until the desired number of leaching periods have been completed. The waste/leachant mixture is normally agitated to promote contact. Kinetic information regarding contaminant dissolution is obtained using the concentrations measured in the leachate from each of the leaching periods. Data from serial batch tests can be used to construct an extraction profile, which can be used to infer the temporal release of leachable constituents.

3.3.2 Flow-around Tests

In flow-around tests, a sample of waste is placed in the leaching vessel and the flow of fresh leachant around the waste provides the driving force to maintain leaching. The liquid-to-solid ratio is expressed as the volume of leachant divided by the surface area of the solid sample. Samples are usually monolithic, although nonmonolithic or crushed waste may be used if it is confined in some manner. Agitation is generally not performed. Leachant flow is either continuous, in which case it is sampled and analysed periodically, or it is intermittently renewed. The latter method is generally simpler from an experimental point of view, but the renewal frequency must be sufficient to prevent a build-up of contaminants at the waste/leachant interface, which may inhibit further leaching.

3.3.3 Flow-through Tests

In a flow-through test, an open container is packed with a porous solid through which leachate is passed, either continuously or intermittently. The effluent is sampled periodically and analysed for the parameters of interest. The results are used to examine contaminant removal in which the primary transport mechanism is advection. There are two basic types of flow-through tests characterised primarily by the shape and size of the container:

(i) a column test is performed using a small cylindrical container

(ii) a lysimeter test is carried out in a large rectangular or cylindrical container.

Columns may be operated either in an upflow or downflow mode, whereas lysimeters are always operated in a downflow mode. Flow through the waste depends upon the hydraulic conductivity of the waste as well as the hydraulic gradient, and varies with the individual test.

3.3.4 Soxhlet Tests

A soxhlet test can be used to continuously contact the waste sample with fresh leachant without adding or removing leachant from the apparatus. In a soxhlet test, the leachant is not renewed volumetrically. Rather, the sample is continuously contacted with leachant, which has the leached constituents removed from it by evaporation and condensation prior to contact. The purpose of the test is not to collect kinetic information but to obtain the maximum amount of a constituent leachable from a waste sample, quickly and under severe conditions.

In a soxhlet apparatus, the leachate is boiled, condensed, and recirculated repeatedly through or around the waste sample, depending upon its physical structure. A soxhlet test permits very high liquid-to-solid ratios and yet concentrates the leached constituents, avoiding analytical detection limitations. It is limited to using low-boiling point liquids as leachants, and cannot be used to study chemical species that are volatile at the boiling temperature of the leachate.

4. SUMMARY OF LEACHING TEST PROTOCOLS

In this section a number of leaching test protocols of extraction or serial batch type are summarised in tabular form. Brief bibliographic details or points considered noteworthy are provided in the accompanying text. Discussion has been restricted to these two types of test at this time as these are seen as most appropriate for the waste concentration problem at hand. For information on other test types interested readers are directed to the Environment Canada Publication EPS-3/HA/7 (1990) – "Compendium of Waste Leaching Tests".

4.1 Agitated Extraction Tests (Environment Canada, 1990)

Table 2 shows a compendium of agitated extraction test protocols. Individual protocols are discussed as follows:

EP Tox (EPA Method 1310). The Extraction Procedure Toxicity Test (EP Tox) was promulgated in 1980, under the Resource Conservation and Recovery Act (RCRA), to classify wastes as hazardous or non-hazardous, based on maximum permissible concentrations for eight metals, four pesticides, and two herbicides. This test was based upon a 95% municipal/5% industrial co-disposal mismanagement scenario, primarily for inorganic wastes. It requires structural integrity testing for monolithic waste samples, and intermittent pH adjustment. The test process is now redundant in the United States.

Promulgation of EP Tox by the United States Environmental Protection Agency (US EPA) has spawned the development of a whole family of similar protocols, including its successor, the Toxicity Characteristics Leaching Procedure (TCLP), the Ontario Reg 309 Leachate Extraction Procedure (LEP), the Quebec R.s.Q. (Q.R.s.Q.), and the California Waste Extraction Test (WET), described below.

LEP. The Leachate Extraction Procedure (LEP) is the regulatory extraction test used in the province of Ontario, Canada. It was promulgated under Regulation 309 of the Ontario Environmental Protection Act in 1985. Only minor changes to EP Tox were made in developing this test. The Provisional Standard for Leachate Extraction Procedure of the Canadian General Standards Board (CGSB) is identical to LEP and has been adopted by the provinces of British Columbia, Alberta, and Manitoba. The minor variations in methodology between LEP and EP Tox result in only one practical difference: in the case of limed wastes with high moisture content and little free liquid, the pH of the final extract is higher using LEP, which results in less dissolution of some metals.

TCLP (EPA Method 1311). The Toxicity Characteristics Leaching Procedure (TCLP) was developed in 1984 under the Hazardous and Solid Waste Amendments to RCRA. It was promulgated under the Hazardous Waste Management System Land Disposal Restrictions, and has replaced EP Tox in its hazard determination role. TCLP was based on the same assumptions as EP Tox, but it includes the following modifications:

(i) Volatiles are prevented from escaping to the atmosphere by using a modified leaching vessel called a zero headspace extractor (ZHE), which eliminates the headspace.

Table 2
A Compendium of Agitated Extraction Test Protocols

Test Name and Proponent	Status of Development	Leaching Vessel	Sample Preparation	Sample Mass	Leachant	Liquid-to-solid Ratio	Agitation	Duration	Leachate Separation
EP Tox[7] US EPA Method 1310	Standard regulatory method (1980)	Unspecified	Nonmonolithic waste: phase separation Monolithic waste: particle-size reduction	100 g	Deionised water 0.5 N acetic acid (max. 2.0 meq H+/g solid)	20:1	Unspecified, continuous	24 to 28 hours	0.45 µm filtration
LEP[8] MOE (Ontario)	Standard regulatory method (1985)	Wide mouth, 1250 mL cylindrical bottle	Phase separation by 0.45 µm membrane filter	50 g of dry solids	Distilled water Acetic acid (2.0 meq H+/g dry solids)	20:1	End over end (10 rpm)	24 hours	0.45 µm filtration
TCLP[9] US EPA Method 1311	Standard method (1986)	Any material compatible with waste, zero headspace extractor (ZHE) for volatiles	Cutting/crushing and grinding Solid/liquid phase separation No structural integrity	100 g (25 g for ZHE)	Buffered acetic acid 1) pH = 4.93 2) pH = 2.88	10:1	End over end (30 rpm)	18 hours	0.6 to 0.8 µm borosilicate glass fibre filter combines liquid phase with extract
Q.R.s.Q[10] MOE (Quebec)	Standard regulatory method (1987)	>1 L bottle	No phase separation Grinding No structural integrity	100 g dry solids 50 g for volatiles	Inorganics: buffered acetic acid (0.82 meq H+/g dry solids) Organics: distilled water	10:1	End over end (10 to 20 rpm)	24 hours	30 min decantation, 0.45 µm filtration
WET[11] California	Standard regulatory method (1985)	Polyethylene or glass container	Milling, 0.45 µm filtration	50 g of dry solids	0.2 M sodium citrate at pH 5.0	10:1	Table shaker Rotary Extractor	48 hours	Centrifugation, 0.45 µm filtration
French Leach Test[12] AFNOR (France)	Proposed standard for hazardous waste (1987)	Straight wall, 1.5 L bottle	Remove free liquid Reduce particle size to <9.5 mm	100 g	Demineralised water	10:1	Roller or shaker	16 hours	0.45 µm filtration
EE(13) Environment Canada	Published research method	Inorganics: wide mouth, plastic sample bottle (250 mL) Organic: glass (500 mL)	Grinding (inorganic) Mortar and pestle (organic)	Inorganics: 40 g Organics: 80 g	Distilled water	4:1	National Bureau of Standards rotary extractor	7 days	0.45 µm vacuum screen
ASTM D3987[14] ASTM	Standard research method	Round, wide mouth bottle	As received	700 g	Distilled water (ASTM Type IV)	4:1	Shaking	48 hours	0.45 µm filtration
MBLP[15] Environment Canada	Published research method	Square, polyethylene or glass bottle, 1 to 2 L	Remove free liquid Reduce particle size to <9.5 mm	Variable to fill 90% of bottle	Distilled water Acidic water buffer to pH 4.5 Synthetic municipal solid waste	4:1 or 2:1	Slow rotary tumbling	24 hours	0.45 µm filtration
MCC-35[16] Materials Characterization Center	Standard regulatory method (radioactive wastes)	Teflon container, 20 mL to 1 L	Crush waste form into two fractions: 74 to 149 µm 180 to 425 µm	>1 g	Choice of high purity water, silicate water, brine, repository water	10:1	Rolling and rocking	Variable: 28 days to several years	N/A

Notes: AFNOR = Agence Française de Normalisation

(ii) Two leachants are employed in the procedure. For highly alkaline wastes, a solution of acetic acid is used (pH = 2.88); whereas for other wastes, use of a buffered leachant (pH = 4.93) eliminates the need for continual pH adjustment. In either case, the maximum amount of acid addition is the same, i.e. 2 meq/g waste. There is no allowance for structural integrity testing of monolithic samples, and all wastes must be ground to a particle size of less than 9.5 mm. Compared with EP Tox, TCLP increases by an order of magnitude the number of organics required to be analysed. Other equipment changes and specifications were made to improve reproducibility and reduce contamination. All of these changes have resulted in only slightly greater metal concentrations in TCLP leachates compared with those in EP Tox leachates.

Quebec R.s.Q. (Q.R.s.Q.) is the regulatory test used in the province of Quebec, Canada. It was first developed in 1980, and promulgated in 1985, with no provision for organics. In 1987, it was included in the Procedure for Evaluating the Physical and Chemical Characteristics of Solid and Liquid Wastes. Both EP Tox and TCLP were referred to in its development, with the resulting test being very similar to the latter. In Q.R.s.Q., however, phase separation is not performed, and the liquid part of the waste becomes part of the extracting fluid or leachant. Also, only distilled water is used as a leachant for organics.

California Waste Extraction Test (WET). First submitted in 1984, WET forms part of an overall legislation to identify hazardous and extremely hazardous wastes. WET was established to determine the amount of extractable substances in a waste that have been identified as being hazardous according to the Persistent and Bioaccumulative Toxic Substances criteria. This test is very similar to EP Tox; however, the addition of sodium citrate in the leachant as a chelating agent may make this test more aggressive towards certain wastes, or waste components.

French Leach Test. This French protocol is a standard regulatory leach test used to determine the soluble fraction of a solid waste in an aqueous solution, under strictly defined test conditions. The test cannot be used to analyse environmental impact or treatment technologies, such as solidification. It is intended for solids, pastes, and particles.

ASTM D3987. D3987 was standardized by the American Society for Testing and Materials (ASTM) in 1981, and a revised version was published in 1985. It still stands as a basic single batch agitated extraction test; however, many modifications have since been proposed or developed. Its intent is to provide a rapid, standard extraction procedure for industry. It is not intended to simulate site-specific conditions. The final pH of the leachate reflects the interaction of the leachant with the buffering capacity of the waste. It has not yet been validated for organic wastes.

LE. The Equilibrium Extraction (EE) protocol was first published in 1986 by Environment Canada. It was based upon ASTM D3987. EE uses distilled water and a low liquid-to-solid ratio to let the waste establish the chemical environment. Also, a long test duration (7 days) and particle-size reduction (<100 mesh) increase the probability of achieving steady-state conditions in the leachate. The procedure is run separately for organic and inorganic wastes, using appropriate containers and crushing methods.

MBLP. The Multiple Batch Leaching Procedure (MBLP) was based on experimentation carried out at Environment Canada's Wastewater Technology Centre between 1980 and 1986. This battery of tests allows for comparisons of the leachability of various wastes by measuring the change in leachability under different test conditions. Single elutions are performed, but under a variety of test conditions: two liquid-to-solid ratios, and three leachants. Two of the extractions are similar to EP Tox, the difference being that less acid is added and there is no phase separation.

MCC-3. The agitated powder extraction test (MCC-3) of the Materials Characterization Centre (MCC) is one in a series of test methods that were developed to help evaluate the chemical durability of nuclear waste forms. This test determines the maximum concentration of elements in solution under steady-state conditions. It is applied to crushed samples of monolithic waste forms. This test is similar in intent to that of MCC-1 and MCC-2 in which monolithic samples are tested, but the increased surface area exposed to the leachant makes it a more rapid test.

4.2 Serial Batch Tests (Environment Canada, 1990)

Details of Serial Batch type protocols are provided in Table 3 below.

MEP (EPA Method 1320). US EPA's Multiple Extraction Procedure (MEP) is based entirely on EP Tox. It was designed to simulate the leaching that a waste would undergo if it were exposed to repeated events of acid precipitation in an improperly designed landfill. It determines the maximum leachate concentration under acidic conditions for solid, liquid, or multiphase wastes. It is conducted entirely according to the EP Tox protocol, except that nine successive elution are performed on the same sample with a synthetic acid rain after initially conducting a distilled water elution.

MWEP. The Monofill Waste Extraction Procedure (MWEP) is a technical resource document (TRD) of US EPA. It is to be used by writers and reviewers of permit applications for hazardous waste land disposal facilities. TRDs provide information on technologies and evaluation techniques, but they are not regulations in themselves. This test is intended to derive reasonable leachate compositions for industrial wastes subjected to monofilling in properly engineered facilities. It gives an indication of which constituents are potentially leachable, as well as the expected relative delay in their release, and it determines the maximum release under mildly acidic conditions. It does not attempt to simulate field leaching. MWEP was previously called the Solid Waste Leaching Procedure (SWLP). The MWEP protocol involves a four-step sequential batch extraction at 18 hours per elution, at a liquid-to-solid ratio of 10:1. It

has been tested for a wide range of contaminants, except volatile organics. Although there is no provision for particle-size reduction, it has been observed that the rotary tumbler tends to crush many monolithic samples.

Graded Serial Batch. The Graded Serial Batch test was developed by the United States Army Corps of Engineers. It is a rapid and versatile research test that can be used to measure the leachability of selected wastes, the attenuation capacity of certain soils, and the effectiveness of various fixation processes for industrial wastes. It has been developed with the remediation of contaminated sites in mind, where more than one kind of waste and more than one kind of soil are present. The procedure involves repeated extractions of a waste, as well as sequential equilibrations of the waste extract with a soil. The same waste sample is extracted with increasing volumes of leachant, and the leachate is equilibrated in succession with the same three soil samples. This procedure can be repeated for any number of extractions.

ASTM D4793-88, SBE. The Sequential Batch Extraction (SBE) of ASTM is a modification of Method D3987 that allows for filtration and separation of the waste sample after the first distilled water elution, and subsequent extractions of the same waste, up to nine times.

WRU Leaching Test. The Waste Research Unit (WRU) at Harwell Laboratory in the United Kingdom has developed a serial batch method, called the repetitive shaker test, to provide a simple method of quantifying the initial leaching of a waste to help assess any limitations on landfill disposal that should be imposed. It expresses the liquid-to-solid ratio in terms of a bed volume, which is the volume of leachant required to just saturate a waste. The protocol calls for first performing an equilibrium shaker test, which is a single elution agitated extraction, with hourly sampling to determine steady-state conditions. A repetitive shaker test is then conducted, which involves five serial batch extractions with one bed volume of leachant and a sixth extraction with 10 bed volumes of leachant (to represent the average leaching over 6 to 15 bed volumes).

SLT Cascade Test. In 1981, the Netherlands recognized the need to develop standardized leaching tests for coal and municipal solid waste (MSW) incinerator ashes. In 1984, a series of standard leaching tests was published with the goal of closely approximating field leaching conditions, to improve the capability to predict the possible environmental effects of ash disposal. The Standard Leaching Test (SLT) in full involves evaluating the total composition of the residue, its behaviour in a column (the composition of the first leachate and the time to peak in a column test), its medium- and long-term leaching behaviour (cascade test), and its maximum leachability (a two-step extraction). The user decides among the various tests that are offered on a decision tree, based on the liquid-to-solid ratio that best approximates the time of leaching and amount of liquid that the waste would encounter in the field. The cascade test is a serial batch extraction with five elutions in which fresh acidified leachant is added at a liquid-to-solid ratio of 20:1, for a cumulative liquid-to-solid ratio of 100:1.

Table 3
A Compendium of Serial Batch Type Protocols

Test Name and Proponent	Status of Development	Sample Preparation	Leachant	Leaching Vessel	Agitation	Sample Mass	Liquid-to-solid Ratio	Contact Time	Number of Elutions	Leachate Separation
MEP(20) US EPA Method 1320	Standard test method (1986)	Same as EP Tox	Acetic acid Synthetic acid Distilled water	Same as EP Tox	Same as EP Tox	Same as EP Tox	Same as EP Tox	Same as EP Tox	10	Same as EP Tox
MWEP(21) US EPA	Technical resource document (1986)	Particle-size reduction to <9.5 mm or structural integrity	Distilled water Site water	Wide mouth sample bottle	Rotary tumbler	Unspecified	10:1	18 hours	4	Setting and filtration
Graded Serial Batch(22) US Army	Research method for waste and soil (1987)	–	Distilled water	Unspecified	Periodic gentle shaking (4/5 times daily)	300 g	2/3/6/12/24/48 96:1	Until steady-state conditions attained	>7	Vacuum filtration
SBE(23) D4793-88 ASTM	Standard method (proposed) (1988)	Drying Phase separation	Reagent water (Type II D1193)	2 L, wide mouth bottle	None	100 g	20:1	24 hours	10	0.45 µm membrane filter
WRU Leach Test(24) Harwell Laboratory United Kingdom	Standard method (1982)	Crushing Vacuum filtration	Distilled water Dilute acetic acid buffered (pH = 5)	50 mL, wide necked flask	Mechanical flask shaker	100 g	One bed volume (first five elutions) 10 bed volumes (more than six elutions)	2 to 80 hours Steady state	5	Vacuum filtration
SLT Cascade Test(25) SOSUV Netherlands	Standard research method for combustion residues (1984)	Crushing/sieving Dry	Distilled water Nitric acid (pH = 4.0)	1 L polyethylene bottle	Shake/roll	40 g	20:1	23 hours	5	Settling and 0.45 µm filtration

5. ASSESSING LEACHING TEST RESULTS. AN IN-DEPTH COMPARISON OF TWO PHILOSOPHIES – THE US EPA TCLP APPROACH AND HARWELL LABORATORY (WRU) APPROACH

The preceding section gave a generic summary of various methodologies that have been employed to assess the extent that contaminants may leach from a waste. Assessing the significance of the test result is the next important step in deciding how a waste is categorised and decisions made on its disposal.

In this section two quite different approaches for assessing leaching tests results are discussed. Although the assessment procedure is very much tied into the associated leaching protocol in each case, it is hoped that this exercise will emphasise some of the important concepts used in deciding if a waste is suitable for landfill disposal. It is hoped also that some of the strengths and weaknesses of the respective processes will be brought to light.

5.1 The United Kingdom Harwell Waste Research Unit (WRU) Approach

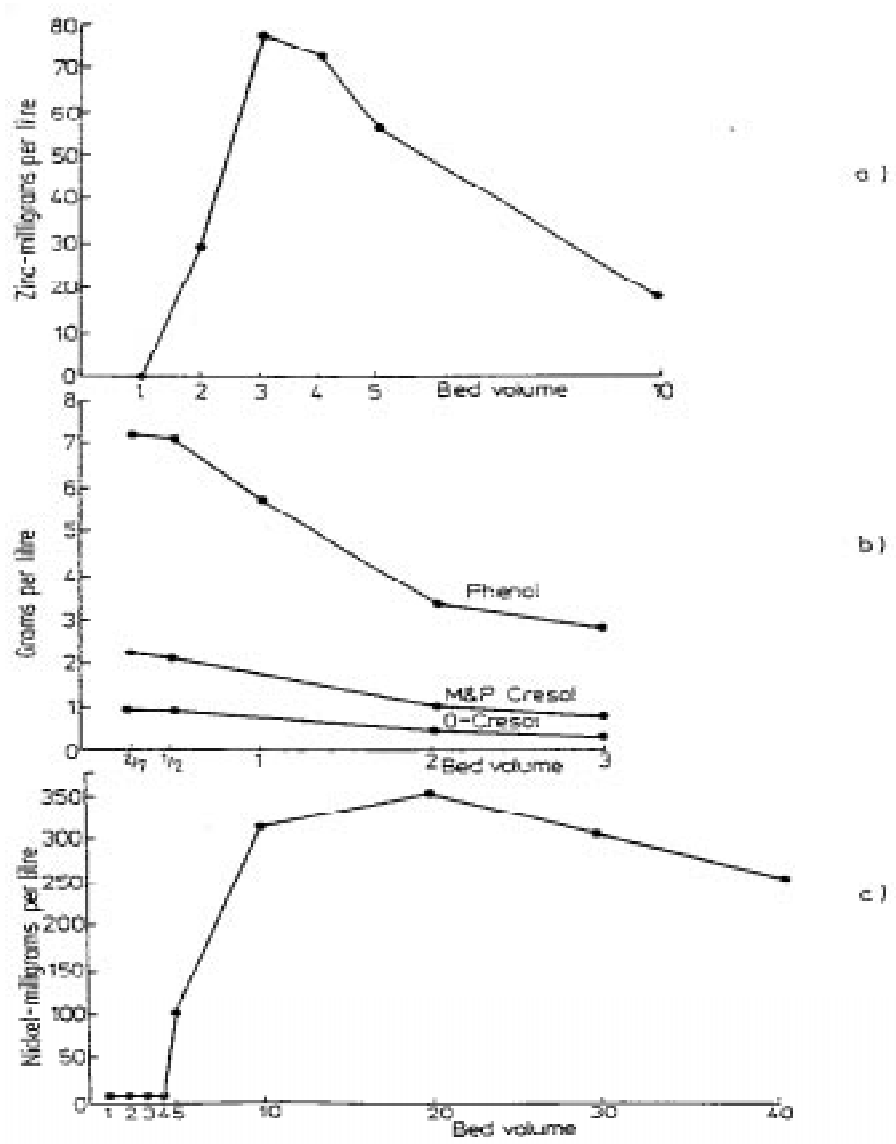
The Waste Research Unit of the Harwell Laboratory in England developed a dynamic leaching test based on the serial batch method (Young et al., 1982). The aim was to assess the tendency of wastes to leach over a period of time and to determine the significance of the extent of leaching in terms of the landfill's ability to attenuate pollutant species to non-hazardous levels.

The WRU method uses the concept of bed volumes of leaching fluid to try and determine the rate at which the waste will leach. A bed volume is that volume of leaching fluid which just covers the waste. In practical terms it probably represents between one and four years production of natural leachate for a landfill – depending on the waste's density, the rainfall and the design of the landfill. The waste is shaken for a sufficient period to establish equilibrium with the leaching fluid, after which the fluid is filtered. The waste solid is then returned to the leaching flask for subsequent leaching with a bed volume of fresh leaching solution, and the above procedure repeated. After analysis of the filtrate from each leaching cycle, it is possible to graph contaminant species concentration versus bed volume number to obtain a temporal pattern of waste leaching and to indicate the anticipated maximum leachate concentration. This can be derived from the point with the highest y co-ordinate. Examples of such graphs for specific waste contaminant species are shown in Figure 3.

The maximum leachate concentration is an important quantity to estimate. This indicates the level of attenuation that is required from the landfill to reduce the concentration to an appropriate level – perhaps the drinking water standard. The actual level of attenuation that can be derived from a landfill is a relatively unknown quantity. It will depend on a number of factors, not least being the depth of the landfill and how mature the refuse is. With this approach, knowledge is also required of the amount of contaminant already in the landfill. Sources include hazardous waste already deposited, the contaminant's presence in municipal waste and possibly its presence as natural geologic background material.

Harwell indicate that dilution with other leachate in the landfill could give rise to an attenuation of 10-300 times.

Figure 3
Contaminant Species Concentration versus Bed Number Graphs
Using the WRU Process (Young et al., 1982)



Lysimiter studies involving laboratory-scale columns of domestic refuse can be used to obtain an approximate estimation of attenuation levels that might be expected for a particular waste, but the method is slow and expensive. In practice it is assumed that most co-disposal sites will be able to attain attenuation factors of 100-200, or even greater (Young et al., 1982).

Whether the waste is adjudged suitable for co-disposal depends then on the landfill's estimated ability to attenuate the waste, and the target concentration in leachate leaving the site. Depending on the point of impact this may vary from a water quality standard protective of sensitive aquatic life, if the leachate were to enter surface water of designated value in a Resource Management plan, to the levels based on a water treatment plant's ability to assimilate and degrade the contaminant.

A criticism of the WRU approach is that attenuation factors used are at best crude estimates of the processes actually occurring in the landfill.

5.2 USEPA Toxicity Characteristic Leaching Procedure (TCLP)

5.2.1 TCLP Procedure

The protocol for conducting the TCLP leaching test (USEPA, 1990), an agitated extraction test type, is shown in flowchart form in Figure 4. The test is designed to simulate leaching that takes place in a sanitary landfill. The particular buffered acetic acid leachant employed (two types are nominated) is dependent on the alkalinity of the solid phase of the waste. A sample of waste is then extracted for 18 hours. The filtered extract from the TCLP is then analysed to determine if any of the thresholds for the 40 Toxicity Characteristic Constituents (listed in Table 4) have been exceeded (USEPA, 1990). If this is the case then the waste possesses the characteristic of toxicity and is considered to be a hazardous waste in the United States.

5.2.2 The Rationale for the Toxicity Characteristic and the Significance of the Regulatory Thresholds of Table 4

The regulatory thresholds shown in Table 4 are not levels at which it is considered safe to dispose of wastes in a sanitary landfill. Rather these levels indicate that there is a high degree of certainty that a waste which contains contaminant species at or above the nominated concentration, needs to be managed in a controlled manner. In the United States disposal in a sanitary landfill is regarded as uncontrolled disposal.

To place the Toxicity Characteristic in perspective it is useful to have a working knowledge of the USEPA's procedure for listing and designating hazardous wastes.
Under Section 3001 of the Resource Conservation and Recovery Act, EPA was charged with identifying those wastes which pose a hazard to human health and the environment if improperly managed. It further called on the USEPA to identify such wastes through development of lists of hazardous waste and through characteristics of hazardous wastes (Federal Register, 1986). (As discussed above the TCLP test allows a waste to be assigned the characteristic of "toxicity". Other characteristics are "flammability", "corrosivity" and "explosivity".) These two methods of identifying hazardous wastes employ fundamentally different approaches.

Figure 4
Flowchart for TCLP Protocol (USEPA, 1990)

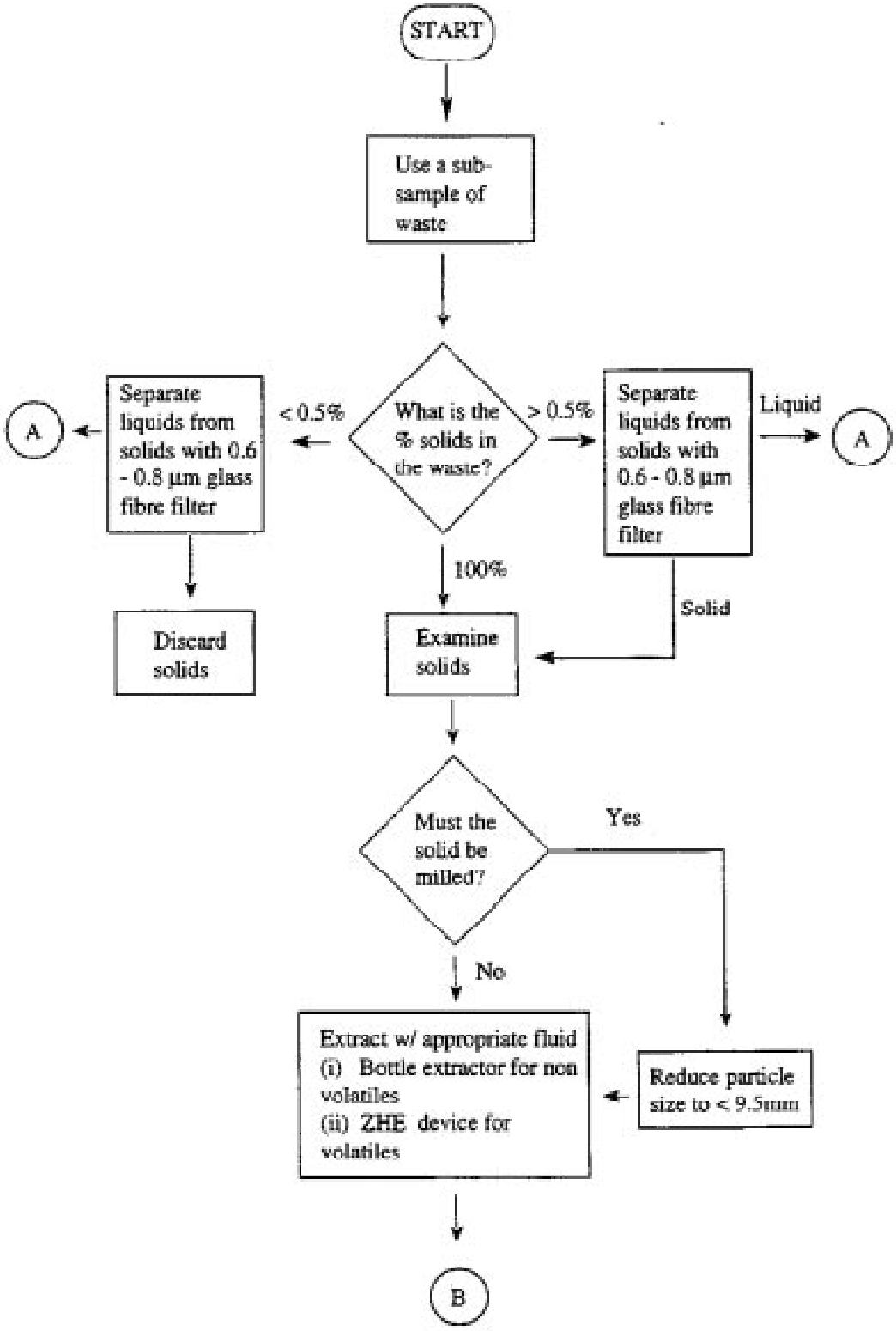


Figure 4 continued
Flowchart for TCLP Protocol (USEPA, 1990)

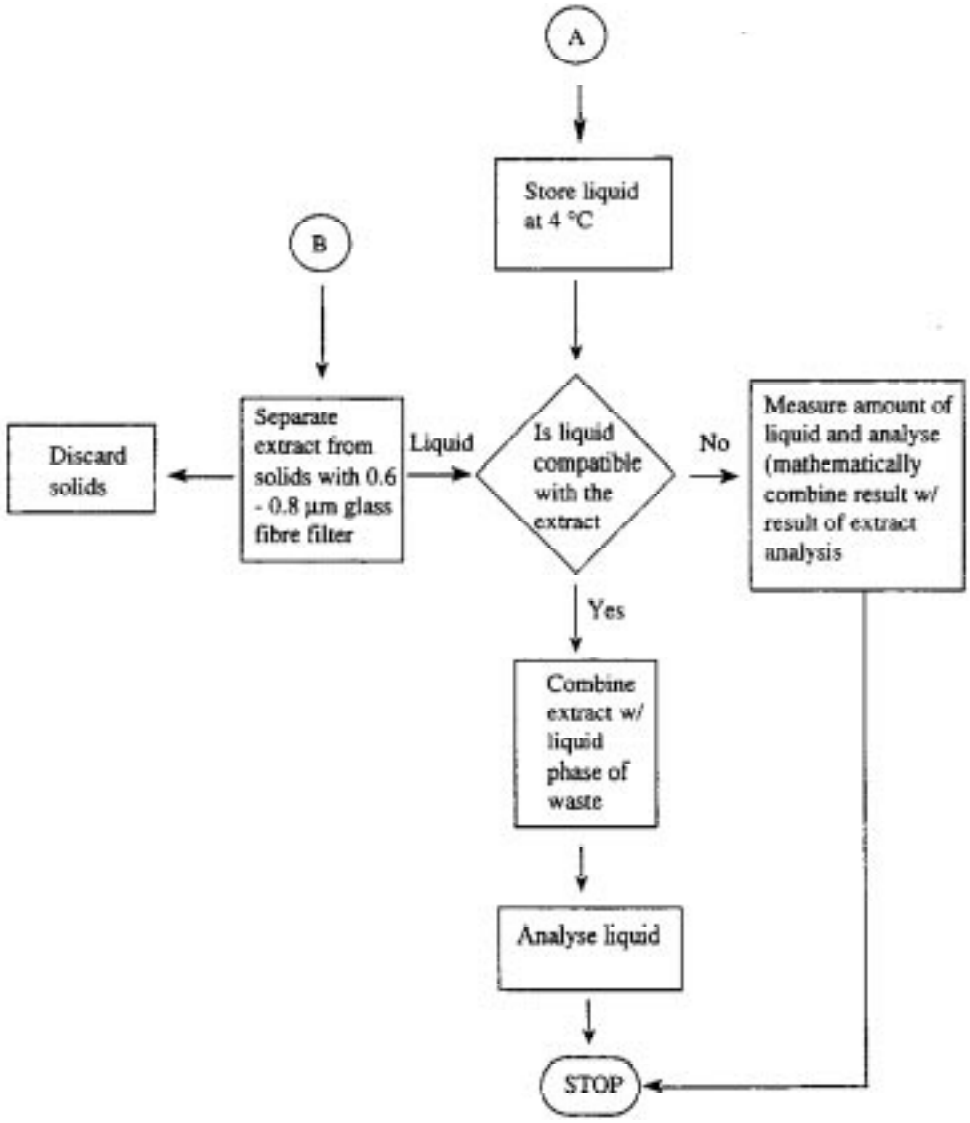


Table 4
Regulatory Thresholds Defining the Toxicity Characteristic

Contaminant	Regulatory Level (mg/L)
Arsenic	5.0
Barium	100.0
Benzene	0.5
Cadmium	1.0
Carbon tetrachloride	0.5
Chlordane	0.03
Chlorobenzene	100.0
Chloroform	6.0
Chromium	5.0
0-Cresol	200.0[1]
m-Cresol	200.0[1]
p-Cresol	200.0[1]
Cresol	200.0[1]
2,4-D	10.0
1,4-Dichlorobenzene	7.5
1,2-Dichloroethane	0.5
1,1-Dichloroethylene	0.7
2,4-Dinitrotoluene	0.13[2]
Endrin	0.02
Heptachlor (and its hydroxide)	0.008
Hexachlorobenzene	0.13[2]
Hexachlorobutadiene	0.5
Hexachloroethane	0.3
Lead	5.0
Lindane	0.4
Mercury	0.2
Methoxychlor	10.0
Methyl ethyl ketone	200.0
Nitrobenzene	2.0
Pentachlorophenol	100.0
Pyridine	5.0[2]
Selenium	1.0
Silver	5.0
Tetrachloroethylene	0.7
Toxaphene	0.5

To list a waste as hazardous, the USEPA conducts a detailed industry study, placing particular emphasis on the hazardous constituents contained in specific wastes from the industry being studied. This process involves literature reviews. engineering analyses, surveys and questionnaires, and site visits, including sampling and analysis

of wastes. As such the listing process may require from 1 to 3 years or more, depending on the complexity of the industry being investigated. On 15 November 1990 the USEPA issued a final rule designating as hazardous three categories of wastes from wood preserving operations that use chlorophenolic, creosote and/or inorganic (arsenical and chromium preservatives) (Federal Register, 1990). The three waste types with the respective designations F032, F034, and F035 are described in Table 5 which is sourced from the Code of Federal Regulations. An earlier listed waste, F027 covering discarded pentachlorophenol formulations, is also relevant to this study. The USEPA have deferred listing of waste F033, relating to wastes from use of sodium pentachlorophenate for surface protection of timber, because they have insufficient information to make a decision.

The process of identifying wastes as hazardous by reason of a characteristic is fundamentally different. Characteristics are those properties which, if exhibited by a waste, identify the waste as a hazardous waste. It is a generic process whereby the USEPA identifies properties that might be possessed by a waste which would cause the waste, if improperly managed, to cause harm to human health or the environment. The USEPA then determines a reasonable mechanism by which such harm might occur, develops a quantitative model to identify hazard levels, and whenever possible, tests methods for use in determining if a specific waste possesses hazardous levels of the property. Any solid waste which exhibits a defined characteristic is a hazardous waste. When treated so that it no longer exhibits the characteristic it is no longer subject to Resource Conservation and Recovery Act regulations governing hazardous wastes. It should however be noted that wastes which do not exhibit a characteristic are not necessarily non-hazardous (Federal Register, 1986). The defining concentrations (see Table 4) are established at levels at which there is a high degree of certainty that the waste requires careful management. In certain cases, on evaluation of specific industry wastes, the USEPA lists wastes as hazardous some wastes that may pass the characteristic test.

5.2.3 Levels of Contaminants in Wastes Acceptable for Land Disposal in the United States in Relation to the TCLP Extract

Part 268.41 of the Code of Federal Regulations (Section 40) identifies a list of restricted waste types. These wastes may be land disposed only if their TCLP extract does not exceed the value specified for a particular contaminant shown in the so-called "Table CCWE" of the Federal Regulations.

The relevant parts of "Table CCWE" for this study are shown in Table 6. Unfortunately it would appear that limiting concentrations have not been developed for the newly listed timber treatment wastes, F032–F035. For waste type F027 (discarded pentachlorophenol formulations) tetra to hexa chlorinated dioxin and furan species are restricted at 1 part per billion and pentachlorophenol at 10 parts per billion.

Table 5
Relevant United States Descriptions of Hazardous Wastes

Industry and EPA hazardous waste no.	Hazardous waste	Hazard code
F027	Discarded unused formulations containing tri-, tetra-, or pentachlorophenol or discarded unused formulations containing compounds derived from these chlorophenols. (This listing does not include formulations containing Hexachlorophene synthesised from prepurified 2,4,5-trichlorophenol as the sole component).	(H)
F028	Residues resulting from the incineration or thermal treatment of soil contaminated with EPA Hazardous Waste Nos. F020, F021, F022, F023, F026, and F027.	(T)
F032	Wastewaters, process residuals, preservative drippage, and spent formulations from wood preserving processes generated at plants that currently use or have previously used chlorophenolic formulations (except potentially cross-contaminated wastes that have had the F032 waste code deleted in accordance with §261.35 of this chapter and where the generator does not resume or initiate use of chlorophenolic formulations). This listing does not include K001 bottom sediment sludge from the treatment of wastewater from wood preserving processes that use creosote and/or pentachlorophenol. (NOTE: The listing of wastewaters that have not come into contact with process contaminants is stayed administratively. The listing for plants that have previously used chlorophenolic formulations is administratively stayed whenever these wastes are covered by the F034 or F035 listings. These stays will remain in effect until further administrative action is taken.)	(T)
F034	Wastewaters, process residuals, preservative drippage, and spent formulations from wood preserving process generated at plants that use creosote formulations. This listing does not include K001 bottom sediment sludge from the treatment of wastewater from wood preserving processes that use creosote and/or pentachlorophenol. (NOTE: The listing of wastewaters that have not come into contact with process contaminants is stayed administratively. the stay will remain in effect until further administrative action is taken.)	(T)
F035	Wastewaters, process residuals, preservative drippage, and spent formulations from wood preserving process generated at plants that use inorganic preservatives containing arsenic or chromium. This listing does not include K001 bottom sediment sludge from the treatment of wastewater from wood preserving processes that use creosote and/or pentachlorophenol. (NOTE: This listing of wastewaters that have not come into contact with process contaminants is stayed administratively.) The stay will remain in effect until further administrative action is taken.	(T)

Table 6
TCLP Criteria
(adapted from 268.41 Table CCWE – Constituent Concentrations in Wastes Extract, US Code of Federal Regulations)

Waste code	Regulated hazardous constituent	CAS no for regulated hazardous constituent	Non-wastewaters concentration (mg/l)
F020-F023 and F026-F028 dioxin containing wastes	HxCDD-All Hexachloro-dibenzo-p-dioxins		<1 ppb
	HxCDF-All Hexachloro-dibenzofurans		<1 ppb
	PeCDD-All Pentachloro-dibenzo-p-dioxins		<1 ppb
	PeCDF-All Pentachloro-dibenzofurans		<1 ppb
	TCDF-All Tetrachloro-dibenzo-p-dioxins		<1 ppb
	TCDF-All Tetrachloro-dibenzofurans		<1 ppb
	2,4,6-Trichlorophenol	88-06-2	<0.05 ppm
	2,3,4,6-Tetrachlorophenol	58-90-2	<0.05 ppm
	Pentachlorophenol	87-86-5	<0.01 ppm

Care must be taken in trying to apply the species concentrations listed in Table 6 to the New Zealand situation. Firstly the values specified apply to a system of waste management which is very much based on the treatment and destruction of hazardous wastes rather than storage or land disposal. The limits shown in Table 6 for waste type F027 are regarded as the minimum level at which regulatory and commercial laboratories can produce quantitative results with acceptable accuracy. It is necessary to question whether such stringent levels are applicable to the current situation in New Zealand where co-disposal of wastes of relatively low levels of contamination should be seen as very much an interim solution prior to the development of management plans. Such plans should establish the appropriate options for destruction and disposal of hazardous wastes. In the interim it will be necessary for New Zealand to develop its own leaching test extract contaminant concentration criteria based on a considered assessment of the likely impact that contaminated leachate emanating from a landfill will have on human health and the environment and the requirements of the Resource Management Act.

5.3 A COMPARISON OF THE HARWELL WRU AND USEPA'S TCLP LEACHING TEST PROCEDURE

Table 7 depicts some of the advantages and disadvantages of the two major leaching schemes described in the previous sections.

Table 7
A Comparison of the Advantages and Disadvantages Associated with the Harwell WRU and USEPA TCLP Leaching Test Procedures

Leaching Test	Advantages	Disadvantages
HWU	Flexibility – able to assess different co-disposal environments. Assesses an important component: the maximum leachate concentration. Identifies how the waste may leach with time.	Approach gives guidance only. Non-regulatory. Landfills ability to cope with waste is based on inexact assessment of attenuation factors. Testing is time consuming. Requires multiple analysis. Instructions imprecise. Current US disposal guidelines are probably unrealistically strict.
TCLP	Instructions for assessing waste clear cut. All landfill sites are treated equally. Highly developed leaching test method. Complete and unambiguous instructions. Widely used and accepted throughout the world.	TCLP test gives no information on max. anticipated leaching conc. or temporal behaviour expected.

6. CONCLUSIONS

A number of different types of leaching test are available which simulate the leaching that occurs when a co-disposed waste comes in contact with leachate from a sanitary landfill. It is important to be aware of the various stages of biochemical and microbiological activity that occur in a landfill's lifetime in order to appreciate the likely effect that the leachate will have on the waste and the desirable stage of refuse maturation required for successful waste deposition (see Section 2).

An overview of the different approaches that can be adopted in devising leaching tests was presented including some of the parameters that can be estimated using specific leaching test types (see Section 3).

The more significant leaching protocols that have been adopted by regulatory bodies throughout the world are listed and a brief summary given of the procedure adopted in each test (see Section 4).

An in-depth outline and comparison of two prominent leaching test approaches – the United Kingdom's Harwell Laboratory Waste Research Unit test and the United States Environmental Protection Agency's Toxicity Characteristic Leaching Procedure was made. On balance it would appear that greatest benefit for New Zealand would occur from the adoption of the USEPA approach, provided the contaminant concentrations at which decisions on disposal are made concur with the current interim disposal situation and the requirements of the Resource Management Act (see Section 5).

7. REFERENCES

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ATTACHMENT – THE TCLP APPROACH IN RELATION TO MONOLITHIC WASTES AND HIGHLY ALKALINE WASTES

1. Monolithic Wastes

The recommended waste leaching protocol – the USEPA TCLP – requires that waste material be cut or ground so that it can pass a 9.5 mm sieve. There is no provision for waste which is monolithic in nature, that is essentially composed of a single part as occurs when waste is encased in concrete or the waste is actually a functional object such as a telegraph pole.

The test protocol which the TCLP superseded, the so-called EP Toxicity test did in fact make provision for leach testing of monoliths. There was, however, an additional test parameter to be employed called the structural integrity test. This involved hitting the waste with a 13 ounce ball hammer from a height of 30 inches a total of 14 times. The test was then conducted on the remaining monolith and any fragments produced during the structural integrity procedure. There may be merit in restoring this test procedure for evaluating monolith leaching, provided the reasons for the USEPA removing its use in the TCLP test are considered unwarranted.

The reasons given by the USEPA in the Federal Register of Friday 13 June 1986 (p 21657) were:

1. The TCLP should be restricted to one testing device and one set of operating conditions.
2. An environmentally conservative approach should be adopted concerning the long-term environmental stability of solidified wastes. Stabilisation processes decrease leaching potential through reduction of surface area and thus the area of potential leachate contact. The USEPA believes that physical stabilisation alone is not enough to insure that components do not leach in significant quantities from wastes. There are two types of actions which may act to reduce the physical integrity of stabilised wastes. First the action of heavy landfill equipment will act to reduce the monolithic blocks to smaller pieces. Secondly, and more importantly, is the effect of natural weathering forces such as wet/dry and freeze/thaw cycles in breaking down the monolith.

2. Highly Alkaline Wastes

The TCLP utilises a more acidic leaching solution for wastes that are moderately to highly alkaline (such as metal-bearing wastes stabilised by addition of lime). In devising the TCLP it was of concern for the USEPA that the conventional pH 5 acetic acid buffered solution (giving an added acidity of 70 milliequivalents of acid per gram of waste) may not adequately mimic the leaching effect of leachate in a landfill because of the competing effect for acidity coming from the waste's alkaline component.

Data gathered at the USEPA's Boone County field site over a period of seven years indicated that the leachate generated by decomposing municipal waste contains approximately 0.14 equivalents of acidity per kilogram of dry refuse. Applying this data to the hypothetical co-disposal environment, the USEPA concluded that 1 gram of industrial waste could potentially be acted upon by 2 milliequivalents of acid. For a hundred gram sample (normal TCLP size) this translated to a total of 200 milliequivalents of acid – the amount that is provided with the alkaline waste leaching medium which has been adopted.

APPENDIX C
LANDFILL CLASSIFICATION LISTS

Note that the classification of these sites into the three classes has been undertaken on the basis of discussions with the engineer or person responsible for the relevant site. The classification therefore relies on verbal opinions and should not be assumed to be correct without further verification. Criteria for the three classification levels are given in Table 7.3 of this document.

New Zealand Landfills Likely to be Able to Conform to Class 1 Criteria

Landfill Name	Operating Authority
Greenmount	Northern Disposal Systems
Redvale	Waste Management NZ Limited
Whitford	Manukau District Council

New Zealand Landfills to be Able to Conform to Class 2 Criteria (including those likely to conform to higher grading)

Landfill Name	Operating Authority
Greenmount	Northern Disposal Systems
Redvale	Waste Management NZ Limited
Whitford	Manukau District Council
Porirua – Spicer Valley	Porirua City Council

New Zealand Landfills Likely to be Able to Confirm to Class 3 Criteria (including those likely to conform to higher grading)

Landfill Name	Operating Authority
Greenmount	Northern Disposal Systems
Redvale	Waste Management NZ Limited
Whitford	Manukau District Council
Porirua – Spicer Valley	Porirua City Council
Palmerston – Falcon Street	Waitaki District Council
Oamaru – Tamar St	Waitaki District Council
Timaru – Redruth	Timaru District Council
Nelson – York Valley	Nelson District Council
Napier-Hastings – Omaranui	Hastings District Council
Palmerston North – Awapuni	Palmerston North City Council

APPENDIX D
TOXICITY EQUIVALENCY FACTORS

In these guidelines, the term dioxins has been used to refer to a large group of chemically similar compounds more correctly known as polychlorinated dibenzodioxins and polychlorinated dibenzofurans. There are 75 chemically distinct isomers of the polychlorinated dibenzodioxins (PCDDs) and 135 chemically distinct isomers of the polychlorinated dibenzofurans (PCDFs). However, only a relatively small number of the PCDDs and PCDFs, i.e. those with chlorine atoms substituted at the 2, 3, 7, 8 positions, exhibit toxicity analagous to 2, 3, 7, 8 TCDD, which has been identified as the most toxic dioxin.

To assist in assessing the significance of dioxin contamination, Toxicity Equivalancy Factors (TEFs) that allow the concentration of each congener to be expressed in terms of the concentration of 2, 3, 7, 8 TCDD that would exhibit approximately the same toxicity, have been adopted. The combined toxicity of the PCDD/PCDF mixture may then expressed in terms of the equivalent concentration of 2, 3, 7, 8 TCDD, which then can be used for comparison with appropriate guidelines.

The estimation of the 'Toxic Equvalent' concentration of a dioxin mixture may be summarised as follows:

$$\text{Toxic Equivalent Concentration} = S\,(TEF_i) \times (\text{Concentration of Congener 'i'})$$

Where 'i' refers to each 2,3,7,8 substituted congener.

TEFs for dioxins have been developed by a range of organisations; however, for the purposes of these guidelines the TEFs developed by NATO (NATO, 1988) have been adopted. The TEFs developed by NATO are the most commonly used internationally, and are presented in Table 1.

Where a sample is analysed using the OCDD screening method, the toxic equivalent concentration shall be reported as the sum of contributions of the 2, 3, 7, 8 substituted hepta and octa chlorinated dibenzodioxins and dibenzo furans. No allowance shall be made for the contribution of dioxin and furan congeners not quantified using the OCDD screen.

Where a particular chlorinated dioxin or furan isomer is not detected, the concentration is assumed to equal one half of the detection limit, for the purpose of estimating the toxic equivalent concentration. The results of any dioxin analysis should be carefully reviewed to determine the contribution of "non-detects" to the overall toxic equivalent concentration. Where the "non-detects" contribute a significant proportion of the estimated toxic equivalent concentration, less reliance should be placed on the value reported. For example, "non-detects" may contribute approximately 90% of the toxic equivalent concentration, where dioxin concentrations in water approach 1 pg/L (THE).[1] Similarly, "non-detects" may

[1] Based on full congener analysis.

contribute in the order of 10 to 20% of the toxic equivalent concentration for sediment samples in the 1 to 5 ng/kg (TE) range (Gifford et al., 1993). The actual contribution of "non-detects" will depend on the concentration of dioxins and furans, the matrix and conditions within the laboratory.

Table 1
Toxicity Equivalency Factors (TEF) for Dioxins

Compound	TEF
Mono-, Di-, and Tri-CDDs	0
2, 3, 7, 8-TCDD	1
Other TCDDs	0
1, 2, 3, 7, 8-PeCDD	0.5
Other PeCDDs	0
2, 3, 7, 8 substituted-HxCDD	0.1
Other HxCDDs	0
1, 2, 3, 4, 6, 7, 8-HpCDD	0.01
1, 2, 3, 4, 6, 7, 9 HpCDDs	0
OCDD	0.001
Mono-, Di-, and Tri-CDFs	0
2, 3, 7, 8-TCDF	0.1
Other TCDFs	0
1, 2, 3, 7, 8-PeCDF	0.05
2, 3, 4, 7, 8-PeCDF	0.5
Other PeCDFs	0
2, 3, 7, 8 substituted-HxCDF	0.1
Other HxCDFs	0
2, 3, 7, 8 substituted-HpCDF	0.01
Other HpCDFs	0
OCDF	0.001

References:

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