GUIDELINES FOR THE SAFE APPLICATION OF BIOSOLIDS TO LAND IN NEW ZEALAND

AUGUST 2003

Ministry for the
Environment
Manatū Mō Te Taiao

nzwwa
New Zealand Water & Wastes Association

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FOREWORD

Protecting and enhancing the quality of our environment is essential for sustainable development. There are many matters where local government, industries and communities can establish and implement environmental improvement that will benefit us all. The land application of biosolids is one such example.

Biosolids have valuable fertilising and soil conditioning properties. They contain micronutrients and organic material and, worldwide, they are commonly applied to land as a means of restoring degraded soils or boosting the productivity of soils within areas of low natural fertility. Examples of beneficial use include the restoration of depleted cropping soils, maintenance or enhancement of soil fertility in forests, rehabilitation of mine tailings and quarry sites, landfill cover, golf courses, parks and gardens.

The application of biosolids to land can be controversial. Biosolids contain microbiological and chemical contaminants, and there are legitimate concerns about public health, environmental and economic risks.

Traditionally, sewage sludge has been regarded as a waste product, and most commonly managed by disposal to the ocean or to landfill. Ocean discharge is simply not an acceptable practice, whilst disposal to landfill is becoming increasingly expensive and the production of methane gas from sludge in landfills is inconsistent with our Kyoto Protocol commitments. The conversion of sewage sludge into biosolids and the controlled application of biosolids to land provide an opportunity to take advantage of the fertilising and soil conditioning properties of this resource whilst avoiding the disposal problem.

The key to beneficial use is risk management. In order to minimise the risks associated with the land application of biosolids, quality control and management practices are required. These guidelines, and the supporting technical manual recommend a framework for biosolids management in New Zealand for local government implementation.

The guidelines have been produced as a joint initiative of the wastewater industry, central and local government and other key stakeholders. A broad range of views has been canvassed during their development. But I note that these guidelines are just the starting point. My Ministry is looking at developing a national environmental standard for biosolids that will provide even more certainty for the wastewater industry, thereby encouraging further investment and development in this area.

Barry Carbon

Barry Carbon
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Ministry for the Environment

DEPARTMENTAL STATEMENT

The Ministries of Environment, Health, and Agriculture and Forestry were represented on the Steering Committee overseeing the development of the *Guidelines*, but the *Guidelines* are not published by these Ministries and are not government policy. However, the Ministries consider that the *Guidelines* improve and expand on the best practice set out in existing guidelines and, in supporting a nationally consistent approach to biosolids management, see value in their adoption and use in New Zealand.

The *Guidelines* have been developed to encourage producers, end users and regulators of biosolids, as well as local community, iwi and interest groups to adopt current best practice for the application of biosolids to land. The best available information has been used throughout, but it should be noted that their adoption cannot be taken as a guarantee of public health or food safety when biosolids are applied to land. Local authorities may wish to set more stringent requirements than those recommended here if this is appropriate to their local environment and community requirements, or indicated in regional policy objectives.

The present *Guidelines* supersede those parts of the Department of Health's *Public Health Guidelines for the Safe Use of Sewage Effluent and Sewage Sludge on Land* (1992) that govern the use of sewage sludge application to land. Those parts of the 1992 guidelines should no longer be used, and are withdrawn by the Ministry of Health.

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GUIDELINES FOR THE SAFE APPLICATION OF BIOSOLIDS TO LAND IN NEW ZEALAND

VOLUME 1: GUIDELINES

CONTENTS

OVERVIEW

These *Guidelines for the Safe Application of Biosolids to Land in New Zealand* contain information and recommendations to assist producers, dischargers and regulators (regional councils) to manage the discharge of biosolids to land in New Zealand. The recommendations contained in these *Guidelines* will only gain "force" if councils choose to incorporate them into their regional plans (e.g. by way of rules) or in resource consent conditions. They do not have any legal status on their own, and only provide guidance. Users of these *Guidelines* should note, however, that the Ministry for the Environment is proposing to develop *national environmental standards* for the application of biosolids to land, under the Resource Management Act. National environmental standards are regulations that have legal standing above regional plans unless the regional plans set more stringent requirements.

The aims of these *Guidelines* are to:

- Safeguard the life-supporting capacity of soils
- Promote the responsible use of biosolids
- Protect public health and the environment
- Identify the risks associated with biosolids use and promote best practice for minimising such risks
- Encourage local authorities to adopt a consistent approach to regulating the application of biosolids to land
- Create awareness within the community of the benefits and risks of biosolids use
- Minimise the risk to the economy.

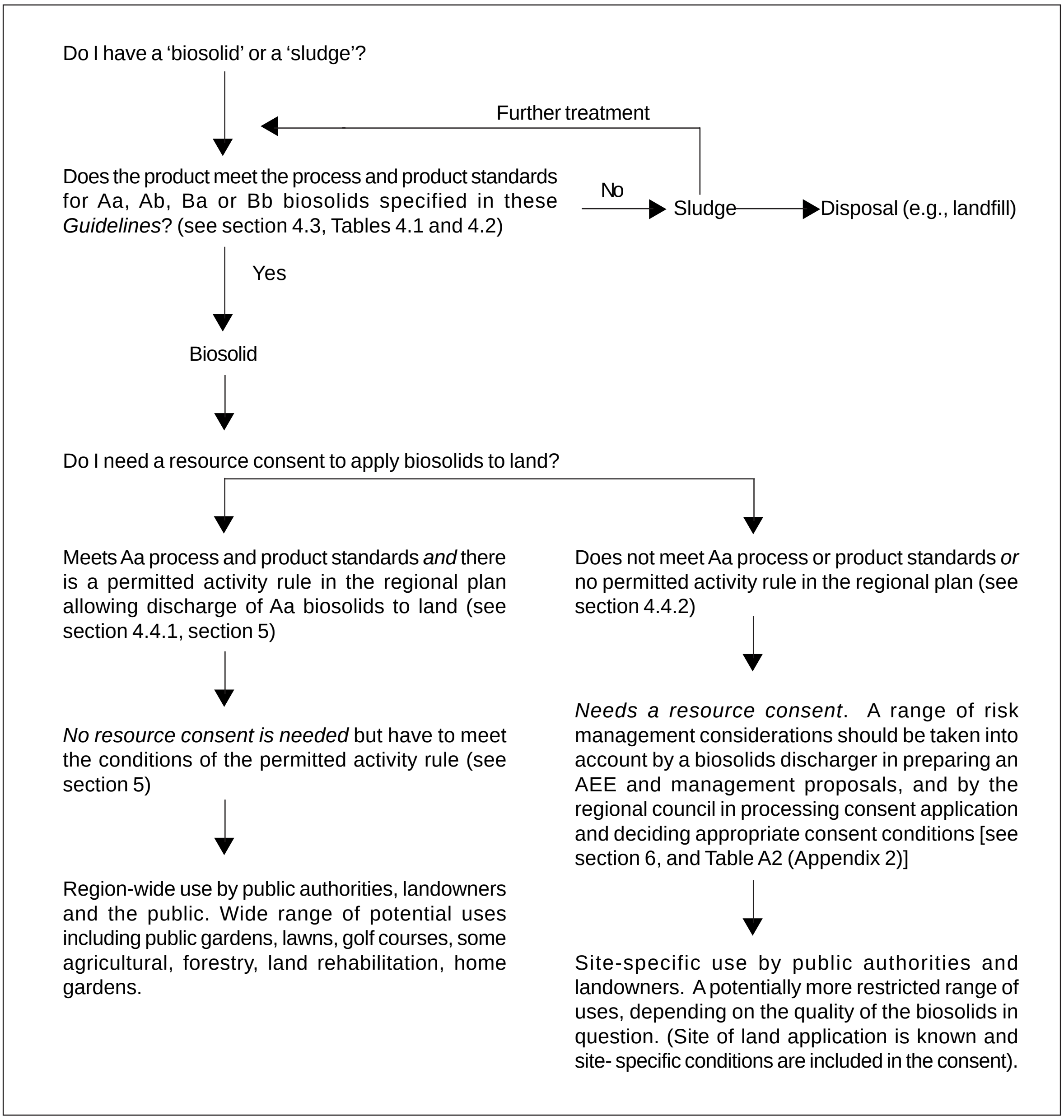
The Guidelines propose a grading system whereby biosolids are assigned a stabilisation (microbiological) grade 'A' or 'B', and a chemical contaminant grade 'a' or 'b' (see section 4). An 'A' grade biosolid is one in which pathogens and vector-attracting compounds, such as volatile solids, have been substantially reduced or removed by an "acceptable" pathogen reduction process. Grade 'B' biosolids have a lesser degree of stabilisation and will contain pathogens.

To achieve contaminant Grade 'a' the concentrations of *all* the contaminants (i.e. metals and organochlorine compounds) within the biosolids must be at, or below, specified limits. A biosolid is classified as Grade 'b' even if only one of the contaminants exceeds the limit specified for a Grade 'a' biosolid.

If a biosolid does not meet the process and product standards for Aa, Ab, Ba, or Bb biosolids, the biosolid should be considered a "sludge" rather than a biosolid and it should (therefore) be treated or blended with another substance in order to meet the biosolid product standard, or disposed of.

The *Guidelines* propose that the discharge of Aa biosolids to land be handled by way of a *permitted activity* rule in regional plans and that these biosolids carry a registered Biosolids Quality Mark (BQM) as a means of providing independent third party accreditation that the biosolids meet all the relevant process and product standards (see section 5). It is proposed that the discharge of Ab, Ba or Bb biosolids to land be treated as a *discretionary activity* requiring a resource consent.

The proposed framework for managing biosolids discharges to land is summarised below:



Do I have a ‘biosolid’ or a ‘sludge’?

Does the product meet the process and product standards for Aa, Ab, Ba or Bb biosolids specified in these *Guidelines*? (see section 4.3, Tables 4.1 and 4.2)

Do I need a resource consent to apply biosolids to land?

Meets Aa process and product standards *and* there is a permitted activity rule in the regional plan allowing discharge of Aa biosolids to land (see section 4.4.1, section 5)

No resource consent is needed but have to meet the conditions of the permitted activity rule (see section 5)

Region-wide use by public authorities, landowners and the public. Wide range of potential uses including public gardens, lawns, golf courses, some agricultural, forestry, land rehabilitation, home gardens.

Does not meet Aa process or product standards *or* no permitted activity rule in the regional plan (see section 4.4.2)

Needs a resource consent. A range of risk management considerations should be taken into account by a biosolids discharger in preparing an AEE and management proposals, and by the regional council in processing consent application and deciding appropriate consent conditions [see section 6, and Table A2 (Appendix 2)]

Site-specific use by public authorities and landowners. A potentially more restricted range of uses, depending on the quality of the biosolids in question. (Site of land application is known and site- specific conditions are included in the consent).

Section 6 of the *Guidelines* contains detailed guidance on the matters which dischargers of biosolids should consider when selecting a site for biosolids application and when preparing their consent applications, and which consent authorities should take into account when deciding whether or not to grant a consent and, if granted, the conditions which should be attached to a consent.

Recommended quality assurance procedures and management practices, including environmental management systems, record keeping, labelling, monitoring and sampling protocols, test methods, data reporting, and heath and safety precautions, are given in section 7 of the *Guidelines*.

1. INTRODUCTION

1.1 What are biosolids?

Biosolids are sewage sludges or sewage sludges mixed with other materials that have been treated and/or stabilised to the extent that they are able to be safely and beneficially applied to land. Biosolids have significant fertilising and soil conditioning properties as a result of the nutrients and organic materials they contain.

The term 'biosolids' does not therefore include untreated raw sewage sludges or sludges solely from industrial processes.[1] Neither does it include animal manures, or food processing and abattoir wastes. However, as shown in Figure 1.1, septic tank sludges may become biosolids depending on the degree of treatment they have received.

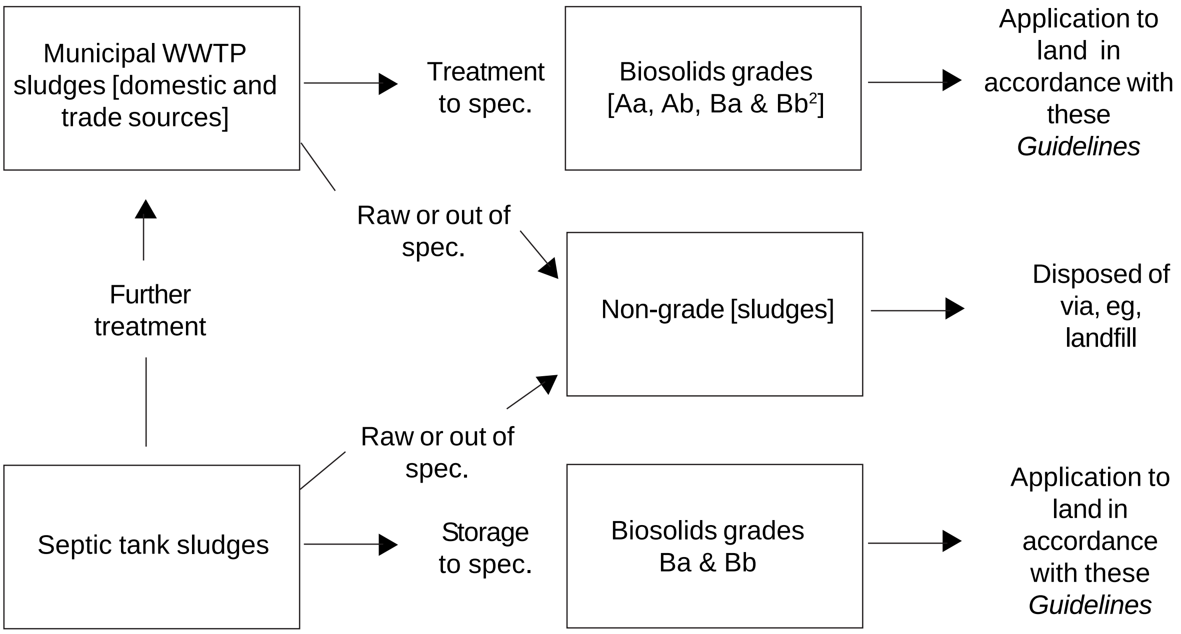


Figure 1.1: The relationship between sludges and biosolids

Sewage sludges or sewage sludge mixes that do not achieve the process or product standards required to attain the status of biosolids (as specified in these *Guidelines*) should be disposed of, for example by way of landfilling.

Septic tank sludges not meeting biosolids standards should be either subjected to further treatment (e.g., discharge to a sewage treatment plant, or storage) so that they do meet those standards, or disposed of (e.g., via landfilling).

Blended biosolid products (e.g., treated sewage sludge mixed with green waste, sand or pumice) are covered by these *Guidelines*. This *Guideline* does not cover radioactive waste; the National Radiation Laboratory should be consulted on all issues to do with radioactive wastes.

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[1] While biosolids contain material delivered from industrial inputs to sewers, such material is diluted by the organic material in domestic sewage inputs. Industrial sludges, on the other hand, may contain high concentrations of specific chemicals, little or no organic material, and the organic material this is present may be of a different nature to that contained in sewage. Consequently, any proposal to discharge industrial sludges to land should be treated on its merits.

[2] See section 4.3 for a discussion of the biosolids grading system.

1.2 Purpose of the *Guidelines*

This document aims to provide national guidelines for managing the application of biosolids to land, and to promote a more consistent approach to the management of biosolids throughout New Zealand.

The *Guidelines* have been produced by a joint committee established by the New Zealand Water and Wastes Association representing a range of interests including the wastewater industry, user groups, central and local government, and environmental groups. It is anticipated that the *Guidelines* will be used by territorial local authorities, biosolids manufacturers and distributors, biosolids users, regulatory agencies (regional and district councils, government agencies), environmental groups, iwi and the broader community.

The *Guidelines* are designed to provide a framework for biosolids management that enables the land application of biosolids in New Zealand in a way that maximises the benefits[3] and minimises the risk of adverse effects on human health, the environment and the economy. They also support beneficial use in line with the *New Zealand Waste Strategy* (Ministry for the Environment, 2002). More specifically, the *Guidelines* are aimed at:

- safeguarding the life-supporting capacity of soils
- promoting the responsible use of biosolids
- protecting public health and the environment
- identifying the risks associated with biosolids use and promoting best practice methods for minimising such risks
- encouraging regulatory authorities to adopt a consistent approach to regulating the application of biosolids to land
- creating an awareness within the community of the benefits and risks of biosolids use
- minimising risks to the economy.

The *Guidelines* are accompanied by a Technical Manual (Volume 2), which provides detailed information about how the limit values in this document were decided and how to implement some of its recommendations. Other useful background information is also included.

The *Guidelines* are intended to be a 'living document'. They are based on current knowledge about the use of biosolids in New Zealand and overseas, and they will be regularly reviewed in the light of future research findings and management experiences (see section 8).

1.3 Background

1.3.1 General

Biosolids contain a wide variety of macro-and micro-nutrients essential for plant growth, including nitrogen, phosphorus, calcium, magnesium, potassium, sodium, manganese, copper, zinc, molybdenum and boron. They also contain organic matter, which improves soil structure, water storage and microbial health.

However, some of the elements referred to above can cause damage to plant or animal health if taken up in excessive amounts. There are toxic, persistent and bioaccumulative heavy metals and trace organic compounds with the potential to accumulate in animals – and ultimately humans – via the food chain. In addition, biosolids can contain pathogenic micro-organisms (bacteria, viruses, helminths, protozoa and fungi), depending on the degree of treatment they have received. Consequently, there are both environmental and public health risks associated with the application of biosolids to land.

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[3] The application of biosolids to land is not the only beneficial use option. Other potential re-use options need to be considered.

These risks need to be managed by:

- improved source management (control over what goes down the pipe, in terms of both tradewaste and domestic inputs)
- biosolids manufacturing techniques (reducing levels of pathogens/contaminants)
- regulatory control over biosolids discharges to land
- land application techniques
- adequate monitoring and feedback systems.

In recent years new treatment methods and technologies have improved the quality of biosolids. There is also increased awareness of the value of biosolids products and greater demand for their use. Factors such as source control, through tradewaste management and increasingly stringent regulation, will produce further improvements in quality, thereby increasing the quantity of biosolids suitable for beneficial use.

1.3.2 Previous guidelines

The first guide to the disposal of sewage sludge in New Zealand was a circular issued by the Department of Health in 1973, entitled *Disposal of Sewage Effluent and Sewage Sludge on Land*.[4] In 1975 this circular was published as a guidelines document. In 1984 the Department issued a new memorandum, *Disposal of Sewage Sludge on Land*.[5]

The Department continued to monitor the development of guidelines and regulations around the world, and in 1990 initiated further research leading to the 1992 *Public Health Guidelines for the Safe Use of Sewage Effluent and Sewage Sludge on Land* (referred to hereafter as the DOH guidelines).

1.3.3 Current production and use of biosolids in New Zealand

In New Zealand approximately 250 public wastewater treatment plants (WWTPs) treat domestic sewage from about 80% of the 4 million population, plus tradewastes from industry. About 55% of the population is connected to 25 high-rate WWTPs, many of which have full secondary treatment. These plants tend to produce digested, dewatered sludge as a minimum. The majority of the remaining WWTPs are small waste stabilisation ponds (WSPs), which accumulate sludge for infrequent removal. Estimates of New Zealand municipal wastewater sludge quantities are given in Table 1.1 below:

Table 1.1: Estimated New Zealand wastewater sludge quantities

Source	Tonnes of dry solids per year
High-rate plant sludges	55,000
Waste stabilisation pond residues	22,000

Source: Walmsley, 2001

The figures in Table 1.1 indicate that there is the potential for some 77,000 tonnes of dry solids to be beneficially used: 55,000 tonnes a year are produced on a regular (weekly/monthly) basis, plus an average of 22,000 tonnes are periodically removed from WSPs every year. At present there are five significant beneficial use schemes for biosolids, at Christchurch, Nelson, Wellington, New Plymouth and Rotorua. Approximately half the biosolids are discharged to production forests and half to agricultural land. A small quantity is also sold through garden centres and other retail outlets. The combined throughput of these schemes comprises less than 15% of the total potential biosolids, or 20% of the biosolids produced on a regular basis.

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[4] Circular Memorandum No. 1973/132

[5] Circular Memorandum No. 1984/93

The potential scale of biosolids application to land in New Zealand can be illustrated as follows. If biosolids have an average 3% total nitrogen content, and were applied to land at 200 kg total nitrogen/ha/year, a total of 11,550 ha of land could benefit. This is less than 1% of the land currently in production forest or 0.1 % of the agricultural land, which means there is significant opportunity to use biosolids beneficially without trade and social conflicts over land use. These areas could have repeated applications. This is a significant advantage over other countries with higher population densities.

1.3.4 Global perspectives

Biosolids production and use

The European Community produces about 8 million tonnes dry weight of sludge a year. Of this, approximately 40% is applied to agricultural land, 6% to forest, and the rest is disposed of to landfills or incinerated.

The US produces about 5.5 million tonnes dry weight of sludge per year, of which about 40% is applied to land (67% of this to agricultural land, 12% to domestic gardens/lawns, 9% to public parks/gardens, 9% to reclamation and 3% to forests), and the rest is disposed of via landfilling (17%), incineration (22%) or other methods (21%).

Australia produces about 300,000 tonnes dry weight of sludge per year. New South Wales is leading Australia in the beneficial use of biosolids. Over 90% of Sydney's biosolids are being beneficially used, with about half going to agriculture (pastoral, cropping and forestry land) and 25% to the domestic market via composting.

The application of biosolids to land has not been without controversy. In parts of the US, for example, there have been consumer-led boycotts on the purchase of products grown in biosolids-amended soils, and some states have placed moratoria on the application of biosolids to land in response to public concerns or to scientific debate about the adequacy of regulatory standards. Some countries prohibit the application of biosolids to land, for a variety of reasons.

However, the general trend has been towards increased use of biosolids on land, driven by a number of factors:

- discharge of sludge to the sea has been banned in most countries
- the costs of landfilling sludge can be high – and are increasing
- there are pressures to prevent or minimise the disposal of organic material to landfills to reduce the generation of greenhouse gases
- there is a shortage of space for new landfills
- incineration can cause atmospheric pollution, and ash from incineration requires disposal
- there is greater awareness of the agronomic and economic benefits from applying biosolids to land.

The fertilising and soil-conditioning properties of biosolids are receiving increasing recognition in the context of attempts to rehabilitate degraded soils, particularly in areas that have been subject to intensive cropping. Biosolids are also being widely used for capping landfills and reclaiming contaminated land such as mine-tailing sites.

Accompanying (and largely as a result of) the trend towards increased use of biosolids there has been a trend, at least in developed countries, towards the reduction of contaminant levels in sewage sludge as a result of improved source control and tradewaste pre-treatment initiatives.

Regulating biosolids

Internationally, heavy metals have been the principal impetus behind regulations governing the land application of biosolids because of their potential toxicity and persistence in the soil. Some countries regulate pathogen content, and a few regulate organic contaminants.

Two quite different approaches have been adopted in the US and in Europe. The United States Environmental Protection Agency (USEPA) has adopted a risk assessment approach, which involves analysing risks to plants, animals, humans and soil organisms for exposure to contaminants for 14 different pathways for land applied biosolids (USEPA, 1995). These assessments provided the basis for the *Standards for the Use and Disposal of Sewage Sludge* (USEPA, 1993), commonly referred to as the Part 503 Rule. This rule establishes maximum permissible soil limits for various contaminants and recognises four classes of biosolids, including an exceptional quality or 'EQ' biosolid (with low contaminant concentrations and virtually no pathogens), which can be used by the public with the minimum of regulation. There has been some scientific debate over the adequacy of the metals risk assessment work underlying Rule 503 (e.g., McGrath et al., 1994), but the Rule 503 approach to pathogen and vector attraction reduction is widely accepted.

In contrast to the US approach, which is designed to facilitate the application of biosolids to land, most European countries have adopted the so called 'LOAEC approach', which involves setting soil limits at the *lowest observed adverse effects concentrations*, notwithstanding the fact that some of these observations derive from laboratory trials involving the dosing of soils with high bio-availability metal salts rather than from field trials involving land application of biosolids containing metals (McGrath et al., 1994; 1995). This approach has resulted in recommendations for very low (relative to the US) soil and biosolids limits in the CEC Directive[6] which controls the application of sewage sludge to agricultural land (Commission of the European Communities, 1986). The LOAEC approach is inherently precautionary.

In Australia, a national set of biosolids guidelines, *Guidelines for Sewage Systems: Biosolids Management*, are in preparation (Natural Resource Management Ministerial Council, 2003). New South Wales has produced *Environmental Management Guidelines: Use and Disposal of Biosolid Products* (New South Wales EPA, 1997), and other states have also produced guidelines. All Australian guidelines are based to varying degrees on the (precautionary) European approach, so the adopted limits for metal concentrations in soils and biosolids are considerably lower than in the USEPA's Rule 503.

In New Zealand the 1992 DOH guidelines (see section 1.3.2) were largely based on the 1986 CEC Directive. The soil limits recommended in the current *Guidelines* are based on a European-type LOAEC approach (see section 4.2) and are similar to those adopted in Australia.

1.4 Underlying principles

The principles underlying the approach adopted in these *Guidelines* are outlined below.

1.4.1 Biosolids can and should be used beneficially

Biosolids possess significant soil conditioning and fertilising properties. In fact the wide array of elements they contain that are essential for plant growth and the organic content of biosolids have led some authors to suggest that biosolids are a more complete fertiliser than most other proprietary fertilisers. However, the presence of a wide variety of chemicals in biosolids (albeit in trace amounts) with the potential for uptake by plants and animals, together with the potential presence of pathogens, means that biosolids cannot be treated like other fertilisers.

A fundamental premise of these *Guidelines* is that biosolids *can* be beneficially recycled to land, providing that both the process of manufacture and the process of applying the biosolids to land are subject to adequate management control, and providing the biosolids are applied at a rate that does not exceed the agronomic nitrogen needs of crops.[7]

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6 This Directive is currently (2003) being reviewed.

7 This is a conversative position because it does not take account of all the nitrogen loss mechanisms from the soil (see Section 6.3).

In other words, rather than viewing biosolids as waste products that should be disposed of (e.g., by way of landfilling, which has its own adverse environmental effects), they are viewed as a valuable resource.

1.4.2 Protection of public health

Biosolids, being derived from sewage, have the potential to contain pathogenic micro-organisms, including salmonella, campylobacter, giardia and entero-viruses. The risk of infection must be eliminated or reduced to an acceptable level by:

- employing appropriate pathogen-killing techniques during the biosolids production process
- taking precautions with the handling of biosolids
- the careful choice of end use
- temporary exclusion of people from areas where biosolids have been applied.

Biosolids also contain contaminants (e.g., metals, organic compounds) with the potential to enter the human food chain. The risk of this occurring can be eliminated or minimised by placing appropriate controls on the way biosolids are applied and the levels that contaminants are permitted to build up to in soils.

1.4.3 Consistency with legislation

National guidelines for biosolids management need to be consistent with statute law. In this respect the sustainable management purpose of the Resource Management Act (RMA) and its effects-based approach (see section 3.2) have been important considerations in the preparation of these *Guidelines.*

Section 8 of the RMA requires that, in achieving the purpose of the Act, all persons exercising functions and powers under it shall take into account the principles of the Treaty of Waitangi.

1.4.4 Consistency with central government policy

Although government policy directions may change over time, the position adopted here is that the *Guidelines* should not be inconsistent with current government policy.[8]

Government waste policy is detailed in *The New Zealand Waste Strategy* (Ministry for the Environment, 2002). This strategy was developed by the Ministry for the Environment and Local Government New Zealand in consultation with the waste sector. It takes a preventive approach, changing the emphasis from end-of-pipe management (which focuses on waste disposal) to a focus on waste prevention. To achieve this, the Waste Strategy promotes materials and resource efficiency at every stage of production and consumption. The beneficial use of biosolids fits within these principles.

The Waste Strategy contains specific practical targets for various wastewater streams. One of these targets is:

By December 2007, more than 95% of sewage sludge currently disposed to landfill will be composted, beneficially used or appropriately treated to minimise the production of methane and leachate.

Targets are also set for waste minimisation and improved tradewaste management. The Waste Strategy recognises that the beneficial use of biosolids provides an incentive for improved tradewaste management.

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[8] Provision has been made for regular review of these Guidelines (see Section 8).

1.4.5 The biosolids management framework needs to be flexible and enabling

The *Guidelines* provide a broad generic framework for managing biosolids. While endeavouring to promote national consistency in the way the application of biosolids to land is managed, they do not provide a list of rules to be applied in any situation. This is because it is important to take into account local circumstances and community aspirations in deciding what conditions should be applied to specific proposals.

1.4.6 Risk management

Risk management is the key to successfully managing biosolids application to land in New Zealand. The main way to reduce contaminant loads in biosolids is via cleaner production initiatives and/or improved tradewaste management; that is, by way of source control or waste minimisation. Both regulation (tradewaste bylaws) and education have a role to play.

Risks can be managed by:

- pathogen reduction/destruction during biosolids manufacture (e.g., via an appropriate heat/ time regime, or by pH control), and monitoring of this process
- controlling vector attraction (disease transmission, odour risk)
- diluting chemical contaminants via blending (e.g., soil, sand, wood waste, pumice mixes)
- storage requirements
- labelling requirements
- setting biosolids contaminant limits and monitoring these requirements
- establishing contaminant soil limits (maximum permissible levels in the receiving environment) and soil monitoring requirements
- soil testing prior to application
- control over the application rate (loading) of biosolids discharges to land
- control over end uses
- soil incorporation requirements
- public exclusion/stock withdrawal periods (pathogen risk)
- soil monitoring requirements
- soil pH adjustment
- requirements for environmental management systems (EMS)
- quality assurance requirements.

Most of these risk management techniques are amenable to regulation by way of the conditions on permitted activity rules and/or via the conditions on resource consents; guidance is provided in sections 5 and 6 of this document.

1.4.7 A precautionary approach

It is generally agreed that a precautionary approach needs to be adopted to managing the risks associated with biosolids use. This stance recognises that there are gaps in scientific knowledge about the long-term effects of applying biosolids to land and acknowledges the importance of healthy soils to the New Zealand economy and the future wellbeing of New Zealanders.

1.5 Approach

Chapter 2 of the *Guidelines* identifies and discusses the risks that need to be managed, including risks to:

- human health
- groundwater quality
- habitat/biodiversity values
- soil micro-organisms
- plant health
- animal health
- domestic and international trade.

The main way to control the risks arising from the presence of chemical contaminants (heavy metals, organic compounds) in biosolids is to establish a conservative *soil limit*, or maximum permissible level in the soil, for each contaminant of concern. While the location of discharge or 'end use' of biosolids will determine the exposure pathways, it is the concentration of contaminants in the soil that determines the *degree* of exposure of soil organisms, plants, animals or humans at a given site.

There is, however, scientific uncertainty about the levels of contaminants in soils that are 'safe' (various levels are reported in the literature). The evidence suggests that the level of a particular contaminant considered safe in one soil type may not be safe in others due to differences in physico-chemical factors such as soil organic carbon content and soil pH. Consequently, in accordance with the precautionary principle, a soil limit value at the lower end of the range within which effects have been reported has been selected for these *Guidelines* (see section 5 of the accompanying *Technical Manual*).

There are three ways to control the risks to animal or human health arising from the pathogen content of sewage sludges:

- the biosolids manufacturing process (including storage)
- the choice of end use
- stock or human 'exclusion periods' following the application of biosolids to land.

In these *Guidelines*, biosolids have been divided into two classes on the basis of microbiological and chemical quality (see section 4.4). The so-called 'unrestricted use' biosolids are those that have been treated or processed to the extent that they contain very low levels of pathogens and can be used by the general public without risk to human health. They also have low contaminant levels, so there is a very low risk of the soil limit being reached at a given site even in the unlikely event that biosolids were to be applied at that site for decades. The 'restricted use' biosolids are of lower quality in terms of either pathogen content, contaminant content, or both, and therefore need to be subject to tighter management control.

The discharge of biosolids to land is governed by the requirements of the RMA and regional councils are able to include rules in regional plans (see section 3.2.2). It is recommended that:

- The discharge of *unrestricted use biosolids* to land be controlled only by conditions on a permitted activity rule, in much the same way that the application of fertilisers is regulated in New Zealand (see sections 4.4.1 and 5). This would enable high- quality biosolids to be sold in garden centres, for example, and people would be able to apply them to their gardens without the need for a resource consent and without the need to monitor environmental effects. This approach is consistent with the aim of promoting the beneficial use of biosolids, and with the 'enabling' principle referred to previously. It also provides an incentive for local authorities to reduce contaminant levels in sewage sludges by way of cleaner production, education and/or improved tradewaste management.

- *Restricted use biosolids*, on the other hand, would only be able to be discharged to land if a resource consent is first obtained for a specific site (see sections 4.4.2 and 6). The resource consent process provides an opportunity for full public scrutiny of a land application proposal, and consent conditions allow appropriate site-specific risk management controls, including monitoring requirements, to be placed on the discharge.

In some countries restricted use biosolids are divided into sub-classes depending on their pathogen or contaminant content, and each sub-class is identified as being suitable for specified end uses. This is a risk management technique in the sense that low-quality biosolids can be directed towards the less sensitive end uses (e.g., where there is less potential for humans to be exposed to pathogens or no potential for contamination of the human food chain). This approach has not been adopted in these *Guidelines* because there are other options for controlling health risks and it is the *mass loading* of contaminants in the discharge (i.e., the concentration of contaminants in the biosolids x the rate of application) that will determine the speed at which the soil limits are approached, and the potential for adverse effects (see section 6.3). Putting the emphasis on mass loading rates rather than on biosolids quality tied to a specific end use is consistent with the ‘flexible and enabling’ principle referred to earlier.

Control over end use can still be used as a risk management tool (e.g., to avoid application to particularly sensitive sites or the application of low-quality biosolids to food-producing soils), but the extent to which this is necessary is best determined either by the biosolids producer (when evaluating beneficial use options) or by the consent authority after hearing community submissions and considering all information relevant to a specific proposal.[9]

As noted in section 1.4 above, in addition to establishing soil limits and possible constraints on loading rates and end uses, there are various other methods to reduce the contaminant levels in biosolids and/or the degree of exposure of soil organisms, plants, animals and humans to risk. These methods should be taken into account by biosolids producers when deciding their treatment/biosolids manufacturing systems, and in developing land application proposals (the lower the risk, the greater the likelihood of gaining consents). More particularly, they should be taken into account by regulators when deciding what conditions should be applied to discharge consents (see section 6).

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[9] Guidance on end-use considerations is contained in Section 6.2 of these Guidelines.

2. MANAGEMENT ISSUES

2.1 Introduction

Any proposal to manufacture biosolids and to apply them to land will need to address a number of management issues. Some of these relate to the actual or potential risks associated with the use of biosolids, and others to social and economic matters that need to be taken into account when planning or assessing a biosolids project.

This section identifies and briefly overviews the main risk management issues (the actual *means* of managing these risks are covered in section 6) and social considerations. Readers are referred to section 3 of *Volume 2: Technical Manual* for further details.

2.2 Risks to human health

2.2.1 Risks from pathogen content

Sewage sludge can contain a wide variety of pathogens, including bacteria, viruses, parasites (helminths/protozoa), and fungi.

The pathogens that are generally subject to regulation are salmonella, enteric viruses, helminth ova, and ocysts and oocysts of protozoa.

Escherichia coli (*E. coli)* is used as a general indicator for the presence of bacterial pathogens found in sewage sludges.

Salmonella spp. bacteria are subject to regulation because outbreaks of gastro-enteritis have in the past been relatively common and experience has shown that reducing salmonella to a low level, as with faecal coliforms, provides a high degree of assurance that other bacterial pathogens do not constitute a health risk. *Salmonella* spp. are so common in the environment that the proper control of salmonellosis is through food hygiene; sewage sludge properly used in agriculture is not involved in the transmission of human salmonella (Smith, 1996).

The high rate of campylobacteriosis disease in New Zealand and its prevalence in the environment make this bacterium of greater concern than *Salmonella* spp., and its measurement has been included in these *Guidelines* (see Table 4.1). There is potential for regrowth of these organisms in sewage sludge and biosolids in storage or after application to land.

There are numerous types of enteric viruses that occur in sewage sludge including Hepatitis A virus, Small Round Structured Viruses including Norwalk virus, and Adeno virus. Some enteric viruses, such as the Adeno virus, are more resistant to physical and chemical agents than others. The types and concentrations of viral agents in sewage sludge are dependent on the level of viral illness present in the associated community.

Helminths found in sewage sludge include nematodes (roundworms and hookworms), cestodes (tapeworms) and trematodes (flukes). Helminth ova are very resistant to the environmental factors that reduce the numbers of bacterial indicators or animal viruses in sludge. Because of their resistance, the presence/absence of viable helminth ova is used as a criterion for monitoring for the presence of helminths in sludge to be applied to land (USEPA, 1993).

Taenia saginata, the beef tapeworm, is a helminth of potential concern. It produces eggs that are resistant to environmental stresses. *T. saginata* has a cyst stage in cattle and a worm stage in humans. Although present in New Zealand, it appears not to be a significant problem for either human or animal health. Overseas, problems have generally only arisen where cattle have been grazed on pasture irrigated with untreated human sewage sludge (thereby becoming infected with the larval stage of the tapeworm) and then people have eaten the raw or undercooked beef, becoming infected with viable larvae. There is a very low incidence of *T. saginata* in New Zealand cattle, and people in New Zealand generally do not eat raw meat. It is concluded that the transmission risks associated with biosolids use are minimal.

Protozoa found in sewage include *Entamoeba histolytica* (amoebic dysentery) *Giardia entestinalis* (gastro-enteritis), *Cryptosporidium* (gastro-enteritis) and *Balantidum coli* (gastro-enteritis). These organisms generally occur as cysts, which are inactivated by heat and/or lime treatment.

Biosolids that have been treated to the point where they are deemed safe for use by the general public (see section 4.4.1) are essentially free of pathogens and there is a very low risk to people handling them.

However, biosolids that do not attain the unrestricted use, or Grade A stabilisation standards (see section 4.3.1), potentially contain pathogens at infectious levels, so they cannot be sold or given away direct to the public and the risks need to be managed by reducing the pathogen levels prior to land application (e.g., by storage), by soil incorporation, by restricting end uses to areas with low public exposure, or by imposing exclusion periods following land application.

In addition to the risk of infection arising from direct contact with biosolids that do not attain Grade A stabilisation, there is a potential food safety issue. The survival of some pathogens in the soil, most notably parasites and viruses, is not well understood. Because there is no final barrier (i.e., a kill step such as cooking) with crops that are eaten raw (e.g., salad crops or root crops such as carrots), biosolids not attaining the Grade A standard should not be applied directly to these crops without a further waiting period (see Table 6.2).

2.2.2 Risks from metal content

The main metals of concern, from a human health perspective, are cadmium, lead and mercury (Smith, 1996). The USEPA found that the direct ingestion of biosolids by children is the most critical pathway of exposure to cadmium, lead and mercury, with the greatest risk being for domestic gardens. (The USEPA limits were set accordingly, using conservative assumptions about intake rate).

Neither lead nor mercury are taken up to any extent by crops and consequently do not pose a risk through the dietary intake of plant foods.

Cadmium, on the other hand, can accumulate in crops. In practice it has been found that humans are protected from (sludge-related) cadmium toxicity because the high ratio of zinc to cadmium in sludge inhibits the uptake of cadmium in plants; plants take up zinc in preference to cadmium (Chaney and Ryan, 1994). The US National Research Council (1996) found that although levels of cadmium in crops grown in sludge-amended soils can be elevated, there are no documented cases of human or animal poisoning from this source.

Copper, nickel and zinc can be translocated to the edible parts of crops. However, humans, like animals, are protected by the so called ‘soil–plant barrier’ (before metals in crops reach levels potentially harmful to humans, growth is so severely stunted that the crop or vegetable is not harvestable). The US National Research Council (1996) noted that no adverse human acute or chronic toxicity effects had been reported resulting from the ingestion of food plants grown in soils amended by sludges.

The New Zealand Total Diet Survey 1997/98 (Ministry of Health, 2000) contains useful background information on the toxicology and dietary intake of various metals, including cadmium, lead and mercury.

2.2.3 Risks from organic contaminants

The organic contaminants generally considered when assessing risks from biosolids are the persistent organochlorine pesticides (such as DDT and dieldrin), polychlorinated biphenyls (PCBs) and dioxins. These organochlorine chemicals are also known as persistent organic pollutants (POPs).

Human exposure to POP chemicals is primarily via the food chain, especially from grazing animals, where the contaminants accumulate in body fat. An international treaty on POPs, the Stockholm Convention, aims to protect people’s health by reducing further exposures to these chemicals.

There would appear to be minimal risk to human health via dietary intake of organic contaminants from crops grown in biosolids-treated soils because there is little or no plant uptake, except perhaps in the peel of carrot (known to accumulate PCBs under some circumstances), which is generally removed during normal food production (Smith, 1996). Reviews of the possible environmental hazards from organic contaminants in biosolids repeatedly identify surface application of biosolids to grassland as the only potentially significant pathway of human exposure from the agricultural use of biosolids, because of the possibility of persistent lipophilic compounds (e.g., DDT, dieldrin) transferring to the tissue fat and milk of grazing livestock. The main risk arises from surface-applied biosolids adhering to grass and being directly ingested by grazing animals. Currently there would appear to be no reported examples where unacceptable concentrations of organic contaminants in foods are linked to the agricultural use of biosolids (Carrington et al., 1998).

Exposure can be managed by removing or minimising the soil-grazing animal exposure pathway; for example, by ensuring that biosolids are thoroughly incorporated into the soil in situations where animals are to be grazed.

The derivation of dioxins from pentachlorophenol reinforces the need to ensure that wastes such as sawdust and ash from treated timber are kept out of compost products (treated wood can also be a significant source of copper, chromium and arsenic).

Available data (which are limited) indicate that levels of POP chemicals are not unduly high in New Zealand biosolids. Pesticides and PCBs are commonly present below parts-per-million levels, and dioxins at low parts-per-trillion levels; see Ogilvie,1998. There is therefore a low risk of these substances translocating to livestock given the level at which they occur in biosolids and the low probability of extensive use of biosolids on pastoral land in New Zealand.

Setting conservative maximum residue limits of POP chemicals in biosolids that are to be land applied should ensure there are negligible risks to human health and the environment. Note, however, that there are other organic contaminants present in biosolids, about which we know very little, especially with regard to their fate once applied to land, their availability from the biosolids material, and any risk they may pose. These chemicals, which include brominated diphenyl ethers, alkyl phenols, alkylbenzene sulphonates and phthalates, are becoming a focus of increased regulatory consideration overseas.

Further information on this topic can be found in *Volume 2: Technical Manual*.

2.3 Environmental risks

2.3.1 Groundwater quality

The leaching of nutrients (nitrogen and phosphorus), metals or organic substances from biosolids into groundwater is an issue of potential concern.

In Europe concern over the potential effects on human health due to increased concentrations of nitrate in water supplies has resulted in regulatory action to control the application to agricultural land of all nitrogen (N) sources, including sewage sludge (Council of the European Communities, 1991).

Biosolids generally have low N content (1–6%) relative to nitrogenous fertilisers. Relative to raw sewage, the organic matter in biosolids (compost in particular) is highly stabilised, and even high rates of application pose little risk of nitrate leaching (Smith, 1996). Only a small proportion (approximately 10%) of the total N applied is available on an annual basis. The mineralisation of organic N in sewage sludge takes place quite slowly relative to N in other wastes (e.g., poultry litter, pig effluent).

The key to both minimising the risk of nitrate leaching and the efficient use of biosolids is to take into account the rate of mineralisation of the organic N in the biosolids and to match application rates as closely as possible to the agronomic nutrient needs of crops (see section 6.3).

Compared to N, phosphorus (P) is relatively immobile in soils and applying sewage sludge is unlikely to result in P leaching from the soil profile (Lindo et al., 1993). Phosphorus retention mechanisms in most New Zealand soils are believed to result in a low risk of P leaching (White and Sharpley, 1996).

Most studies have found very little movement of metals through the soil profile, with authors generally attributing this to the strong binding effect of organic and inorganic particles in the soil or soil/sludge mix. There are exceptions to this. Dowdy and Volk (1984), after reviewing the literature on heavy metal movement in soils, concluded that it was most likely to occur where heavy applications of sewage sludge are made to sandy, acidic, low organic matter soils receiving high rainfall or irrigation. McLaren and Smith (1996) cite an example of the downward migration of nickel and zinc in a sandy soil treated with sludge at intervals over a period of 25 years.

The POP chemicals (see section 2.2.3) in biosolids have low potential to contaminate groundwater because their concentrations are typically low, they are strongly absorbed onto soil particles, and they have low water solubility.

On the basis of an extensive literature review, Smith (1996) concluded that impacts of sludge contaminants on groundwater quality are very unlikely, and that all the available information indicates that the organic, inorganic and biological constituents of sludge are retained in the surface layers of the soil and exhibit only very limited downward movement through the soil profile.

2.3.2 Surface waters

There is a risk of biosolids being eroded from sloping land into watercourses if placed too close to them. This risk is relatively easily managed by commonsense measures such as not placing biosolids on steeply sloping land and/or near to streams, rivers or the edges of lakes.

2.3.3 Habitat and biodiversity values

As with any fertiliser or soil conditioner, it is inappropriate to apply biosolids to areas where there is a risk of altering important natural habitat values; for example, in native bush reserves, wetlands, or in habitats of rare or endangered species. The increased level of nutrients resulting from the applications of biosolids may affect native plant communities that have adapted to soils of low fertility.

2.3.4 Soil organisms and soil fertility

Biosolids are generally applied to land to enhance the fertility of soils. However, concerns have periodically been raised about the potential for sludge or biosolids to adversely affect soil micro-organisms and/or the long-term fertility of soils. (The proper functioning of the microbial biomass is essential to the intrinsic fertility of agricultural soils because of its role in mineralising nutrients from the soil organic matter to support plant growth).

Although the literature indicates it is unlikely that organic contaminants in sewage sludge impair the fertility of agricultural soils, there is scientific debate over the effects of heavy metals on soil micro-organisms and microbial transformations.

Brookes and McGrath (1984) reported that the microbial biomass was halved in sludge-treated soil at Woburn, in the UK, compared with the control plot. This effect was attributed to elevated concentrations of potentially toxic metals in the sludge-amended plots, with zinc (at 180 mg/kg) being identified as the most likely causative agent. Brookes et al. (1984) and Brookes et al. (1986) reported that nitrification processes, which are performed by specialised groups of free-living heterotrophic bacteria, were suppressed in sludge-amended soil at Woburn, relative to the control.

In 1993 the UK Government commissioned an independent scientific committee to review the implications for soil fertility of the UK rules on sewage sludge application to agricultural land (MAFF/DoE, 1993). The review concluded that the experimental evidence on the effects of heavy metals from sludge on soil micro-organisms was inconsistent and incomplete. It also recognised that there were difficulties in assessing the soil limits required for individual metals to protect soil microbial processes due to the interacting effects of sludge, soil and environmental factors.

There is ongoing research in the UK, USA and New Zealand (an ESR research programme with Lincoln University and Landcare involvement) on the effects of heavy metals from biosolids on soil micro-organisms. Until definitive findings are available, the best way to manage risks to soil organisms and/or long-term soil fertility is by setting conservative soil limits for contaminants, particularly for metals like zinc, copper, nickel and cadmium, which are known to inhibit microbial activity when present in soils at high concentrations (see section 4.2).

2.3.5 Plant health or crop yield

Biosolids are generally applied to soils to enhance plant health and/or crop yield. However, some of the constituents of biosolids can have the opposite effect if applied in excessive quantities.

Zinc, copper and nickel are the principal phyto-toxic metals in biosolids, but the risks can be minimised by setting an appropriately conservative soil limit. Arsenic, chromium, mercury and lead have very low bio-availability to plants. Cadmium is readily taken up by plants but does not appear to cause phyto-toxic effects.

The only real risk to plant health/yield from biosolids occurs when soil pHs are low (< 5.5) and/or when soils are sandy. Metals are generally more mobile in acid soils, and in sandy soils due to the relative lack of binding materials (clay, organic material).

There is, however, some doubt about the extent to which sandy soil represents a risk when applying biosolids. Biosolids add organic and inorganic material, which tends to bind metals. However, the application of raw, undigested sludge to sandy soils in New Plymouth has resulted in pH reduction, metal uptake by plants and metal leaching to the water table (Speir et al., submitted). Some countries impose tighter metal limits on sludge application to sandy soils.

Organic contaminants generally have a very low bio-availability to plants and have no phyto-toxic effects at the concentrations found in biosolids-treated soils.[10]

There is no evidence to suggest that plant diseases are transmitted in sewage sludge (Smith, 1996).

2.3.6 Animal health or production

Stock are potentially exposed to the contaminants in biosolids by way of the soil ⇨ plant ⇨ animal food chain, and via direct ingestion. In the case of copper, nickel and zinc uptake by plants (see above), livestock are protected by the soil–plant barrier (see section 2.2.2).

The direct ingestion of sludge and sludge-treated soil is widely considered to be the principal risk to grazing livestock from sludge application, with copper and cadmium identified as the main metals of concern.

Smith (1996) reviewed the literature on copper uptake by sheep in the UK and concluded that the risk of copper toxicity is actually much smaller than might be anticipated from simple calculations of total intake, and field trials indicate that copper toxicity to sheep under field conditions is very unlikely.

Cadmium and other metals can accumulate in the liver and kidney of animals fed on pasture or crops grown on sludge-amended soils, but there appears to be no evidence of adverse effects on animal health. An oft-cited example is that of Werribee Farm near Melbourne, where cattle have been grazed for many years on wastewater-irrigated pastures. Although cadmium and chromium levels are ‘elevated’ in kidneys and livers, the levels are within the range common to cattle in general (US National Research Council, 1996).

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[10] There has been incidents where specific herbicides have not been destroyed during composting of biosolids with greenwaste.

Both molybdenum and selenium are readily taken up by plants and there are documented cased of livestock suffering poisoning from these elements (US National Research Council, 1996). However, molybdenum and selenium are present only in small amounts in municipal sewage sludge.

There is no evidence that organic contaminants applied to soil in sludge are toxic to grazing animals (Smith, 1996). The principal concern here is for human dietary intake resulting from accumulation in animal tissues and milk (see section 2.2.3).

2.3.7 Odour generation

Odour generation from biosolids-production facilities, or from biosolids after they have been applied to land, is one of the more significant environmental issues in the US and Europe.

The risk of odour generation can be minimised by selecting an appropriate biosolids manufacturing process, quality control and installing biofilters (at production facilities), an appropriate choice of vector attraction reduction (VAR) method and land application method, and by soil incorporation (see sections 4.3.1, 6.10, 6.11).

2.4 Trade risks

There are potential risks to domestic and international trade (commodity markets) as a result of applying biosolids to food-producing land. These include the risk of consumer-led boycotts on the purchase of produce grown in or on biosolids-amended soils (domestic or overseas markets) and the risk of overseas governments placing a trade embargo on products containing chemical residues and diseases (such as taeniasis in export beef cattle).

There are a variety of ways of managing trade risks, including measures aimed at reducing the potential for residues in export produce, such as soil incorporation, soil pH control, stock exclusion periods, and non-application on leaf or salad crops.

For any proposal to apply biosolids to horticultural or agricultural land, the relevant sector marketing or industry group should be consulted before commencing application. Some sector groups have already established a position on biosolids.

2.5 Social considerations

2.5.1 Site selection

When selecting a suitable site for applying biosolids it is important to consider social as well as bio-physical issues. Projects involving the application of large quantities of biosolids to large areas of land have the potential to cause nuisances (e.g., odour, transport noise, road spillages) and/or disruption to local community activities (e.g., exclusion of access to recreation areas following biosolids application). There may also be adverse reactions based on the *perception* of public health risk.

2.5.2 Community involvement and public consultation

Because of the types of issues referred to above, it is important that the instigators of restricted use biosolids[11] projects involve the community from the outset, preferably at the stage of assessing the available options for land application sites. A public awareness and involvement programme may be warranted.

A high degree of public acceptance is essential for biosolids projects. There is generally good will towards the concept of beneficial use provided that procedures for managing the risks are in place and the community is kept well informed.

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[11] See Section 4.4.2 of these Guidelines for a definition of restricted use of biosolids.

The choice of consultation method can be critical, and regional council advice should be sought on this. The preparation of a formal consultation strategy, including distributing an information package and/or an issues and options paper (with invitation for feedback), is one way of facilitating community involvement.

When preparing a resource consent application under the RMA, it is advisable to invite community involvement in the scoping of the assessment of environmental effects (AEE) and to seek community or interest group comment on the draft of the AEE before it is finalised (see section 3.2.3).

2.5.3 Iwi consultation

Maori protocol generally favours recycling waste material to land rather than discharge to water. However, there are some locations or sites at which the application of wastes of human origin, including biosolids, may be considered unacceptable by iwi.

Iwi issues in relation to the application of biosolids to land could include, but may not be restricted to (Marawhenua Group, Ministry for the Environment; personal communication):

- potential for contamination of food sources
- proximity to sites of food preparation, harvesting and processing
- potential contamination of water bodies
- the need for potential mitigation measures (e.g., riparian planting)
- avoiding applying biosolids on, or in the vicinity of, wahi tapu (sacred sites)
- potential constraints on future land-uses as a consequence of biosolids applications (e.g., land subject to Treaty of Waitangi claims)
- monitoring requirements.

Consequently, it is recommended that iwi be consulted at the earliest stage in the development of a biosolids project and that appropriate constraints be placed on any discharge consents. Iwi should be included in public education programmes.

During the development of rules in plans, including rules governing the discharge of biosolids to land, it is a requirement under the RMA for regional councils to consult with iwi.

3. THE REGULATORY FRAMEWORK

3.1 Introduction

The primary legislation governing the application of biosolids to land in New Zealand is the Resource Management Act 1991 (RMA). Other legislation (e.g., the Agricultural Compounds and Veterinary Medicines Act, the Health Act, the Land Transport Act) may have a direct or indirect bearing on a given biosolids manufacturing or distribution project depending on the project.[12]

This section of the *Guidelines* outlines the relevant provisions of the key statutes relating to biosolids management in New Zealand, with particular emphasis on the provisions of the RMA. It also provides guidance to regional councils on the nature and content of the rules that may be applicable to the regulation of biosolids discharges to land in their region.

3.2 Resource Management Act 1991[13]

The discharge of contaminants (and hence biosolids) to land in New Zealand is controlled by regional councils under the provisions of the RMA.

The purpose of the RMA is to promote the sustainable management of natural and physical resources, which include land, water, plants and animals. ‘Sustainable management’ is defined in terms of sustaining the potential of natural and physical resources to meet the reasonably foreseeable needs of future generations; safeguarding the life-supporting capacity of water, soil and ecosystems; and avoiding, remedying or mitigating any adverse effects of activities on the environment.

The Act focuses on the *effects* of activities rather than the activities themselves. Effects are defined to include both positive and adverse effects, and any cumulative effect that arises over time or in combination with other effects.

Discharges to land are controlled by section 15 of the RMA. If any contaminant in a discharge may enter water, or if the contaminants are from industrial or trade premises, then the person responsible for the discharge must obtain a resource consent, or must act in accordance with a rule in a regional plan or with regulations promulgated by central government.

The contaminants in biosolids originate at wastewater treatment plants, which are defined in the RMA as industrial or trade premises. Some contaminants in biosolids *may* enter water. Therefore, unless the relevant regional plan contains a permitted activity rule allowing the discharge, a resource consent will be required.

3.2.1 Resource management objectives and policies

Regional policy statements contain objectives and policies that promote the integrated management of the natural and physical resources of the region.

The objectives and policies of regional policy statements and regional plans prepared under the RMA are important because they establish the local decision-making framework, and in determining consent applications consent authorities (in this case regional councils and the Environment Court) are required to have regard to relevant objectives or policies of the regional policy statement, regional plans or proposed plans (see section 3.2.3).

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[12] These Guidelines do not purport to contain definitive legal advice. If there is any doubt about legal issues surrounding a specific project, seek legal advice.

[13] This section paraphrases the requirements of the RMA; if the need arises, or if in doubt, readers should refer to the full text of the relevant sections of the Act.

The RMA requires that councils adopt all provisions in their policy statements and plans in consultation with the community. Objectives state the resource management outcomes that councils and their communities are endeavouring to achieve. Polices provide the direction for *how* the objective is to be achieved. For example, in achieving the sustainable management of a regions' soil and water resources, a policy could be adopted that promotes practices such as reusing and recycling materials.

Encouraging the treatment of sewage sludge to a sufficient quality that allows it to be used as a soil conditioner and fertiliser will reduce environmental effects at landfills, improve soil quality and, if properly managed, recognise Maori culture and traditions. This helps promote the sustainable management of natural resources in New Zealand, while working towards achieving the zero waste objective of many councils.

3.2.2 Regional rules

Rules are key components of resource management plans, because once plans are approved they have the force of regulations. Regional councils can include rules in regional plans declaring the discharge of (specified or unspecified) biosolids to be a permitted activity, a controlled activity, a discretionary activity, a non-complying activity or a prohibited activity.

If the discharge is a *permitted activity* it is allowed without the need for a resource consent, providing it complies in all respects with any conditions specified in the rule.

If the discharge is a *controlled activity* it needs a consent and it has to comply with any standards or terms specified in the plan. A consent application is assessed in accordance with the matters council has reserved control over in the plan, and consent cannot be declined provided the activity complies with the standards and terms specified in the rule.

If the discharge is a *discretionary activity* it is allowed only if a resource consent is obtained, and the consent authority can decline the consent application. The consent authority has full discretion in respect of the conditions it attaches to the consent.

3.2.3 The consent process

The process of applying for a resource consent, including the requirement for preparing an assessment of environmental effects (AEE), is set out in section 88 of the RMA. The Fourth Schedule of the Act contains a list of the matters that should be included in an AEE and those that should (at least) be considered when preparing an AEE. The definitions of 'environment' in section 2 and 'effect' in section 3 of the Act are important.

Although not explicitly required under the Act, it is good practice for the applicant to undertake a well-structured consultation process with neighbours, the wider community interest groups, iwi, etc. This helps to identify issues that need to be addressed in the AEE (see also section 2.5.2).

The pathway for processing a resource consent application is set out in sections 92–120 of the RMA.

The matters the consent authority is to consider when determining whether or not a consent should be granted, and the conditions that should apply, are set out in section 104 of the Act and include:

- the nature of the discharge and the sensitivity of the proposed receiving environment to adverse effects, and the applicant's reasons for making the proposed choice
- any actual and potential effects on the environment of allowing the activity
- any possible alternative methods of discharge, including discharges into any other receiving environment
- any relevant objectives, policies, rules or other provisions of a plan or proposed plan
- any relevant regulations
- any other matters the consent authority considers relevant.

3.3 Agricultural Compounds and Veterinary Medicines Act 1997

The Agricultural Compounds and Veterinary Medicines (ACVM) Act 1997,[14] administered by the New Zealand Food Safety Authority (a semi-autonomous body within the Ministry of Agriculture and Forestry), is narrowly focused on the application of substances (agricultural compounds) to agricultural land. The Act does not cover environmental effects or human health effects other than those in respect of food residues.

The purpose of the Act is to:

(a) Prevent or manage risks associated with the use of agricultural compounds, being:
  - *(i) risks to trade in primary produce, and*
  - *(ii) risks to animal welfare, and*
  - *(iii) risks to agricultural security*

(b) Ensure that the use of agricultural compounds does not result in breaches of domestic food residue standards

(c) Ensure the provision of sufficient consumer information about appropriate compounds.

The ACVM Act provides that no person may *sell* or *use* any agricultural compound within New Zealand unless that agricultural compound is a registered ‘trade name product’ or is exempt by regulations made under section 75 of the Act.

Biosolids fall within the definition of ‘agricultural compounds’ in the ACVM Act. Under current (2003) wording of the ACVM Act and regulations (regulation 9, schedule 5; and regulation 4, schedule 1) it would appear[15] that all biosolids defined in these *Guidelines* as unrestricted use biosolids (see section 4.4.1 and the definition in the Glossary) are exempt from registration as a fertiliser under the Act (provided prescribed conditions are met), but that restricted use biosolids (apart from composts), which will constitute the majority of biosolids, are not. The New Zealand Food Safety Authority (NZFSA) may amend the regulations such that all biosolids will be exempt from registration as a fertiliser under the ACVM Act.

It is important to note that even if a biosolid is exempt from registration, it is still subject to the jurisdiction of the ACVM Act, and the NZFSA could still get involved in biosolids management if an issue arises in connection with any of the risks referred to above. For example, section 28 of the Act provides for the Director-General of Agriculture to issue codes of practice for manufacturing, selling or using any agricultural compound, and section 75 for the making of regulations relating to the quality, manufacturing and use of agricultural compounds, irrespective of whether or not they are registered products.

All would-be dischargers of restricted use biosolids should seek early advice from the NZFSA via the Ministry of Agriculture and Forestry (Head Office, Wellington) as to their obligations and responsibilities under the ACVM Act and its regulations.

3.4 Hazardous Substances and New Organisms Act 1996

The Hazardous Substances and New Organisms (HSNO) Act provides comprehensive regulatory coverage of everything to do with hazardous substances management in New Zealand, including their import, manufacture, storage, transport, use and disposal. The HSNO Act, section 25, prohibits the import or manufacture of a hazardous substance other than in accordance with an approval under Part V of the Act.

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[14] The ACVM Act replaced the previous Fertiliser Acts 1960, 1982; the Stock Foods Act 1946; and the Animal Remedies Act 1967.

[15] There are ambiguities associated with the wording of the regulations.

The Act only covers substances that are hazardous. ‘Substances’ are defined in the Act to include:

(a) *any element, defined mixture of elements, compound or defined mixture of compounds, either naturally occurring or produced synthetically, or any mixture thereof … etc*

It is clear that biosolids or their constituents are substances.

‘Hazardous substances’ are defined as any substances that have one or more of six specified ‘intrinsic’ properties, including toxicity and eco-toxicity (the HSNO Act is not concerned with pathogen content or public health risk). The Hazardous Substances (Minimum Degree of Hazard) Regulations 2001 prescribe threshold levels for each property. If the concentration of a substance, or one of its constituents, exceeds a specified threshold or trigger level (for, say, toxicity or eco-toxicity) in the regulations, it is classified as toxic (class 6) or eco-toxic (class 9), or both. Within these classes there are sub-classes for the *types* of hazard (e.g., hazardous to soil organisms) and categories for the *degree* of hazard presented by a substance. In general, the higher the degree of hazard, the stricter the level of control imposed under regulations.

Biosolids or their constituents are unlikely to trigger any of the hazardous property criteria. If this is the case, there would be no need for biosolids producers to obtain approval for use of their products under the HSNO Act.

However, hazardous substances (such as heavy metals) may still be subject to certain HSNO Act controls, if in other products they trigger any of the thresholds. In particular, the Act provides for the setting of national (receiving environment) standards for hazardous substances by the Environmental Risk Management Authority (ERMA); the latter include *environment exposure limits* (EELs), being the allowable limit of hazardous substances in air, soils and water; and *tolerable exposure limits* (TELs) for allowable exposure to humans from the different exposure routes in the environment.[16]

3.5 Health Act 1956

The Health Act 1956 defines the functions and powers of the medical officer of health. The medical officer of health’s primary concern with respect to biosolids management is to ensure that the manufacture, distribution or use of biosolids does not create a ‘nuisance’ (in terms of the definition in section 29 of the Health Act), and that these activities are not injurious to health. The Health Act is currently under review.

The medical officer of health currently does not have any statutory approvals in relation to the discharge of biosolids to land.

3.6 Health and Safety in Employment Act 1992

This Act requires employers to protect the health and safety of employees in the workplace. It therefore applies to all biosolids producers, distributors, dischargers, and the owners of land to which biosolids are applied.

The Occupational Safety and Health Division (OSH) of the Department of Labour administers the Act. OSH recognises adherence with certain codes of practice as satisfying the requirements of the Act.

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[16] ERMA has the mandate to develop EELs and TELs as a result of Part V applications and as part of the transfer process. It may be necessary to review the soil limits adopted in these Guidelines as EELs and TELs are developed.

3.7 Land Transport Act 1993

The Land Transport Safety Authority regulates the transport of goods on public roads in New Zealand under the provisions of the Land Transport Act 1993. The Authority makes and administers land transport rules, including a rule governing the transport of dangerous goods (Land Transport Rule No. 45001: Dangerous Goods 1999).

The Dangerous Goods Rule contains some basic safety requirements relating to secure containment, proper labelling, etc., but the Rule also requires compliance with *NZS 5433:1999 Transportation of Dangerous Goods on Land*, which contains detailed technical standards for labelling, loading, placarding, segregation, containerisation and documentation.

The Rule covers ‘infectious material’. Under the biosolids grading scheme in these *Guidelines* (see section 4.3), Grade A biosolids (being essentially free of pathogens) would not be classified as ‘infectious’ whereas Grade B biosolids (being biosolids with a potentially significant pathogen content) would be. That is, the transport of Grade B biosolids would be subject to the Dangerous Goods Rule and NZS 5433.

Under the Rule, the onus is on the owner of the infectious material to advise the cartage contractor of the relevant regulatory requirements, and the contractor must be licensed to carry infectious goods. The cartage contractor or employee needs to have a correctly detailed Material Safety Data Sheet[17] (MSDS), plus his/her licence must be endorsed for Class D and the vehicle must be correctly registered. The penalty for non-compliance can be severe, with both the owner of the material and the cartage contractor being potentially liable for fines in excess of $200,000. The rule is enforced by the police and local authority dangerous goods inspectors.

Territorial local authorities and Transit NZ have the capacity to make bylaws controlling the transport of hazardous substances on roads.

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[17] An example MSDS is included in Appendix IV.

4. NATIONAL BIOSOLIDS GRADING AND DISCHARGE CONTROL SYSTEM

4.1 Introduction

This section of the *Guidelines* sets out a recommended approach to the management of biosolids discharges to land in New Zealand. It relies for its implementation on the provisions of the Resource Management Act 1991 (see section 3.2).

The principal risk management instrument is the establishment of a conservative *soil limit* for each of the key contaminants found in biosolids.

A simple biosolids grading system has been developed to differentiate:

- biosolids that are capable of being beneficially applied to land from sludges (whose degree of stabilisation and/or contaminant content is such that they should not be applied to land)
- biosolids that are of sufficiently high quality they can be applied to land with the minimum of regulation (unrestricted use biosolids) from biosolids that are of lower quality and whose application to land therefore needs to be subject to site-specific management controls (restricted use biosolids).

The system is based on a biosolids':

- stabilisation grade (which depends on pathogen content, and whether or not an approved pathogen reduction procedure and an approved vector attraction reduction method have been implemented)
- contaminant grade (which depends on the levels of metals and organochlorine contaminants).

Depending on its stabilisation and contaminant grades, a biosolid is classified as an unrestricted use biosolid or a restricted use biosolid. If a product fails to meet either or both of the minimum stabilisation or contaminant grade requirements, it is not a biosolid and should be referred to as a sludge.

4.2 Soil limits

The second column of Table 4.2 lists soil limits for metals and organic compounds. Only one soil limit is given; it is not land-use dependent and is recommended for all soils in New Zealand where biosolids are applied.[18] A discussion of the rationale behind the soil limit numbers is contained in section 5 of *Volume 2: Technical Manual*.

The proposed limits are considered to be inherently conservative because they have been selected from the lower end of the range within which effects have been reported and are based on the assumption that all of the contaminants are bio-available, which they are not (studies have shown that metals and organic contaminants can have a very low level of bio-availability, depending on the retention properties of soils).

The soil limit should generally not be exceeded for any of the parameters listed. This means that restricted use biosolids should not normally be applied at locations where the levels of the elements or compounds in question are at or above the soil limits. However, there may be some situations (e.g., a high natural background level of a metal in a soil and/or a demonstrable low bio-availability of that

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[18] This approach is different to other soil criteria established in New Zealand for environmental contaminants, which recommend different soil limits for different land uses (e.g. agriculture vs residential soils). The approach adopted in these Guidelines, which is based on the most sensitive land use, is viewed as a risk management measure to cover the possibility of future land-use changes.

metal) where a case can be made to a consent authority for relaxation of the soil limit for a specific contaminant or contaminants. Technical justification for such action would need to be provided in the assessment of environmental effects.

4.3 Biosolids grading system

The biosolids grading system is made up of two parts. The first part, which is denoted by a capital 'A' or 'B', represents the stabilisation grade. The second part, denoted by a lower case 'a' or 'b', represents the chemical contaminant grade.

4.3.1 Stabilisation grade

Biosolids are stabilised by treating them in a way that reduces or eliminates the potential for putrefaction and which, as a result, reduces pathogens, vector attraction and the potential for offensive odours. Further discussion on the rationale behind the stabilisation requirements is contained in section 6 of *Volume 2: Technical Manual*.

Vectors such as flies, mosquitoes, birds and rodents are potential carriers of disease. They can transmit pathogens to humans and other hosts physically through contact, or biologically by playing a specific role in the lifecycle of the pathogen. Reduction of the attractiveness of biosolids to vectors reduces their potential for transmitting diseases. This process is known as vector attraction reduction (VAR). VAR can be achieved by either:

- subjecting the biosolids to specific physico-chemical processes or conditions, or
- preventing access to the biosolids by vectors, usually by incorporating the biosolids into soil.

A high-quality biosolid 'A' is one in which pathogens and vector-attracting compounds, such as volatile solids, have been substantially reduced or removed. Some pathogen reduction processes are also effective at reducing vector attraction. Table 4.1 summarises the process and product standard (pathogen content) requirements to achieve each stabilisation grade. To achieve stabilisation Grade A the biosolids must have an accredited quality assurance system *and* meet at least one of the accepted pathogen-reduction processes, *plus* one of the accepted vector attraction reduction methods and all of the pathogen standards.

To achieve stabilisation Grade B, the biosolids need to meet a lesser degree of stabilisation *plus* one of the VAR requirements for Grade A; no pathogen reduction processes or product standards are applicable. (Note: for Grade B, specified storage periods and/or access restrictions are recommended controls; see Table 6.2).

4.3.2 Contaminant grade

Table 4.2 gives the maximum contaminant concentrations that are applicable to each of the biosolid contaminant grades (refer to section 7, *Volume 2: Technical Manual* for justification of these concentration limits).

To achieve contaminant Grade 'a' the concentration of all the contaminants within the biosolids must be at or below the level indicated in column 3 of Table 4.2. Two sets of values are given for this column, allowing for a 10-year transition period to achieve contaminant Grade 'a' status (see section 4.3.3).

A biosolid has to be classified as chemical contaminant Grade b even if only one contaminant falls within that category. If any contaminant concentration is above the limit given for Grade b (column 4), then the product is to be considered a sludge rather than a biosolid and the sludge has to be treated, or blended with another substance, in order to become a biosolid, or disposed of.

Table 4.1: Stabilisations requirements[1]

Grade	Acceptable pathogen reduction processes	Acceptable vector attraction reduction methods	Product pathogen standard [2]
Grade A	Accredited quality assurance ***plus*** one pathogen reduction process from the 3 options below:- **1. Time–temperature processes** **(a) $\geq$ 7% DS** Within the relationship $t = \frac{131,700,000}{10^{0.14T}}$; t = days, T = °C where $T \geq 50^{\circ}$C and $t \geq 15$ seconds, or **(b) < 7% DS** Within the relationship $t = \frac{50,070,000}{10^{0.14T}}$; t = days, T = °C where $T \geq 50^{\circ}$C and $t \geq 30$ minutes, or **(c) Composting**[3] (i) In-vessel: $T \geq 55^{\circ}$C for $\geq$ 3 days, or (ii) Windrow: $T \geq 55^{\circ}$C for $\geq$ 15 days with a minimum of 5 turnings during this period[4]. **2. High pH – high temperature process** pH > 12 (measured at 25°C) for $\geq$ 72 hours and maintain T > 52°C for 12 consecutive hours within the 72 hours, all from the same chemical application, and drying to > 50% DS afterwards. **3. Other processes** Demonstration by agreed comprehensive process and product monitoring that the Grade A pathogen levels can be consistently met.	Accredited quality assurance ***plus*** at least one VAR/odour method from the list below: 1. mass of volatile solids in biosolids shall be reduced by a minimum of 38%[5]; or 2. biosolids $\geq$ 90% DS if heat dried at T > 80 °C; or 3. $T \geq 40^{\circ}$C for $\geq$ 14 days and $T_{ave} \geq 45^{\circ}$C; or 4. SOUR @ 20°C $\leq$ 1.5 g/m³ for liquid sludges from aerobic processes; or 5. pH $\geq$ 12 @ 25°C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours; or 6. soil incorporation.	Accredited quality assurance ***plus both:*** **1. Verification sampling** showing that:· ■ *E. coli* < 100 MPN/g ■ *Campylobacter* < 1/25g ■ *Salmonella* < 1/25g ■ enteric viruses < 1 PFU/4g ■ helminth ova < 1/4g and **2. Routine sampling** showing that: *E. coli* < 100 MPN/g
Grade B	Verified quality assurance ***plus*** storage/exclusion period (see Table 6.2).	Verified quality assurance ***plus*** one of the VAR requirements from Grade A.	Not applicable

[1] *Table 4.1 is derived from USEPA Rule 503 requirements and the NSW EPA stabilisation grading.*

[2] *See section 7.5 for product monitoring requirements and Table 7.1 for sampling frequencies. In the event that one of the samples fails to meet the product verification standards specified, all of the pathogen tests for that sample must be repeated. 100% compliance must be achieved in order to meet the stabilisation grade standard. (In the case of biosolids manufacturing facilities in existence prior to the publication of these* Guidelines, *it is acceptable to use data up to 12 months old for the purposes of product verification. Older data cannot be used).*

[3] *All compost must have 30 days maturation pre-use.*

[4] *5 x 3 days at $T \geq 55$°C plus time periods to reach 55°C after each turning.*

[5] *Based on representative samples before and after the reduction process (see section 7).*

Table 4.2: Soil limits and biosolids classification by contaminant levels

Parameter	**Soil limit or ceiling concentrations[1]** **(mg/kg dry weight)**	**Biosolids limits[2]**		
		Grade a max. concentration[3] (mg/kg dry weight)		**Grade b max. concentration[3] (mg/kg dry weight)**
Metals		***Until 31/12/12***	***After 31/12/12***	
Arsenic	20	20	20	30
Cadmium	1	3	1	10
Chromium	600	600	600	1500
Copper	100	300	100	1250
Lead	300	300	300	300
Mercury	1	2	1	7.5
Nickel	60	60	60	135
Zinc	300	600	300	1500
Organics				
DDT/DDD/DDE	0.5	0.5	0.5	0.5
Aldrin	0.02	0.02	0.02	0.2
Dieldrin	0.02	0.05	0.02	0.2
Chlordane	0.02	0.02	0.02	0.2
Heptachlor & Heptachlor epoxide	0.02	0.02	0.02	0.2
Hexachlorobenzene (HCB)	0.02	0.02	0.02	0.2
Hexachlorocyclohexane (Lindane)	0.02	0.02	0.02	0.2
Benzene hexachloride (BHC)	0.02	0.02	0.02	0.2
Total polychlorinated biphenyls (PCBs)	0.1	0.2	0.2	0.2
Total dioxin TEQ[4]	0.00001	0.00003	0.00003	0.00005

[1] *Refer to section 6.11 and Appendix III for advice on compliance.*

[2] *See section 7.5 for product monitoring requirements, and Table 7.2 for sampling frequencies. In the case of biosolids manufacturing facilities in existence prior to the publication of these* Guidelines *it is acceptable to use data up to 12 months old for the purposes of* product verification*. For the purposes of determining compliance at the 95 percentile for* routine sampling*, the age of the data set shall be no more than 2 years for metals and organochlorines and no more than 5 years for dioxins. This avoids the scenario of old data masking upward trends in contaminant concentration.*

[3] *Compliance against the biosolid limit values are to be 95 percentile from sampling results, with a maximum of 20% above the limits.*

[4] Note: 'dioxin' means the seventeen 2,3,7,8-chlorinated congeners of the polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans. The residue limit is expressed as 2,3,7,8-tetrachlorodibenzo-p-dioxin toxic equivalents (TEQ) using the toxic equivalent factors (TEF) assigned by the World Health Organisation at a 1997 WHO consultation (Van den Berg et al, 1998). The TEQ is calculated by multiplying the concentration of each 2,3,7,8-chlorinated congener measured in an analysis by its appropriate TEF (as assigned by the WHO), and adding the result from these calculations together. If any 2,3,7,8-chlorinated congener is not detected in the analysis, that dioxin must be assumed to be present and an amount equal to one half of the analytical detection limit must be used in the TEQ calculation.

4.3.3 Reason for a transitional period

Table 4.2 has two sets of contaminant limits for Grade 'a' biosolids. The first set allows some contaminants to be present in biosolids in concentrations greater than the soil limits for a 10-year transitional period. After 31 December 2012 no contaminants in Grade 'a' biosolids may exceed the soil limits (although the toxic equivalents for dioxin may be greater than the allowable soil maximum). The transitional figures have been developed in the context of these *Guidelines;* they are not to be used for any other purpose than the ones they were intended for.

The transitional period – with higher limits for cadmium, copper, mercury, zinc and dieldrin – has been proposed to give wastewater treatment plant operators time to set up and implement programmes for cleaner waste streams entering the plant, and to develop sludge treatment facilities. According to the limited information available about contaminant concentrations in New Zealand sewage sludge (Ogilvie 1998), few plants could comply with the more stringent Grade 'a' contaminant limits for biosolids without a transitional period.

It is emphasised that the transition period has been included to provide time for producers to implement the waste management measures necessary to achieve the tighter standards that will apply after 31 December 2012. For example, few treatment plants could currently achieve a standard of 1 mg/kg mercury in their sludges, but some may currently achieve a 2 mg/kg standard; the transitionary period will enable biosolids producers to put in place a programme of action targeting point source discharges of mercury (e.g. dental clinics) aimed at meeting the 1 mg/kg standard by 2013. It is proposed that BQM accreditation of unrestricted use biosolids which relies on attainment of the transitionary standards be issued *subject* to biosolids producers demonstrating that they have a clear policy and plan of action aimed at meeting the post-transition standards (see section 5.1).

International data suggest that a high proportion of copper and zinc in sewage sludge comes from the domestic waste stream. Most of the copper is from plumbing (pipes and hot water cylinders). Some of the zinc is from plumbing (galvanised pipes and brass fittings), but a substantial proportion comes from toiletries, cosmetics, cleaning products and faeces. Mercury is thought to derive largely from the premises of dental practitioners, but some also derives from certain medical and pharmaceutical products. Cadmium in sewage sludge is generally associated with industrial sources. The source of dieldrin is unknown, although considerable amounts have been used for insect control, both in agriculture and domestically (e.g., borer control). Lowering the levels of these contaminants in the incoming sewage is difficult, and some wastewater treatment plant operators will need time to develop ways to do this.

Allowing these five contaminants to be present in biosolids in higher concentrations than those allowed in soil will not cause biosolids-amended soil to approach the soil contaminant limits in the short term, provided biosolids are applied in accordance with the model rule (see section 5 of these *Guidelines*). There are two reasons for this. First, condition 2 of the model rule limits biosolids applications to a nitrogen (N) loading rate of no more than 200 kg N per hectare per year (averaged over three years). With an N proportion generally in the range of 1–6%, this means the highest application rate would be 40 tonnes per hectare (assuming 1% N), or 4 kg per square metre. If biosolids with a cadmium concentration of 3 mg/kg are applied every second year for 10 years, the soil cadmium concentrations will only increase by 0.3 mg/kg.[19]

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[19] 4 kg biosolids/m^2 gives 4 x 3 mg Cd/m^2. If the soil bulk density is assumed to be 1.0, 12 mg/m^2 = 12 mg/200kg (if biosolids incorporated to 20 cm; i.e., 1 m^2 x 0.2 m = 0.2 m^3 = 0.2 tonnes = 200 kg). This means that Cd in the soil after one application = 12/200 mg/kg = 0.06 mg/kg. If applied 5 times (10 years), total Cd in soil would be 0.3 mg/kg.

A second safeguard is that the transitional figures are not to be used for topsoil replacement products, as this will cause the soil limits to be exceeded.

4.3.4 Use of blending to achieve contaminant grades

The blending of sewage sludge with other substances (e.g., pumice, sand, sub-soils, bark, sawdust, green waste) before, during or after treatment is an acceptable way of diluting contaminant levels in order to attain either the ‘b’ or ‘a’ biosolid contaminant grades.

The potential for adverse environmental effects is governed by the quality of the product. Cleaner production and improved tradewaste management (i.e., waste minimisation) will still be needed if biosolids producers are to consistently meet contaminant grade standards and/or avoid the need to obtain large quantities of (costly) blending materials.

The use of blending to get rid of other contaminated materials is not an acceptable practice, and the chemical nature or ‘contaminant content’ of blended materials should be declared by biosolids producers. Control can be exercised over the composition of blended materials via the quality assurance system (see section 7), and in the case of restricted use biosolids a regional council can impose conditions relating to the type or composition of the blending materials used.

4.4 Biosolids classification

Biosolids are classified according to their stabilisation and contaminant grades as follows.

- Unrestricted use biosolids: Aa
- Restricted use biosolids: Ab, Ba, Bb.

4.4.1 Unrestricted use biosolids (Aa)

Only biosolids that have achieved both an A grade for stabilisation and an ‘a’ contaminant grade are classified as unrestricted use biosolids.

Biosolids that have demonstrably achieved these standards are considered to be of sufficiently high quality that they can be safely handled by the public and applied to land without risk of significant adverse effects. That is, control over their discharge to land can be exercised by way of a *permitted activity rule* in a regional resource management plan in much the same way that fertiliser application is currently regulated.[20]

It is important to note that the term ‘unrestricted use’ is largely a misnomer, a shorthand way of indicating that no consent is required. Under a permitted activity rule of the type envisaged (section 5), the application of Aa biosolids to land would still be regulated. The discharge would be subject to meeting a number of conditions designed to minimise any risks to public health and environmental values.

From a public policy perspective there are two good reasons for having a permitted activity rule for high-quality biosolids:

- sewerage authorities have an incentive to improve their tradewaste management practices and source controls to achieve an ‘a’ grade contaminant standard (consistent with waste minimisation and recycling objectives)
- the biosolids can be distributed or sold in retail outlets like other fertilisers or soil conditioners (individual users, or ‘dischargers’, do not need to obtain a resource consent, and the range of potentially beneficial uses is increased).

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[20] This is consistent with the approach adopted in both Australia and the US. National guideliens and rules in these countries recognise ‘unrestricted use’ categories of biosolids, which, having met the specified product and process standards, can generally be applied as freely as any other fertiliser or soil amendment to any type of land. They can be sold or given away in bags or other containers, or used in bulk.

4.4.2 Restricted use biosolids (Ab, Ba, Bb)

These *Guidelines* classify all biosolids graded Ab, Ba, or Bb as restricted use biosolids. These grades should be applied to land only with site-specific controls imposed in accordance with a resource consent.

If a council classifies the discharge of these grades as a discretionary activity, it can treat each proposal on its merits, applying conditions that are relevant to the particular grade and its proposed receiving environment (Section 6). The council can decline the application if it is not satisfied that the effects of the discharge can be effectively controlled.

4.4.3 Sludges

Sewage sludges that do not achieve the standards necessary to be classified as either unrestricted use biosolids or restricted use biosolids either require further treatment to achieve such standards or should be disposed of via, for example landfilling (see section 1.1 of these *Guidelines*).

5. REGULATION OF GRADE Aa BIOSOLIDS DISCHARGES

The following model is advanced as a basis from which a regional council can develop a permitted activity rule, under the Resource Management Act 1991, for the discharge of unrestricted use biosolids (Grade Aa) to land should it consider it appropriate to include such a rule in its discharge to land plan (Table 5.1).

Table 5.1: Model Permitted Activity Rule

Discharge of Unrestricted Use (Aa) Biosolids to Land

The discharge of biosolids to land is a permitted activity provided:

1 *The biosolids carry the registered Biosolids Quality Mark (BQM)*[1] *accreditation; and*

2 *Biosolids application rates do not exceed a three-year average of 200 kg total N/ha/year; and*

3 *The soil pH where the biosolids are discharged is not less than pH 5.5; and*

4 *The discharge is:*

- *(a) more than 20 metres from any river, lake, wetland or artificial watercourse or from the coastal marine area; and*
- *(b) not onto land where there is a likelihood of the biosolids entering water; and*

5 There is no discharge to any of the following (sensitive) areas: wahi tapu, archaeological sites, habitats of rare or endangered species; and

6 There is no discharge onto grazed land.[2]

<u>Explanation</u>

Biosolids are defined in Section __ of this Plan.

The purpose of the Rule is to enable biosolids with low contaminant concentrations and which have been treated to the extent that they are essentially pathogen free to be discharged to land without the need for individual dischargers to obtain a resource consent.

The biosolids in question are the 'unrestricted use' or Aa category of biosolids recognised in the Guidelines for the Safe Application of Biosolids to Land in New Zealand (the New Zealand Guidelines). Unrestricted use biosolids are able to be sold or given away to the general public by biosolids producers.

Condition 1 requires the product to carry the registered BQM trademark. This provides an assurance that the biosolids in question meet the specified product standards for unrestricted use biosolids, have been produced or manufactured in accordance with one of the 'acceptable pathogen reduction processes' and one of the 'acceptable vector attraction reduction methods' for stabilisation Grade A biosolids specified in the New Zealand Guidelines, and that they meet the labelling requirements for unrestricted use biosolids stipulated in the BQM Code of Practice.

Condition 2, which requires that biosolids application rates do not exceed a three-year average of 200 kg total N/ha/yr, is a conservative estimate of the maximum application rate which is unlikely to result in acidification of the soil (via nitrification and the uptake of alkalinity) with consequent risk of metals becoming more bio-available and/or leaching to groundwater. Experience with both fertiliser and biosolids applications has shown that adherence to a 200 kg total N/ha/yr limit will ensure that biosolids are not applied at a rate that exceeds the nitrogen needs of crops and/or that results in the leaching of N to groundwater. The use of a three-year average figure allows, for example, for a single application of a larger amount of biosolids every three years.

Condition 3 is aimed at avoiding biosolids application to low pH (acid) soils, thereby minimising the potential for leaching of metals into groundwater, and minimising the bio-availability of metals.

Condition 4 is aimed at avoiding the contamination of water bodies as a result of biosolids application to land.

Condition 5 is self-explanatory.

Condition 6, relating to non-application of biosolids to grazed land, is aimed at avoiding the ingestion of contaminants by grazing animals, and hence the potential for accumulation of metals or organic compounds in human tissues or animal produce. The condition relates to land that has an established pasture or fodder crop cover and is not intended to preclude the incorporation of biosolids into soil prior to the establishment of new pasture or fodder crops.

[1] *See Glossary for a definition of 'BQM'.*

[2] *See Glossary for a definition of 'grazed land'.*

5.1 Discussion

The requirement for unrestricted use biosolids to carry a registered Biosolids Quality Mark (BQM) is a means of providing independent third-party confirmation (accreditation) that the biosolids meet all the relevant process and product standards for unrestricted use biosolids (i.e., the standards referred to in Tables 4.1 and 4.2). A BQM must be registered by the New Zealand Water Environment Research Foundation or any other agency that gains accreditation. Public health protection is a major issue for the use of biosolids by the public. The BQM provides assurance that the biosolids have been produced in accordance with one of the acceptable pathogen-reduction processes and one of the acceptable vector attraction reduction methods for stabilisation Grade A biosolids. A BQM will also certify that the biosolids meet the labelling requirements for unrestricted use biosolids specified in these *Guidelines* (section 7.4), which are consistent with the labelling requirements for agricultural compounds exempt from registration under the Agricultural Compounds and Veterinary Medicines Act.

Regular audits will be commissioned (from accredited auditing agents) of the process and product standards being achieved by producers of Aa biosolids. The BQM audit will be against a code of practice for the manufacture of Aa biosolids, which will essentially be a stand-alone compendium of the process, product and labelling requirements for Aa biosolids specified in these *Guidelines*. The results of the audits will be made available to all interested parties, including regional councils and the Ministry of Health. Failure to achieve the Aa product or process standards, and hence to attain BQM certification, will mean the biosolids in question cannot be used by the public or otherwise applied to land without a discharge consent. If necessary, the Ministry of Health is able to issue a repair notice or closing notice to the biosolids producer under the Health Act.

The rule does not contain a requirement for soil incorporation of unrestricted use biosolids because the risk of biosolids entering the food chain by way of ingestion by grazing animals is addressed by condition 6 of the model rule.

In addition to the BQM safeguard, the labelling requirements mean that producers will effectively have to certify or sign off that the key consent parameters, including product quality and process controls, have been met. Any discrepancy between labelling and product quality would amount to a breach of the Consumer Guarantees Act 1993.

Under the rule there would be no constraint on the volume of biosolids applied and/or the frequency of application. The justification for this is that the contaminant Grade 'a' criteria are designed to provide protection when biosolids are used as a complete topsoil replacement. The contaminant grade limits applying after the transition period are, with the exception of PCBs and dioxins, the same as the soil contaminant limits or 'ceiling concentrations' (see Table 4.2). The only constraint is that the maximum application rate shall not exceed a three-year average of 200 kg N/ha/year (see condition 2). Overseas experience with unrestricted use biosolid products has been that the cost of the product mitigates against the recommended application rates being exceeded.

Under a permitted activity rule there can be no requirement to monitor the effects of the discharge to land. This is consistent with the underlying premise that there is little or no potential for adverse effects from the activity.

It is anticipated that many biosolids producers will need time to be able to achieve the high ‘a’ grade contaminant standards applying to unrestricted use biosolids. However, the standards should be achievable either by virtue of a locally low industrial base, the separation of industrial wastes from domestic wastes, the implementation of tight tradewaste bylaws, or via dilution of sewage solids with materials such as green waste, pumice, sand or depleted soil.

6. REGULATION OF GRADE Ab, Ba AND Bb BIOSOLIDS DISCHARGES

6.1 Introduction

Restricted use biosolids are likely to need a resource consent. They are of lower quality than unrestricted use biosolids – in terms of their pathogen content, contaminant content or both – and so it is appropriate that any proposal to apply such biosolids to land should be subject to a resource consent application for a Discretionary Activity. This carries with it a requirement to prepare an assessment of environmental effects (AEE) and provides an opportunity for full public scrutiny of the proposal, with provision for objection and appeal (see section 3.2.3).

The discharger of the biosolids would seek and hold the consent whether the discharger is the producer or manufacturer of the biosolids or the landowner.

Developing a discretionary activity rule governing the discharge of restricted use biosolids to land is relatively straightforward as there is no need or requirement to specify conditions on the rule, and the rule simply captures all those biosolids discharges not meeting the conditions on the permitted activity rule, including those relating to product quality or process control. It is envisaged that the rule would be constructed along the following lines (Table 6.1).

Table 6.1 Model Discretionary Activity Rule

Discharge of Restricted Use Biosolids to Land

Any discharge of Aa biosolids to land where any of the permitted activity conditions specified in Rule x cannot be met, or any discharge of restricted use (Ab, Ba, Bb) biosolids to land, is a discretionary activity.

Assessment criteria include:

- *sensitivity of receiving environment*
- *N content and proposed application rate*
- *contaminant content*
- *pathogen content*
- *process of biosolids manufacture*
- *storage conditions/duration*
- *labelling*
- *transport issues*
- *application methods*
- *soil incorporation*
- *exclusion periods*
- *monitoring proposals*
- *record keeping requirements*
- *site management proposals*
- *environmental management systems*
- *quality assurance procedures*
- *any other criterion considered relevant by the regional council.*

Discretionary activity rules in plans frequently include assessment criteria, so a list of the matters that regional councils will need to take into account when considering applications for discharge of unrestricted use biosolids to land is included in the model rule.

In addition to being issues that regional councils will need to take into account when appraising a specific proposal and/or deciding appropriate consent conditions, some of the assessment criteria referred to are relevant to site selection and all of them are relevant to scoping an AEE.

An AEE should clearly identify the risks associated with a land application proposal, and how it is intended to manage them to avoid or mitigate any adverse environmental effects. Preparing a draft set of conditions is a useful way of consolidating and making clear the applicant's risk management intentions.

The rest of this section provides guidance to would-be dischargers of restricted use biosolids and to regulators on the relevance of each of the assessment criteria referred to below, from a risk management perspective.

6.2 Sensitivity of the proposed receiving environment

Under section 104(3) of the RMA the consent authority is required, when considering an application to discharge contaminants to land, to have regard to the sensitivity of the proposed receiving environment to adverse effects.

Matters relating to the sensitivity of the receiving environment, and hence to the nature and magnitude of environmental effects, include:

- soil chemical and physical properties
- soil pH
- depth and proximity of local aquifers
- slope of land and proximity of surface water bodies
- presence/absence of sensitive areas
- existing land use(s)
- social considerations.

These will now be considered in turn.

6.2.1 Soil chemical and physical properties

Soils vary greatly in their ability to sustain both the short- and long-term application of biosolids, and a number of individual soil chemical and physical properties will determine the suitability of a soil to receive applications of biosolids (see *Volume 2: Technical Manual* for details).

Soils should be tested prior to deciding on the suitability of a site for the *bulk* application of biosolids. If any of the chemical elements or compounds already occur in the soil at levels at or above the recommended soil limits (Table 4.2), then biosolids should generally not be applied, although these *Guidelines* recognise that there may be situations where a case can be made to consent authorities for a variation of specific soil contaminant limits (see section 4.2), e.g. for remediation of contaminated land. If the concentrations of any of the chemical parameters are close to any of the soil limits, the receiving soils should be considered 'sensitive' and it may be necessary for the consent authority to control application rates on the basis of contaminant loading (see section 6.3).

The ability of the soil to tightly retain contaminants is important, and this mainly depends on the amount of oxide (iron, aluminium, manganese), clay and organic matter in the soil and its pH. In general, the greater the amounts of these constituents and the higher the pH (see below), the greater the contaminant retention capacity. For metals, the cation exchange capacity of the soil provides a useful estimate of retention capacity.

The physical characteristics of the soil, such as soil texture (porosity and permeability), soil depth, and the presence or absence of iron pans, determine the drainage characteristics of the soil. Poorly drained, fine-textured soils should be avoided because application machinery may cause compaction of the soil (with subsequent ponding and/or anaerobic conditions). Also, decomposition of organic material in the biosolids is likely to be slow, and there will be a greater risk of surface run-off. On the other hand, excessively drained, coarse-textured soils may not be suitable due to their potential for leaching and/or greater bio-availability (see sections 2.3.1 and 2.3.5).

6.2.2 Soil pH

Soil pH is a major factor affecting both the retention and the solubility of metals (and some organic chemicals) in the soil. Therefore it has an important effect on both the *bio-availability* and the *mobility* of metals in soils amended with biosolids. The main metal contaminants (cadmium, chromium, copper, nickel, lead and zinc) are present in the soil predominantly as cations, and these tend to become more soluble and bio-available as soil pH decreases (the soil becomes more acid).

Consent authorities should attach a condition to a consent relating to maintaining soil pH. The scientific literature indicates that maintaining the pH value of biosolids-treated agricultural soils above pH 5.5 is likely to minimise the risk of phyto-toxicity or excessive metal uptake by crops. Normal agronomic liming practices should ensure this, but soils tend to become more acid with time so regular pH monitoring of biosolids-amended soils should be carried out.

Unlike agricultural soils, forestry soils are not usually limed to maintain soil pH and some forestry soils are naturally acidic, with pH values of less than 5. Although applying biosolids to forestry soils under low pH conditions would appear to represent a potential problem in terms of high metal availability and mobility, this is not necessarily the case. Since food crops are not involved, plant uptake of metals is not a health concern and trees do not appear to be particularly sensitive to metal toxicity. However, the drainage regime of forest soils and the depth and proximity of local aquifers may be an important consideration. The potential for future land-use change should also be taken into account.

6.2.3 Depth and proximity of local aquifers

Generally, if recommended application rates are followed and any necessary soil pH adjustments made, there is low potential for biosolids to contaminate groundwater (see section 2.3.1). However, as indicated above, in some situations the potential for leaching of contaminants, including nitrogen, may become an issue and information on the depth and proximity of local aquifers will need to be presented by the applicant, along with information relating to the drainage characteristics and pH of the soil. This information should be brought together with information relating to biosolids content and other relevant considerations (e.g., rainfall data) to provide an overall assessment of the likelihood of groundwater contamination.

6.2.4 Slope of land and proximity of surface water bodies

The AEE should identify the slope of the land and the proximity of rivers, streams, lakes, wetlands or the coastal marine area to the proposed area of application.

Resource consents for applying restricted use biosolids to land should generally contain a condition to mitigate against the possibility of contaminating surface waters such as prohibiting application on a specified slope of land or within a specified number of metres from a river or stream bank or from the edge of a lake, wetland or the sea (see proposed condition 4 for permitted activity rule, above).

6.2.5 Presence of ‘sensitive’ areas

The AEE should identify the presence of any sensitive areas such as wahi tapu, archaeological sites, wildlife habitats and areas of importance from a biodiversity perspective within or in the vicinity of the proposed area of application. The consent authority can then specifically prohibit the application of biosolids to any such areas.

6.2.6 Existing land uses

The existing land use (e.g., agriculture, forestry, public recreation) may be an important site selection consideration in that land use can influence land application costs, risk management considerations (e.g., soil pH trends and/or the ability to lime, the ability to incorporate with soil), permissible loading rates (if leaching is a potential issue), public health risks (exposure pathways), and the likely degree of local opposition to a proposal.

The use of biosolids as a fertiliser/soil conditioner for the growth of annual crops, where it can be incorporated into the soil prior to sowing (see section 6.11), is probably one of the best ways of utilising biosolids. However, suitable cropping land will not always be available and there will be situations where the bulk application of biosolids to forestry, public gardens, golf courses and the like, or the use of biosolids for rehabilitation of degraded land (e.g., quarry sites or mine tailings), will be considered a better option.

The primary concern for the consent authority will not be with the land use as such, but with the risk profile of the site (its sensitivity), and whether or not the risks can be adequately managed via consent conditions.

6.2.7 Social considerations

The sensitivity of the receiving environment extends beyond bio-physical matters to include social considerations such as the proximity of neighbours/dwellings, the presence of wahi tapu, existing public use and access to the proposed land application area or adjacent areas, and existing traffic movements in the area. Such considerations are all relevant to site selection, effects assessment and/or the conditions that should be applied to a consent. The RMA consultation process provides an opportunity to identify and address community concerns.

6.3 N content of biosolids and application rate

The nitrogen (N) content and the proposed application rate of the biosolids are, along with soil characteristics such as texture and pH, relevant to the potential for contamination of groundwater via the leaching of nitrate or metals (see section 2.3.1). Worldwide, the general approach to this issue is to recommend or require that biosolids be applied in accordance with the agronomic N needs of crops. This is considered to be a conservative approach because it does not take into account all of the nitrogen mechanisms, such as mineralisation and denitrification, which will also occur.

Different types of biosolids have different N contents (typically 1–6%, depending on the N content in the original sludge and the biosolids manufacturing process) and different N release characteristics. The form of N will affect the soil retention characteristics; for example, if it is present in an organic form or as ammonium, soils will retain it strongly whereas they will not retain N if it is present as nitrate. Different crops have different N requirements and the agronomic rate of application may be dependent on pasture management practices. Consequently, advice should be sought from an agronomist to determine specific rates for agricultural and forestry purposes and the management regime during incorporation.

Due to the complexity of estimating agronomic rates and because such estimates rely on a detailed knowledge of local site conditions, a default value of 200 kg total N/ha/yr is often adopted in New Zealand for pasture (see condition 2 of the model permitted activity rule and associated explanation, section 5), and it will often be appropriate for a resource consent governing the application of restricted use biosolids to land to contain a similar condition. In some situations it may be appropriate to express the N loading as an average, enabling larger amounts to be applied at less frequent intervals.

Although 200 kg total N/ha/yr is a reasonable default value, it is preferable that application rates be based on site-specific assessments including assessment of soil characteristics, mineralisation rates, climate, and the agronomic N needs of the crops in question.

It is undesirable, from a risk management perspective, to allow biosolids to be applied at a rate that results in soil contaminant limits being reached after one or two applications. This is why biosolids applications should not cause contaminant limits to be reached for at least a 20-year application period. In most cases N will be the limiting factor determining the maximum application rate of biosolids. However, in some cases, where one or more metals are already close to the soil limits, consent authorities may wish to impose an application rate limit based on 20 years of average annual application. This involves a simple mass balance calculation relating to the soil limit for that metal, its existing concentration in the soil, and its concentration in the biosolids (see Appendix III for a worked example).

6.4 Contaminant content of biosolids and soil

Applicants for discharge consents should fully characterise the contaminant content (metals and organic chemicals) of the biosolids. A consent authority needs this information to satisfy itself that the biosolids are not abnormal in any way (e.g., an unusually high herbicide or pesticide content) and that they are in fact biosolids, not sludges (see section 4.3.2)

The consent authority should also ensure there is sufficient head room between the measured “background” soil contaminant concentrations at a site (see section 6.2.1) and the soil limit (Table 4.2) when considering the composition of the biosolids. As noted above, the contaminant content of the biosolid and its background soil concentration may determine whether or not a consent authority sees fit to apply an application rate limit and the level of that limit.

The content of contaminants in the biosolids is relevant to the rate of build-up in the soil, and hence potentially to application rate constraints. But given the proposals to establish conservative soil limits for heavy metals and organic compounds (section 4.2) and to monitor the rate of build-up of contaminant levels in the soil (section 7.5.6), there will normally be no need for consent authorities to regulate the content of contaminants in restricted use biosolids, other than to stipulate that concentrations not exceed the maximum concentrations for Grade b biosolids specified in Table 4.2. This is consistent with the loading rate philosophy (see section 1.5).

6.5 Pathogen content

Applications for discharge consents for restricted use biosolids should specify whether or not the biosolids have attained or will attain (e.g., following proposed treatment) either the A or B stabilisation grades (section 4.3.1).

If the biosolid will attain the A grade prior to discharge, and the discharger would be relying on attainment of that standard to avoid public or animal health risks, then the consent conditions should require that:

- the discharge is subject to the biosolids meeting the processes and product requirements for stabilisation Grade A, as specified in these *Guidelines*
- the biosolids are prepared or manufactured under a certified quality assurance system.

In this situation, the biosolids can be considered safe (i.e., essentially pathogen free) and no other risk management measures (other than precautionary handling instructions on the label, see sections 6.8 and 7.4) are necessary from a human health perspective.

The consent for discharge of a B grade product (Ba or Bb class biosolid) should contain conditions requiring that the relevant processing, storage, VAR and exclusion period requirements be met (see Tables 4.1 and 6.2).

If a biosolids product is Grade B, it potentially contains pathogens at levels constituting a risk to human health. The consent authority will therefore need to consider implementing other risk management techniques to minimise the risk of human infection, having regard to the proposed land use. Table 6.2 sets out recommended controls that should be implemented by way of conditions on resource consents.

Table 6.2: Recommended controls for stabilisation Grade B biosolids, depending on end use

Land use	VAR requirement (see Table 4.1)	Recommended controls
Salad crops, fruit, other crops for human consumption that may be eaten unpeeled or uncooked	■ Mass of volatile solids in biosolids shall be reduced by a minimum of 38%; or ■ SOUR @ 20°C $\leq$ 1.5 g/m^3 for liquid sludges from aerobic processes; or ■ pH $\geq$ 12 @ 25°C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours.	May be applied immediately ***Plus*** Soil incorporation (see section 6.11) ***Plus*** A further waiting period of at least 1 year before crops are sown (the land may be used for other purposes in the meantime).
	■ Storage/exclusion period.	Store or lagoon for at least 1 year prior to application. ***Plus*** Soil incorporation (see section 6.11). ***Plus*** A further waiting period of at least 1 year before crops are sown (the land may be used for other purposes in the meantime).
Public amenities, sport fields, public parks, golf courses, play grounds, land reclamation	■ Mass of volatile solids in biosolids shall be reduced by a minimum of 38%; or ■ SOUR @ 20°C $\leq$ 1.5 g/m^3 for liquid sludges from aerobic processes; or ■ pH $\geq$ 12 @ 25°C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours	Store or lagoon for at least 6 months prior to application. ***Plus*** Soil incorporation (see section 6.11). ***Plus*** Restriction on public access for period of time necessary to establish a full vegetation cover on the land (see section 6.12).
	■ Storage/exclusion period.	Store or lagoon for at least 1 year prior to application. ***Plus*** Soil incorporation (see section 6.11) ***Plus*** Restriction on public access for period of time necessary to establish a full vegetation cover on the land (see section 6.12).
Fodder crops and pasture, orchards where dropped fruit is not harvested, turf farming, industrial or non-edible crops, crops that will be peeled or cooked before eating	■ Mass of volatile solids in biosolids shall be reduced by a minimum of 38%; or ■ SOUR @ 20°C $\leq$ 1.5 g/m^3 for liquid sludges from aerobic processes; or ■ pH $\geq$ 12 @ 25°C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours.	May be applied immediately. ***Plus*** Soil incorporation (see section 6.11). ***Plus*** Fruit and turf should not be harvested or pastures grazed for at least 6 months after applications. ***Plus*** Crops that will be peeled or cooked should not be harvested for at least 6 months after application.

Land use	VAR requirement(see Table 4.1)	Recommended controls
	■ Storage/exclusion period.	Store or lagoon for at least 1 year prior to application ***Plus*** Soil incorporation (see section 6.11). ***Plus*** Fruit and turf should not be harvested or pastures grazed for at least 6 months after applications. ***Plus*** Crops that will be peeled or cooked should not be harvested for at least 6 months after application.
Forest, trees or bush scrubland	■ Mass of volatile solids in biosolids shall be reduced by a minimum of 38%; or ■ SOUR @ $20^{o}C \leq 1.5$ g/m^3 for liquid sludges from aerobic processes; or ■ pH ≥ 12 @ $25^{o}C$ for at least 2 hours and pH ≥ 11.5 for 22 more hours.	May be applied immediately. ***Plus*** Public access restricted for 6 months (see section 6.12). ***Plus*** Buffer zones should be fenced and signposted.
	■ Storage/exclusion period.	Store or lagoon for at least 1 year prior to application. ***Plus*** Public access restricted for 6 months (see section 6.12). ***Plus*** Buffer zones should be fenced and signposted.

Source: Updated from Department of Health, 1992.

Note: SOUR = standard oxygen uptake rate.

If a product does not attain Grade B stabilisation, it is not a biosolid and is therefore not covered by these *Guidelines* for land application.

6.6 Process of biosolids production

The application should also describe the process of biosolids production or manufacture. There is a wide range of treatment processes available that enable biosolids products to meet the A or B stabilisation requirements (see section 6.2 of *Volume 2: Technical Manual*).

In addition to the treatment process, the application should describe the vector attraction reduction (VAR) method because this also helps to establish the risk profile of the biosolid in terms of its potential to attract vectors and/or to generate odours.

Different treatment processes can achieve different degrees of pathogen reduction with different degrees of consistency. Also, some of the treatment processes are equally effective at both reducing pathogens and vector attraction, whereas others may be better at one than the other. Consequently, providing biosolids treatment details in consent applications – including time-temperature relationships, the frequency of turning (composts), and pH conditions (lime stabilisation) – also helps to establish the risk profile of the product, and may be relevant to the setting of consent conditions.

Consent authorities should not place conditions on consents requiring biosolids dischargers to achieve stabilisation Grade A when the application is for discharge of a Grade B product. This is because achieving the Grade A standard is expensive, may not be affordable to many sewerage authorities in New Zealand, and may not be warranted if the biosolids contaminant levels are such that the unrestricted use classification (Aa) cannot be achieved. More significantly, international experience has shown that Grade B biosolids can be safely applied to land providing other risk management measures, such as soil incorporation and exclusion periods, are implemented.

6.7 Storage conditions and duration of storage

The storage of biosolids for extended periods can be used as a (low-energy) risk management technique to reduce the level of pathogens in biosolids prior to land application. The term ‘cold storage’ is often used, meaning storage under ambient conditions.

Storage can be viewed either as an adjunct to other pathogen reduction or VAR methods, or as a treatment process in its own right. As well as reducing health risk, it can help to achieve a better-quality product in terms of appearance, humus content and minimisation of the potential for releasing plant-damaging ethylene gas later.

Storage can be undertaken for practical reasons (e.g., if there is continuous biosolids production but a seasonal land application cycle, or if desludging of an oxidation pond produces large volumes of biosolids which cannot immediately be disposed of), with the added benefit of pathogen reduction.

Consent applications should indicate what storage conditions will be applied to the biosolids in question.

If the producer of the biosolids is also the discharger, it may be appropriate to attach storage conditions to a consent but, as with other process controls, it is probably preferable for the consent authority to focus on product quality at the point of discharge, rather than how best to achieve this quality. If the landowner is the discharger it will usually be inappropriate for the consent authority to specify storage conditions on the consent because storage is usually outside the landowner’s control.

If the requirements for stabilisation Grade A biosolids are met in full, there is no basis for recommending a storage period[21] and the issue of storage is essentially a marketing one. However, where there is doubt about the ability of a new biosolids production facility to consistently achieve Grade A stabilisation requirements, it may be prudent for producers to begin with an initial storage period, then gradually reduce or phase it out entirely when it becomes clear that pathogen standards are being met.

In the case of stabilisation Grade B biosolids, such as those produced by digestion, storage can be used as a way to reduce pathogen levels, but it is not recognised as a method of achieving Grade A stabilisation. This is because, unlike the methods referred to in column 2 of Table 4.1, where there is a high degree of confidence that they will effectively eliminate pathogens from the product, there can be no guarantee that storage will reduce pathogen content to any specified level. Consequently, the degree of pathogen reduction needs to be established by measurement, and this may become a relevant consent condition consideration if the discharger is also the producer of the biosolids (see above).

If a producer is relying on storage rather than some other method to achieve an acceptable level of public health risk reduction, then the guideline storage periods in Table 6.2 should be applied.

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[21] As distinct from a ‘maturation period’ for certain products such as composts (see column 2 of Table 4.1).

6.8 Labelling

Labelling is an important risk management tool. All biosolids, including restricted use biosolids,[22] should be appropriately labelled (see section 7.4).

In the case of biosolids supplied in bulk, it is the responsibility of the biosolids producer to ensure that the discharger of the biosolids receives a leaflet providing the same information that would appear on the label of a packaged product, at the time of delivery of the bulk order.

The application for a discharge consent should detail the information that will appear on the label of the product to be discharged so that the consent authority can see how labelling provisions fit into the overall risk management strategy.

If the producer is also the discharger of the biosolids, it may be appropriate for the consent authority to impose labelling requirements on the consent as per section 7.4 of these *Guidelines*. An alternative approach is to require the discharge to comply in all respects with the definition of restricted use biosolids, which incorporates labelling requirements.[23]

If the landowner is the discharger, it is inappropriate for the consent to specify labelling requirements, although a link with labelling (as above) may still be desirable.

All consents should include a condition requiring the discharger to apply biosolids in accordance with the producer’s or manufacturer’s instructions.

6.9 Transport issues

The application for a resource consent to discharge biosolids to land should describe the proposed method of transport of biosolids to the land application site, and identify any environmental effects, hazards or risks arising from this activity.

The current situation in respect of biosolids transportation is discussed in section 3.7. It is recommended that the transport of Grade B biosolids be regulated under the Dangerous Goods Rule (Land Transport Act) and *NZS 5433 Transportation of Dangerous Goods on Land*.

Regional councils may wish to impose consent conditions covering issues such as action in the event of spillage (prevention of contamination of waterways), avoidance of nuisance odours, or the timing of trucking movements.

6.10 Application methods

The method of applying biosolids to land (e.g., spraying, ploughing, surface application, soil injection) can determine the range of potentially adverse environmental effects and can therefore be an important risk management consideration. For example, a spraying operation may increase the potential for aerosol and/or odour generation.

The application should detail the proposed method of applying the biosolids. In some circumstances (e.g., where there are close neighbours) the consent authority may wish to control or at least influence the method of application, and this should be the subject of early discussions between the applicant and the consent authority.

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[22] *Reistricted use biosolids may have a significant pathogen content.*

[23] *See Glossary.*

6.11 Soil incorporation

Soil incorporation (e.g., via ploughing, rotary hoeing, injection) is an important risk management technique. It has the effect of:

- reducing exposure to pathogens (public or animal health risk)
- reducing the uptake of contaminants by grazing animals and transfer to the human food chain (animal health, public health and trade risks)
- removing vector access to pathogens (public health risk)[24]
- maximising dilution and dispersal of contaminants, enhancing immobilisation reactions in soil
- minimising losses after application from wind blow or surface run-off, and effects on non-target areas.

Soil incorporation should take place on the same day the biosolids are applied, and biosolids should be incorporated to a depth of at least 100 mm, and preferably 200 mm. (It is acknowledged that a full 200 mm incorporation depth may not be practical unless the land is ploughed). Soil samples for analysis should be taken to the depth of incorporation up to a maximum of 200mm.

Consent applications should detail (any) soil incorporation proposals (see guideline in Table 6.2).

As noted above, soil incorporation before planting annual crops is an ideal way of using biosolids as a fertiliser and soil conditioner, and will therefore tend to occur as a matter of course. However, consent authorities should attach a condition requiring soil incorporation (pre-planting) on all consents relating to discharges of biosolids to horticultural or cropping land. This will negate the undesirable practice of applying biosolids to leaf, salad, or root crops where there is a risk of direct transfer to animals or humans.

For pastoral land, soil incorporation is a vital risk management tool. As noted in sections 2.2.3 and 2.4, there is concern about the potential to contaminate meat or dairy produce via direct ingestion of pasture and/or surface soil. Consequently, consent conditions relating to discharges of biosolids to pastoral land should also include a requirement for soil incorporation (biosolids can be applied to pastoral land when pasture is resown).

6.12 Exclusion periods

Imposing public exclusion or stock withdrawal periods following the application of biosolids to land is a potentially important – and sometimes essential – risk management technique for restricted use biosolids. It allows time for the natural die-off of pathogens in the soil and the re-establishment of vegetation cover on the land following soil incorporation.

Such exclusion periods are particularly relevant where the biosolids are surface-applied and are known to have a high pathogen content. Adequate publicity needs to be given to exclusion periods, particularly where there has been a history of unconstrained public access and/or recreational use of the area in question. Consideration also needs to be given to the desirability of establishing buffer zones around the application area and/or erecting fences and warning signs.

Consent applications should contain details of public exclusion or stock withdrawal proposals (see Table 6.2).

The optimal exclusion period can depend on factors such as the sludge treatment system, climate, soil type, and land use, but usually ranges from 6 to 12 months.

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[24] Soil incorporation is one of the acceptable VAR methods (see Table 4.1).

Consent authorities should consult with the local medical officer of health and/or the NZ Food Safety Authority before deciding on appropriate public exclusion and/or stock withdrawal conditions.

6.13 Monitoring proposals

Monitoring of both the process of biosolids production (i.e., the key parameters such as time, temperature, pH), and product quality is an essential part of quality assurance and hence of risk management. Monitoring the build-up of contaminants in the receiving soil (relative to soil limits) is also an important risk management measure.

The requirements for monitoring Grade B stabilisation products are not as onerous as for Grade A products (see section 7).

The producer of the biosolids should develop a process and product monitoring programme in accordance with the guidelines in this document. If the producer is also the discharger of the biosolids, it is appropriate to include details in the consent application and for the consent authority to incorporate process and product monitoring requirements in the consent.

Irrespective of whether the producer or the landowner is the discharger, a condition should be attached to the consent requiring soil monitoring in accordance with section 7.5.6 of these *Guidelines*.

6.14 Record keeping requirements

Record keeping requirements are an important component of quality assurance and risk management. The consent authority should place a condition on consents requiring the maintenance of records in accordance with section 7 of these *Guidelines*.

6.15 Site management plan

A land application site management plan (LASMP) is a useful way of co-ordinating an approach to managing the risks associated with a specific land application project. The idea of the plan is to provide the detail of how it is proposed to implement, or ensure compliance with, the consent conditions. It also provides an opportunity to integrate consent requirements with other desirable site management practices such as public liaison and complaints procedures.

Consent authorities should attach a consent condition requiring the consent holder to prepare an LASMP for submission to the consent authority within an appropriate period following the granting of the consent.

An LASMP may form part of an environmental management system (see section 7.2).

7. QUALITY ASSURANCE

7.1 Introduction

In order to minimise risks associated with the use of biosolids, quality control and management practices are required. Their collective use and documentation provide quality assurance. The most effective way of ensuring a consistently high-quality product is to implement control and monitoring mechanisms at different stages in the biosolids life cycle, rather than just one quality check of the final product. This means that if one check or control fails, any problems should be picked up by the other mechanisms.

The importance of having quality assurance procedures and management practices in place cannot be over-emphasised and councils should consider these recommendations when assessing resource consent applications. The best management plans are site-specific, and the specific life cycle of each batch of biosolids should be considered when devising a management plan.

The amount of measurement and record keeping recommended differs for producers and dischargers. It is also different for unrestricted grade (Aa) biosolids and restricted grades (Ab, Ba, Bb). It is important that the principles of adequate measurement, maintaining records and ensuring the accuracy of the data are appropriately used in each case.

7.2 Environmental management systems (EMS)

It is recommended that all producers and bulk dischargers of biosolids have a system of formalised (auditable) management plans in place. The ISO management systems are probably the best known, but other in-house systems can be equally useful provided they can be audited and verified by an external agency.

An environmental management system, or EMS (which is often based on an ISO 14001 operating system), is a pro-active approach to environmental management. It is used to control environmental performance through continuous feedback and process improvement. The approach relies on a philosophical and personal commitment from every employee.

The basic foundation of an EMS is a written plan, which includes a description of responsibilities for managers, production workers, those responsible for land application, and the regulators. The plan usually has five sections: environmental policy, planning, implementation, monitoring and corrective action, and management review.

An EMS would cover *all* biosolids activities relevant to the producer or discharger. If the producer is also the discharger, the EMS would cover all production, storage, transport and discharge activities.

Beyond ensuring full compliance with all resource conditions and other regulatory requirements, an EMS can address such issues as odour, noise and other potential community concerns. An EMS can have significant public relations value and economic benefits.

In the US, the National Biosolids Partnership (NBP)[25] has begun pilot testing its EMS for biosolids programmes. It is on record as saying that it believes independent third party audits are "critical to increasing public acceptance of effective biosolids management and to the overall integrity of the NBP's EMS program".

An EMS can be prepared at the instigation of the biosolids producer/discharger, or at the request of the consent authority. Ideally it involves a collaborative effort between the two parties.

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25 An alliance of the Water Environment Federation, the Association of Metropolitan Sewerage Agencies and the US Environmental Protection Agency.

7.3 Record keeping

Appropriate records should be kept by manufacturers of biosolids, dischargers and any third parties involved in the bulk transport of biosolids. The following sections detail the information that is required at each stage in a biosolid's lifecycle. These sections relate to bulk and commercial operations and are not intended for purchasers and dischargers of bagged biosolids for domestic use.

The type of records that should be kept will vary depending on whether they are related to the production, transport, storage or discharge of the biosolid. The same person or organisation may be responsible for more than one of these activities; if this is the case, they should keep records relating to all the functions they perform or control.

Records should be kept and maintained for as long as possible. In the case of records relating to contaminant loads applied to a site, they should be maintained for at least as long as biosolids are applied to that site and for a five-year period after application ceases.

7.3.1 Records to be maintained by producers of biosolids

All producers of all grades of biosolids shall maintain the following records:

- batch number (if batched)
- date of production
- stabilisation grade, including:
    - microbiological results
    - frequency of sampling
    - method of achieving stabilisation grade
    - VAR method
- contaminant grade, including:
    - contaminant concentrations
    - frequency of sampling
- nature of any co-blended material (e.g. green waste in compost)
- quantity of biosolids produced in that batch
- treatment processes employed
- monitoring of treatment processes and results
- documented procedures for operations, sampling, monitoring and auditing
- concentrations of N (P and K are also useful).

The following *additional* information should be recorded if the biosolids produced are within the restricted use category (Grades Ab, Ba, Bb) or for *bulk use* (> 50m^3) of unrestricted use (Grade Aa) biosolids:

- address of destination for discharge of biosolids.

7.3.2 Records to be maintained by storers of biosolids

The following records are to be maintained by storers of loose bulk biosolids:

- complaints received and responses made
- inputs and outputs
- information on any monitoring carried out, including insect nuisance and odour
- storage times.

7.3.3 Records to be maintained during transfer of biosolids

This section applies to information that should be collected and passed on during the transfer of bulk biosolids (all grades) from the producer to the end user. It is best collected on a *chain of custody* form. If the biosolids are not being transferred during their lifecycle (the producer is also the discharger), then this phase of record keeping is not necessary.

The records to be maintained during transfer are:

- source of the biosolids, batch number and date of dispatch
- mass of product
- stabilisation grade
- contaminant grade
- location the biosolids are transferred to.

7.3.4 Records to be maintained by dischargers of biosolids

Dischargers of restricted use biosolids (Ab, Ba and Bb) should collect the following information:

- location of application site
- current land use
- name of site occupier and owner (if different)
- area of application (an accompanying map showing the area would be useful)
- date of application
- soil pH (before biosolids application)
- soil contaminant concentration before biosolids application
- estimated soil contaminant concentration after biosolids application
- biosolids application rate (based on agronomic rate)
- method of application
- source of biosolids and batch number
- stabilisation grade
- contaminant grade
- concentration of N
- monitoring methods
- frequency of soil sampling.

There is no requirement to keep records on the application of unrestricted use biosolids (Grade Aa). However, it is a good management practice for all users of bulk quantities (> 50 m^3) of biosolids to keep records of application, as above.

7.4 Labelling

A label/information leaflet detailing the following as a minimum should accompany all grades of biosolids produced:

- trade name (if it has one)
- name and address of the producer/manufacturer

- information on the product's origins (i.e., that it includes treated human waste) and precautionary handling instructions; an example of wording for a compost is: "made from the natural composting of garden trimmings and biosolids (treated human waste) that has been subject to temperatures of $\geq 55^{o}C$
- health warning – generally a health statement is printed on the packaging of all garden mixes and should similarly be used for bagged biosolids products; suitable wording is:

  "This biosolid product may contain a variety of living micro-organisms, some of which on rare occasions can cause illness in humans. Serious infection is rare but can happen for older people and those with reduced immunity. Please take the following precautions:

  - Avoid opening the bag in enclosed areas.
  - Avoid inhaling the mix.
  - Always wear gloves and wash hands after use.
  - Work with damp or wet mix/soil to reduce the dust potential.
  - See your doctor if you develop a high fever, chill, breathlessness or cough."
- grade of product – stabilisation and contaminant grades
- batch number (if applicable)
- order number (if applicable)
- delivery date (if applicable)
- the process the biosolids have been subjected to (e.g., digestion, pH adjustment)
- nutrient (N) content and modifying pH value (if the latter is applicable)
- placement of material in relation to water bodies
- requirement to incorporate into soil if applied to agricultural land
- recommended application rates.

While labelling biosolids is useful for providing information to the end user, it can also have the effect of improving general public perception of the product. This has been demonstrated in a trial conducted in Austria, where it was found that labelling biosolids increased their use in agriculture and also increased the perception of the product as a high-quality fertiliser (EC 2001).

7.5 Monitoring

7.5.1 Responsibility for monitoring

The producer of the biosolids is responsible for conducting any monitoring that relates to production/storage and transportation. The holder of the discharge consent is responsible for any monitoring required by the consent in relation to land application activities. Consent holders may contract others to undertake activities for them, including the monitoring, but they themselves will remain ultimately responsible.

7.5.2 Sludge monitoring

The quality of the biosolid is dependent on the quality of the sewage sludge used to produce it. While these *Guidelines* focus on the quality and monitoring of the final treated product, it is important to note that monitoring of the raw ingredients is also useful to the producer of biosolids.

Monitoring contaminant concentrations in sludge is necessary to determine if there is potential for the sludge to be converted into biosolids. If the raw product contains levels of chemical contaminants higher than those acceptable in either contaminant Grade 'a' or 'b' biosolids, then there may be potential to convert the sludge into biosolids by way of dilution with less contaminated sludge and/or other material. Also, if monitoring shows that chemical contaminant levels are high, this may indicate that trade discharge controls in the catchment need to be introduced or tightened in order to improve the quality of the sludge.

7.5.3 Process monitoring

Process monitoring is undertaken by the producer to determine if the required conditions for pathogen reduction and vector attraction reduction (VAR) are occurring. That is, it provides assurance that stabilisation objectives are being achieved. Process monitoring should be managed within an auditable EMS (see section 7.2).

Biosolids producers should follow the management practices detailed below regardless of whether they are monitoring VAR, pathogen reduction, or both.

- All process conditions, design diagrams and calculations should be recorded.
- Any changes to the process and/or equipment should be noted and retained for future reference.
- Equipment should be operated and calibrated in accordance with the manufacturer’s instructions, with independent test records/certificates kept up to date.
- Records should be maintained for all key process conditions, such as time, temperature and pH.

If the above information is recorded then the producer will be in a strong position to demonstrate that the appropriate pathogen and VAR requirements are being met. Production records should be maintained for at least two years.

7.5.4 Biosolid product monitoring

The final biosolid product is monitored by the producer to determine the extent of vector attraction reduction, pathogen numbers and chemical contaminant concentrations. Section 7.6 details the number of samples required and the frequencies at which they should be taken to determine compliance with both the stabilisation and the chemical contaminant grades.

Records should be kept of these results and results of any other testing conducted on the final product (e.g., percentage solids, reduction in volatile solids, pH). These can then be used to demonstrate compliance with the VAR requirements, pathogen requirements and chemical contaminant concentrations specified in these *Guidelines*.

7.5.5 Monitoring of unrestricted use biosolids

As noted in section 5, the requirement for unrestricted use (Aa) biosolids to carry the registered Biosolids Quality Mark (BQM) is a means of providing independent third party confirmation that the biosolids meet all the relevant process and product standards.

While the monitoring of process and product standards will still be the responsibility of the biosolids producer, regular audits of the process and product standards achieved by producers of Aa biosolids will be commissioned under the BQM accreditation process.

7.5.6 Soil monitoring

Soil should be tested *before* the application of biosolids to determine the existing contaminant concentration. For bulk users, soil samples should be taken before the initial application of biosolids and then every five years after this. In this way the accumulation of chemical contaminants in the soil can be monitored. Regular monitoring of soil is only recommended for the application of restricted use biosolids, as chemical contaminant limits in unrestricted use biosolids are low enough to prevent rapid accumulation. However, periodic monitoring of soil that has had Aa grade biosolids applied to it would be useful information for long-term reviews. This data should be collected and held available as a restricted record for review. It is not necessary to monitor pathogen concentrations in the soil as biosolid application will not cause a cumulative increase in pathogens.

The discharger of restricted use biosolids will be responsible for monitoring soil contaminant levels via consent conditions.

7.6 Sample frequency

7.6.1 Biosolids product sampling

Biosolids product sampling is used as an important check that the primary process controls are effective. Product sampling is not a stand-alone control and therefore need not be statistically representative in its own right. This principle has been used in developing the recommended sampling frequency presented below.

The frequency at which samples are taken as part of a product monitoring programme is dependent on the objectives of sampling. There are two types of sampling:

- verification sampling
- routine sampling.

Verification sampling is required to demonstrate that a treatment process is producing a final product of consistent quality. This phase of monitoring is typified by a high-frequency sampling regime. Verification monitoring should occur not only when a new process is commissioned but also when changes are made to an existing process, and also if any of the routine samples exceed the limits set for pathogens or chemical contaminants; in other words, whenever there could be a possible change to the quality of the final product.

Routine sampling is typified by a less onerous monitoring regime than that required for verification. This is because verification monitoring is used to demonstrate ability and stability of the process and/or the quality of the product. Once this is determined to be satisfactory the number of samples taken and the number of monitoring periods can be reduced.

Full details of the sampling requirements are in section 8 of *Volume 2: Technical Manual*. Note that when one or more contaminants exceed the limits specified then a return to verification monitoring is only required for those contaminants. However if one or more pathogens exceed the limits specified a return to verification monitoring is required for all pathogens.

Tables 7.1 and 7.2 detail the frequency and number of samples recommended for stabilisation grade monitoring and contaminant monitoring, in relation to the potential grade of the biosolid, and whether the samples are being taken for verification or routine sampling.

7.6.2 Soil sampling

Soil from sites to which biosolids have been applied should be sampled using an unbiased pattern such as a rectangle or grid. Samples should be taken from areas representative of the site as a whole. Avoid taking samples from non-representative areas such as fence lines, dung patches, areas where burning has taken place, fertiliser storage areas, swampy areas, and near roads or buildings.

The number of samples required depends on the area of application. However, a minimum of 10 core samples, taken to a depth to which the biosolids were incorporated into the soil, up to a maximum of 200 mm (see section 6.11), should be collected per hectare of biosolids application. If there is no soil incorporation, the sampling depth should be 200mm. All samples should be combined to form a composite and then tested for the contaminants listed in Table 4.2. The pH of the sample should also be determined.

Table 7.1: Stabilisation grade sampling frequencies

Grade	Monitoring type	Sampling regime	Parameters to be monitored
A	Product verification[1,2]	≥ 15 evenly dispersed grab samples per month for a 3-month period with ≤ 3 failures. If > 3 failures then the 15 following consecutive grab samples must comply.	■ *E. coli* ■ *Salmonella* ■ *Campylobacter* ■ enteric viruses ■ helminth ova ■ VAR
	Routine sampling	≥ 1 grab sample per week	■ *E. coli* ■ VAR
B	Product verification[2]	Not applicable for pathogen testing	■ VAR[3]
	Routine sampling	Not applicable for pathogen testing	■ VAR[3]

[1] *No more than three samples should be taken per day during this period.*

[2] *In the case of biosolids manufacturing facilities in existence prior to the publication of these* Guidelines *it is acceptable to use data up to 12 months old for the purposes of* product verification.

[3] *If a barrier is to be used for VAR no monitoring is required at the production stage.*

Table 7.2: Contaminant grade sampling frequencies

Grade	Monitoring type	Number of samples
a ***and*** b	Product verification[1]	■ Metals: 1 composite[2]/week over a 3-month period. ■ Organics: 1 composite sample[2]/month over a 3-month period ■ Dioxins: 1 composite[3]/3 months
	Routine sampling[1]	■ Metals: ≥ 1 composite[2]/2 weeks ■ Organics: 1 composite/2 months ■ Dioxins: 1 composite/year[4]

[1] *In the case of biosolids manufacturing facilities in existence prior to the publication of these* Guidelines *it is acceptable to use data up to 12 months old for the purposes of* product verification. *For the purposes of determining compliance at the 95 percentile for* routine sampling, *the age of the data set shall be no more than 2 years for metals and organochlorines and no more than 5 years for dioxins. This avoids the scenario of old data masking upward trends in contaminant concentrations.*

[2] *Samples tested should be made up from daily composites. For organochlorines there shall be no exceedence over 3 consecutive samples.*

[3] *The dioxin verification composite should be made up of 1 sample taken per day during the verification period.*

[4] *Composite to be made up of 1 sample per week over a year-long period.*

7.7 Test methods

Analytical methods for contaminant and pathogen testing are detailed in section 8 of *Volume 2: Technical Manual*. It should be noted that pathogen testing requires the specific procedures detailed and testing laboratories will not be accredited for them. Special care should be taken to follow the instructions.

7.8 Health and safety precautions

Biosolids may contain micro-organisms, so it is good practice to exercise care when working with them. Individuals handling biosolids should ensure that:

- hands are washed and nails scrubbed well with soap before eating, drinking or smoking, and at the end of the working day
- cuts and skin abrasions are covered with waterproof dressings
- no food or drink is consumed or smoking permitted by employees while working with biosolids or biosolids products
- a suitable change of clothing is worn during work, and safety footwear and gloves are worn to protect against injury from sharp objects
- showering facilities are available to biosolids production workers
- eye protection is worn to protect against dust
- if dust or aerosols are considered a problem, masks conforming to a recognised standard should be worn to prevent inhalation. Wherever possible, workers should be upwind of the land application process.

In addition to the above it is good practice for biosolids producers and transporters to produce an occupational health and safety plan to ensure workers handling biosolids are adequately protected from, and informed of, risks (see section 3.6).

Employees working with biosolids and/or wastewater residuals should also review their immunisation history. As a minimum, anyone working in this type of environment should be immunised against tetanus, but should also consider immunisations against Hepatitis A and B.

Useful additional guidance can be found in a National Institute of Occupational Safety and Health publication (NIOSH, 2002).

8. ONGOING RESEARCH AND REVIEW

8.1 Biosolids research

There is a need for ongoing research into the effects of biosolids application on soil resources, particularly in respect of the bio-availability of metals and the effects of biosolids on soil ecology and fertility. A considerable amount of research is taking place overseas, and much of this information will become available here. Some New Zealand research is under way (see section 2.3.4) but more information specific to New Zealand's soils, climate and production systems is needed. The involvement of research agencies in well-designed long-term monitoring programmes and ongoing funding of biosolids research programmes is an important means of gaining such information.

8.2 Review of this document

The review of these *Guidelines* will be co-ordinated by the New Zealand Water Environment Research Foundation on a five-yearly basis in light of overseas and New Zealand research findings and management experiences. The review process will include consultation with interested parties.

If the need arises, the *Guidelines*, or parts of them, will be reviewed more frequently.

9. GLOSSARY

Agricultural land: Horticultural, cropping and pastoral land.

Agronomic rate: The agronomic rate for biosolids application is designed to provide the amount of nutrients needed by a crop or vegetation to attain a defined yield, while minimising the amount of nitrogen that will pass below the root zone of the crop or vegetation to the groundwater.

Bio-availability: The availability of substances for uptake by plant and animal species.

Biosolid: A sewage or sewage sludge derived from a sewage treatment plant that has been treated and/or stabilised to the extent that it is able to be safely and beneficially applied to land and does not include products derived from industrial wastewater treatment plants. The term biosolid/biosolids is used generically throughout this document to include products containing biosolids (e.g., composts).

Biosolids Quality Mark: The registered biosolids quality mark accreditation, which provides an assurance that the biosolids in question meet all the relevant manufacturing process standards (pathogen reduction and vector attraction reduction), product standards, and labelling requirements applying to unrestricted use (Aa) biosolids in the *Guidelines for the Safe Application of Biosolids to Land in New Zealand.*

BQM: Biosolids Quality Mark.

Bulk use: Application of biosolids to land equalling or exceeding 50 m^3 by volume.

Composting: A biosolids manufacturing process that biologically stabilises dewatered sludge. It is ordinarily an aerobic process taking place at thermophilic temperatures (about 55°C) because of heat released by biochemical transformations.

Contaminant: Any substance (including heavy metals, organic compounds and micro-organisms) that, either by itself or in combination with other substances, when discharged onto or into land or water, changes or is likely to change the physical, chemical or biological condition of that land or water. [RMA Definition].

Contaminant concentration limits: The maximum permissible amount of a given contaminant in biosolids or a biosolid blend (see Table 4.2 of these *Guidelines*).

Discharger of biosolids: The party responsible for applying biosolids to land; the discharge consent holder.

DS: Dry solids.

EMS: Environmental management system.

Grazed land: Land that is being grazed or will be grazed in the next 12 months. Grazed land may have a cover of pasture or fodder crops.

Groundwater: Sub-surface water from which wells or springs are fed; strictly, the term applies only to water below the water table.

Heat drying: A biosolids manufacturing process whereby sewage sludge is dried by direct or indirect contact with hot gases to reduce the moisture content to 10% or lower.

Helminth: Parasitic worm-like invertebrate.

Horticultural land: Land used for process food crops, leaf crops, root crops.

Lime stabilisation: A biosolids manufacturing process involving the addition of sufficient lime or lime mixtures to the sewage sludge to raise the pH of the sludge to 12 after 2 hours of contact.

MAF: Ministry of Agriculture and Forestry.

Maturation: The conversion and amendment of the rapidly biodegradable components in the organic material (of sludges and biosolids) to substances similar to soil humus that slowly decompose. Compost that is insufficiently mature will reheat and generate odours in storage and upon rewetting. It may also inhibit seed germination by generating organic acids and inhibit plant growth by removing nitrogen as it decomposes in the soil.

Most probable number (MPN): A sample analysed by dispersion in an extracting solution, by excessive dilution, then using statistical analysis based on the positive or negative growth for each sample.

Nuisance: Something which is noxious, dangerous or offensive.

NZWERF: New Zealand Water Environment Research Foundation.

Pastoral land: Grazed land, including land used for dairy, beef, sheep and deer production.

Pathogens: Disease-causing micro-organisms such as certain bacteria, viruses and parasites.

PCB: Polychlorinated biphenyl.

PFU: Plaque-forming unit.

pH: A measure of the hydrogen ion concentration in a solution. On the pH scale of 0–14, a value of 7 represents a neutral condition; decreasing values (below 7) indicate increasing hydrogen ion concentration (acidity); increasing values, above 7, indicate decreasing hydrogen ion concentration (alkalinity).

Phyto-availability: The availability of substances (e.g., metals, nutrients) for plant uptake.

Phyto-toxic effects: Adverse toxic effects of contaminants on plant growth and development.

Producer of biosolids: A person who either produces biosolids by operating a biosolids manufacturing facility (e.g., a composting, heat-drying, lime stabilisation or digestion plant) or who manufactures a blended product from biosolids.

Protozoa: Small, single-celled animals including amoebae, ciliates and flagellates.

Restricted use (Ab, Ba, Bb) biosolids: Biosolids that:

(i) meet the process and product requirements for Ab, Ba or Bb biosolids as specified in the *Guidelines for the Safe Application of Biosolids to Land in New Zealand*; and

(ii) have labelling supplied with the product at the point of sale or give away, such labelling specifying:

- (a) trade name (if it has one)
- (b) name and address of the producer/manufacturer
- (c) information relating to the product’s origins (i.e., that it is biosolids-based) and precautionary handling instructions
- (d) a health warning
- (e) grade of product – stabilisation and contaminant grades
- (f) batch number (if applicable)
- (g) order number (if applicable)
- (h) delivery date (if applicable)

    - (i) process the biosolids have been subjected to (e.g., digestion, pH adjustment)
    - (j) nutrient (N) content and modifying pH value (if the latter is applicable)
    - (k) placement of material in relation to water bodies
    - (l) requirement to incorporate into soil if applied to agricultural land
    - (m) recommended application rates.

Sensitive sites: Sites at which biosolids should not be applied due to the ecological, social or cultural values associated with them.

Sewage sludge: The unstabilised organic solid material settled out from domestic and industrial wastewater during the treatment process. It contains pathogens, organic material, nutrients, metals and other chemicals from residential (human waste) and commercial properties, and tradewaste discharges. Sewage sludge is an unavoidable product of advanced wastewater treatment. Untreated sewage sludge would not meet the stabilisation and/or contaminant grades defined in these *Guidelines* and cannot be beneficially used without further treatment and stabilisation to produce biosolids.

SOUR: Standard oxygen uptake rate.

Unrestricted use (Aa) biosolids: Biosolids that:

- (1) meet the process and product requirement for stabilisation Grade ‘A’ and contaminant Grade ‘a’ as specified in the *Guidelines for the Safe Application of Biosolids to Land* in *New Zealand* (note: the microbiological and chemical standards are to be met at the time the biosolids product is prepared for sale); and
- (2) carry the registered Biosolids Quality Mark (BQM) accreditation; and
- (3) have labelling supplied with the product at the point of sale or give away, such labelling specifying:
    - (a) trade name (if it has one)
    - (b) name and address of the producer/manufacturer
    - (c) information relating to the product’s origins (i.e., that it is biosolids-based) and precautionary handling instructions
    - (d) a health warning
    - (e) certification that the product has been manufactured in accordance with one of the ‘accepted process methodologies’ for stabilisation Grade ‘A’ biosolids recognised in the *Guidelines for the Safe Application of Biosolids to Land in New Zealand*
    - (f) batch number (if applicable)
    - (g) order number (if applicable)
    - (h) delivery date (if applicable)
    - (i) the process the biosolids have been subjected to (e.g., composting, heat drying, pH adjustment)
    - (j) nutrient (N) content and modifying pH value (if the latter is applicable)
    - (k) placement of material in relation to water bodies
    - (l) requirement to incorporate into soil if applied to agricultural land and to avoid application to grazed land
    - (m) recommended application rates.

Urban land: Domestic gardens, lawns, public parks and gardens, golf courses, sports fields, turf farming, land rehabilitation.

VAR: Vector attraction reduction (see below).

Vectors: Organisms such as rodents and insects that are attracted to putrescible organic matter and that may spread disease by carrying and transferring pathogens.

Vector attraction reduction: Processes by which biosolids are treated to remove or reduce substances that attract vectors.

Verified: Independently checked or audited.

Wahi tapu: Maori sacred site.

WSP: Waste stabilisation pond.

WWTP: Wastewater treatment plant.

10. REFERENCES

Brookes, P.C. and McGrath, S.P. (1984). Effects of metal toxicity on the size of the soil microbial biomass. *Journal of Soil Science* 35, 341–346.

Brookes, P.C., McGrath, S.P. and Heijnen C.E. (1986). Metal residues in soils previously treated with sewage sludge and their effects on growth and nitrogen fixation by blue-green algae. *Soil Biology and Biochemistry* 18, 345–353.

Brookes, P.C., McGrath, S.P., Klein, D.A. and Elliott, E.T. (1984). Effects of heavy metals on microbial activity and biomass in field soils treated with sewage sludge. In: *Environmental Contamination*, international conference, London. CEP Consultants Ltd, Edinburgh.

Carrington, E.G., Davis, R.D. and Pike E.B. (1998). *Review of the Scientific Evidence Relating to Controls on the Agricultural Use of Sewage Sludge. Part 1: The Evidence underlying the 1989 Department of the Environment Code of Practice for Agricultural Use of Sludge and the Sludge (Use in Agriculture) Regulations*. Report No DETR 4415/3.

Chaney, R.L. and Ryan, J.A. (1994). *Risk Based Standards for Arsenic, Lead and Cadmium in Urban Soils*. DECHMA, Frankfurt/Main.

Commission of the European Communities (1986). Council directive (86/278/EEC) on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture. *Official Journal of the European Communities* 181, 6–12.

Council of the European Communities (1991). Council directive of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources (91/676/EEC). *Official Journal of the European Communities* 375, 1–8.

Department of Health (1992). *Public Health Guidelines for the Safe Use of Sewage Effluent and Sewage Sludge on Land*. Department of Health, Wellington.

Dowdy, R.H. and Volk, V.V. (1984). Movement of heavy metals in soils. In: D.W. Nelson, D.E. Elrick and K.K. Tanji (eds). *Chemical Mobility and Reactivity in Soil Systems*, pp. 229–40. Soil Science Society of America, Madison, USA.

European Commission (2001). *Disposal and Recycling Routes for Sewage Sludge: Part 1 – Sludge Use Acceptance.* DG Environment, Sede, Arthur Andersen.

Lindo, P.V., Taylor, R.W. and Shuford, J.W. (1993). Accumulation and movement of residual phosphorus in sludge-treated decatur silty clay loam soil. *Commun. Soil Sci. Plant Anal.* 24, 1805–1816.

MAFF/DoE (Ministry of Agriculture, Fisheries and Food/Department of the Environment) (1993). *Review of the Rules for Sewage Sludge Application to Agricultural Land: Soil Fertility Aspects of Potentially Toxic Elements*. Report of the Independent Scientific Committee. MAFF Publications, London.

McGrath, S.P., Brookes, P.C. and Giller, K.E. (1988). Effects of potentially toxic metals in soils derived from past applications of sewage sludge on nitrogen fixation by *Trifolium repens* L. *Soil Biology and Biochemistry* 20, 415–424.

McGrath, S.P., Chang, A.C., Page, A.L. and Witter, E. (1994). Land application of sewage sludge: Scientific perspectives of heavy metal loading limits in Europe and the United States. *Environmental Review* 2, 108–118.

McGrath, S.P., Chaudri, A.M. and Giller, K.E. (1995). Long-term effects of metals in sewage sludge on soils, micro-organisms and plants. *Journal of Industrial Microbiology* 14, 104.

McLaren, R.G. and Smith, C.L. (1996). Issues in the disposal of industrial and urban wastes. In: R. Naidu, R.S. Kookana, D.P. Oliver, S. Rogers and M.J. McLaughlin (eds). *Contaminants and the Soil Environment in the Australasia–Pacific Region: Proceedings of the First Australasia–Pacific Conference on Contaminants and Soil Environment in the Australasia–Pacific Region*, Adelaide, Australia, 18–23 February 1996. Kluwer Academic Publisher, Dordrecht.

Ministry for the Environment (2002). *The New Zealand Waste Strategy*. Ministry for the Environment, Wellington.

Ministry of Health (2000). *The New Zealand Total Diet Survey*. Ministry of Health, Wellington.

Natural Resource Management Ministerial Council (2003). *Guidelines for Sewage Systems: Biosolids Management.* Department of Agriculture, Fisheries and Forestry, Canberra, Australia.

New South Wales EPA (1997). *Environmental Guidelines: Use and Disposal of Biosolids Products*, Sydney NSW.

NIOSH (2002). *Guidance for Controlling Potential Risks to Workers Exposed to Class B Biosolids*. Publication No. 2002-149. National Institute for Occupational Safety and Health.

Ogilvie, D. (1998). *National Study of the Composition of Sewage Sludge. Prepared for the New Zealand Water and Wastes Association*. Drainage Managers Group of NZWWA.

Smith, S.R. (1996). *Agricultural Recycling of Sewage Sludge and the Environment*. CAB International, Oxford.

Speir, T.W., van Schaik, A.P., Percival, H.J., Close, M.E. and Pang, L. Heavy metals in soil, plants and groundwater following high-rate sewage sludge application to land. *Water, Air, and Soil Pollution* (submitted).

USEPA (1993). Standards for the use and disposal of sewage sludge: Final rules, 40 CFR Parts 257, 403 and 503. *Federal Register* 58, 32, 9248–9415.

USEPA (1995). *A Guide to the Biosolids Risk Assessment for the EPA Part 503 Rule*. EPA 832-B-93-005. USEPA Office of Waste Management.

US National Research Council (1996). *Report of the Committee of the Use of Treated Municipal Wastewater Effluents and Sludge: Use of Reclaimed Water and Sludge in Food Crop Production.* National Academy Press, Washington DC.

Van den Berg, M., Birnbaum, L., Bosveld, A.T.C., Brunstrom, B., Cook, P., Feeley, M., et al. (1998). Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environmental Health Perspectives* 106, 775–792.

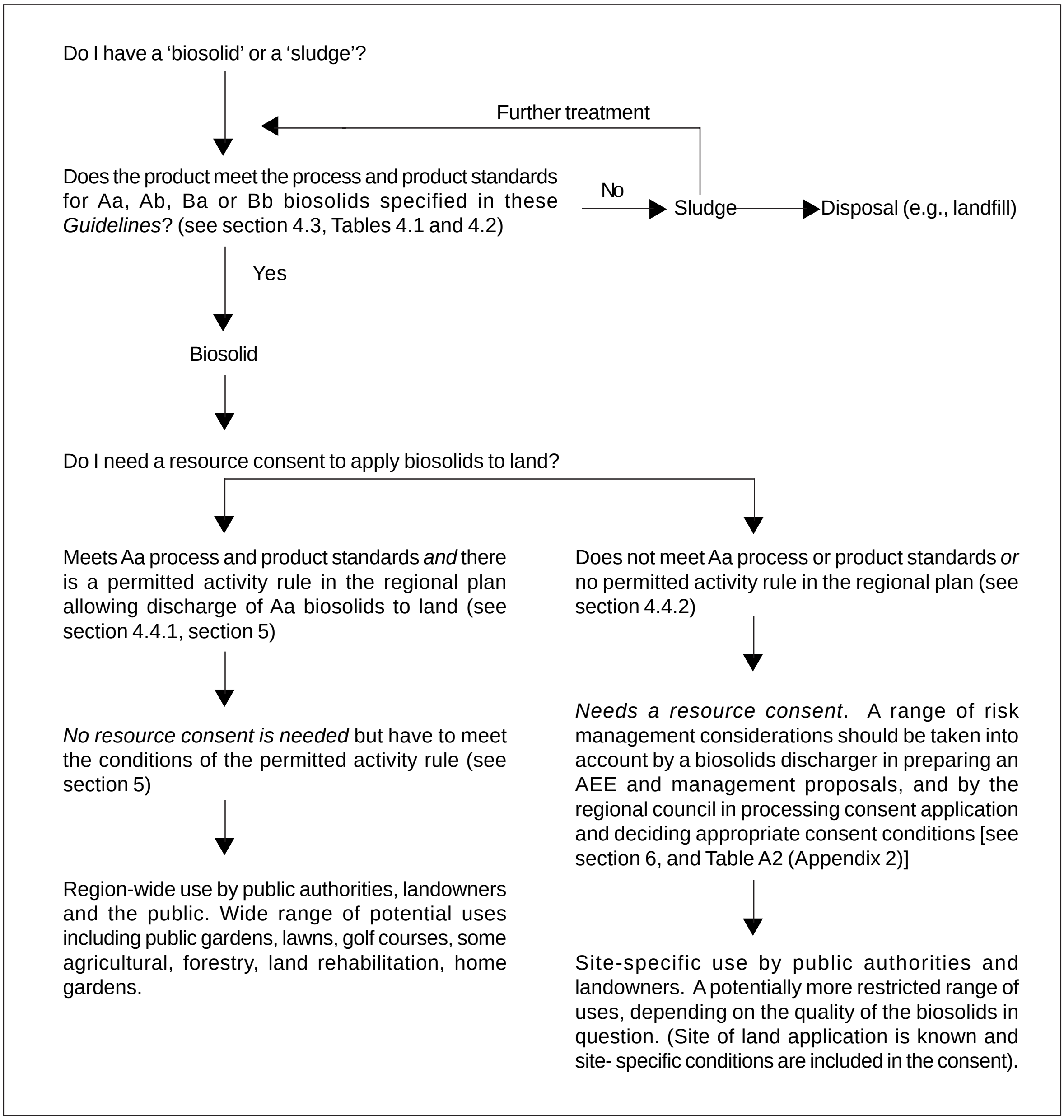
Walmsley, N.A. (2001). Recent changes in New Zealand biosolids production, *Proceedings of the NZWWA Conference 2001*, Wellington.

White, R. R. and Sharpley, A.N. (1996). The fate of non-metal contaminants in the soil environment. In: R. Naidu, R.S. Kookana, D.P. Oliver, S. Rogers and M.J. McLaughlin (eds). *Contaminants and the Soil Environment in the Australasia–Pacific Region: Proceedings of the First Australasia–Pacific Conference on Contaminants and Soil Environment in the Australasia–Pacific Region*, Adelaide, Australia, 18–23 February 1996. Kluwer Academic Publisher, Dordrecht.

APPENDICES

APPENDIX I: THE FRAMEWORK FOR MANAGING BIOSOLIDS DISCHARGES TO LAND

This appendix provides an overview of the management framework for biosolids discharges described in these *Guidelines*, and gives a quick way to see how to manage any biosolids you are dealing with. References to capital ‘A’ or ‘B’ relate to a biosolid’s stabilisation grade, and references to a small ‘a’ or ‘b’ relate to its chemical contaminant grade (see section 4.3 for an explanation of the grading system).



Do I have a ‘biosolid’ or a ‘sludge’?

Does the product meet the process and product standards for Aa, Ab, Ba or Bb biosolids specified in these *Guidelines*? (see section 4.3, Tables 4.1 and 4.2)

Do I need a resource consent to apply biosolids to land?

Meets Aa process and product standards *and* there is a permitted activity rule in the regional plan allowing discharge of Aa biosolids to land (see section 4.4.1, section 5)

No resource consent is needed but have to meet the conditions of the permitted activity rule (see section 5)

Region-wide use by public authorities, landowners and the public. Wide range of potential uses including public gardens, lawns, golf courses, some agricultural, forestry, land rehabilitation, home gardens.

Does not meet Aa process or product standards *or* no permitted activity rule in the regional plan (see section 4.4.2)

Needs a resource consent. A range of risk management considerations should be taken into account by a biosolids discharger in preparing an AEE and management proposals, and by the regional council in processing consent application and deciding appropriate consent conditions [see section 6, and Table A2 (Appendix 2)]

Site-specific use by public authorities and landowners. A potentially more restricted range of uses, depending on the quality of the biosolids in question. (Site of land application is known and site- specific conditions are included in the consent).

Figure A1: Framework for managing biosolids discharges to land

APPENDIX II: SUMMARY OF THE REQUIREMENTS FOR GRADING AND MANAGING BIOSOLIDS

Table A1 summarises the standards or requirements applying to production of the various grades of biosolids recognised in these *Guidelines.*

Table A1: Summary requirements for production of biosolids grades

Requirement	**Biosolids grade**				**Text reference**
	Aa	**Ab**	**Ba**	**Bb**	
Pathogen reduction	✓	✓	✓	✓	Tables 4.1 & 6.2
VAR	✓	✓	✓	✓	Tables 4.1 & 6.2
Contaminant limits:					
■ pathogens	✓	✓	✗	✗	Table 4.1
■ metals	✓	✓	✓	✓	Table 4.2
■ organics	✓	✓	✓	✓	Table 4.2
Process monitoring	✓	✓	✓	✓	Table 4.1, section 7.5.3
Product monitoring	✓	✓	✓	✓	Tables 7.1, 7.2, section 7.5.4
Quality assurance	✓	✓	✓	✓	Section 7
Labelling	✓	✓	✓	✓	Section 7.4
Record keeping	✓	✓	✓	✓	Section 7.3

Table A2 summarises the management controls recommended for the various grades of biosolids recognised in these guidelines.

Table A2: Summary of recommended management controls

Requirement	**Biosolids grade**				**Text reference**
	Aa	**Ab**	**Ba**	**Bb**	
Process monitoring	✓	✓	✓	✓	Table 4.1, section 7.5.3
Product monitoring	✓	✓	✓	✓	Tables 7.1 & 7.2, section 7.5.4
Permitted activity rule	✓	✗	✗	✗	Sections 4.4.1 & 5
Resource consent	✗	✓	✓	✓	Sections 4.4.2 & 6
Transportation	✗	✗	✓	✓	Sections 3.7 & 6.9, Appendix IV
VAR	✓	✓	✓	✓	Tables 4.1 & 6.1
Storage	✗	✗	✓	✓	Table 6.2, section 6.7
N application rate	✓	✓	✓	✓	Section 6.3
Contaminant loading rate	✓	✓	✓	✓	Section 6.3
Soil incorporation	✓	✓	✓	✓	Table 6.2, section 6.11
Exclusion periods	✗	✗	✓	✓	Table 6.2, section 6.12
Soil monitoring	✓	✓	✓	✓	Sections 7.5.6 & 7.6.2
Labelling	✓	✓	✓	✓	Sections 6.8 & 7.4
Record keeping	✓	✓	✓	✓	Sections 6.14 & 7.3
Site management plan	✓	✓	✓	✓	Section 6.15
Environmental management system	✓	✓	✓	✓	Section 7.2

APPENDIX III EXAMPLE OF MASS LOADING CALCULATION

EXAMPLE BIOSOLIDS APPLICATION RATE (MASS LOADING CALCULATION) FOR RESTRICTED USE BIOSOLIDS

ASSUMED BIOSOLIDS DETAILS

Total nitrogen content of biosolids (dry matter)	3	% w/w	Item [1]
Maximum agronomic total nitrogen application rate	200	kg TN/ha/yr	Item [2]
Nitrogen limited biosolids application rate	6.7	dry tonnes/ha/yr	Item [3] = Item [2] / (Item [1] * 10)

ASSUMED SOIL DETAILS

Soil bulk density	1.0	dry tonnes/m^3	Item [4]
Soil depth	200	mm	Item [5]
Incorporated soil mass per hectare (dry tonnes/ha)	2,000	(dry tonnes/ha)	Item [6] = Item [4] * Item [5] * 10

CHECK CONTAMINANTS

Contaminant	Soil limit or ceiling concentrations (mg/dry kg) (Item [7])	Soil contaminant concentration measured in situ (mg/dry kg) (Item [8])	Available capacity of soil to assimilate contaminants (mg/dry kg) (Item [9]) = Item [7] - Item [8])	Biosolids contaminant concentration (mg/dry kg) (Item [10])	Contaminant limited biosolids total application (dry tonnes/ha) (Item [11]) = Item [6] * Item [9] / Item [10])	Contaminant limited biosolids application rate for 20 years (dry tonnes/ha) (Item [12]) = Item [11] / 20
Metals						
Arsenic	20	1	19	8	4,750.0	237.5
Cadmium	1	0.05	0.95	2	950.0	47.5
Chromium	600	30	570	125	9,120.0	456.0
Copper	100	5	95	359	529.2	26.5
Lead	300	15	285	85	6,705.9	335.3
Mercury	1	0.05	0.95	4	475.0	23.8
Nickel	60	3	57	45	2,533.3	126.7
Zinc	300	15	285	867	657.4	32.9
Organics						
Aldrin	0.5	0.025	0.475	0.25	3,800.0	190.0
Dieldrin	0.02	0.001	0.019	0.150	253.3	12.7
Chlorodane	0.02	0.001	0.019	0.150	253.3	12.7
Heptachlor & heptachlor epoxide	0.02	0.001	0.019	0.150	253.3	12.7
Heptachlorobenzene (HCB)	0.02	0.001	0.019	0.150	253.3	12.7
Hexachlorocyclohexane (Lindane)	0.02	0.001	0.019	0.150	253.3	12.7
Total polychlorinated biphenyls (PCBs)	0.3	0.015	0.285	0.15	3,800.0	190.0
Total dioxins PCDD, PCDF	0.0001	0.0000005	0.0000095	0.0001	1,900.0	95.0
Contaminant limited biosolids application rate (minimum individual contaminant rate)						12.7
Nitrogen limited biosolids application rate (refer item [3])						6.7
Overall limit on biosolids application rate (lesser of contaminant and nitrogen limits)						**6.7**

APPENDIX IV: EXAMPLE MATERIALS SAFETY DATA SHEET

Statement of hazardous nature	Hazardous to health. Biological pathogens

Company details

Company:	BNFCL Biosolids Ltd
Address:	
Telephone number:	
Emergency telephone number:	
Product name:	Restricted grades Ba, Bb biosolids
Other names:	Processed sewage sludge
Manufacturer’s product code:	N/A
UN number:	2814
Dangerous goods class:	6.2
Hazchem code:	2X
Toxic Substances Schedule:	N/A
Uses:	Fertiliser, soil conditioner, landfill material
Physical description / properties:	Sludge, approx. 15–20% solids. Non-corrosive. Non-toxic. Potentially objectionable odour. Potential biohazard.
Appearance:	Dark grey/black. Mud-like structure.
Boiling point / melting point (C^{o}):	N/A
Vapour pressures (Pascals or mm of Hg at 25^{o}C):	N/A
Specific gravity:	1.1 (approx.)
Flash point (^{o}C):	N/A
Flammability limits (%):	N/A
Solubility in water (g/l):	N/A

Other properties:

Ingredients:			Wastewater treatment plant waste solids
Chemical name	Gas No:	Proportion:	N/A

Page 1 of 2

Materials Safety Data Sheet - continued

Health effects:	
Acute swallowed:	Potential pathogenic content.
Eye:	No direct effect
Skin:	No direct effect
Inhaled:	No direct effect
Chronic:	
First aid	
Swallowed:	Water rinse, oral hygiene
Eye:	Water rinse
Skin:	Wash
Inhaled:	N/A
First aid facilities:	
Advice to doctor:	Material potentially contains organisms present in faecal matter and wet industry waste.
Exposure standards:	N/A
Engineering controls:	N/A
Personal protection:	Gloves. Minimise contact with exposed skin. Wash hands immediately after contact.
Flammability:	N/A
Storage and transport:	Covered vehicle to minimise odours. Store as non-volatile solids, minimise body contact during handling.
Spills and disposal:	Ensure complete dry clean up of solids with transfer to landfill. Wash contaminated surfaces of spill site.
Fire/explosion:	No

Other information:

Contact point:

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GUIDELINES FOR THE SAFE APPLICATION OF BIOSOLIDS TO LAND IN NEW ZEALAND

VOLUME 2: TECHNICAL MANUAL

CONTENTS

1 INTRODUCTION

1.1 What are biosolids?

Biosolids are sewage sludges, or sewage sludges mixed with other materials, that have been treated and/or stabilised to the extent that they are able to be safely and beneficially applied to land. Biosolids have significant fertilising and soil conditioning properties as a result of the nutrients and organic material they contain.

The term 'biosolids' does not therefore include untreated raw sewerage sludges or sludges solely from industrial processes.[1] Neither does it include animal manures, food processing or abattoir wastes. However, septic tank sludges may constitute biosolids depending on the degree of treatment they have received (see Volume 1, section 1 and Figure 1.1).

Blended biosolid products (e.g., treated sewage sludge mixed with green waste, sand and pumice) are also covered by these *Guidelines*.

1.2 Purpose of the *Guidelines*

This document aims to provide national guidance for managing the application of biosolids to land, and to promote a more consistent approach to the management of biosolids throughout New Zealand.

The *Guidelines* have been produced by a joint committee established by the New Zealand Water and Wastes Association representing a range of interests, including the wastewater industry, user groups, central and local government, and environmental groups. It is anticipated that the *Guidelines* will be used by territorial local authorities, biosolids manufacturers and distributors, biosolids users, regulatory agencies (regional and district councils, government agencies), environmental groups, iwi and the broader community.

The *Guidelines* are designed to provide a framework for biosolids management that enables the land application of biosolids in New Zealand in a way that maximises the benefits and minimises the risks of adverse effects on human health, the environment and the economy. More specifically, the *Guidelines* are aimed at:

- safeguarding the life-supporting capacity of soils
- promoting the responsible use of biosolids
- protecting public health and the environment
- identifying the risks associated with biosolids use and promoting best practice methods of minimising such risks
- encouraging regulatory authorities to adopt a consistent approach to regulating the application of biosolids to land
- creating an awareness within the community of the benefits and risks of biosolids use
- minimising risks to the economy.

The *Guidelines* comprise two volumes: Volume 1, which provides background information and discusses management issues and the recommended regulatory and grading framework; and Volume 2 (this document), which provides detailed information about how the limit values were decided and about how to implement some of the *Guidelines* recommendations.

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[1] While biosolids contain material delivered from industrial inputs to sewers, such material is diluted by the organic material in domestic sewage inputs. Industrial sludges, on the other hand, may contain high concentrations of specific chemicals, little or no organic material, and the organic material that is present may be of a different nature to that contained in sewage. Consequently, any proposal to discharge industrial sludges to land should be treated on its merits.

The *Guidelines* are intended to be a ‘living document’. They are based on current knowledge on the use of biosolids in New Zealand and overseas, and they will be regularly reviewed in the light of future research findings and management experience (see Volume 1, section 8). *Volume 2: Technical Manual* will be a significant resource for these reviews as a record of the reasons supporting the current recommendations.

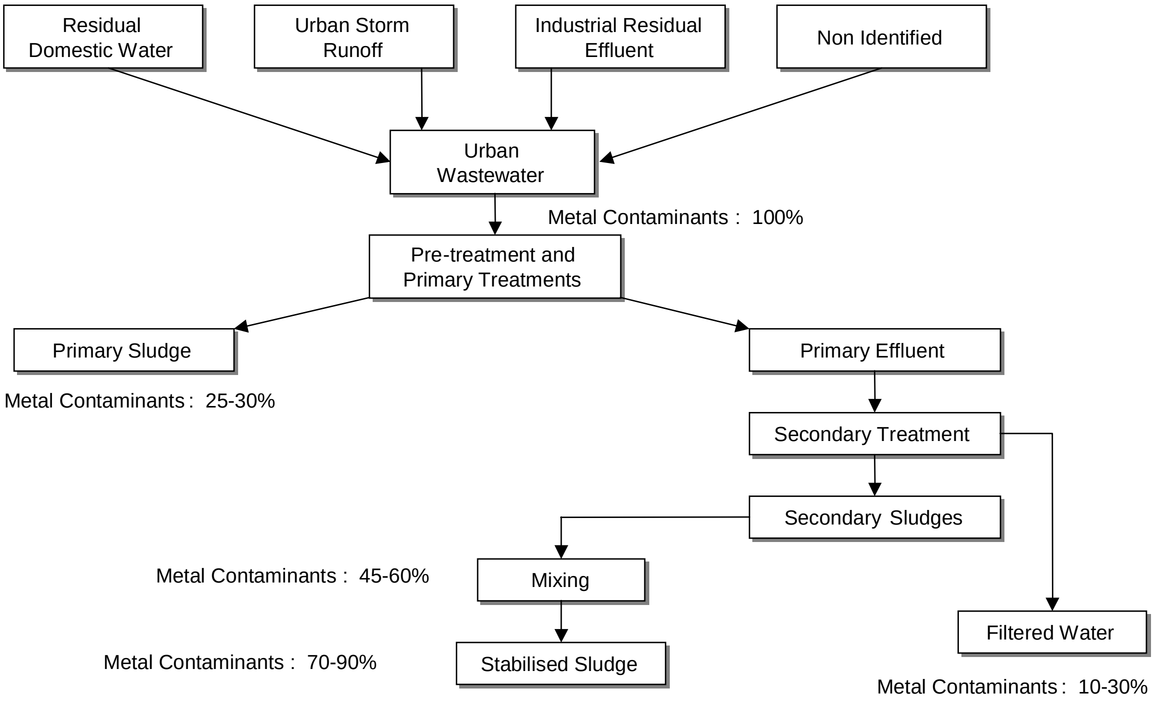
2 SOURCES OF CONTAMINANTS IN SLUDGE

A large range of contaminants are discharged to sewer. These are transferred during the processes of sewage treatment into sludge, which forms the base ingredient for biosolids. Sewage treatment destroys few of these contaminants, merely transferring them from the liquid to the solid phase. To improve waste management practices, an important aim must be to reduce inputs of contaminants entering the wastewater system in the first instance.

This chapter is largely based on a report for the European Commission, *Pollutants in Urban Wastewater and Sewage Sludge* (IC Consultants, 2001), which provides background information on the sources of contaminants in sewage sludge. There have been no comparable New Zealand studies published.

2.1 Metals and contaminants

The majority of metals in raw sewage are transferred to sewage sludge during treatment (see Figure 2.1). However, significant quantities may be lost in the treated effluent depending on the solubility of the metal concerned.



Source: ADEME, 1995.

Figure 2.1: Origin and fate of metals during treatment of wastewater

Average concentrations of metals in German domestic and commercial wastewater are given in Table 2.1. The maximum concentrations found in commercial wastewater are generally greater than those in domestic wastewater.

Table 2.1: Concentrations of metals in domestic and commercial wastewater

Metal	Domestic wastewater (g/m^3)	Commercial wastewater (g/m^3)
Lead	0.1	13
Copper	0.2	0.04–26
Zinc	0.1–1.0	0.03–133
Cadmium	< 0.03	0.003–1.3
Chromium	0.03	20
Nickel	0.04	7.3

Source: Wilderer and Kolb, 1997

Typically, the origin of up to 50% of the metals present in wastewater cannot be accounted for. Better source inventory data are therefore essential in order to effectively target reductions in emissions from all the different sources. Identifying some of the industrial sources may require increased trade effluent discharge controls, while domestic and urban run-off sources may require different types of action, such as changes in the use of products containing these metals.

Emissions of contaminants from industrial point sources used to be the major source of pollution to urban wastewater for most industrialized countries in the northern hemisphere. However, stringent and more widespread tradewaste limits applied to these larger industries have considerably reduced the levels of metals that they discharge into urban wastewater. In many countries there has been a general decline of metals discharged from industrial sources since the 1960s, due to factors such as cleaner industrial processes, trade effluent controls and heavy industry recession.

2.1.1 Domestic sources of metals

Domestic sources of metals in wastewater are rarely quantified because of the difficulty in isolating them from other waste streams. Domestic sources include those metals discharged from the household, as well as corrosion from materials used in distribution and plumbing networks, tap water and detergents. A study by the RIVM (Dutch Institute of Public Health and the Environment) in the Netherlands (Speed, 1993) quantified the waterborne emissions of metals from household sources, dentistry and utility buildings in the urban environment. Table 2.2 shows this data in tonnes per annum.

Table 2.2: Emissions of metals from urban sources in the Netherlands

Metal	Gross waterborne emissions to surface water in 1993 (tonnes/year)		
	Household sources	Dentistry	Utility buildings
Copper	94	0.6	27
Zinc	118	-	26
Lead	113	-	3.1
Cadmium	0.7	-	0.2
Nickel	7.3	-	0.9
Chromium	2.9	-	0.3
Mercury	0.3	2.3	0.01

Source: Adapted from Speed, 1993

Domestic products containing metals used on a regular basis at home and/or at work were reviewed by Lewis (1999). The main domestic sources of metals in wastewater were estimated by WRc (1994) to be (in order of importance):

Cadmium: faeces > bath water > laundry > tap water > kitchen
Chromium: laundry > kitchen > faeces > bath water > tap water
Copper: faeces > plumbing > tap water > laundry > kitchen
Lead: plumbing > bath water > tap water > laundry > faeces > kitchen
Nickel: faeces > bath water > laundry > tap water > kitchen
Zinc: faeces > plumbing > tap water > laundry > kitchen

The following lists the principal metals and the products containing them that can enter urban wastewater.

Cadmium

This is predominantly found in domestic rechargeable batteries (nickel–cadmium batteries), in paints and in photographic chemicals. The main sources in urban wastewater are food products, detergents and bodycare products, and stormwater (Ulmgren, 2000a,b).

Copper

This comes mainly from corrosion and leaching of plumbing, fungicides (cuprous chloride), pigments, wood preservatives, larvicides (copper acetoarsenite) and anti-fouling paints.

Mercury

Most mercury compounds and uses are now (or about to be) banned, although elemental mercury is still used in thermometers and dental amalgam. Mercury can still also be found as an additive in old paints for waterproofing and marine anti-fouling, in old pesticides (including fungicides and insecticides), in wood preservatives, in embalming fluids, in germicidal soaps and antibacterial products, as mercury-silver-tin alloys, and in ‘silver mirrors’.

Nickel

This element can be found in alloys used in food processing and sanitary installations, in rechargeable batteries (nickel–cadmium), and in protective coatings.

Lead

The main source of lead in Europe is from old lead piping in the water distribution systems. This may also be true for New Zealand. It can also be found in old paint pigments (as oxides, carbonates), solder, pool cue chalk (as carbonate), in certain cosmetics, in glazes on ceramic dishes and porcelain (although this use is now banned), and in ‘crystal glass’. Lead has also been found in wines, possibly from the lead-tin capsules used on bottles and from old wine-processing installations.

Zinc

This comes from corrosion and leaching of plumbing, water-proofing products, anti-pest products (including insecticides and fungicides, rat poison, rabbit and deer repellents, and anti-moth agents), wood preservatives, deodorants and cosmetics, medicines and ointments, paints and pigments, printing inks and artist’s paints, a colouring agent in various formulations, a UV absorbent agent in various formulations, and ‘health supplements’.

Arsenic

Arsenic is one of the most toxic metals found in urban wastewaters, and is important because of its ability to cause deleterious effects on human/animal health. Arsenic come from natural background sources and from household uses such as washing products, medicines, garden products, wood preservatives, old paints and pigments. It is present mainly in urban effluents and sewage sludge as dimethylarsinic acid and as As (III) (arsenite) (Carbonell-Barrachina et al., 2000).

2.1.2 The contribution of household products

Several studies have investigated household products as sources of metals entering the sewer (Comber and Gunn, 1996; WRc report, 1994).

There can be a great deal of variability in metal content between products and between types of the same product. The high variability of cadmium concentrations found in big-box washing powders, for example, can be explained by the differences in the composition of phosphate ores used in their production. Reducing the amount of phosphate in washing powders, or choosing phosphate ores with low cadmium concentration, could lead to a reduction in cadmium in wastewater from diffuse sources. In Sweden the amount of cadmium in sewage sludge was reduced from 2 mg/kg dry solids to 0.75 mg/kg dry solids (Ulmgren, 1999), and cadmium discharges from households in the Netherlands have been substantially reduced due to the switch to phosphate-free detergents (Speed, 1993). The ‘ultra’ washing powders, usually phosphate-free, have lower amounts of toxic metals than the traditional powders, and are designed to be used in smaller quantities. A shift to these newer products will reduce the overall metal load from this source.

The products with the highest metal contents include medicated (e.g., anti-dandruff) shampoos, which contain zinc pyrithione. Cosmetics may also contain high levels of zinc, and several of these products will enter the wastewater system. One study in France (ADEME, 1995a) identified the main sources of metals in domestic wastewater as cosmetic products, medicines, cleaning products and liquid wastes (including paint), which were directly discharged from the household sink.

2.1.3 Domestic water and heating systems

Studies in the US (Isaac et al., 1997), and Europe (WRc, 1994) show that corrosion of the distribution/plumbing/heating networks contribute major inputs of lead, copper and zinc. Lead concentrations, for instance, can vary between 14 µg/L at the household input and 150 µg/L at the output. It has been found that concentrations of copper in sewage sludge are directly proportional to water hardness (Comber and Gunn, 1996). Hard water (high pH) is potentially more aggressive to copper and zinc plumbing, increasing leaching. However, the opposite is true for lead, which dissolves more readily in soft, acidic water.

The addition of alkaline agents to water at the treatment stage and the replacement of much lead piping has led to reductions in lead concentrations (Comber and Gunn, 1996). Zinc in domestic plumbing comes from galvanised iron used in hot water tanks, but is less problematic than lead and copper because the amount actually decreases with the ageing of the installations. Copper corrosion and dissolution are also greater in hot water than in cold water supplies (Comber and Gunn, 1996). The ‘first draw’ (i.e. initial flow of water in the morning) has higher amounts of copper and lead compared to subsequent draws (Isaac et al., 1997).

2.1.4 The influence of various treatment processes on the fate of metals and their transfer to sewage sludge

The idea of treating wastewater (sewage) is to remove the various solids and contaminants so as to end up with water that is suitable for discharge to the environment. The material removed is sludge, which, if suitably treated, can become biosolids (see Volume 1, Figure 1.1). This raises the issue that the very nature of the removal process acts to concentrate contaminants, including trace metals. It turns out that the degree of concentration varies according to the type of treatment and the type of metal.

Sludges from conventional sewage treatment plants are derived from primary, secondary and tertiary treatment processes. The contaminant load in the raw wastewater is transferred to the sludge as settled solids at the primary stage, and as settled biological sludge at the secondary stage. Contaminants are also removed along with the solids during the primary and secondary sedimentation stages of conventional wastewater treatment. Metal removal during primary sedimentation is a physical process, dependent on the settlement of precipitated insoluble metals, or the association of metals with settleable particulate matter. Little removal of dissolved metals occurs at this stage, and the proportion of dissolved metal to total metal in the effluent increases as a result.

Just how efficiently the suspended solids are removed is the main factor influencing the extent of metal removal during primary wastewater treatment. However, the relative solubilities of different metals present in the wastewater are also important. Thus, nickel shows the poorest removal (24 %) during primary treatment, whereas 40% of the cadmium and chromium in raw influent is transferred to the primary sludge, and more than 50% of the zinc, lead and copper.

The removal of metals during secondary wastewater treatment is dependent on the uptake of metals by the microbial biomass, and the separation of the biomass during secondary sedimentation. Several mechanisms are important here, including physical trapping of precipitated metals in the sludge floc, and binding of soluble metal to bacterial extracellular polymers. The patterns of metal removal from settled sewage by secondary treatment are similar to those recorded for primary sedimentation. However, general surveys of removal efficiencies suggest that secondary treatment (by the activated sludge process) is more efficient at removing certain metals (e.g. chromium) than the primary stage.

Operational experience and metal removal measured by experimental pilot plant systems can indicate the overall likely transfer to sludge of toxic metals from raw sewage during conventional primary and secondary wastewater treatment. This shows that approximately 70–75% of the zinc, copper, cadmium, chromium, mercury and arsenic in raw sewage is removed and transferred to the sludge (Blake, 1979), and concentrations of these metals in the final effluent would be expected to decrease by a similar amount compared with the influent to the water treatment plant. Up to 80% of lead may be removed, whereas the smallest overall reductions are obtained for nickel, approximately 40% of which may be transferred to the sludge.

Most of the metals in raw sewage are partitioned during wastewater treatment into the sewage sludge or the treated effluent. However, atmospheric volatilisation of mercury as methylmercury, formed by aerobic methylation biotransformation processes, is also suggested as a possible mechanism contributing to the removal of this element during secondary wastewater treatment by the activated sludge system (Yamada et al., 1959). However, although some of the mercury removal observed in activated sludge may be attributed to bacterially mediated volatilisation, it is unlikely that this is a major route of mercury loss because of the significant quantities of this metal recovered in surplus activated sludge (Lester, 1981).

2.1.5 The New Zealand context

There have been no New Zealand studies comparable to those summarised above. Similar findings would, however, be expected to emerge from any such studies due to similarities in the lifestyles and consumer products used in this country, although New Zealand water supplies tend to be more aggressive (corrosive) than many European waters and this may increase the tendency for metal dissolution into the wastewater stream and thereby into sewage sludge.

2.2 Organic contaminants

2.2.1 Sources and pathways

A large number of organic contaminants from a wide range of sources may enter urban wastewater. Paxéus (1996) identified over 137 organic compounds in the influent of municipal wastewater plants in Stockholm. However, these *Guidelines* are limited to controls on three groups of organic contaminants: persistent organochlorine pesticides, polychlorinated biphenyls (PCBs) and dioxins.

Persistent organochlorine pesticides

Organochlorine pesticides include DDT, aldrin, dieldrin and lindane. They were heavily used in New Zealand from the mid-1940s to the 1970s in agriculture, horticulture, timber treatment and public health. Smaller amounts were also used for amenity purposes and in households. Their use was progressively restricted by a succession of legislation, so that by the mid 1970s their use in agriculture and horticulture had effectively ceased. They were deregistered by the Pesticides Board in 1989. A chronology of organochlorine pesticide use in New Zealand is given in Buckland et al., (1998a).

Polychlorinated biphenyls

There are two main sources of Polychlorinated biphenyls (PCBs).

- Directly manufactured PCBs. These included their use as hydraulic liquids (hydraulic oils), emollients for synthetic materials, lubricants, impregnating agents for wood and paper, flame-protective substances, carrier substances for insecticides and in transformers and condensers. The use and storage of PCBs in New Zealand has been banned under the Toxic Substances Regulations (although a few exemptions have been given).
- PCBs as by-products. The other main source of PCBs in the environment is their emission as a by-product from thermal processes, including emissions from waste combustion and incineration plants.

PCBs are adsorbed by solids and therefore accumulate in sewage sludge. The highly substituted (high chlorine content) PCBs are the main representatives present in sewage sludge, but they amount to just 35% of the total PCBs.

Polychlorinated dibenzo-p-dioxins and dibenzofurans

‘Dioxins’ is a generic term covering a mixture of 210 different polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans. The most well-known and toxic dioxin is 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD). Dioxin concentrations are often expressed as the sum of the toxicity equivalents (TEQ) relative to the most toxic dioxin (TCDD).

Dioxins are created by some industrial processes and when organic material is burnt in the presence of chlorine. Burning wastes, chlorine bleaching of pulp and paper, and some industrial processes can all create small quantities of dioxins. They may also be formed from natural sources such as forest fires.

Burning waste material in incinerators, uncontrolled fires at landfills, domestic backyard fires, burning wood and coal, and processing metal are the major sources of dioxin emissions to air. Industrial sources are estimated to produce approximately 60% of New Zealand’s total emissions to air. Domestic activities (e.g., backyard burning of waste and burning wood to heat our homes) are other important sources.

New Zealand also has reservoirs of dioxin in contaminated soils, waste dumps, and landfills. The dioxins present in these reservoirs can potentially be redistributed back into our environment over a long period. The importance of these reservoirs relative to current ongoing emissions is unknown (Buckland et al., 2000).

The kinds of dioxins present in household wastewater and laundry wastewater are similar to those found in the sediments of sewer collecting systems and sewage sludge. A mass balance indicates that between two and seven times more dioxins in sewage sludge originate from households than from urban run-off. Washing machine effluent is a major source of dioxins in household wastewater. They have also been detected in shower water, and in urban run-off from various human activities (Horstmann et al., 1993; Horstmann and McLachlan, 1995). Buckland (1999) has published a review of the sources of dioxins in sewage sludge.

2.2.2 Partitioning of organic contaminants in wastewater treatment processes

As with metals, the general effect of wastewater treatment processes is to concentrate the organic contaminants (i.e. persistent organochlorine pesticides, PCBs and dioxins) in the sewage sludge. These organochlorines have a very low water solubility, and bind strongly to the particulate and organic matter in the sludge. The overall result of wastewater treatment processes is to discharge a treated wastewater relatively free of organic contaminants, leaving a sewage sludge that contains most of the organic contamination that was present in the feed wastewater.

2.2.3 The New Zealand context

The Ministry for the Environment's Organochlorines Programme has undertaken a series of national studies to identify the sources of dioxins in New Zealand (Buckland et al., 2000), and to measure the levels of organochlorines in our atmosphere (Buckland et al., 1999), terrestrial (Buckland et al, 1998a) and aquatic ecosystems (Buckland et al., 1998b and Scobie et al., 1998), and foods (Buckland et al., 1998c), and in the bodies of New Zealanders (Buckland et al., 2001). Further details of the Organochlorines Programme can be found on the Ministry for the Environments website: http://www.mfe.govt.nz/issues/hazardous/contaminated/organochlorines.html

The studies found that the background levels of dioxins and PCBs in New Zealand are generally low relative to other countries. Also, the levels of discharges identified are at the low end of the scale compared to discharges in the northern hemisphere. These lower discharges are thought to be due to the lower level of industrialisation in New Zealand and our lower population density rather than to the use of better environmental management practices.

The dioxin inventory has assessed the total amount of dioxin that could be present in all New Zealand sewage sludge and discharged to land (Buckland et al., 2000). It estimated a total annual release range of between 0.61–5.2 g TEQ per year.

3 RISK ASSESSMENT

3.1 Introduction

This risk assessment of biosolids is not based on strict quantitative risk because there are insufficient data available for New Zealand. It is estimated (NRMMC, 2003) that at least 10 years' research is required before such an analysis is possible. A qualitative analysis has therefore been used, having regard to the precautionary principle.

The risks associated with the beneficial use of biosolids are described in the following sections (see also, Volume 1, section 2).

3.2 Risks to plant health

Biosolids are applied to plants because the nutrients and trace metals they contain are usually beneficial to crop growth and health. However, some metals present in biosolids are only beneficial to plant health/growth at particular concentrations, and beyond that level may be detrimental to the plant. Copper, nickel and zinc are the main metals that can have toxic effects on plants. The limits set for biosolids in these *Guidelines* (see section 7, and Volume 1, section 4) are designed to ensure that the concentrations of these metals in soils after the application of biosolids do not have any phytotoxic effect on any plants present.

Cadmium, while not apparently phytotoxic, can accumulate in plant tissues to such an extent that it becomes toxic to humans and/or grazing animals. The levels given for cadmium in these *Guidelines* are designed to prevent this from occurring.

Arsenic, mercury, chromium (in the form of chromate [VI]) and lead may also be toxic to plants. However, the majority of plants do not take these metals up easily and so are only likely to be affected by the presence of these metals at high concentrations. Once again, the limits given in these *Guidelines* are designed to prevent this from happening.

The pH of the soil can affect the mobility of certain metals, with mobility increasing as the soil becomes more acid. In Europe, several countries (UK, Spain and Portugal) have tried to address this bio-availability issue by giving different soil limits for soils of different pH. The mechanisms behind bio-availability are not well understood and it is felt that more research needs to be undertaken before this type of approach is used in New Zealand. These *Guidelines* assume 100% bio-availability of all metals, which is extremely unlikely and therefore represents a margin of safety built into the recommended limits.

There is no evidence to suggest that plant diseases are transmitted in sewage sludge or biosolids (Smith, 1996).

3.3 Risks to animal health

Animals may be exposed to biosolids if they are used as a fertiliser on paddocks and pastures. Animal fodder may also be grown in fields treated with biosolids. However, the main risk to animal health is the direct ingestion of biosolids by livestock when grazing on treated pasture.

Cadmium, mercury and copper are particularly toxic to animals if ingested, and there is some concern that animals grazing on biosolids-treated grass could be affected by these metals (DEFRA, 1998). However, the likelihood of toxic effects occurring if animals are fed food that has been treated with biosolids, rather than ingesting the biosolids themselves, is thought to be low (Wellington City Council, 1997).

Concern has also been raised regarding the accumulation of some organic compounds in the tissues and milk of grazing animals. While there is no evidence that this type of accumulation is detrimental to the animal (Smith, 1996), it may be harmful to humans who eat the meat and drink the milk of animals who have accumulated these organic compounds.

3.4 Risks to soil microbial processes

There is much debate over the effect of metals present in biosolids on soil micro-organisms and microbial activity, and much of the available literature is contradictory. This is because the toxicity of metals is dependant on many factors, including soil pH, the tolerance of soil micro-organisms to the metal being investigated, the presence of other metals and the soil type. The debate is further complicated by the interactions that can occur between some of these factors.

Initial concern regarding soil microbial processes was raised after an experiment conducted at Woburn in the UK in 1984 indicated that nitrogen-fixing bacteria were adversely affected by the application of sludge to the soil. However, subsequent studies have shown little or no effect (see Volume 1, section 2.3.4).

Further investigation is being conducted into this issue in New Zealand, the US and Europe, and these *Guidelines* will be reviewed in the light of results from this work (see Volume 1, section 8).

3.5 Risks to human health

Risks to humans can come both from direct exposure to biosolids and from eating food that has been grown on land to which biosolids have been applied. The risks posed to humans can be divided into three categories: pathogens, metals and organics. These are discussed in turn below.

3.5.1 Pathogens

All sludges that are used to produce biosolids will contain pathogens in varying numbers. The number of pathogens in the final product will depend on the treatment used to produce the biosolids. The methods used to apply the sludge to land can further decrease the risk to humans by affecting the rate of die-off, reducing the numbers present and/or decreasing the likelihood of human contact.

Pathogen numbers can be reduced directly or indirectly by:

- sunlight
- ambient temperature
- desiccation
- soil pH
- soil characteristics
- presence of competing organisms
- quantity of sludge spread.

Irrespective of the numbers or type of pathogens present in the final product, the application of biosolids to the surface of the land is more effective at reducing pathogen numbers than incorporating the product into the soil, due to the effects of sunlight, air temperature, etc. However, although it may not reduce pathogen numbers to the same extent, incorporating biosolids into the soil has the effect of reducing the probability of human contact, so in terms of reducing the risk to human health it can be as (or more) effective than surface application. For this reason and others (see section 3.5.5) soil incorporation is preferred for all biosolids applications.

Workers involved in the production of and/or application of biosolids are particularly at risk from exposure to pathogens and should follow the appropriate safety procedures.

3.5.2 Metals

The US National Research Council (1996) has reported that no adverse human acute or chronic toxic effects have been reported as the result of ingesting foods grown in soils amended by sludges and/or biosolids. This is probably because any plants that contain metals at a level harmful to humans would be so damaged themselves they would be unsuitable for sale.

In terms of human exposure, the main metals of concern are cadmium, lead and mercury. Direct ingestion is considered to be the most critical pathway for these metals, and this is based on children playing in domestic gardens to which biosolids have been applied (USEPA, 1995). Even though the USEPA approach (USEPA, 1993) is based on conservative assumptions, the biosolids limits given for metals in the US standard are often much higher than those set in Europe and in Australia.

3.5.3 Organic compounds

The risk to humans from exposure to organic compounds found in biosolids is thought to be minimal. However as stated in section 3.3, some organic compounds, including the PCBs and persistent organochlorine pesticides, have been found to accumulate in the meat and milk of livestock, and these may therefore be passed on to human consumers of animal products.

It should also be noted that, historically, some wood treatments used pentachlorophenol (which is contaminated with dioxins) and other organochlorine pesticides, so sawdust and wood chips used as a co-product in the composting process must come from non-treated wood.[2]

The USEPA is currently undertaking research to determine an acceptable level of dioxins in biosolids, the results of which are expected sometime in 2003. In response to this the NSW EPA has omitted standards for dioxins from its guidelines until the USEPA issues further guidance. However, this approach was not thought suitable for New Zealand and a dioxin limit has been set in these *Guidelines* (see Section 7.2).

3.5.4 Risk to groundwater and surface water

The leaching of nutrients such as nitrogen and (to a certain extent) phosphorus is the main risk posed to groundwater by the application of biosolids. Limiting application rates by linking them to the agronomic rate of nitrogen uptake should help to resolve this issue (see Volume 1, sections 2.3.1 and 6.3). Nitrogen leaching can also be reduced by applying biosolids at or as close to the time when maximum crop growth and nitrogen uptake occur. Compared to nitrogen, phosphorus is relatively immobile in soils and will not leach at the same rate.

Metals are unlikely to move through the soil and into groundwater because of the binding mechanisms in soil. However, some movement may occur through acidic, sandy soils under conditions of high biosolids application, coupled with either irrigation or high rainfall.

The organic contaminants covered by these *Guidelines* are unlikely to move from biosolids to groundwater because of their low water solubility and the binding properties of the soil.

Risks to surface water are similar to those discussed above for groundwater. They can be reduced by ensuring that biosolids are not spread too close to watercourses, on waterlogged or steeply sloping land or during periods of heavy rainfall (see section 8.4).

3.5.5 Risks to air quality

The main risk to air quality from the application of biosolids is odour. This can be controlled by ensuring that the biosolids are incorporated into the soil within a few hours of application. Incorporation of biosolids will also reduce ammonia emissions, and therefore the amount of nitrogen lost to the atmosphere.

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[2] This requirement will also prevent biosolids being contaminated with copper, chromium and arsenic from CCA treated timber.

4 SOIL CHARACTERISTICS AND INTERACTION WITH BIOSOLIDS CONTAMINANTS

4.1 Introduction

The interaction and fate of contaminants in biosolids applied to the soil are fundamental to the short- and long-term effects of biosolids use. This chapter summarises the behaviour of contaminants in soil in sufficient detail to enable the user to understand the issues and complexities. It also discusses the effects of land management practices on contaminant uptake and mobility.

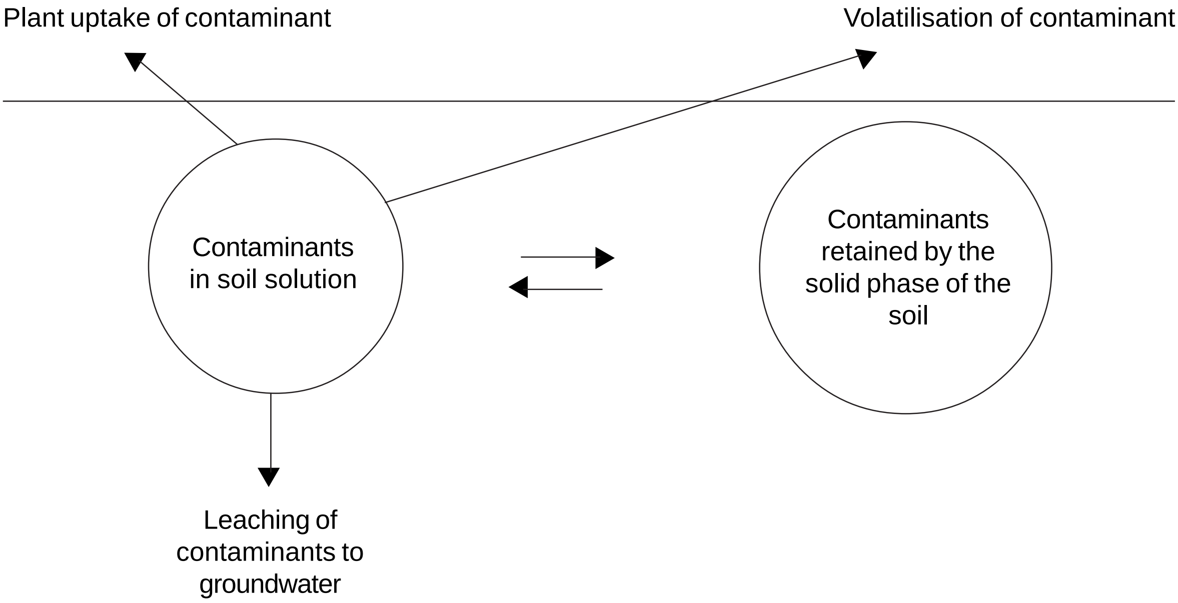


Figure 4.1: Simplified depiction of the fate of contaminants in the soil

Contaminants can be involved in many different reactions and processes in the soil, but their ultimate fate can be summarised as shown in Figure 4.1. Essentially, contaminants can:

- react with and become retained by the solid phase of the soil
- be volatilised into the atmosphere as a result of various physical, chemical and biological transformations
- be taken up by plants
- be leached out of the soil into drainage water
- be removed in soil ingested by grazing animals
- be transported to another location in surface run-off.

Not every contaminant is subject to all of the above processes, and in some cases a single process may dominate the fate and potential adverse effects of a contaminant. For example, most metals are not subject to volatilisation losses, and the fate of nitrogen is often dominated by leaching from the soil.

4.2 Retention processes

Contaminants entering the soil, whether metals or organic compounds, are subject to numerous chemical and biological processes that have implications for contaminant bio-availability and mobility. Some of the most important processes are those responsible for the accumulation of contaminants in the solid-phase components of the soil, processes collectively referred to as ‘contaminant retention’. Metals or organic chemicals retained by the soil are generally considered to be less bio-available and mobile than those remaining in the soil solution.

4.2.1 Metal retention

Exchangeable metals

Metals that occur as cationic species might be expected to take part in normal cation exchange mechanisms in the soil (i.e., non-specific, electrostatic attraction of cations to negatively charged soil colloids). Similarly, metals that occur in anionic forms may interact with anion exchange sites. However, it seems unlikely that ion exchange is a particularly important mechanism for the metallic elements (Swift and McLaren, 1991; Barrow, 1999). The concentrations of major nutrient ions (e.g., Ca^{2+}, Mg^{2+}, K^+, NO_3^-, SO_4^{2-}) in the soil solution are several orders of magnitude greater than those of the metals. Thus, mass action (competition) effects should prevent the occurrence of significant concentrations of metals on exchange sites, except in grossly contaminated soils. This is confirmed by the observation that only extremely small concentrations of exchangeable metals are determined in soils using standard techniques of extraction; i.e., extraction with salt solutions (Tiller et al., 1972; McLaren and Crawford, 1973a; Shuman, 1979).

Metals specifically sorbed by inorganic soil colloids

The sorption of trace metals by inorganic soil colloids such as alumino-silicate clays and iron, aluminium and manganese oxides and hydroxides are considered to occur mainly by specific sorption mechanisms (Swift and McLaren, 1991). This means that the soil colloid shows a preference for the sorbed substance, so that metal ions such as Cu^{2+} or Zn^{2+} may be sorbed despite the presence of much higher concentrations of Ca^{2+} or Mg^{2+} (Barrow, 1999). Unlike the mechanism of ion exchange that involves electrostatic bonding only, specific sorption involves the formation of a stable chemical bond between a trace element ion and certain functional groups at the surface of the soil colloid. Evidence for the formation of these bonds has been obtained by both indirect and direct methods (McBride, 1989). However, it should be appreciated that as a result of the large number of chemical species, surfaces and bonding mechanisms involved, specific sorption processes in soil are extremely complex.

Metals complexed or chelated by organic colloids

As with inorganic colloids, soil organic colloids are able to bind substantial concentrations of metals, and with some metals (e.g., copper) the bonds formed are very strong. It is generally assumed that bonding is the result of complex formation involving the various functional groups (carboxyl, hydroxyl, etc.) found in soil organic matter in general, and humic substances in particular (Stevenson, 1982). Where ring structures are formed by this type of reaction, the element is said to be chelated. The complexation of trace metals by organic colloids can also be regarded as an example of a specific sorption mechanism.

Metals occluded by, or as structural components of, secondary minerals and other inorganic compounds

Trace elements sorbed on to the surfaces of iron, aluminium and manganese oxides may eventually be occluded by further growth of the oxides. Alternatively, metals may be substituted in the structures of oxide minerals during their formation in soils. For example, there is good evidence for the substitution of trace metals in the structures of a range of iron oxides and oxyhydroxides (Gilkes and McKenzie, 1988; Cornell and Schwertmann, 1996). In some cases it may also be possible that metals are precipitated in the form of simple inorganic compounds such as carbonates, phosphates, sulphides or hydroxides. However, this is most likely to happen only in heavily contaminated soils, under waterlogged or high pH conditions, or in calcareous soils. For most soils there is little evidence for the existence of such compounds (Swift and McLaren, 1991).

4.2.2 Factors affecting metal retention

Soil composition

Since metals are retained predominantly by soil organic matter, soil oxides and clay minerals, those soils with high contents of these constituents will have a larger capacity to retain metals. Conversely, soils with low organic matter, low oxide and low clay contents are likely to have a limited ability to retain metals.

Soil pH

Numerous studies have examined the effect of pH on metal retention processes. In general it has been observed that pH has a profound effect on metal retention behaviour by all major soil constituents (Swift and McLaren, 1991). For example, metal retention by soil organic matter increases substantially from pH 2 to 6 (Kerndorff and Schnitzer, 1980; McLaren and Crawford, 1973b; McLaren et al., 1986). Similarly, for those metals occurring as cations, retention by inorganic soil constituents (oxides and clay minerals) also increases with an increase in soil pH.

For metals or metalloids occurring as anions, the picture is more complex (e.g., molybdenum, selenium and arsenic). In some cases, retention of these metals decreases with an increase in pH (e.g., Hingston et al., 1972), and in other cases retention increases to a maximum and then decreases with a further increase in pH (e.g., Smith et al., 1999). The exact nature of the pH effect in these cases will be determined by factors such as the types of soil constituent involved and the presence of co- or competing ions.

Soil redox conditions

Redox (reduction–oxidation) conditions affect metal retention in two main ways. Firstly, under waterlogged, anaerobic conditions oxides of iron and manganese become reduced and are solubilised, releasing any bound metals back into solution. Secondly, a small group of metals or metalloids (arsenic, selenium and mercury) are capable of being volatilised from the soil under reducing conditions (section 4.3).

4.3 Contaminant volatilisation

Under anaerobic conditions, both bacteria and fungi have the ability to transform arsenic into volatile forms, predominantly dimethlyarsine and trimethylarsine gases (Tamaki and Frankenberger, 1992). Similarly, selenium volatilisation from soils can occur as a result of microbial methylation reactions (Haygarth, 1994). Numerous bacteria and fungi appear to be responsible for this process, and the main volatile species produced is the non-toxic dimethylselenide (Frankenberger and Losi, 1995).

Mercury, like arsenic and selenium, can also undergo microbial methylation reactions in the soil to produce methyl mercury species (Kabatas-Pendias and Pendias, 2001). Another naturally occurring process is microbial reduction of mercuric ions (Hg^{2+}) to elemental mercury (Hg^0), which can then be volatilised from the soil. There is good evidence that Hg can be volatilised from soils contaminated with a variety of mercury-containing materials, including municipal sewage sludge (Carpi and Lindberg, 1997).

Some organic compounds (e.g. some pesticides) will also volatilise, a process determined predominantly by the solubility and vapour pressure of the particular compound. In addition, all organic molecules will undergo some biological and/or chemical decomposition or transformation in the soil. For those organic compounds that are easily decomposed, their retention in the soil will not be a major issue. However, for the persistent pesticides, PCBs and dioxins that are addressed by these *Guidelines*, they decompose only extremely slowly, and as they are not lost by volatilisation or leaching to any significant extent, they will be retained in the soil for considerable periods of time (years to decades).

The nitrogen added in biosolids can also be lost from the soil to the atmosphere through volatilisation as ammonia (NH_3), or by denitrification, predominantly in the form of nitrous oxide (N_2O), but also possibly as nitric oxide (NO) and nitrogen gas (N_2) (McLaren and Cameron, 1996). Some biosolids are rich in ammonium-N, and substantial volatilisation of ammonia gas may take place immediately following application to the soil, particularly under conditions of high pH and with surface applications. Denitrification is most likely to take place under wet, anaerobic conditions.

4.4 Contaminant bio-availability

4.4.1 Metal bio-availability

To be available for uptake by plants (or soil micro-organisms), metals must be present in the soil solution. Solution metal concentrations are controlled in two main ways:

- by the solubility of solid-phase compounds containing the metal of interest
- by sorption/desorption reactions at the surfaces of soil colloids.

Solubility of their respective oxides is probably the main factor controlling solution concentrations of aluminium, iron and manganese, but for all other trace metals sorption/desorption reactions are likely to be the major types of mechanism involved (Swift and McLaren, 1991). Those trace metals present in the crystal structures of primary and secondary silicate minerals, or occluded by oxides (unless soil conditions favour their dissolution), will clearly be unavailable for plant uptake.

To a large extent metal concentrations in the soil solution will be inversely related to the metal retention properties of the soil solid phase; that is, the greater the retention, the lower the metal concentration in solution. Thus any factor that affects metal retention (see above) will also affect metal solution concentrations (e.g., pH and soil composition).

However, other factors may also influence solution metal concentrations and hence bio-availability. For example, the role of the soil biomass, although by no means clear, is likely to be important. Only small amounts of trace metals will be present in the biomass, but the release of decaying organic materials and the continual cycling of trace metals through the soil microbial population could have a significant effect on maintaining solution metal concentrations.

Chemical speciation of trace metals in the soil solution

There is considerable evidence that the chemical speciation of trace metals in solution affects their availability and/or toxicity to plants (Parker et al., 1995). For example, many studies have shown a high level of correlation between plant uptake of trace metals and the activity of free, uncomplexed metal ions in solution, such as Cu^{2+} (Graham, 1981) and Cd^{2+} (Cabrera et al., 1988), rather than with total metal concentrations. Such studies have been interpreted as demonstrating that complexed forms of trace metals in solution are unavailable for plant uptake. However, other studies have suggested that this is not necessarily the case. For example, DeKock and Mitchell (1957) demonstrated increased uptake of aluminium and other trivalent metals by chelators such as EDTA and DTPA. More recently, Smolders and McLaughlin (1996) and Weggler-Beaton et al. (2000) have demonstrated that Cl-complexed Cd may also be available for plant uptake (in addition to Cd^{2+}).

The importance of complexed trace metal species for plant uptake from soils is difficult to assess at this stage. Much of the research has been carried out in solution cultures rather than in soil. Whether some complexed species are indeed taken up by plants, or whether complexation can enhance availability primarily by increasing the diffusion of trace metals to plant roots, remains to be determined.

Soil pH

Soil pH is recognised as having a major influence on the availability of trace metals. For those trace metals that occur predominantly as cations (e.g., Cu^{2+}, Co^{2+}, Pb^{2+}), their availability to plants is highest in acid soils, and decreases as the soil pH increases. Conversely, those trace metals such as arsenic, molybdenum and selenium that occur as anions are most available in soils of high pH and least available in acid soils.

The influence of soil pH on trace metal availability is due mainly to its effect on the reactions controlling trace metal concentrations in the soil solution. Under acid conditions, sorption of trace metal cations by soil colloids is at a minimum, and thus solution concentrations are relatively high. In addition, the solubilities of iron and manganese oxides are high under low pH conditions. As the soil pH rises, the sorption of trace metals increase and the solubility of oxides decrees. The sorption of those metals that occur in anionic forms decrees with increasing soil pH, and hence solution concentrations and their availability increase.

A complicating factor with certain trace metal cations (e.g., Cu^{2+} and Pb^{2+}) is that as soil pH increases, metal solubility reaches a minimum between pH 6 and 7 and then rises again at even higher pH (e.g., McBride and Blasiak, 1979; Bruemmer et al., 1986). This is mainly due to increased solubility of organic matter at high pH, causing the retention of trace metals in solution in the form of soluble organic complexes. However, as discussed above, the extent to which trace metals occurring as complexes will be available for plant uptake is unclear.

Metal desorption

The immediate bio-availability of trace metals depends primarily on their concentration and speciation in the soil solution. However, continuing bio-availability depends on the soil's ability to release trace metals from the solid phase to replenish those removed by plant uptake. It is now generally accepted that in the medium to long term, solution concentrations of trace metals are most likely to be controlled by sorption–desorption reactions at the surfaces of soil colloids (Swift and McLaren, 1991).

Desorption of soil-retained trace metals shows a variety of trends, depending on the nature of both the surface and the trace metal being studied. These trends range from (a) complete desorption, to (b) significant desorption but with a proportion of trace metal retained, to (c) minimal desorption, with a high proportion retained by the surface. Many trace metals fall into category (c), appearing to be strongly retained by soils and showing limited reversibility of sorption (desorption).

As with sorption, desorption of trace metals from soils is affected markedly by soil pH. In particular, desorption of both native (i.e. naturally occurring) and applied trace metals has been observed to decrease with increasing soil pH (e.g., McLaren et al., 1997; Gray et al., 1998).

Ageing effects

The issue of whether the bio-availability of contaminant metals added to soils decreases with time is somewhat contentious, particularly in relation to metals added in biosolids. However, there is good evidence from laboratory studies using simple metal salts that increasing the contact time between soil and added metal can decrease the metal's subsequent ability to desorb from the soil (Barrow, 1986; Hogg et al., 1993; Gray et al., 1998). Such findings have been linked to observations that following the initial rapid sorption of metals by soil oxide materials, continuing slow reactions between the metal and the oxides take place (Benjamin and Leckie, 1981; Bruemmer et al., 1988; Backes et al., 1995). Whether such processes involve solid diffusion of metals into the lattice structure of oxides (which would be expected to be extremely slow) or diffusion of ions into very small pores and inter-particle spaces remains to be determined (McBride, 1991).

Whatever the mechanism, there is evidence to link such slow reactions with decreased availability of metals to plants. For example, Brennan et al. (1980) have reported reduced copper availability to plants with increasing contact time between added copper and the soil. Similar observations have been made by Brennan (1990) in relation to the availability of zinc to plants. More recently, Hamon et al. (1998) have produced evidence that cadmium added to soils in superphosphate fertiliser becomes less bio-available with time. These researchers estimated that cadmium was being fixed very slowly in non-bio-available forms at a rate of 1–1.5% of the total cadmium added per year. This observation is supported by the work of Gray et al. (1999), who, using a sequential fractionation technique, determined that a substantial proportion of cadmium applied to a soil in superphosphate over a 44-year period had been incorporated into a residual soil fraction. McLaren and Ritchie (1993) have made similar observations in relation to the long-term fate of copper in soil.

The evidence referred to above suggests that the bio-availability of metals, in simple forms and at relatively low levels of contamination, may well decrease significantly over time. However, with metals added in biosolids the picture is far from clear. Research in this area has been discussed in detail by

Smith (1996). In summary, whereas some studies have indicated a decrease in metal bio-availability with time after biosolids application has ceased, others have not. A major complicating factor with biosolids studies is that in addition to metals, large amounts of organic matter are also added to the soil, sometimes along with substantial quantities of other inorganic materials. The effect of the decomposition of this organic matter and the presence of inorganic metal sorbents on the long-term bio-availability of metals is probably the major issue facing sludge researchers at present.

4.4.2 Bio-availability of organic chemicals

Compared with research on metals in biosolids, comparatively little work has been done on the fate of the organic chemicals present in such material. However, O'Connor et al. (1991) carried out a detailed review of the bio-availability to plants of sludge-borne toxic organics. They concluded that the vast majority of these chemicals in sludges occur initially in sludge-amended soils at low concentrations, and are so strongly sorbed in the sludge–soil matrix as to have low bioavailabilities to plants. In addition, the large size of many organic molecules precludes their uptake by plants.

4.5 Contaminant mobility

Contaminants added to soil in biosolids can be mobilised by two main processes: surface run-off and leaching downwards through the soil profile. Surface run-off involves bulk movement of material (biosolids or a soil/biosolids mixture) and associated contaminants over the surface of the land in water that is unable to infiltrate readily into the soil. If such material finds its way into surface water bodies (e.g., streams), serious contamination of the water may result. Management strategies to minimise contaminant mobility are discussed in sections 4.8 and 8.4, and Volume 1, section 6.2.

Leaching through the soil profile is probably more complex and difficult to control, and occurs because water moving through the soil generally transports any dissolved solutes with it. These solutes may be trace metals, organic molecules or nutrient ions such as nitrate nitrogen, all of which are added to the soil in biosolids applications. In some cases such transport can result in the pollution of groundwater.

4.5.1 Principles of solute transport

Solute movement occurs through a combination of three main mechanisms: convection, diffusion and dispersion (McLaren and Cameron, 1996). Convective transport results from the movement of solutes with the mass flow of water in soil, and can be described by a modified form of Darcy's Law. Convective transport is often referred to as 'piston displacement', and the distance of transport per unit time depends on the average pore water velocity. In reality, the solute does not remain as a sharp band but tends to spread throughout the profile due to the processes of diffusion and hydrodynamic dispersion.

Diffusion occurs when there is an uneven distribution of solutes in a solution, causing a diffusive flux of solute from areas of high concentration to areas of lower concentration. Hydrodynamic dispersion is caused by the mechanical action of a solution flowing through soil, which tends to cause mixing and equalises the solute distribution. This process enhances the dispersive effect of diffusion and during flow it usually completely masks it.

Hydrodynamic dispersion occurs because (i) the flow velocity within a single pore is not uniform, (ii) the large variation in pore size in soil causes an extremely wide range of pore water velocities, and (iii) the tortuosity of pores results in a range of flow path lengths (McLaren and Cameron, 1996).

Solute transport is also affected by a number of other factors, including macropore effects, reactivity with and transformations in the soil, and plant uptake of solutes. Earthworm activity, root growth, freezing and thawing, and wetting and drying cycles can lead to the development of surface-connected macropores in the soil. Water flow through these pores can have two distinct effects on leaching:

- when water is applied immediately after a solute, macropore flow may lead to extensive leaching at a faster rate than normal
- when solutes are present within aggregate micropores they may be bypassed by the bulk of the flowing water and thus protected from leaching.

Reaction of solutes with the soil may also reduce the rate of leaching. Cations (including metals), which are adsorbed by soil surfaces, are generally less prone to leaching than non-adsorbed anions such as nitrate and sulphate. Some solutes are also involved in biological transformation processes that can either decrease (immobilisation) or increase (mineralisation) solute concentrations. Plant uptake of solutes decreases their concentration in the soil solution and therefore reduces their rate of leaching loss from the soil.

4.5.2 Nutrient leaching

Although the nutrient elements present in biosolids (e.g., nitrogen, sulphur, phosphorus, potassium, calcium and magnesium) are considered beneficial for plant growth, some of them can be regarded as contaminants if they leach into groundwater. The main element in this category is nitrogen, which in the form of nitrate (NO_3^-) is extremely mobile and has a high potential for leaching. Indeed, in the short term, the potential for nitrate to leach is the main limitation on the amount of biosolids that can be applied to the soil. Applying amounts of nitrogen in excess of crop requirements is likely to result in increased concentrations of potentially leachable nitrogen in the soil.

Phosphorus contamination of water is also a potential problem in relation to eutrophication, although little research has been reported on phosphate leaching from biosolids-amended soils. There is no doubt that phosphorus is likely to accumulate in soils that receive repeated applications of biosolids, and the implications of this in relation to phosphate leaching require attention. There is currently considerable concern worldwide regarding the build-up of phosphate in soils as a result of inorganic fertiliser applications, with the realisation that phosphate leaching from such soils may be environmentally significant (Haygarth and Jarvis, 2000).

4.5.3 Leaching of organic chemicals

There is little data on the leaching of organic chemicals from biosolids-treated soils. However, for the organochlorine pesticides, PCBs and dioxins, their extremely low water solubility and strong binding ability with soil means they are unlikely to be measured in groundwater. However, it should be noted that some specific pesticides are known to be reasonably mobile in the soil and have the potential to contaminate groundwater.

4.5.4 Leaching of metals

With some exceptions, trace metals tend to be sorbed relatively strongly by soils, and only small concentrations are present in the soil solution. They are therefore generally considered to be rather immobile in soils and their leaching down through the soil profile minimal (Dowdy and Volk, 1983; Ellis et al., 1983). However, evidence is accumulating that the leaching of both trace metal cations and anions does occur, and in some circumstances may result in significant movement of trace metals down the soil profile and into groundwater.

The suggestion by many researchers that no substantial long-term leaching losses of trace metals occur from surface soils contaminated by biosolids is often based on the observation that little trace metal accumulation is observed in the subsoil. For example, studies by Chang et al. (1984) and Williams et al. (1985) both showed that despite large applications of sewage sludge to soils, most trace metals did not appear to move below the depth of sludge incorporation. However, this assumption ignores the possibilities that (i) drainage leachate may move through preferential flow channels in the subsoil, thus bypassing most of the soil mass, and (ii) sorption of some trace metals in the subsoil may be minimal due to the dominance of organically complexed or colloid-associated forms.

The possibility that trace metals might be transported to lower depths in the soil through cracks and macropores has been suggested by Dowdy et al. (1991). Such a process could be conducive to significant leaching of trace metals without markedly increasing concentrations in the subsoil (Sidle and Kardos, 1977; Camobreco et al., 1996). It is interesting to note that at many sites where trace metals have been added in sewage sludge treatments, mass balances calculated several years after application have been unable to account for all the trace metals applied (e.g., McGrath and Lane, 1989; McBride et al., 1997). It seems possible that these deficits could be explained, at least in part, by leaching.

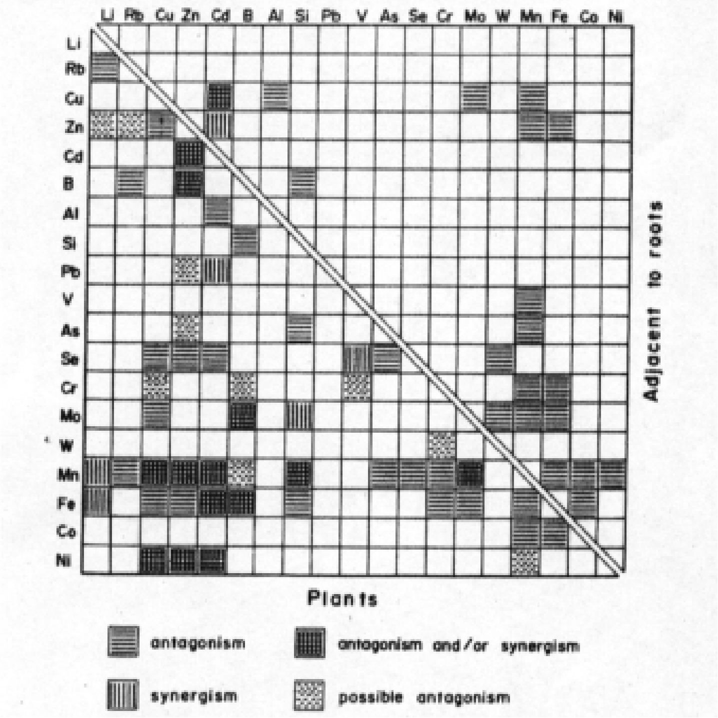
Apart from field study data, knowledge of metal movement through soils is based primarily on studies using homogenised, repacked soil columns (Camobreco et al., 1996). Such studies, even when using coarse-textured soils, have tended to support the contention that cationic trace meals are basically immobile (e.g., Giordano and Mortvedt, 1976; Gerritse et al., 1982). In recent years, however, studies using undisturbed soil columns (or lysimeters) have demonstrated the potential importance of preferential flow for trace metal leaching. For example, Camobreco et al. (1996) compared the movement of cadmium, zinc, copper and lead through undisturbed and homogenised soil columns. The homogenised columns retained all added metals, whereas substantial amounts of all four metals were present in the effluent from the undisturbed columns.

More recently, McLaren et al. (1999) have demonstrated increased metal leaching from undisturbed soil lysimeters treated with metal-spiked sewage sludge. In this study there were clear examples of the preferential flow of metals taking place, with increased metal concentrations in leachates occurring well before a pore volume of water had passed through the lysimeters. The study also revealed considerable differences between different soil types in the extent of metal leaching. It should be noted that the amounts of metals leached represented a very small fraction of the metals applied to the soil. Nevertheless, the cumulative effect of several decades of leaching has the potential to substantially influence the concentrations and distribution of trace metals in the soil.

4.6 Interactions between contaminants

Plants require a balance of elements for their proper growth and development, and imbalance between elements may cause chemical stress. Imbalances may result from both antagonistic and synergistic effects operating within the plant, or may influence absorption of elements from the soil. Antagonism between elements occurs when the combined effect of two or more elements is less than the sum of their independent effects, and synergism occurs when the combined effects of the elements is greater. Interaction processes are controlled by several factors and the exact mechanisms are still poorly understood, although some data are available (Kabata-Pendias and Pendias, 2001).

Interactions have been observed between major elements and trace metals, and between different trace metals. However, the interrelationships between elements are extremely complex, being at times both antagonistic and synergistic (see Figure 4.2), and often difficult if not impossible to predict. Unfortunately, many of the studies in which metal interactions have been observed have been carried out under conditions, or at metal concentrations, that have little practical relevance.



Source: Kabata Pendias and Pendias, 2001

Figure 4.2: Interactions of trace elements within plants and adjacent to plant roots

4.6.1 Implications of metal interactions for biosolids applications

When recommendations for the regulation of zinc, copper and nickel in biosolids-amended soils were first established in the UK (Chumbley, 1971), they implied that the phytotoxic responses of these three metals were additive (the zinc equivalent concept). However, subsequent experimental work has suggested that the toxicity to these metals was not additive but acted independently for each metal below the critical plant tissue concentration values (Beckett and Davis, 1982). Certainly most recent reviews of research in this area have concluded that synergistic interactions between trace metals are not commonly observed (e.g., Smith, 1996; Kabata-Pendias and Pendias, 2001).

In contrast, antagonistic effects are likely to be more common. In particular, zinc is commonly documented as being a competitive inhibitor of the plant uptake of cadmium (e.g., Chaney and Oliver, 1996). Since zinc concentrations in biosolids are usually much higher than cadmium concentrations, the antagonism between these metals is generally regarded as likely to restrict plant uptake of excessive cadmium. Other antagonistic reactions may also occur between iron, manganese, copper and zinc (Kabata-Pendias and Pendias, 2001), but would appear to be impossible to predict.

One type of interaction that takes place in soil may be of particular significance for biosolids applications. It has been proposed that the iron, manganese and aluminium present in biosolids may be oxidised in the soil to form oxide or hydrous oxide minerals (if not already present in these forms), which are then capable of adsorbing other metals, thus reducing their solubility, mobility and bio-availability (Corey et al., 1987). Some data from long-term biosolids plots have indeed been interpreted in this way (Chaney and Oliver, 1996). However, data for New Zealand sewage sludges (Ogilvie, 1998) would suggest that iron, manganese and aluminium concentrations are generally not high enough for such an effect to be significant.

4.7 Biosolids properties

Biosolids can vary greatly in their properties and the type and concentrations of contaminants present, depending on the sources of wastewater entering the treatment plant and the nature of the treatment process. Biosolids material applied to the land can range in physical composition from liquid sludges with less than 10% solids, to dewatered materials (essentially solid, but still with a high moisture content), to completely dried and pelleted material. Biosolids may also be composted with other organic wastes to produce materials with a similar physical composition to garden compost. Clearly the physical nature of the material will affect the ease of handling, and influence the machinery used to apply it to the land.

In addition to the variation in the types and concentrations of contaminants present, there are also substantial variations in the concentrations of beneficial nutrients (e.g., nitrogen, phosphorous, sulphur, potassium, calcium and magnesium). Thus, not surprisingly, the usefulness of particular types of biosolids as fertiliser material will be affected by their actual composition. There is also considerable variability in the inorganic components of biosolids: most materials will contain a substantial proportion of fine silt and clay particles, and possibly iron, aluminium and manganese either as oxides or that become oxidised in the soil. The potential effect of these constituents on metal bio-availability and mobility has been discussed earlier. The forms in which trace metal contaminants occur in the biosolids can also vary markedly (e.g., Steinhilbler and Boswell, 1983; Oake et al., 1984), depending on the other properties of the materials. However, there is considerable contradiction in the literature concerning the importance of such differences in modulating the bio-availability and mobility of metals in treated soils (Smith, 1996).

Some research would suggest that in the short term metal bio-availability may well be significantly influenced by biosolids properties such as pH, organic matter content, inorganic constituents, and the forms of metals (e.g., John and Laerhoven; 1976; Jing and Logan, 1992). However, other research suggests that with time, differences between biosolids disappear as the material becomes incorporated into, and reacts with, the soil (e.g., Mitchell et al., 1978; Berrow and Burridge, 1984). Without considerable further research in this area, our ability to predict likely metal bio-availability or mobility trends on the basis of biosolids properties will remain extremely limited.

4.8 Effect of land management on contaminant bio-availability and mobility

Land management can have a significant effect in minimising the potential for adverse effects from the application of biosolids to land. Sound management strategies must be based on a good understanding of the way the contaminants present in biosolids react with the soil (as discussed above). They should be aimed at minimising the bio-availability and mobility of contaminants and thus reducing their likely movement into the food chain, or transfer into ground or surface waters.

4.8.1 Application strategies

Biosolids should only be applied to soils in ways, and under conditions, that will ensure they remain in place and do not move off-site (se Section 8.4). Sufficient buffer zones should be left to ensure that sensitive areas, like waterways, are not directly affected. The climatic conditions during application also need to be taken into account to ensure that biosolids cannot be blown or washed onto non-target areas. Clearly this will depend to a certain extent on the actual method used for application, and the physical nature of the biosolids material, particularly its moisture content. Ideally, incorporation of biosolids into the soil will minimise losses during application and ensure good contact with the soil, thus placing contaminants in close proximity to sites where immobilisation reactions can occur. Biosolids applications remaining on the land surface may be subject to run-off, so steeply sloping sites should be avoided.

4.8.2 Nature of the soil

Soils in New Zealand are extremely variable and range greatly in their ability to sustain both the short-term and long-term application of biosolids. There are a number of individual soil chemical and physical properties that will determine the suitability of a soil to receive applications of biosolids. The ability of the soil to tightly retain contaminants is clearly important, and this predominantly depends on soil organic matter content, oxide (iron, aluminium and manganese) content, clay content and soil pH. In general, the greater the amounts of these constituents in the soil, and the higher the soil pH, the greater the contaminant retention capacity. For metals, the cation exchange capacity of the soil provides a useful estimate of retention capacity.

Poorly drained soils should be avoided since application machinery is likely to cause structural damage to the soil, and the possibility of surface run-off of material will be greater than for freely drained soils. In addition, decomposition of the organic matter in the biosolids is likely to be slow under saturated soil moisture conditions. At the other end of the scale, excessively drained coarse-textured soils may need to be avoided when considering sites for biosolids application. Such soils usually have low contaminant retention capacities, so that leaching of contaminants down through the soil profile is a distinct possibility. However, the importance of this will depend on the extent of drainage through the soil (in low rainfall areas this may not be a problem), and whether significant concentrations of contaminants are likely to reach aquifers or other water bodies used as sources of potable water, or that have particular environmental significance.

4.8.3 Land use

The use of biosolids as a fertiliser/soil conditioner for the growth of annual crops, where the biosolids can be incorporated into the soil prior to sowing, is probably the most advantageous way of utilising this material. However, incorporation into the soil is not usually an option for permanent crops such as forests and orchards, or pastures. In these situations, care must be taken to avoid the potential problems associated with surface application discussed above.

An additional problem at forest sites is often the low pH of the soils, since this promotes metal solubility and therefore the potential leaching of metal contaminants. Clearly, the drainage regime of these soils and the depth and proximity of local aquifers will be important considerations determining the use of biosolids in forest soils.

4.8.4 Soil pH management

As discussed above, soil pH affects both the retention and solubility of metals in the soil. Therefore pH has an important effect on both the bio-availability and mobility of metals in biosolids-amended soils. Indeed, it has been argued that soil pH is the single most important soil property controlling the availability of metals in biosolids-treated soils (Smith, 1996). The main metal contaminants (cadmium, chromium, copper, nickel, lead and zinc) are present in the soil predominantly as cations, and thus tend to become more soluble and bio-available as soil pH decreases (soil becomes more acidic). For this reason, overseas guidelines and regulations for biosolids sometimes restrict application of biosolids to agricultural soils above a particular pH value, or adjust the maximum permissible metal loadings according to soil pH (e.g., UK Statutory Instrument, 1989). There is some debate as to what constitutes the appropriate lowest soil pH for biosolids application, with values generally ranging from pH 5.0 to 6.5. Research in this area has been reviewed in detail by Smith (1996), and suggests that plant bio-availability of cadmium, nickel and zinc in particular can increase substantially below pH 5.5.

Maintaining the pH values of biosolids-treated agricultural soils between pH 5.5 and 6.5 is likely to minimise the risk of phytotoxicity, or excessive metal uptake by crops. Normal agronomic liming practices should ensure this, but soils do tend to become more acid with time so that regular pH monitoring of biosolids-amended soils should be carried out. In the short term, because some biosolids may have a high pH (7.0 and above), their application to the soil may actually increase soil pH. However, decomposition of the organic matter in biosolids and nitrification of mineralised ammonium will eventually have an acidifying effect.

In contrast to the metal cations, the bio-availability of the anionic metals molybdenum, arsenic and selenium generally increases with pH. However, the concentrations of these metals in New Zealand biosolids are not usually high, so that maintaining soil pH values above 5.5 is not likely to create problems.

Unlike agricultural soils, forestry soils are not usually limed to maintain soil pH, and many forest soils are naturally acidic, with pH values less than 5.0. Although, applying biosolids to forest soils under such low pH conditions could appear to represent a potential problem in terms of high metal bio-availability and mobility, this is not necessarily the case. Since food crops are not involved, plant uptake of metals is not a concern and trees do not appear particularly sensitive to metal phytotoxicity. Some of the more mobile metals (e.g., nickel and zinc) may be leached from biosolids-treated forest soils (Sidle and Kardos, 1977; Cameron et al., 1994) but this seems unlikely to represent a serious environmental threat. In a recent lysimeter study in which metal-spiked biosolids were applied to the surface of three forest soils at the Department of Health's 1992 guideline soil metal limits, the leachate metal concentrations, although increased by the biosolids application, generally remained below maximum acceptable values for drinking water (McLaren et al., 1999).

4.8.5 Animal grazing

The management of grazing animals on land treated with biosolids is a controversial issue. Clearly, to avoid pathogen problems, withholding periods are required before allowing grazing animals on biosolids-treated land. However, there is also the potential for animals to ingest contaminants by consuming plants that contain elevated concentrations of contaminants, or by ingestion of the soil itself. Soil intake occurs either by (i) ingesting soil (or biosolids) present on the surface of plant leaves, or (ii) ingesting soil (or biosolids) directly from the soil surface or attached to plant roots (Healy, 1973). As a result, surface application of biosolids to established pastures is unlikely to be a realistic option. Incorporation of biosolids into the soil when pastures are re-sown should substantially reduce the ingestion of contaminants by grazing livestock.

The potential for adverse effects of metal contaminants on animals grazing on biosolids-treated soils has been reviewed by Smith (1996). He concluded that there was little evidence that animal health could be seriously affected by direct ingestion of sludge-amended soil. However, some caution should probably be applied to this conclusion, which was made in the absence of any substantial long-term field studies involving grazing animals. Irrespective of the potential or actual effects of metal-contaminated soils on animal health *per se*, there appears to be general agreement on the accumulation of metals in animal tissues. Most studies with animals have shown no accumulation of contaminant metals in muscle tissues, but increased concentrations of copper, lead and cadmium in animal livers or kidneys appear to be quite common. For example, Hill et al. (1998a,b) observed the accumulation of cadmium and lead in the livers and kidneys of sheep fed diets that included soil contaminated with sewage sludge. Similarly, Roberts et al. (1994) have reported the accumulation of cadmium in sheep kidneys from low-level soil contamination by phosphate fertilisers in New Zealand. This has resulted in the New Zealand Meat Industry automatically condemning kidneys of slaughtered animals over 2.5 years of age (Roberts et al., 1994), in spite of the fact that the consumption of offal products probably forms a negligible proportion of most people's diet.

The direct ingestion of biosolids-treated soil by grazing livestock is also considered to be the principal route of organic chemical accumulation in the food chain (e.g., Fries, 1982). However, reliable data in this area are somewhat scarce and contradictory. According to Smith (1996), "more data is needed to fully quantify the potential risk to grazing animals of organic pollutants in sludge surface-spread on grassland, although current information suggests the risk is likely to be small". Smith also noted that "it is widely recognised, however, that sludge injection *into the soil* can virtually eliminate problems of animal ingestion of organic contaminants".

4.8.6 Climate

Some of the risks associated with biosolids discussed above will undoubtedly be affected by climate. Rainfall, in particular, can be an important consideration. Application of biosolids to the land under excessively wet conditions will increase the potential for surface run-off, and also possibly macropore flow through soils. Damage to soil structure may also occur as a result of compaction due to the passage of heavy application machinery over the soil. This in turn may lead to surface ponding and anaerobic conditions. Clearly, the application of biosolids under such conditions should be avoided.

Annual rainfall and associated soil drainage may also be factors to consider when assessing the suitability of land for biosolids application. For example, to minimise contaminant leaching it may be necessary to avoid shallow or coarse-textured (sandy) soils in areas with high rainfall. The same types of soil, however, may be quite suitable for biosolids application, and indeed provide substantial benefits in drier regions.

5 SOIL CONTAMINANT LIMITS

5.1 Introduction

This chapter provides the background to the soil metal limits given in these *Guidelines* (see section 7; and Volume 1, section 4). The recommended limits represent a further development from the Department of Health's 1992 soil limits, which were in turn derived from limits used in the UK. These original limits have been revised using more up-to-date data.

The recommended soil limits are based on our current state of knowledge, and it should be noted that more attention has been taken of research carried out with actual biosolids than of studies where metals have been added to soils in some different form (e.g., as simple metal salts). It is also important to realise that as the results of current and future research are reported, further changes to the soil limits may be required. In this regard, it should be noted that in the short to medium term, heavy metal accumulation in soils as a result of biosolids application is unlikely to be a significant issue. As long as biosolids are applied at agronomic rates (based on nitrogen application rates), the accumulation of metals will be very slow, and research should have answered the present uncertainties regarding permissible metal loadings in soils long before the current soil limits are reached.

5.1.1 Methodology

Three different methodologies are commonly used in setting soil limits:

- *analysis of pollutant pathways* – which involves an assessment of the effect of a particular metal on a particular target organism (this is the method used by the USEPA)
- *lowest observed adverse effect concentrations (LOAEC)* – where limits are set based on actual occurrences of effects due to metals (this is the method used in setting the UK and hence the Department of Health's 1992 limits)
- *metals balance approach* – where metal limits are set based on losses of metals due to crop removal, soil erosion and leaching.

Detailed risk assessment approaches, such as that used by the USEPA (1995), although appearing more sophisticated than the LOAEC approach, have their limitations. They can only ever be as good as the assumptions made in their development and the data used to estimate the various input parameters required (e.g., soil–plant transfer coefficients). In particular, the USEPA (1993) soil limits for biosolids-treated soils, estimated using a risk assessment process, have been the subject of considerable criticism. For several metals the derived soil limits are clearly much higher than those at which phytotoxicity has been observed, or would produce plant metal concentrations above current food maximum permissible concentrations (MPCs). In addition, different risk assessment procedures can result in vastly different soil metal limits. This raises questions regarding the use of this type of approach. For example, compare the soil copper limit estimated using the USEPA (1995) procedure (approximately 750 mg/kg) and that estimated using the Ministry for the Environment and Ministry of Health (MfE and MoH, 1997) procedure (40 mg/kg). In this case, these differences occur due to the use of different assumptions, exposure pathways and receptors in the risk model.

The metal balance approach to estimating soil metal limits essentially reflects a philosophy of allowing no metal accumulation in soils above background levels, irrespective of any evidence of their potential adverse effects. This approach effectively prevents recycling of biosolids to land.

Therefore, in keeping with the original New Zealand limits, the LOAEC approach has been retained in defining the current limits, which is thought to be appropriate given the effects-based approach of the Resource Management Act 1991.

5.1.2 Bio-availability

The issue of metal bio-availability requires considerable attention. McLaughlin et al. (2000) have recently reviewed the advantages and disadvantages of a bio-availability-based approach to the control of metal contamination of soils, and have suggested improvements to avoid both over- and under-protective measures. Current soil metal guidelines are all estimated solely on the basis of total soil metal concentrations. While the procedure for determining total metal concentrations in soils is relatively simple, this measure does not reflect the fact that many soil properties (including soil pH, texture, organic matter content) greatly influence the bio-availability of metals in any given soil (see section 4.4). Metal bio-availability will also vary between different types of biosolids material, depending on actual biosolids properties. In effect, metal bio-availability to plants and soil micro-organisms is a function of the:

- bio-availability of dissolved metal species in the soil solution
- ability of the soil to buffer (maintain) metal concentrations in the soil solution.

A whole range of different soil and environmental factors can affect these two parameters, yet we are some way from completely understanding, and being able to predict, metal bio-availability. In particular, considerable research is still required to determine the most appropriate and practical method for assessing metal bio-availability. Currently, several different approaches are being examined, but agreement on this issue, and acceptance by regulators, is probably some way off. However, it is envisaged that the outcomes of future research in this area will almost certainly affect the way in which soil metal limits are set.

5.2 Individual metal limits

5.2.1 Arsenic

Soil metal limit = 20 mg/kg dry weight

Background arsenic (As) levels in topsoils worldwide range from < 1 to 95 mg/kg, although mean As concentrations for individual countries or soil types are usually less than 10 mg/kg (Kabata-Pendias and Pendias, 2001). Data for background As concentrations in New Zealand soils are scarce, with probably the most comprehensive data set being provided by Roberts et al. (1996). These authors surveyed a total of 398 sites throughout New Zealand, including 86 native (unmodified) and 312 agricultural soils (0–7.5 cm topsoils). Mean As concentrations for individual soil types were all less than 8 mg/kg, with the exception of peat soils, which had a mean of 12 mg/kg. There were no significant differences between the native and agricultural soils. Although the mean As concentrations were relatively low, As concentrations at some individual sites were above 30 mg/kg (0.2–30 mg/kg is the background range provided in the ANZECC [1992] guidelines for contaminated sites).[3] The data of Roberts et al. (1996) are also supported by a recent survey by the Auckland Regional Council (1999), who reported a mean background As concentration of 4.5 mg/kg in 99 soils from the Auckland region (range 0.02–24.02 mg/kg).

The soil limit of 20 mg/kg is set primarily to avoid phytotoxic effects on plants, while taking note of the above background levels of As in New Zealand soils. The rationale for basing the limit on phytotoxic effects is that:

- the accumulation of As in the edible parts of most plants is generally low (O'Neill, 1995)
- phytotoxicity of As to plants is often reached before it accumulates to levels that would be toxic to wildlife or humans ingesting the plants (Carbonell-Barracina et al., 1997; Smith et al., 1998).

The most comprehensive summary of phytotoxic levels of soil As has been carried out by Sheppard (1992). This author has clearly demonstrated the very narrow margin between background and toxic

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[3] The background concentrations given in the ANZECC (1992) guidelines are for Austalian soils only.

concentrations of As in soil. Both Sheppard (1992) and Smith et al. (1998) noted that the levels of soil As reported to be toxic to plants span a broad range, and overlap with those reported as being background, no-effect or beneficial levels. This degree of variability is due to the many factors that are known to influence the bio-availability of soil As to plants, including the plant species, form of As, soil pH, soil type, and the presence of other ions (e.g., O'Neil, 1995; Smith et al., 1998).

However, according to the summary of data by Sheppard (1992), soil type (texture) appears to be a dominant factor controlling As phytotoxicity, with much lower (five-fold) toxicity thresholds in sands and loams than in clay soils. The geometric mean As toxicity threshold for sandy soils was determined as 40 mg/kg (Sheppard, 1992), but clearly, since this value is a mean the threshold in many sandy soils will be substantially below this level. The soil limit of 20 mg/kg takes note of this fact, and should therefore be protective for most sandy soils and certainly for finer-textured loam and clay soils. However, there are no data specifically relating to phytotoxicity of As in New Zealand soils, and research in this area would assist in confirming whether this value is indeed appropriate.

The potential for As to leach from soils has been recognised for many years, and there are many reports of As movement down through the soil profile following addition of As-containing materials to the soil (e.g., Johnson and Hiltbold, 1969; Steevens et al., 1972; Merry et al., 1983; McLaren et al., 1998). Such studies indicate that the extent of As leaching is strongly influenced by many factors, including soil texture, the rate and form of As applied, and the amount of rainfall. However, certainly for relatively low levels of contamination (e.g., below the 20 mg/kg soil limit), the data available would suggest that the "possibility of contamination of groundwater with leaching of As is very insignificant" (Sandberg and Allen, 1975). This view is supported by the Dutch, who gave consideration to leachability of soil contaminants in developing their soil quality guidelines (MHSPE, 1994). Their target value[4] for As in soil is 29 mg/kg (MHSPE, 2000).

The soil limit of 20 mg/kg is compatible with several other sets of guidelines and regulations. For example, 20 mg/kg has been set as the guideline As concentration in soils for reuse of biosolids in both South Australia (SA EPA, 1996) and New South Wales (NSW EPA, 1997). The USEPA (1993) has a similar limit of 41 kg/ha, which equates to approximately 20 mg/kg in the topsoil. However, Europe (CEC, 1986) does not appear to have a guideline value for As in soils for re-use of biosolids.

A 20 mg/kg value was used as the investigation level (the level above which an environmental assessment would be necessary) in the ANZECC (1992) guidelines for the assessment of contaminated sites. This has since been superseded by a soil investigation level for residential land with a garden/accessible soil of 100 mg/kg, set by the Australian National Environmental Protection Council (NEPC, 1999). The *Health and Environmental Guidelines for Selected Timber Treatment Chemicals* (MfE and MoH, 1997), which used an accepted risk assessment approach, indicate an interim level for As in the soil of 30 mg/kg for residential land use. Similar risk assessment calculations for agricultural land produced a level of 4.2 mg/kg. However, it was noted that the calculated criteria for agricultural land use was low and impractical given the relationship to background concentrations. Pending additional research and information, the interim value for agricultural soils contaminated with timber treatment chemicals has also been set at 30 mg/kg (MfE and MoH, 1997).

One of the more recently established risk-based soil limits for As for contaminated site investigations has been set in the UK. This specifies a limit of 20 mg/kg dry weight for residential land (with and without plant uptake) and for land to be used for garden allotments (DEFRA, 2002a). The Canadian government has also established an As soil quality guideline using a multi-pathway risk assessment methodology (CCME, 1999). This derives a soil limit of 12 mg/kg dry weight for both agricultural and residential land.

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[4] Target values give an indication of the benchmark for environmental quality in the long term on the assumpton of negligible risks to the ecosystem (MHSPE, 2000)

5.2.2 Cadmium

Soil metal limit = 1 mg/kg dry weight

Mean background cadmium (Cd) concentrations for surface soils worldwide range from 0.06 to 1.1 mg/kg (Kabata-Pendias and Pendias, 2001). ANZECC (1992) guidelines for contaminated sites indicated typical background Cd concentrations in the range 0.04 to 2 mg/kg, and set an investigation level of 3 mg/kg (the level above which an environmental assessment would be necessary).

A survey of 86 native (unmodified) soils in New Zealand determined an overall average background Cd concentration of 0.20 mg/kg, with a range of 0.02–0.77 mg/kg (Roberts et al., 1994). In the complementary survey of 312 pastoral soils, the average Cd concentration was 0.40 mg/kg, with a range of 0.04–1.53 mg/kg (Roberts et al., 1994). The elevation of Cd in pastoral soils was attributed to the long-term application of phosphate fertilisers. Several other New Zealand studies have confirmed the build-up of Cd in soils as a result of phosphate fertiliser use (e.g., Loganathan et al., 1995; Loganathan et al., 1997; Gray et al., 1999).

Cd is regarded as one of the most bio-available of the heavy metals, and at relatively low concentrations in the soil can accumulate in the edible part of crop plants, while having no detrimental effects on crops themselves (Smith, 1996). Even with current New Zealand pasture soil Cd levels (mean 0.4 mg/kg), a substantial percentage of ovine and bovine kidneys exceed the maximum residue level of 1 mg/kg for the Cd content of meat and offal for human consumption. As a result, the New Zealand meat industry now automatically condemns kidneys of slaughtered sheep over 2.5 years of age (Roberts et al., 1994). In addition, a survey of wheat crops in Mid Canterbury (Roberts et al., 1995) revealed that 15% of grain samples had Cd concentrations above the current maximum permissible concentration (MPC) of 0.1 mg/kg fresh weight (Australia New Zealand Food Authority Standards). A more recent investigation by Gray et al. (2001) found 10% of the wheat grain samples examined were non-compliant with the MPC. The Cd concentrations of the soils on which these wheat crops were grown were all below 0.45 mg/kg (Gray et al., 2001).

The soil limit of 1 mg Cd/kg soil is set primarily to minimise Cd concentrations in animal and crop products and to avoid barriers to international trade (failure to meet maximum residue levels and MPCs in food products set by importing countries). It should also be noted that although the current European Community maximum soil limit is 3 mg/kg (CEC, 1986), this value is under review (EC, 1999), and several European countries are already using much lower limits. The most recent European Community draft directive proposes cadmium levels between 0.5 and 1.5 mg/kg depending on soil pH (EC, 2000). It is seen as important that our limits for Cd, in particular, are not substantially higher than that of the European Community, which is an important export market for New Zealand produce.

Using a risk assessment methodology, the UK has set a contaminated land guideline level for Cd in soil that is pH dependent (DEFRA, 2002b). For residential land where plants are grown for food and for garden allotments, the soil guideline values are 1 mg/kg dry weight (at pH 6), 2 mg (pH 7) and 8 mg (pH 8). The Canadians have a risk-based soil quality guideline of 1.4 mg/kg dry weight for agricultural and 10 mg/kg for residential land (CCME, 1999).

Setting a guideline level of 1 mg Cd/kg soil should also protect the microbial population of the soil (Chander et al., 1995), and should not result in significant leaching of Cd from the soil into groundwater. The Dutch soil quality objectives, which give consideration to leachability of soil contaminants set the target value for Cd in soil at 0.8 mg/kg (MHSPE, 2000). However, data on Cd leaching from field soils are scarce, and this is an area requiring further research.

In the short term, adding biosolids-containing materials to soil may, as a result of increased organic matter levels, actually decrease Cd bio-availability to plants (e.g., McLaren and Gray, 1999; Bolan et al., 2001; Buchan, 2001). However, the long-term situation, as the organic matter decomposes, is unclear and requires further research. Some research does suggest that the bio-availability of added Cd may be permanently reduced over time as a result of interactions with inorganic soil components (e.g., Hamon et al., 1998; Gray et al., 1999; Li et al., 2001). Currently the evidence for this effect is far from conclusive so a precautionary approach has been taken in setting the soil Cd limit.

5.2.3 Chromium

Soil metal limit = 600 mg/kg dry weight

The soil limit is for total chromium (Cr), where it is assumed that all chromium present in the soil is chromium (III).

Worldwide background soil Cr concentrations vary widely from as little as 1 mg/kg to 1,500 mg/kg (Kabata-Pendias and Pendias, 2001), mainly depending on soil parent material. The ANZECC (1992) guidelines for the assessment of contaminated sites indicates that typical background soil Cr concentrations in Australia are in the range 0.5–110 mg/kg. However, soils in New Zealand formed on mafic or ultramafic rocks can have Cr concentrations well above this range (e.g., Wells, 1960; Percival et al., 1996; Auckland Regional Council, 1999). Apart from these cases, the bulk of soils in New Zealand have background Cr concentrations less than 60 mg/kg, with many less than 30 mg/kg (Wells, 1962).

Cr can occur in two main oxidation states: as trivalent Cr (Cr^{3+}) or as hexavalent Cr (Cr^{6+}). Whereas Cr^{6+} is regarded as extremely toxic to humans (it is a highly potent carcinogen), animals and ecosystems, the trivalent form (Cr^{3+}) is regarded as relatively inert and much less toxic. Fortunately, it has been shown that most soil Cr occurs as Cr^{3+} and is within mineral structures or forms of mixed Cr^{3+} and Fe^{3+} oxides. Cr^{3+} is only slightly mobile in very acid media, and at pH 5.5 is almost completely precipitated. Its compounds are considered to be very stable in soils (Kabata-Pendias and Pendias, 2001). In addition, when added to soils Cr^{6+} is rapidly reduced to Cr^{3+} by reaction with soil organic matter under nearly all soil conditions (e.g., McGrath, 1995; Chaney et al., 1997). The reverse process (i.e., oxidation of Cr^{3+} to Cr^{6+}) appears to take place only when environmental circumstances fit together in a very narrow and delicately balanced optimum (Bartlett, 1997). Indeed, Chaney et al. (1997) have suggested that the inert nature of Cr^{3+} in soils, with very low concentrations in the soil solution, means that the oxidation process does not proceed.

Given the strong reducing capacity of wastewater solids, it is generally assumed that the chromium present in biosolids will be present as Cr^{3+}. This assumption is supported by the research of Grove and Ellis (1980), who demonstrated that sludge Cr did indeed exhibit the chemistry of trivalent Cr. In addition, the lack of Cr uptake by plants or leaching of Cr at sites treated with biosolids might be taken as indirect evidence for the absence of Cr^{6+}. However, as noted by Bartlett (1997), soil Cr^{6+} has not actually been analysed in these studies.

The soil metal limit of 600 mg/kg for Cr is based on the former Department of Health 1992 guidelines, and on the *Health and Environmental Guidelines for Selected Timber Treatment Chemicals* (MfE and MoH, 1997). The latter guidelines used a risk assessment approach, and set a soil limit of 600 mg/kg for Cr (III) for agricultural and residential land use. This limit is set on the basis of the protection of plant life. For Cr (VI), the timber treatment guidelines derive lower limits of 4 mg/kg for agricultural land and 25 mg/kg for residential land with 10% home-grown produce consumption. The recent UK contaminated land guidelines have set a value of 130 mg/kg dry weight for Cr (VI) for residential land with plant uptake and for garden allotments, which are designed to be protective of young children (DEFRA, 2002c).

A limit of 600 mg/kg is considered likely to be extremely protective of phytotoxic effects. Chaney et al. (1997) could not identify any studies in which the cumulative Cr application from biosolids at rates as high as 3,000 kg Cr/ha (approximately 1,500 mg/kg in topsoil) resulted in yield reduction of crops. Similarly, an expert work group advising the USEPA determined that there were no field studies, conducted with appropriate methods, that showed adverse effects of biosolids Cr on plant growth, or other evidence of adverse effects of Cr in land-applied biosolids on humans or on the environment (see Chaney et al., 1997).

Findings such as these are no doubt the reason why neither the current US (USEPA, 1993) nor European guidelines (CEC, 1986) have set a soil limit for Cr. It would be very hard to justify setting a limit any lower than 600 mg/kg with our present state of knowledge. The environmental investigation level of 50 mg Cr/kg soil in the ANZECC (1992) contaminated site guidelines appears to be extremely anomalous, unless it is based on Cr (VI), in which case it is reasonably comparable with the New

Zealand timber treatment guideline (MfE and MoH, 1997), the UK contaminated land guideline (DEFRA, 2002c) and the Australian National Environment Protection Council Cr (VI) value of 100 mg/kg for residential land (NEPC, 1999).

Because of the very low solubility of Cr (as Cr^{3+}) in soils, significant leaching of Cr into groundwater from biosolids-treated soil is considered extremely unlikely, even with a total soil Cr concentration of 600 mg/kg. For example, a lysimeter leaching study by McLaren et al. (1999) was unable to detect significant increases in Cr in drainage water from soils to which metal-spiked sewage sludge had been added, for a period of three years after the sludge addition. In this study the concentration of added Cr was approximately 600 mg Cr/kg in the top 10 cm of soil.

5.2.4 Copper

Soil metal limit = 100 mg/kg dry weight

Worldwide background soil copper (Cu) concentrations vary widely, from as little as 1 mg/kg to over 300 mg/kg (Kabata-Pendias and Pendias, 2001), mainly depending on soil parent material and soil texture. The median Cu concentrations in uncontaminated soil were reported to average 30 mg Cu/kg, with a range of 2–250 mg/kg (Bowen, 1985). Shacklette and Boerngen (1984) analysed soil samples from various locations in the US, finding that copper concentrations ranged from < 1 to 700 mg/kg, with an average of 25 mg/kg. Kabata-Pendias and Pendias (1984) reviewed the worldwide literature on Cu in uncontaminated surface soils and reported mean concentrations ranging from 6 to 80 mg Cu/kg (dry weight).

ANZECC (1992) guidelines for contaminated sites indicated that typical background soil Cu concentrations in Australia are in the range 1–190 mg/kg. Background Cu concentrations in New Zealand soils are typically up to 30 mg/kg, but higher concentrations occur on soils formed from basaltic and andesitic parent materials (Wells, 1957; Percival et al., 1996). A substantial survey of 400 soils reported by Roberts et al. (1996) found 10% of topsoils had Cu concentrations above 40 mg/kg, with a highest value of 76 mg/kg. Similarly, Auckland Regional Council (1999) determined a mean soil Cu concentration of 27 mg/kg, with individual concentrations ranging from 1 to 111 mg/kg.

The main issue regarding Cu application to agricultural land is the potential phytotoxic effect on crops and pastures. Introducing excessive amounts of Cu into the food chain is regarded as extremely unlikely, since in most plant species Cu concentrations seldom exceed 30 mg/kg even when very large additions of Cu have been made to the soil (Jarvis, 1981). According to the ‘soil–plant-barrier’ concept of Chaney (1980), Cu is in the group of metals that are readily absorbed by plant roots, but the plant suffers severe yield reduction before humans, livestock or wildlife are at any risk from chronic lifetime consumption of the crop. This no doubt partially explains why Food Standards Australia and New Zealand (FSANZ) have no maximum food standard for Cu: essentially, Cu concentrations in plants never reach levels that constitute a serious risk to human health.

The soil Cu concentration at which plant toxicity occurs will vary widely depending on the plant species, and soil properties such as pH, texture and organic matter content (Tiller and Merry, 1981). The properties of the biosolids are also likely to influence Cu bio-availability and toxicity (Smith, 1996). It is therefore not surprising that it has been impossible to find a common soil Cu threshold value in the literature above which Cu phytotoxicity occurs. Smith (1996) has reported one of the better reviews of phytotoxicity data for Cu, which shows that even in some of the most susceptible situations (pot trials involving sandy soils treated with Cu salts), phytotoxicity would seem unlikely to occur below a total soil Cu concentration of approximately 100 mg/kg. In many situations, particularly where biosolids have been applied to soils in the field, the Cu toxicity threshold is likely to be much higher. There is good evidence that the results of metal salt and pot studies greatly overestimate phytotoxicity and the bio-availability of metals in biosolids (e.g., Corey et al., 1987; Smith, 1996). However, this hypothesis has not been tested rigorously and further research in this area is needed.

On the basis of the above discussion, the metal limit for Cu of 100 mg/kg for biosolids-amended soils should be protective of both plant and human health. Such a value should also be protective of the soil microbial population (MAFF/DoE, 1993b; Chander et al., 1995). However, as with the phytotoxicity limit, this requires further testing. Similarly, significant leaching of Cu into groundwater from biosolids-treated soil is considered extremely unlikely at a total soil Cu concentration of 100 mg/kg. For example, a lysimeter leaching study by McLaren et al. (1999) was unable to detect significant increases in Cu in drainage water from soils to which metal-spiked sewage sludge had been added, for a period of three years after the sludge addition. In this study, the concentration of added Cu was approximately 135 mg Cu/kg in the top 10 cm of soil. Indeed for some soils, addition of biosolids actually reduced Cu leaching compared to control treatments (no sludge added).

The limit of 100 mg/kg for Cu is much more restrictive than the USEPA (1993) guidelines (approximately 750 mg Cu/kg soil), which are based on a detailed risk assessment procedure, but are generally in agreement with several European countries and New South Wales (NSW EPA, 1997). However, the soil limit for Cu in these *Guidelines* is substantially less restrictive than the corresponding limit for agricultural soils (40 mg/kg) given in *Health and Environmental Guidelines for Selected Timber Treatment Chemicals* (MfE and MoH, 1997). This difference requires some consideration.

The *Health and Environmental Guidelines for Selected Timber Treatment Chemicals* recognise a soil threshold for phytotoxicity of 130 mg/kg, which is similar to that used in the current guidelines. However, the risk assessment procedure used for deriving the guideline values identified agricultural produce as the limiting pathway for soil Cu, with a limiting soil value of 39 mg/kg. In light of the soil–plant barrier concept discussed earlier in relation to plant Cu levels and food chain implications, it is extremely difficult to understand the logic behind this decision.

This perhaps emphasises one of the problems associated with risk assessment procedures: the values derived from their use are only as good as the assumptions and data used. For example, to assume 100% consumption of produce from contaminated land does not seem to be a particularly realistic scenario. In the case of arsenic, the *Health and Environmental Guidelines for Selected Timber Treatment Chemicals* noted that the calculated criterion for arsenic for agricultural land use was low and possibly impractical, given the relationship to background concentrations. Why the same reasoning was not also applied to Cu should perhaps be questioned. It could be argued that a lower limit is appropriate for sites contaminated by timber treatment chemicals, since the source of contamination is Cu in the form of a simple metal salt compared to Cu bound strongly to biosolids components. However, as noted above, the recommended biosolids soil Cu limit is based largely on phytotoxicity criteria that have been determined for simple inorganic Cu salts.

World Health Organisation ADI for copper

Cu is an essential nutrient, deficiencies of which lead to well-documented disease states. Therefore, the traditional risk assessment practice of applying 10-fold safety factors to experimental NOAELs may lead to 'safe' intakes that are, in fact, incompatible with adequate nutrition. The WHO's *Environmental Health Criteria for Copper* (WHO, 1998) proposes an acceptable range of oral intake for copper of between 0.02 mg/kg/day (adult) and 0.05 mg/kg/day (infant) to maintain nutrition status, and "several but not many" times greater mg/day ('several' meaning more than 2–3 mg/day) to prevent toxicity. The upper limit on this range is driven by acute gastrointestinal effects from drinking-water studies. There were no clear indications of adverse effects having stemmed from food residues of copper. Clearly, the WHO felt that setting precise numerical limits on the upper acceptable intake was inappropriate, and instead opted for a semi-qualitative statement. There is a WHO (1996) reference to a value for a safe range of mean population intake (12 mg Cu/day for men).

The upper end of the range of dietary intakes for Cu in children is 0.1 mg/kg/day (WHO, 1998). The New Zealand timber treatment guidelines use a reference does (RfD) which is five times this level. The basal mean Cu intake level proposed by the WHO is 1.2 mg/day (approximately 0.017 mg/kg/day assuming a 70kg adult), which presumably is intended to prevent nutritional deficiency from occurring.

Cu is neither a carcinogen, nor a systemic toxicant to humans at low doses.

That the WHO, in its 1998 reassessment, found that there was insufficient justification for setting a firm upper limit on Cu intake indicates that the previous level of 0.5 mg/kg/day should be taken as a rough guideline as opposed to a well-defined risk value. However, it turns out that the 0.5 mg/kg/day value is about 10-fold higher than the recommended basal level of 2–3 mg/day.

Air and water contributions to copper intake

The WHO states that, worldwide, the intake of Cu (occupational exposures excepted) is driven by exposure through the food chain. Drinking water can contribute from 0.1 to 0.3 mg/day depending on the presence of copper water pipes in the household, but this is less than 10% of a person's intake from the diet. Estimated mean intake from airborne exposure, based on mean Canadian values, is less than 0.00028 mg/day, which essentially makes the inhalation route insignificant for the general population (ICRP, 1974).

Bio-availability considerations

Cu binds more strongly to soil organic matter than most other metals, and as a result is less influenced by pH. Cu is therefore a good deal less bio-available to crops from sludge amendments than cadmium, nickel and zinc, but somewhat more mobile and bio-available than lead (WHO, 1998).

The timber treatment guidelines

The *Health and Environmental Guidelines for Selected Timber Treatment Chemicals* (MfE and MoH, 1997) use the previous WHO (1982) ADI of 0.5 mg/kg/day for copper as the reference dose (RfD). The guidelines make the assumption that 10% of this threshold dose should be attributable to contaminated soil, including soil ingestion, inputs into the food supply and dust inhalation. Some of the default parameters in the guidelines are based on USEPA estimates, and may require re-examination. California, in their recent 2001 *Multipathway Risk Assessment Guidelines*, use 10 mg/day for this soil ingestion parameter for adults > 15 years old, and 64 mg/day for children 0–15 years old. These compare with the use of older USEPA values of 25 mg and 100 mg, respectively, for these parameters in the timber treatment guidelines (TTG).

The resulting calculations in the timber treatment guidelines (MfE and MoH, 1997) derive a Cu soil criterion for agricultural land of 39 mg/kg.

Dietary intake data in the timber treatment guidelines

Dietary intakes of leafy and root vegetables and fruit are taken from the 1985 Australian Market Basket Survey. However, the 1991 Life in New Zealand Survey (LINZ, 1991) gives values for adults that are slightly different and possibly influence the final resulting soil concentrations. Ideally, even more precise estimates for this could be calculated from the 1999 National Nutrition Survey, but this has not been done here.

Values from LINZ 1991 are different from those of the 1985 Australian Market Basket Survey, and are as follows:

- leafy vegetables (brassicas, lettuce, silverbeet, beans, tomatoes): 19% of basket
- root vegetables (potatoes, onions, beetroot): 34%
- fruits (oranges, apples, others): 47%.

In addition, the amount of vegetable matter consumed per day in the LINZ survey was 0.51 kg for an adult as compared with the 0.45 kg per day assumption used in the TTG.

Table 5.1: Levels of copper in foodstuffs (mg/kg wet weight)

Foodstuff	Mean	Minimum	Maximum	n
Meat:				
■ beef	0.8,1.1	0.74	1.6	39
■ pork	0.9,1.4	0.44	7.22	150
■ lamb	1.6	1.1	1.9	24
Liver:				
■ beef	39	8.0	87	7
■ pork	9.0	0.9	29	126
■ lamb	97	28	195	32
Kidney:				
■ beef	3.7	2.8	4.2	6
■ pork	6.1	2.9	15	75
Fruit:				
■ apples	0.25	0.21	0.31	6
■ pears	0.81	0.48	2.7	24
■ bananas	0.95,0.96	0.70	1.2	12
Vegetables:				
■ potatoes	0.75,0.96	0.26	2.2	40
■ carrots	0.40,0.61	0.26	0.95	30
■ lettuce	0.47,0.72	0.20	1.4	40
■ tomatoes	0.36,0.55	0.29	1.1	26
Fish:				
■ cod	0.19	0.12	0.28	5
■ tuna	0.64	0.48	0.80	9
Wheat:				
■ flour	1.5	0.95	2.9	56
■ bread (white)	1.5	0.89	2.2	32
Milk:				
■ cow	0.06	Trace	0.14	31
■ human	0.54	0.22	0.90	28
Coca Powder	36.4	33.0	410	9

Source: Adapted from Jorhem and Sundstrom (1993) for Sweden, and ANFA (1993) for Australia (WHO, 1998)

The food concentrations listed in Table 5.1 would result in average New Zealand intakes for an adult in the order of 15 mg per day (0.21 mg/kg/day), with about half of this coming from meats and dairy, and half from vegetable and grain sources (using a combination of New Zealand intake data (LINZ, 1991), UN Global Environmental Monitoring System data, and Australian Market Basket Survey data (ANFA, 1996).

Influential assumptions in the copper soil guideline values

While the above individual parameters in the TTG may require further scrutiny, they are unlikely to result in major changes to soil concentrations (beyond several-fold). However, there are two relatively influential but simultaneously incompatible assumptions used in the TTG. One is that only 10% of the RfD can come from contaminated soils, and the second is that 100% of an individual's produce consumption is from soils used as agricultural land. The intake from diet is at least 80–90% of an individual's Cu intake, and about half of this comes from produce. The agricultural land soil value of 40 mg/kg therefore drives the Cu soil concentration, but because the contribution to total intake is restricted to 10% of the RfD, this translates into an artificially low soil value.

The WHO ADI value of 0.5 mg/kg/day as a firm threshold is not wholly justified from the recent WHO reassessment of Cu. The dose–response is not characterised at low doses, and there has been no evidence in the toxicology literature of adverse effects on consumers from residues of Cu in foods. While it could be argued that this is simply a data gap in the toxicology literature, it is not treated so by the WHO, who essentially consider the toxicity of low doses of Cu to be of minor consequence (certainly this is so in comparison to many of the other metals).

The soil limit of 100 mg/kg for Cu therefore better reflects a value that balances concerns over metal accumulation in soils with ensuing health risks, and that allows for a more realistic exposure/risk scenario to be developed.

5.2.5 Lead

Soil metal limit = 300 mg/kg dry weight

Worldwide background soil lead (Pb) concentrations vary widely, from as little as 3 mg/kg to over 200 mg/kg (Kabata-Pendias and Pendias, 2001), with mean values for soil types averaging approximately 30 mg/kg. The ANZECC (1992) guidelines for contaminated sites indicated that typical background soil Pb concentrations in Australia are in the range < 2 to 200 mg/kg. Data for New Zealand soils indicate that background Pb concentrations lie towards the lower end of this range, with mean concentrations for groups of soils typically between 10 and 30 mg/kg (Roberts et al., 1996; Percival et al., 1996; Auckland Regional Council, 1999). A relatively small number of sites examined in these surveys had soil Pb concentrations in excess of 30 mg/kg.

The most serious risk associated with Pb contamination has been identified as the direct ingestion of Pb-contaminated soil or biosolids by children or grazing animals (e.g., Smith, 1996; USEPA, 1995). Derivation of a soil ingestion metal limit for Pb using the Ministry for the Environment and Ministry of Health's (MfE and MoH, 1997) approach gives a soil metal concentration of 280 mg/kg for human health, assuming residential land use. Similarly, the USEPA (1995) risk assessment procedure estimated a Pb limit of 300 mg/kg to protect children from the direct ingestion of Pb in sludge. It is therefore recommended that the soil metal limit for Pb remain unchanged from the Department of Health's 1992 limit of 300 mg/kg.

The soil limit for Pb is the same as that used in the UK (Smith, 1996). Some European countries and individual Australian states use lower limits for Pb (McLaughlin et al., 2000; Kabata-Pendias and Pendias, 2001), ranging from 50 to 200 mg/kg. However, the justification for these lower limits is obscure. The USEPA, although using a limit of 300 mg/kg for exceptional grade (EQ) biosolids, have a lower limit of approximately 150 mg/kg for biosolids-treated soil (USEPA, 1993). It would appear from this that the US limit for Pb is more stringent than the limit given in these *Guidelines* or the current UK regulations. However, as pointed out by Smith (1996), it may be more appropriate to draw a comparison between the USEPA pollutant concentration for Pb in biosolids and the UK (or New Zealand) maximum permissible soil concentration, on the assumption that the risks from ingesting Pb in biosolids or biosolid-treated soil are broadly similar. On this basis, the US and UK/New Zealand regulations are in agreement with respect to Pb limits.

The UK have set a soil guideline level for lead-contaminated land using a risk assessment model of 450 mg/kg dry weight for residential land and garden allotments, which is designed to protect young children. Similarly, the Australian National Environment Protection Council soil guideline value is 300 mg/kg for residential land with a garden/accessible soil (NEPC, 1999). These guideline levels are slightly higher than the residential soil quality guideline for human health derived by the Canadian government of 140 mg/kg dry weight (CCME, 1999).

In the UK a review of the rules for sewage sludge application to agricultural land was undertaken in relation to food safety and relevant animal health aspects (MAFF/DoE, 1993a). The review concluded that current evidence was insufficient to warrant a recommendation that the limit (300 mg/kg) for Pb in sludge-amended soils be reduced. However, the review group noted that research was under way on heavy metal concentrations in tissues of animals being fed on sludge-amended diets, and recommended that the situation be kept under review in the light of these studies.

The results of this now completed research (Stark and Wilkinson, 1994; Stark et al., 1995), have been discussed in detail by Smith (1996). According to Smith, under worst-case scenarios the studies showed some increase of Pb in offal products (liver and kidney) of animals, but there is apparently little overall risk to the human food chain from Pb applied in sludge to grassland soils under normal agricultural management. In publishing some of their results more widely, the authors of the reports were a little more cautious in their interpretation, but acknowledged that confirmation of their results is required in the grazing situation (as opposed to "laboratory" feeding trials) (Hill et al., 1998a,b).

It is generally accepted that the uptake of Pb by plants is not a significant pathway in the transfer of Pb from the soil to humans or grazing animals. Although Pb can be absorbed by plant roots, it is not translocated to shoots in high enough concentrations to comprise any risk (Chaney and Oliver, 1996). For example, as reported by Smith (1996), a summary of 117 sludge trials compiled by Vigerust and Selmer-Olsen (1986) showed no increased crop uptake of Pb for sludge-treated soil. In cases where significant increases in plant Pb concentrations have been observed, they have occurred in soils very highly contaminated with Pb (e.g., Rooney et al., 1999), at concentrations well above the 300 mg/kg soil limit.

It is extremely unlikely that phytotoxic effects will occur at soil Pb concentrations of 300 mg/kg or below. As noted by Kabata-Pendias and Pendias (2001), although the levels of Pb in soils that are toxic to plants are not easy to evaluate, most research would suggest that they are well in excess of 300 mg/kg (e.g., Purves, 1985; Rooney, 1996). For example, in the study by Rooney, yields of plants were unaffected by soil Pb at concentrations up to 1,000 mg/kg.

Lead is tightly held by soils, and concentrations of Pb in the soil solution are low (Kabata-Pendias and Pendias, 2001), so that leaching of Pb into groundwater is not generally considered an important issue. The Dutch soil remediation criteria (MHSPE, 2000), which give consideration to the protection of groundwater, have a target value (negligible risk to the ecosystem) for Pb in soil of 85 mg/kg and an investigation level of 150 mg/kg. Such values would appear to be extremely over-cautious in the New Zealand situation. For example, a lysimeter leaching study by McLaren et al. (1999) was unable to detect significant increases of Pb in drainage water from soils to which metal-spiked sewage sludge had been added, for a period of three years after the sludge addition. In this study, the concentration of added Pb was approximately 280 mg Pb/kg in the top 10 cm of soil.

5.2.6 Mercury

Soil metal limit = 1 mg/kg dry weight

Background soil mercury (Hg) concentrations are not easy to estimate due to widespread Hg pollution. However, data reported for various soils on a worldwide basis show that mean concentrations of Hg in surface soils do not exceed 0.4 mg/kg, and in many soils are less than 0.1 mg/kg (Kabata-Pendias and Pendias, 2001). The ANZECC (1992) guidelines for contaminated sites indicated that typical background soil Hg concentrations in Australia are in the range 0.001 to 0.1 mg/kg. Data on Hg levels in New Zealand soils are extremely scarce. However, a recent survey of 99 soils in the Auckland region produced an overall geometric mean of 0.12 mg/kg, which is above this range (Auckland Regional Council, 1999). Over 50% of the samples analysed had Hg concentrations above 0.1 mg/kg, although with the exception of two outliers, all samples analysed in the survey had concentrations below 0.42 mg/kg. No explanation was given for the two outliers (concentrations of 2.3 and 1.6 mg/kg).

The risk assessment model developed by the USEPA (1995) for their biosolids regulations suggests that the direct ingestion of sludge by children is the most critical pathway of exposure to Hg. Phytotoxicity due to Hg is not considered a problem, since even soils quite heavily contaminated with Hg do not appear toxic to plant growth (Smith, 1996). For example, Estes et al. (1973) showed that the yield of turf grass was unaffected by concentrations of Hg in soils as high as 455 mg/kg. In addition, like lead, Hg is not translocated to plant shoots at high enough rates to create a significant risk to the food chain (Chaney and Oliver, 1996).

The Hg concentration limit for EQ biosolids for the biosolids → human pathway was estimated as 17 mg/kg, with a cumulative soil loading limit of 17 kg/ha (approximately 8.5 mg/kg) (USEPA, 1993). Soil

Hg limits estimated for other pathways, including the biosolids → soil → plant → human and biosolids → soil → plant → animal → human pathways, ranged from approximately 90 to 750 mg/kg (USEPA, 1995). However, virtually all other sets of biosolids guidelines, including those developed by Europe and in Australia, have set soil Hg limits at around 1 mg/kg.

The soil limit for Hg of 1 mg/kg has been set to keep the New Zealand biosolids guidelines in line with those of Europe and Australia, and represents no change from the current Department of Health 1992 soil limit value. It should be noted, however, that the justification for this limit appears somewhat obscure, other than preventing an accumulation of Hg above normal background levels.

The soil Hg limit is above the target value of 0.3 mg/kg dry weight of the Dutch soil remediation criteria (MHSPE, 2000), but below their intervention value of 10 mg/kg. However, it is well below the Australian National Environment Protection Council soil guideline value of 15 mg/kg for residential land with a garden/accessible soil (NEPC, 1999).

The UK and Canada have also derived contaminated land criteria for mercury using a comprehensive risk assessment framework. Thus, the UK have established guideline values of 8 mg/kg dry weight for residential with home-grown produce and allotments, and 15 mg/kg dry weight for residential with no home-grown produce (DEFRA, 2002e). The Canadian soil quality guideline for both agricultural and residential settings is 6.6 mg/kg dry weight (CCME, 1999).

5.2.7 Nickel

Soil metal limit = 60 mg/kg dry weight

Soils throughout the world contain nickel (Ni) within the broad range of 0.2 to 450 mg/kg (Kabata-Pendias and Pendias, 2001). Similarly, the ANZECC (1992) guidelines for contaminated sites indicated that typical background soil Ni concentrations in Australia are in the range 2 to 400 mg/kg. Mean background concentrations for most soils within New Zealand tend to be at the lower end of the ANZECC (1992) range. In the survey of data for Canterbury soils, Percival et al. (1996) noted that the highest mean Ni concentration reported was 31 mg/kg. Similarly, in a survey of 99 soils in the Auckland Region, the median Ni concentration was reported as 11.8 mg/kg (Auckland Regional Council, 1999). However, in this survey, volcanic soils had a much higher median value of 128 mg/kg, with individual concentrations ranging from 4.4 to 320 mg/kg.

The USEPA (1995) biosolids risk assessment procedure indicates that phytotoxicity is the most critical issue in relation to Ni contamination of soils. This view appears to be generally accepted worldwide. Although Ni is readily absorbed by plant roots and translocated to shoots, the plant suffers severe yield reduction before humans and livestock are at any risk from consumption of the crop. Thus phytotoxicity essentially protects the food chain (Chaney, 1980; Chaney and Oliver, 1996).

The soil limit for Ni has been set at 60 mg/kg because there is good evidence that, for soils with pH 5.5 and above, a Ni level of 60 mg/kg protects against phytotoxicity (Smith, 1996). This value is the same as the draft phytotoxicity investigation level (PIL) proposed by the NSW EPA (1998). As noted by McLaughlin et al. (2000), the PILs are primarily designed to protect native species used for revegetation of contaminated soils in urban areas, although the values may also be used in agricultural areas because plant growth is the targeted exposure pathway. A Ni soil limit of 60 mg/kg is used in both the South Australia (SA EPA, 1996) and New South Wales (NSW EPA, 1997) biosolids guidelines, and the upper limit of the European range is 75 mg/kg. UK regulations for Ni range between 50 and 110 mg/kg depending on soil pH (UK Statutory Instrument, 1989). The soil concentrations are: 50 mg/kg ($5.0 \leq$ soil pH < 5.5), 60 mg/kg ($5.5 \leq$ pH < 6.0), 75 mg/kg ($6.0 \leq$ pH < 7.0) and 110 mg/kg (pH ≥ 7.0).

Lower soil Ni limits (30–50 mg/kg) are used in some other European countries' biosolids guidelines, and in Tasmania (Tasmanian Department of Environment and Land Management, 1999), but the justification for these lower limits is unclear. The US soil Ni limit is much higher (approximately 210 mg/kg, USEPA, 1993), but appears to be based on the premise that a 50% decrease in crop yield is acceptable.

As noted above, the soil limit for Ni of 60 mg/kg will align New Zealand with several other sets of guidelines, but it does represent an increase on the Department of Health's 1992 guidelines of 35 mg/kg. However, this lower limit is difficult to justify given the current knowledge of soil Ni concentrations at which phytotoxicity is likely to occur (see above). In addition, as noted by Smith (1996), under field conditions and with normal sludges, phytotoxicity is rarely seen, and then only when soil nickel concentrations greatly exceed the UK limit in sludge-treated agricultural land (of 75 mg/kg). However, there are currently no good New Zealand data on Ni phytotoxicity, and further investigation is required.

In addition to protecting against phytotoxicity, the soil limit for Ni should also be protective of the soil microbial biomass (e.g., Chander et al., 1995). A review of the UK rules for sewage sludge application to agricultural land (MAFF/DoE, 1993b) came to the conclusion that, on the evidence available, the current UK limit for Ni (75 mg/kg) adequately protects soil micro-organisms. However, the issue is by no means clear-cut, and further research on New Zealand soils is required.

Contaminated land values for nickel similar to the soil limit in these *Guidelines* have been set in the UK and Canada. The UK Environment Agency has used a risk assessment model to derive guideline values for residential land (with home-grown produce consumption) and garden allotments of 50 mg/kg dry weight, and 75 mg/kg for residential land with no produce consumption (DEFRA, 2002f), while the Canadian soil quality guideline for nickel is 50 mg/kg dry weight (CCME, 1999).

Compared to most other metals, Ni can be relatively mobile in the soil, and can be leached into groundwater. However, the soil limit for Ni should protect against significant leaching. Dutch soil remediation criteria (MHSPE, 2000), which give consideration to the leachability of soil contaminants, have a target value (negligible risk to the ecosystem) for Ni in soil of 35 mg/kg and an intervention value of 210 mg/kg.

In a lysimeter leaching study by McLaren et al. (1999), significant increases of Ni in drainage water were detected from some soils to which metal-spiked sewage sludge had been added. However, apart from some relatively short-term peaks, Ni concentrations were generally below the current New Zealand maximum acceptable values (MAV) for drinking water, and, with further attenuation and dilution in the vadose zone would seem unlikely to pose a serious health or environmental threat. It should also be noted that the metal-spiked sludge used in this study was applied in a single application to give a Ni concentration in the top 10 cm of the soils of approximately 40 mg/kg. The soils in which most Ni leaching occurred were also very acid, so the results from the study can be considered very much a worst-case scenario.

5.2.8 Zinc

Soil metal limit = 300 mg/kg dry weight

Soils throughout the world contain zinc (Zn) within the broad range from 17 to 125 mg/kg (Kabata-Pendias and Pendias, 2001). However, the ANZECC (1992) guidelines for contaminated sites indicated a somewhat wider range of typical background soil concentrations in Australia of 2–180 mg/kg. Data for New Zealand soils generally fit within this range, although occasional higher values do occur. In the survey of 398 soils by Roberts et al. (1996), mean Zn concentrations for different soil types ranged from 6.5 to 97 mg/kg, with a an overall mean of approximately 68 mg/kg. Percival et al. (1996) and Auckland Regional Council (1999) have reported similar values.

As with nickel, the USEPA (1995) risk assessment procedure for their biosolids regulations indicates that phytotoxicity is the most critical issue in relation to Zn contamination of soils. Again, this view appears to be generally accepted worldwide, although some concerns have been expressed about the potential effects of Zn on soil biota and biological processes in the soil (MAFF/DoE, 1993b).

Zn is readily absorbed by plant roots and translocated to shoots, but the plant suffers severe yield reduction before humans and livestock are at any risk from consumption of the crop. Thus phytotoxicity again protects the food chain (Chaney, 1980; Chaney and Oliver, 1996). The soil limit for Zn has been set at 300 mg/kg because there is good evidence that such a level is protective against phytotoxicity in soils treated with biosolids (Smith, 1996). As noted by Smith, some studies indicate that Zn toxicity may occur at soil Zn concentrations lower than 300 mg/kg, although such studies are mainly limited to pot trials. Logan and Chaney (1983) have discussed the potential limitations of using greenhouse pot experiments to assess the effects of toxic metals on plants. In his review of the subject, Smith (1996) reported that under field conditions with normal sludges, phytotoxicity has rarely been seen and then only when soil metal concentrations greatly exceeded the UK statutory limits for metals (300 mg/kg for Zn). Nevertheless, some countries or states have taken a more precautionary approach and use lower soil limits for Zn, ranging from 140 to 250 mg/kg (e.g., SA EPA, 1996; NSW EPA, 1997; Tasmanian Department of Environment and Land Management, 1999). Data for Zn phytotoxicity in New Zealand soils would help to confirm the soil limit.

According to Chander et al. (1995) a soil metal limit for Zn of 300 mg/kg also provides for protection of the soil microbial biomass. However, because of concerns about the effects of Zn on rhizobia bacteria, an independent Scientific Committee (MAFF/DoE, 1993b) recommended that the UK limit for Zn be reduced to 200 mg/kg. This decision was made despite the Committee reporting that the experimental evidence on the effects of metals from sludge on soil micro-organisms was inconsistent and incomplete (Smith, 1996). Preliminary studies from a New Zealand field experiment with metal-spiked sewage sludge (Speir et al., 2000) suggest that Zn is unlikely to affect soil biological activity at concentrations below the soil Zn limit (300 mg/kg). However, the data are far from conclusive at this stage.

Zinc can be relatively mobile in the soil and can be leached into groundwater. However, the soil limit set for Zn in these *Guidelines*, should protect against significant leaching. Dutch soil remediation criteria (MHSPE, 2000), which give consideration to leachability of soil contaminants, have a target value (negligible risk to the ecosystem) for Zn in soil of 140 mg/kg, with an intervention value of 720 mg/kg.

In a lysimeter leaching study by McLaren et al. (1999), significant increases of Zn in drainage water were detected from some soils to which metal-spiked sewage sludge had been added. However, the increases in concentration were relatively short term, and with further attenuation and dilution in the vadose zone would seem unlikely to pose a serious health or environmental threat. It should also be noted that the metal-spiked sludge used in this study was applied in a single application to give a Zn concentration in the top 10 cm of the soils of approximately 255 mg/kg. The soils in which most Zn leaching occurred were also very acid, so the results from the study can be considered very much as a worst-case scenario.

6 STABILISATION ISSUES

6.1 Introduction

Stabilisation of biosolids is achieved by treating them in such a way as to reduce or eliminate the potential for putrefaction, which as a result reduces pathogens, vector attraction and offensive odours. Therefore, although only one letter (A or B) is used to represent the stabilisation grade (see Volume 1, section 4.3 for a background to the biosolids grading system), to achieve this grade, a combination of pathogen, vector attractant reduction and odour reduction must have taken place. This can be achieved by the use of just one treatment process or a combination of different processes. However, the length of time a process needs to be operated for, or the temperature maintained, may be different for pathogen reduction than for vector attractant reduction.

The following section discusses what is meant by pathogen and vector attractant reduction, as well as giving the treatment requirements for achieving the stabilisation Grade A or B. The different types of treatment process commonly used to produce biosolids in New Zealand are also discussed, as are their probable effectiveness in achieving a reduction in pathogens and/or the vector attractant properties of the final product.

6.2 Pathogen reduction requirements

A pathogen is an organism that can cause disease in humans. There are five main types of pathogens observed in biosolids: bacteria, viruses, fungi and yeast, parasitic worms and protozoa (EC, 2001a). Many different processes can be used to reduce the number of pathogens present in biosolids prior to their use.

These *Guidelines* do not seek to recommend specific processes to reduce pathogens, but instead concentrate on proven relationships (e.g., time/temperature) that need to occur during the biosolids production process. How producers of biosolids choose to meet these requirements is up to them (see column 2 of Table 6.1). This approach should enable the development of new procedures and not limit producers to specific technologies.

6.3 Vector attractant reduction (VAR)

In these *Guidelines*, the term ‘vector’ refers to potential carriers of disease, such as flies, mosquitoes, birds and rodents. In order to meet both Grade A and B stabilisation standards, the biosolids must have been treated in such a way as to reduce their attractiveness to these disease carriers; this process is known as vector attractant reduction (VAR). In the context of biosolids, VAR can be achieved by either:

- reducing the attractiveness of the biosolid to vectors, by biological processes or specific chemical and physical conditions; or
- by removing access to the biosolid from vectors, usually by incorporation of the biosolid into soil shortly following application (within a matter of hours).

High-quality biosolids are those in which vector-attracting compounds, such as volatile solids, have been reduced or removed. This is because reducing vector attraction effectively decreases the risk to public health presented by the biosolid.

VAR methods can apply to both biosolid manufacturing processes and land application processes. In terms of the unrestricted use category (see Volume 1, sections 4.4 and 5), the VAR treatment of the biosolids should occur during the manufacturing process rather than during the application process. This is because the application to land of biosolids in the unrestricted use category is, by definition, uncontrolled.

VAR should take place either at the same time as pathogen control or just after it. VAR control measures should also be taken during storage of biosolids. Adhering to this management practice will also reduce regrowth of pathogenic bacteria. Six ways are suggested for reducing the attractiveness of biosolids to vectors (Table 6.1).

6.4 Biosolids stabilisation requirements

Pathogen reduction and vector attraction reduction requirements necessary to achieve either stabilisation Grade A or B are summarised in Table 6.1. These have been derived from the USEPA Rule 503 requirements (USEPA, 1993) and the NSW EPA (1997) stabilisation grading.

To achieve a stabilisation Grade A, the biosolids must meet one of the pathogen reduction criteria *and* one of the VAR criteria as well as having an *accredited* quality assurance process (refer Volume 1, section 7). The biosolids must also meet the pathogen levels (given in Table 6.2) after processing but before application.

To achieve a stabilisation Grade B the biosolids must meet at least *one* of the six VAR criteria specified for Grade A. There is no requirement for a producer of Grade B biosolids to demonstrate compliance with a pathogen reduction criterion. This is because there are no numerical pathogen limits set for Grade B biosolids, which is in line with NSW EPA requirements. There is also no requirement for the quality assurance process to be accredited in order for the product to achieve a Grade B. However, it is recommended that documented quality assurance procedures be independently verified for Grade B. Good management practices strongly support quality assurance procedures for all production facilities, and these *Guidelines* support and recommend this approach.

Table 6.1: Stabilisations requirements

Grade	Acceptable pathogen reduction processes	Acceptable vector attraction reduction methods
Grade A	Accredited quality assurance ***plus*** one pathogen reduction process from the 3 options below:- **1. Time–temperature processes** **(a) $\geq$ 7% DS** Within the relationship $t = \frac{131,700,000}{10^{0.14T}}$. ; t = days, T = °C where $T \geq 50$°C and $t \geq 15$ seconds, or **(b) < 7% DS** Within the relationship $t = \frac{50,070,000}{10^{0.14T}}$; t = days, T = °C where $T \geq 50$°C and $t \geq 30$ minutes, or **(c) Composting**[a] (i) In-vessel: $T \geq 55$°C for ≥ 3 days, or (ii) Windrow: $T \geq 55$°C for ≥ 15 days with a minimum of 5 turnings during this period[b] **2. High pH – high temperature process** pH > 12 (measured at 25°C) for ≥ 72 hours and maintain T > 52°C for 12 consecutive hours within the 72 hours, all from the same chemical application, and drying to > 50% DS afterwards. **3. Other processes** Demonstration by agreed comprehensive process and product monitoring that the Grade A pathogen levels can be consistently met.	Accredited quality assurance ***plus*** at least one VAR/odour method from the list below: 1. mass of volatile solids in biosolids shall be reduced by a minimum of 38%[c]; or 2. biosolids $\geq$ 90% DS if heat dried at T > 80 °C; or 3. $T \geq 40$°C for ≥ 14 days and $T_{ave} \geq$ 45°C; or 4. SOUR @ 20°C ≤ 1.5 g/m^3 for liquid sludges from aerobic processes; or 5. pH $\geq$ 12 @ 25°C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours; or 6. soil incorporation.
Grade B	Verified quality assurance ***plus*** Storage and/or restricted access (see Table 6.3)	Verified quality assurance ***plus*** one of the VAR requirements from Grade A

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[a] *All compost must have 30 days maturation pre-use.*

[b] *5 x 3 days at $T \geq 55$°C plus time periods to reach 55°C after each turning.*

[c] *Based on representative samples before and after the reduction process (see section 9).*

6.5 Pathogen standards

The pathogen standards detailed in these *Guidelines* have been determined after a review of the pathogen requirements of the USEPA (1993), NSW EPA (1997), NRMMC (2003), Department of Health (1992) and the Wellington Regional Council Living Earth Joint Venture (LEJV) consent.[5] In the case of a guideline proposing more than one standard (e.g. USEPA), the standard for the highest-quality biosolid was used in the review.

Standards have been set for faecal coliforms, salmonella, campylobacter, enteric viruses and helminth ova.

Table 6.2: Pathogen standards[1]

Pathogen	Verification sampling	Routine sampling
E. coli	< 100 MPN[2]/g	< 100 MPN/g
Campylobacter	< 1/25 g	N/A[4]
Salmonella	< 1/25 g	N/A
Enteric viruses	< 1 PFU[3]/4g	N/A
Helminth ova	< 1/4g	N/A

[1] In the event that one of the samples fails to meet the product verification standards specified, all of the pathogen tests for that sample must be repeated. One hundred percent compliance must be achieved in order to meet the stabilisation grade standard. (In the case of biosolid manufacturing facilities in existence prior to the publication of these Guidelines, *it is acceptable to use data up to 12 months old for the purposes of product verification. Older data cannot be used).*

[2] MPN = most probable number.

[3] PFU = plaque forming unit.

[4] Not applicable.

The rationale for each pathogen standard set is as follows.

6.5.1 Escherichia coliforms (*E. coli*)

The LEJV consent for the Wellington biosolids plant is set at < 200 MPN/g, which mirrors the Department of Health (1992) Category II requirement. The NRMMC (2003) guideline sets a lower limit of < 100 MPN/g, while USEPA and NSW EPA set a much higher limit of < 1,000 MPN/g. Only the USEPA gives a reason for the setting of its limit, which is that this level of coliforms has been shown to correlate with low numbers of salmonella. The USEPA is therefore the only agency to give the option of measuring *either* coliforms *or* salmonella, but there is no requirement to monitor both.

6.5.2 Campylobacter

Whereas *Salmonella* spp. are traditionally used as an indicator for pathogen removal, the high incidence of campylobacter infection in the New Zealand community makes it a greater risk. For this reason campylobacter is required for verification sampling.

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[5] It should be noted that the EU directive (CEC, 1986) does not contain any standards for pathogen numbers, although four member states have chosen to implement standards for several pathogens in their own domestic legislation.

6.5.3 Salmonella

The limit given in these *Guidelines* is based on the requirement of the LEJV consent figure of 1/25g. The limits given in USEPA (1993), NRMMC (2003) and NSW EPA (1997) are less stringent than this, but no reasoning could be found for any of the standards reviewed. For this reason it was determined that the LEJV consent requirement was effectively a New Zealand-based standard and that this should be reflected in these *Guidelines*. The requirement to analyse the biosolids for salmonella applies only to verification sampling.

6.5.4 Enteric viruses

The limit of < 1 PFU/4g biosolid is based on USEPA (1993) and NSW EPA (1997) requirements. The LEJV consent is very similar at 1 PFU/4g. In these *Guidelines* the requirement to analyse the biosolids for enteric viruses only applies during the initial verification period. If the pathogens are present at numbers less than the standard (Table 6.2), the requirement to monitor them during routine sampling should be dropped. This is in line with the USEPA requirements and the LEJV consent. This type of test is expensive and the results can take up to four weeks to obtain, which is significant given that the biosolids cannot be sold until the sample results are known.

6.5.5 Helminth ova

The limit of < 1/4g has been derived from the USEPA (1993). The LEJV consent has been set at 1/4g and the NSW EPA (1997) is similar with 1 PFU/4g. The use of PFU in conjunction with helminth ova was considered unusual, which is why it has not been used here. As with enteric viruses, the requirement to analyse the biosolids for helminth ova only applies during the initial verification period. If the pathogens are present at numbers less than the standard (Table 6.2), the requirement to monitor them during routine sampling should be dropped. This is in line with the USEPA requirements and the LEJV consent. This type of test is expensive and the results can take up to four weeks to obtain, which is significant given that the biosolids cannot be sold until the sample results are known.

6.5.6 *Cryptosporidium/giardia*

These are a known problem in New Zealand, and process controls should render them non-viable. Current test methods are not yet sufficiently reliable to warrant setting standards for biosolids. If reliable test methods are established then standards should be considered.

6.6 Treatment process options

Biosolids producers have access to a wide range of treatment processes to enable them to meet the different stabilisation grades of biosolids recommended in these *Guidelines*. Some of these processes are effective at reducing both pathogens and vector attraction, whereas others may be better at one or the other. In the latter case it may sometimes be necessary to combine different treatment processes to ensure that the final product achieves a specific stabilisation grade. These treatment processes do little to change the mass of contaminants, but concentrations may increase due to reductions in volatile solids, and/or decrease due to the addition of chemicals (e.g., lime).

There are five broad approaches to controlling pathogens in biosolids:

- high temperatures
- radiation
- chemical disinfectants
- reducing volatile organic content
- removing moisture.

Within these five approaches there are a number of different types of processes and technologies, some of which can be categorised under more than one approach. These processes and the approaches they are based on are briefly discussed below.

6.6.1 Pasteurisation

Pasteurisation involves heating the sludge to a temperature of 70–80°C for approximately 30 minutes. This treatment will reduce the number of pathogens, but it cannot be considered a stabilisation process in its own right and is usually used in conjunction with other processes, such as mesophilic digestion.

Pasteurisation can be achieved by using heat exchangers or steam injection. In theory the addition of quicklime (an exothermic or heat-producing reaction) could result in pasteurisation, but this is a very difficult process to control as it requires thorough mixing. Odour can be an issue with pasteurised biosolids because the process does not stabilise organic matter but increases the soluble volatile content.

6.6.2 Irradiation

Irradiation of sewage sludge reduces pathogens by disrupting their cell content, which either destroys the organism or prevents it from reproducing. The irradiation used can be gamma or beta ray-based, and its effectiveness relates to the length of dose applied to the sludge. Although irradiation is commonly used as a pathogen reduction technique in several countries in Europe, it is not used in the US or New Zealand.

6.6.3 Lime stabilisation

Lime stabilisation of sewage sludge to form biosolids works by raising the pH to 12 or more, which has the effect of destroying or inhibiting the pathogens present. However, the effectiveness of this treatment process is related to the length of time the pH is constantly above 12. Lime can be added to liquid sludge or to dewatered sludge, but effective treatment depends on adequate mixing of the lime with the sludge to ensure the pH is raised uniformly. The addition of lime to sludge also has the benefit of increasing the level of dry matter present and making handling of the final product easier.

The main problem associated with lime stabilisation is the development of odours if the pH falls below 10.5; this is particularly an issue following land application. Soil incorporation of the biosolids as part of the application process can overcome this.

Lime stabilisation is effective in reducing bacteria and viruses as well as reducing vector attractant properties. Protozoan cysts may be inactivated, but the addition of lime is not thought to be effective on helminth ova, unless combined with heat (USEPA, 1999).

6.6.4 Composting

Composting is an aerobic process, which involves mixing treated sludge with a co-product such as sawdust, greenwaste or wood chips[6]. The co-product provides a source of carbon, increases porosity (and therefore oxygen flow) and absorbs moisture. Heat generated as a result of the aerobic biological activity that takes place destroys the pathogens in the sludge.

Organic material will compost on its own over time without any control, but the use of composting as a biosolids treatment process must be carefully designed, controlled and managed to ensure appropriate time/temperature criteria are achieved throughout the process. There are three types of composting process.

- *Windrow* – the sludge/co-product mixture is placed in long rows, which are turned periodically to introduce air, reduce moisture levels and maintain even temperatures.
- *Aerated static piles* – the sludge/co-product mixture is laid over perforated pipes, through which air is blown or sucked. This introduces oxygen, which aids decomposition and also reduces moisture. If the air is sucked through the compost rather than blown it can also help odour control.

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[6] Treated timber must not be used as a co-product (see section 3.5.3).

- *In-vessel systems* – there are many different types of vessel system in use, but all operate under carefully controlled conditions, including active aeration.

Composting, if conducted under the right conditions, is effective at reducing bacteria, viruses, helminth eggs, protozoa, vector attraction and odour. Composting also reduces the water content in the final product to as much as 60% dry matter, which makes handling much easier and results in a commercially attractive product.

Research by the USEPA (1999) has found that windrow composting may not be as effective at pathogen reduction as in-vessel and aerated static-pile composting. Specific guidance has been given by the USEPA (1999) to improve the effectiveness of the windrow process.

While pathogen and VAR can be achieved without a curing process, such a process can enhance odour reduction.

6.6.5 Anaerobic digestion

Anaerobic digestion is undertaken in closed tanks in the absence of oxygen and with or without additional heating. There are two types: high-rate, which involves mechanical mixing and heating of the sludge, and standard-rate, which is generally conducted at ambient temperature with no mechanical mixing. The majority of plants use high-rate digestion. Both types of anaerobic digestion result in a reduction of volatile solids of between 40 and 50% according to EC (2001b) and between 35 and 60% according to USEPA (1999). This reduction in volatile solids reduces the likelihood of regrowth of pathogens after treatment.

Depending on how the system is managed, anaerobic digestion can be operated in either a mesophilic (~35°C) mode or a combination thermophilic (> 50°C) followed by mesophilic mode.

Methane is a by-product of anaerobic digestion, and this can be recovered and used to heat the process, thus reducing costs. Anaerobic digestion reduces bacteria, protozoa and viruses, although viable helminth eggs may not be significantly affected.

6.6.6 Aerobic digestion

Aerobic digestion of sludge is carried out in either open or closed vessels. In order to supply enough oxygen to the bacteria in the sludge, the mixture either has to be agitated or have air injected into it. Heat is generated when the bacteria present break down organic matter to carbon dioxide, nitrate nitrogen and water. Depending on how the system is managed, aerobic digestion can operate in either the mesophilic or thermophilic temperature ranges. If operated correctly, aerobic digestion can result in a 40% reduction of volatile matter in the mesophilic range, and up to a 70% reduction if thermophilic aerobic digestion is used. It is also effective at reducing bacteria, protozoa, viruses and helminth eggs, although the latter is more effective at the higher temperatures associated with thermophilic aerobic digestion.

The operation of an aerobic digester can either be batch or continuous. The length of time the sludge needs to remain within the digester for effective pathogen reduction to take place, irrespective of the process used, is based on the temperature the process is operating at.

6.6.7 Thermal and air drying

In air drying, the sludge is applied to a sand or gravel bed and allowed to dry naturally over a period of months. During drying, biological processes take place, such as decomposition of organic matter, formation of ammonia and reduction in moisture, which in turn reduce bacteria, protozoa and viruses. Viable helminth eggs may also be reduced if the drying temperature is high enough, but this is dependent on the species involved, as some are much hardier than others.

Air drying is a simple process, but it requires a lot of land and is dependent on ambient temperatures for its effectiveness (higher temperatures are better). A dry matter content of up to 50% can be reached depending on local climatic conditions (EC, 2001b). Sludge should also be partially digested before it is applied to the drying bed. The USEPA (1999) has recommend that air drying be used as an additional process to aerobic or anaerobic digestion to meet the 38% volatile solids reduction requirement.

In thermal drying, sludge is dried by direct or indirect contact with heat. If conducted properly this has the effect of reducing bacteria, viruses, protozoa and viable helminth eggs. It is also effective at reducing VAR, provided the biosolids remain dry after treatment.

There are four main types of thermal driers: flash, spray, rotary and steam. All operate at different temperatures, and as a result the percentage dry matter in the final product can range from 35 to 90%. This type of treatment process results in a product that is greatly reduced in volume from the raw sludge and can be applied to land like a mineral fertiliser.

6.6.8 Long-term storage

Long-term storage of sewage sludge will result in the reduction of bacteria and viruses present. However, the effectiveness of the process depends on the type of treatment prior to storage, and the length of storage time. It is unlikely that long-term storage would affect the viability of protozoa. The ambient temperature, with decreasing pathogen reduction at lower temperatures, also affects the efficiency of this process. Nitrogen levels in the sludge will also be reduced, which has the effect of reducing the agricultural value of the product.

Recommended controls for stabilisation Grade B biosolids and/or public protection using storage and exclusion periods (i.e. access restrictions) have been adapted from the Department of Health (1992) guidelines, and are summarised in Table 6.3 (see also, Volume 1, section 6.5).

Table 6.3: Recommended controls for stabilisation Grade B biosolids, depending on end use

Land use	VAR requirement (see Table 6.1)	Recommended controls
Salad crops, fruit, other crops for human consumption that may be eaten unpeeled or uncooked	■ Mass of volatile solids in biosolids shall be reduced by a minimum of 38%; or ■ SOUR @ 20°C $\leq$ 1.5 g/m^3 for liquid sludges from aerobic processes; or ■ pH $\geq$ 12 @ 25°C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours.	May be applied immediately ***Plus*** Soil incorporation (see Volume 1, section 6.11) ***Plus*** A further waiting period of at least 1 year before crops are sown (the land may be used for other purposes in the meantime).
	■ Storage/restricted access.	Store or lagoon for at least 1 year prior to application. ***Plus*** Soil incorporation (see Volume 1, section 6.11). ***Plus*** A further waiting period of at least 1 year before crops are sown (the land may be used for other purposes in the meantime).
Public amenities, sport fields, public parks, golf courses, play grounds, land reclamation	■ Mass of volatile solids in biosolids shall be reduced by a minimum of 38%; or ■ SOUR @ 20°C $\leq$ 1.5 g/m^3 for liquid sludges from aerobic processes; or ■ pH $\geq$ 12 @ 25°C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours	Store or lagoon for at least 6 months prior to application. ***Plus*** Soil incorporation (see Volume 1, section 6.11). ***Plus*** Restriction on public access for period of time necessary to establish a full vegetation cover on the land (see Volume 1, section 6.12).

Land use	VAR requirement (see Table 4.1)	Recommended controls
	■ Storage/restricted access.	Store or lagoon for at least 1 year prior to application. ***Plus*** Soil incorporation (see volume 1, section 6.11) ***Plus*** Restriction on public access for period of time necessary to establish a full vegetation cover on the land (see section Volume 1, 6.12).
Fodder crops and pasture, orchards where dropped fruit is not harvested, turf farming, industrial or non-edible crops, crops that will be peeled or cooked before eating	■ Mass of volatile solids in biosolids shall be reduced by a minimum of 38%; or ■ SOUR @ 20^{o}C $\leq$ 1.5 g/m^3 for liquid sludges from aerobic processes; or ■ pH $\geq$ 12 @ 25^{o}C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours.	May be applied immediately. ***Plus*** Soil incorporation (see Volume 1, section 6.11). ***Plus*** Fruit and turf should not be harvested or pastures grazed for at least 6 months after applications. ***Plus*** Crops that will be peeled or cooked should not be harvested for at least 6 months after application.
	■ Storage/restricted access.	Store or lagoon for at least 1 year prior to application ***Plus*** Soil incorporation (see Volume 1, section 6.11). ***Plus*** Fruit and turf should not be harvested or pastures grazed for at least 6 months after applications. ***Plus*** Crops that will be peeled or cooked should not be harvested for at least 6 months after application.
Forest, trees or bush scrubland	■ Mass of volatile solids in biosolids shall be reduced by a minimum of 38%; or ■ SOUR @ 20^{o}C $\leq$ 1.5 g/m^3 for liquid sludges from aerobic processes; or ■ pH $\geq$ 12 @ 25^{o}C for at least 2 hours and pH $\geq$ 11.5 for 22 more hours.	May be applied immediately. ***Plus*** Public access restricted for 6 months (see Volume 1, section 6.12). ***Plus*** Buffer zones should be fenced and signposted.
	■ Storage/restricted access.	Store or lagoon for at least 1 year prior to application. ***Plus*** Public access restricted for 6 months (see Volume 1, section 6.12). ***Plus*** Buffer zones should be fenced and signposted.

Source: Updated from Department of Health (1992).

Note: SOUR = standard oxygen uptake rate.

7 BIOSOLIDS CONTAMINANT GRADING

7.1 Metals grading

Table 7.1 gives the metals contaminant concentrations in biosolids that are applicable to each of the chemical contaminant grades (see Volume 1, section 4.3.2 for a background to the contaminant grading system).

When grading a biosolid, if any of the contaminant concentrations are Grade b then the biosolid has to be graded b, even if only one of the results falls into this category. If any of the contaminant concentrations are above the limits given for Grade b, then the solids either have to be blended with a co-product to reduce the contaminant level and hence comply with biosolids Grade b limits, or they cannot be called biosolids and must be disposed of as a contaminated sludge.

Table 7.1: Biosolids classification, by metal contaminant levels

Parameters	**Soil limit or ceiling concentrations[1] (mg/kg dry weight)**	**Biosolids limits[2]**	
		Grade a max. concentration[3] (mg/kg dry weight)	**Grade b max. concentration[3] (mg/kg dry weight)**
Arsenic	20	20	30
Cadmium	1	1	10
Chromium	600	600	1500
Copper	100	100	1250
Lead	300	300	300
Mercury	1	1	7.5
Nickel	60	60	135
Zinc	300	300	1500

[1] *Refer to section 5 for a discussion on the soil limit. Refer to Volume 1 section 6.11 and Appendix III for advice on compliance.*

[2] *See section 8 for product monitoring requirements, and Tables8.1 for sampling frequencies. In the case of biosolids manufacturing facilities in existence prior to the publication of these* Guidelines*, it is acceptable to use data up to 12 months old for the purposes of* product verification*. For the purposes of determining compliance at the 95 percentile for* routine sampling*, the age of the data set shall be no more than 2 years. This avoids the scenario of old data marking upward trends in contaminant concentration.*

[3] *Compliance against the biosolid limit values are to be 95 percentile from sampling results, with a maximum of 20% above the limits.*

The metal concentrations for Grade ‘a’ have been set at the soil limits. An interim Grade ‘a’ is proposed for the first 10 years (2003–2012) to allow biosolids producers time to implement changes to bring their products within the more stringent Grade ‘a’ standard (refer Volume 1, sections 4.3.2 and 4.3.3, and Table 4.2). This interim Grade ‘a’ allows slightly higher values for cadmium, copper, mercury and zinc, set at the same values used in the LEJV consent conditions for the Wellington region-wide distribution of composted biosolids.

Given that mass loading limits the amount of contaminants applied to land, there need not (theoretically) be a Grade b maximum concentration. However, a maximum concentration is required for management controls and to reinforce the differentiation of biosolids (as a quality product) from sludges. These limits are given in Table 7.1.

The metal concentrations given for Grade b have been selected as 50% of the NRMMC (2003) Grade C2 values, except for lead, where it was less than the Grade 'a' requirement, in which case the Grade 'a' requirement took precedence. The basis of this reasoning was that the Grade C2 value from NRMMC were considered too high, and would allow the land application of biosolids containing New Zealand sludges with unusually high contaminant levels. The available New Zealand biosolids data (Ogilvie, 1998) show that the Grade b maximum limits adopted should be achievable for the majority of New Zealand wastewater treatment plants.

Most of NRMMC Grade C2 requirements appear to have been taken from the NSW EPA Grade C requirements. The exceptions to this are arsenic and copper, which are higher than the NSW EPA figures. The concentration for chromium is given as a range in the NRMMC guideline, but no information is given in the supporting text as to how the range should be used. The value for chromium adopted for these *Guidelines* is half the top figure (3,000 mg/kg) from the NRMMC (2003).

Both the NRMMC Grade C2 and the NSW EPA Contaminant Grade C are similar to the New Zealand Grade b with respect to their classification and potential biosolids end-uses. Grade C2 is the cut-off limit defining a biosolid in the NRMMC guidelines, whilst contaminant Grade C is the lowest contaminant grade allowed for agricultural use in the NSW EPA (1997) guidelines.

In USEPA Rule 503 (USEPA, 1993), the 'ceiling limit' category is in effect equivalent to the New Zealand contaminant Grade b. However, the actual concentrations given under the ceiling limit category are considerably higher for all metals than those given in other guidelines (e.g., the USEPA ceiling limit for copper is 4,300 mg/kg). According to information published by the USEPA (1995), their limits have been derived using a comprehensive risk assessment process, but because the figures derived are so different to Australian and European figures there has been much discussion in the scientific community regarding their validity (Speir, 1999). Due to the concerns regarding the derivation of the US figures, they were not considered when deriving the New Zealand biosolid limits specified in these *Guidelines*.

7.2 Organic compounds

Limits for organochlorine pesticides and PCBs (Table 7.2) are based on NSW EPA (1997) contaminant grades, and are also consistent with limits set in NRMMC (2003).

Biosolids Grade 'a' is based on the NSW EPA Contamination Grade A, which, along with the appropriate stabilisation grade is classified for unrestricted use in the following applications:

(i) home lawns and gardens
(ii) public contact sites
(iii) urban landscaping
(iv) agriculture
(v) forestry
(vi) soil and site rehabilitation
(vii) landfill disposal
(viii) surface land disposal within boundaries at wastewater treatment plants.

To support beneficial use programmes for biosolids, these G*uidelines* allow for an interim Grade 'a' limit for dieldrin that will apply to December 2012 (refer Volume 1, sections 4.3.2 and 4.3.3, and Table 4.2). Dieldrin is known to be a problematic contaminant at several New Zealand wastewater treatment plants, and the interim Grade 'a' limit should allow producers time to implement changes to bring their biosolids within the more stringent Grade 'a' standard.

Grade b is based on the NSW EPA Contamination Grade B, which, along with the appropriate stabilisation grade, is classified as Restricted Use 1. Land application uses for Grade B are the same as for Grade A except for the exclusion of home lawns and gardens. The Grade B threshold levels are under review by the NSW EPA.

It should be noted that the NSW EPA has two more lenient contaminant grades (C and D), which are approved for beneficial re-use. Contaminant Grade C excludes public contact sites and urban landscaping. Contaminant Grade D further excludes agriculture.

The dioxin limit for Grade ‘a’ biosolids is 0.00003 mg TEQ/kg and for Grade b biosolids it is 0.00005 mg TEQ/kg. These are based on the German federal regulations (mass/load approach) but take into consideration the ranges in nitrogen content and application rates that may be used in New Zealand. The German regulations (German Federal Law Gazette, 1992) were the basis for the dioxin limit in the LEJV consent conditions for region-wide distribution of composted biosolids.

The dioxin limit in the German regulations is the same concentration as proposed in the current draft of the European Commission (EC, 2000) biosolids regulation. The NSW EPA have omitted to set a dioxin limit subject to a review of studies undertaken in the US. The NRMMC (2003) notes that dioxins in biosolids are an emerging issue and an area for possible research.

The organochlorine pesticides listed in Table 7.2 have been deregistered by the Pesticides Board, and as such are no longer available commercially and cannot be imported (see section 2.2.1). The use and storage of PCBs is also prohibited. Consequently, the levels of these contaminants in biosolids can be expected to decrease with time. In contrast, dioxins continue to be emitted from a variety of industrial and domestic activities (sections 2.2.1 and 2.2.3). Nonetheless, steps that have been taken to reduce the release of dioxins to the environment, together with other ongoing dioxin reduction initiatives, can be expected to have a beneficial effect and result in falling biosolid dioxin levels over time.

7.3 Nutrients

If biosolids are to be used beneficially on agricultural land, they will need to be analysed for nitrogen content so that the biosolids application rate can be calculated in accordance with the agronomic loading rate. If the biosolids are to be marketed as a fertiliser it may also be necessary to analyse the phosphorus and potassium contents at the same time as nitrogen, in order to calculate the N:P:K ratio. For New Zealand conditions, a default nitrogen loading of 200 kg total N/ha/year is adequately conservative for most circumstances (see Volume 1, section 6.3).

It is recommended that monitoring for nutrients be conducted at the same frequency as the monitoring for chemical contaminants (excluding dioxins) (see Table 8.2, section 8). This is in accordance with NSW EPA guidelines.

7.4 Annual loadings

Biosolids application should not cause soil contaminant limits to be reached for at least a 20-year application period. This is in line with the USEPA approach, which they consider to be conservative as regular or continuous applications to land for 20 years are unlikely for most agricultural or home garden circumstances. The overall application rates for restricted use biosolids[7] are primarily governed by mass loading limits to give an absolute control of the nominated soil contaminant limits.

Table 7.2: Biosolids classification, by organic contaminant levels

Parameter	Soil limit or ceiling concentrations[1] (mg/kg dry weight)	Biosolids limits[2]	
		Grade a max. concentration[3] (mg/kg dry weight)	Grade b max. concentration[3] (mg/kg dry weight)
DDT/DDD/DDE	0.5	0.5	0.5
Aldrin	0.02	0.02	0.2
Dieldrin	0.02	0.05	0.2
Chlordane	0.02	0.02	0.2
Heptachlor & Heptachlor epoxide	0.02	0.02	0.2
Hexachlorobenzene (HCB)	0.02	0.02	0.2
Hexachlorocyclohexane (Lindane)	0.02	0.02	0.2
Benzene hexachloride (BHC)	0.02	0.02	0.2
Total polychlorinated biphenyls (PCBs)	0.1[4]	0.2[5]	0.2
Total dioxin TEQ[6]	0.00001[7]	0.00003	0.00005

[1] *Refer to Volume 1 section 6.11 and Appendix III for advice on compliance.*

[2] *See section 8 for product monitoring requirements, and Tables 8.2 for sampling frequencies. In the case of biosolid manufacturing facilities in existence prior to the publication of these* Guidelines*, it is acceptable to use data up to 12 months old for the purposes of* product verification *(see Section 8). For the purposes of determining compliance at the 95 percentile for* routine sampling*, the age of the data set shall be no more than 2 years for organochlorine pesticides and PCBs and no more than 5 years for dioxins. This avoids the scenario of old data masking upward trends in contaminant concentration.*

[3] *Compliance against the biosolid limit values are to be 95 percentile from sampling results, with a maximum of 20% above the limits.*

[4] *The NSW EPA guidelines were amended in July 2001, from 0.3 to "no detectable PCBs at a limit of detection of 0.1mg PCB/kg.soil. Biosolids should not be applied to land where PCBs have been detected at a limit of detection of 0.1mg PCB/kg or when the calculated level of PCBs applied to the soil will exceed 0.02 mg PCB/kg.soil."*

[5] *The NSW EPA guidelines were amended in July 2001, from 0.3 to "no detectable PCBs at a limit of detection of 0.2 mg PCB/kg.biosolids."*

[6] *'Dioxin' means the seventeen 2,3,7,8-chlorinated congeners of the polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans. The residue limit is expressed as 2,3,7,8-tetrachlorodibenzo-p-dioxin toxic equivalents (TEQ) using the toxic equivalent factors (TEF) assigned by the World Health Organisation at a 1997 WHO consultation (Van den Berg et al., 1998). The TEQ is calculated by multiplying the concentration of each 2,3,7,8-chlorinated congener measured in an analysis by its appropriate TEF (as assigned by the WHO), and adding the result from these calculations together. If any 2,3,7,8-chlorinated congener is not detected in the analysis, that dioxin must be assumed to be present and an amount equal to one half of the analytical detection limit must be used in the TEQ calculation.*

[7] *The dioxin soil limit value is taken from MfE and MoH (1997).*

8 MONITORING AND QUALITY ASSURANCE

8.1 When to monitor

8.1.1 Monitoring the final product

Ideally, monitoring the quality of biosolids should occur just prior to their use. This practice is in accordance with the USEPA, European Union, NSW EPA and the NRMMC guidelines.

When determining the stabilisation grade, pathogen reduction monitoring should only be undertaken on the final product (just before sale), because pathogenic organisms may regrow after treatment has taken place. Producers of biosolids or products containing biosolids should be aware of this requirement, and ensure they have enough storage space to hold the product while waiting for the results of monitoring. They should also inform the analytical laboratory of the need to complete the analysis as quickly as possible.

When determining the biosolids contaminant grade, it is not as important to undertake monitoring just before sale, as the concentrations of these parameters (i.e. metals and organic contaminants) are unlikely to change after treatment. However, if the biosolids are to be mixed with another material (e.g., as part of a composting process) before sale, then any monitoring should be undertaken on the final product.

8.1.2 Monitoring the raw product

Although these *Guidelines* relate to the beneficial use of biosolids, it is worthwhile for wastewater managers to consider monitoring wastewater influent as a means of determining the chemical contaminant quality of the end product after it has been treated.

These *Guidelines* give maximum limits for a variety of chemical parameters that may be present in biosolids (Tables 7.1 and 7.2). From these limits it should be possible for wastewater managers to calculate what the maximum allowable concentrations of these determinands in the wastewater arriving at a treatment plant should be in order for the resultant biosolids to comply with these limits. By monitoring the influent, and provided that the maximum allowable influent load threshold is not exceeded, there will be a high degree of confidence that the contaminant concentrations in the resulting biosolids will be in compliance with these *Guidelines.*

8.2 Types of monitoring

The biosolids monitoring process in these *Guidelines* is divided into two phases:

- verification monitoring
- routine monitoring.

Initially it was proposed that within these phases there would be distinct subsets of sample numbers relating to whether the biosolids production was a batch process or a continuous process. This is the approach used in the NSW EPA (1997) guidelines. The reason for the different sampling protocols for continuous and batch-produced biosolids is that the quality of each batch is not related to the quality of the other batches produced, whereas there is a form of quality consistency in the continuous process.

However, if a similar approach were adopted in these *Guidelines*, it would mean that batch producers would be required to take significantly more samples than producers who use a continuous process. As well as this being prohibitively expensive, it was also envisaged that confusion could arise as to which sampling regime was appropriate for those producers who use a semi-continuous process. For this reason it was decided to use a simplified system that would apply to all processes used for producing biosolids.

8.2.1 Verification monitoring

Verification monitoring is the name given to the phase of monitoring undertaken when:

- a new plant is commissioned
- process or equipment changes are made to an existing plant
- pathogen or chemical contaminant levels in the biosolids exceed the limits specified in these *Guidelines* (see Tables 6.2, 7.1 and 7.2).

8.2.2 Routine monitoring

Routine monitoring is typified by a less onerous sampling regime than that required for verification monitoring. This is because the product quality verification monitoring is used to demonstrate the ability and stability of the process and/or the quality of the product. Once these are determined to be satisfactory, the number of samples taken and the number of monitoring periods can be reduced.

8.3 Number of samples

The minimum number of samples that should be taken in each monitoring phase and for each grade are detailed in Table 8.1 and 8.2. Note that the sample numbers given are not designed to result in statistically representative data. This approach was felt unnecessary as the quality assurance controls comprise both process and product monitoring, so the product monitoring is a supporting indicator that the process is working correctly. In addition, the analytical cost of monitoring biosolids at a statistically representative frequency would be prohibitively high, effectively stymieing any beneficial use applications.

Table 8.1: Stabilisation grade sampling frequencies

Grade	Monitoring type	Sampling regime	Parameters to be monitored
A	Product verification[1,2]	≥ 15 evenly dispersed grab samples per month for a 3-month period with ≤ 3 failures. If > 3 failures then the 15 following consecutive grab samples must comply.	■ *E. coli* ■ *Salmonella* ■ *Campylobacter* ■ enteric viruses ■ helminth ova ■ VAR
	Routine sampling	≥ 1 grab sample per week	■ *E. coli* ■ VAR
B	Product verification[2]	Not applicable for pathogen testing	■ VAR[3]
	Routine sampling	Not applicable for pathogen testing	■ VAR[3]

[1] *No more than three samples should be taken per day during this period.*

[2] *In the case of biosolids manufacturing facilities in existence prior to the publication of these* Guidelines *it is acceptable to use data up to 12 months old for the purposes of* product verification.

[3] *If a barrier is to be used for VAR no monitoring is required at the production stage.*

Table 8.2: Contaminant grade sampling frequencies

Grade	Sample type	Number of samples
a ***and*** b	Product verification[1]	■ Metals: 1 composite[2]/week over a 3-month period. ■ Organics: 1 composite sample[2]/month over a 3-month period ■ Dioxins: 1 composite[3]/3 months
	Routine sampling[1]	■ Metals: $\geq$ 1 composite[2]/2 weeks ■ Organics: 1 composite/2 months ■ Dioxins: 1 composite/year[4]·

[1] *In the case of biosolids manufacturing facilities in existence prior to the publication of these* Guidelines *it is acceptable to use data up to 12 months old for the purposes of* product verification*. For the purposes of determining compliance at the 95 percentile for* routine sampling*, the age of the data set shall be no more than 2 years for metals and organochlorines and no more than 5 years for dioxins. This avoids the scenario of old data masking upward trends in contaminant concentrations.*

[2] *Samples tested should be made up from daily composites. For organochlorines there shall be no exceedence over 3 consecutive samples.*

[3] *The dioxin verification composite should be made up of 1 sample taken per day during the verification period.*

[4] *Composite to be made up of 1 sample per week over a year-long period.*

8.3.1 Stabilisation grade sampling

When monitoring for pathogens it is important that the samples taken are grab samples (see Section 9.2.1). Composite samples are not used because the risk of exposure to pathogens is not cumulative. For the same reason, the actual values from each grab sample need to be reported – not the average. This is in alignment with the USEPA recommendations for pathogen monitoring of a Class A product.

During the process verification period a total of three failures is allowed. If this number is exceeded, then the next 15 grab samples must comply. If any of the failures occur during the last month of verification sampling (i.e., there are fewer than 15 samples left to take), samples must continue to be taken at the verification frequency until 15 consecutive compliant samples have been obtained. It is therefore possible that the verification period may involve more than 45 samples.

Once the verification monitoring has been completed, the sampling regime can change to the one specified under the routine monitoring regime (i.e., at least one grab sample per week). If any of these samples fail, then, for stabilisation Grade A, a return to the verification monitoring regime for all pathogens is required to ensure product quality.

The samples taken during the verification period must be analysed for *E.* coli, salmonella, campylobacter, enteric viruses, helminth ova and vector attraction reduction. Once it has been confirmed that the product is of a consistent quality, then, for Grade A, routine samples only need be analysed for *E. coli* and vector attraction reduction. For Grade B, there is only a requirement for VAR monitoring during routine biosolids production, unless a barrier is to be used for VAR in which case no monitoring is required.

8.3.2 Contaminant grade sampling

A review of existing biosolids literature indicates that there are three approaches taken to contaminant monitoring.

- *EU* – after initial sampling to determine a baseline (no numbers given), one sample every six months for metals listed in the directive. No organics monitoring is required. (CEC, 1986).

- *USEPA* – the number of samples taken for contaminant monitoring is based on the amount of biosolids produced. Sampling rates vary from once to 12 times per year (USEPA, 1993).
- *NSW EPA* – the number of samples is based on a combination of production process (batch or continuous) and previous results obtained (NSW EPA, 1997).

To determine the contaminant concentrations, the sampling regime detailed in Table 8.2 should be followed. This has been loosely based on the requirements of the NSW EPA. The approach taken in the NSW guidelines is very well documented and has been devised to ensure that biosolids products are graded and classified with an acceptable degree of accuracy. However, the actual methodology is very detailed and the number of samples required was considered excessive in the New Zealand context, particularly since the majority of New Zealand wastewater treatment plants are small (< 10 tonnes ds/day). The aim in these *Guidelines* is not to have sampling that is statistically representative on its own, but to use sampling as a means to show that process controls are working and that the *average* levels of contaminants in the biosolids are below the limits specified.

The USEPA approach was not thought appropriate for these *Guidelines*, as the bands used to classify biosolid production are very broad and not applicable to the scale of production in New Zealand. The bands only relate to the number of monitoring periods, not the number of samples. The only advice given on the number of samples to be taken is that they should be representative. The philosophy behind this approach is to ensure that monitoring requirements for small producers are not overly burdensome. No explanation could be found in the literature reviewed to indicate that there was any statistical reasoning behind the monitoring regime suggested. Similarly, no documentation was found that discussed the reasoning behind the EU approach.

Unlike the samples taken for stabilisation grade monitoring, samples taken for contaminant grade monitoring should be composite samples (see section 9.2.2). Each composite sample should be made up of a number of grab samples taken from different locations and/or at different times. Composite samples are used because the chemical contaminants do not degrade and (compared with grab samples) this type of sample generally provides results that are more representative of the overall quality of the final biosolids product.

There is a less stringent requirement for dioxin sampling than there is for metals and the other organic contaminants, mainly because of the expense of this analysis. During the verification period one dioxin sample should be prepared which is made up of one sample per day taken over the three-month period. If this sample is compliant with the limits given in Table 7.2, then only one dioxin sample needs to be analysed annually. The sample taken under the routine monitoring regime should be a composite of one sample taken weekly over a year-long period. If the sample is not compliant, then full verification must be undertaken.

Irrespective of whether the samples are taken as part of verification monitoring or of routine monitoring, they should all be analysed for the contaminants given in Table 7.1 (metals) and Table 7.2 (organic contaminants).

If any contaminant fails (i.e. its concentration exceeds those specified in Tables 7.1 or 7.2), then a return to verification monitoring is required only for that contaminant. All other contaminants should continue to be monitored at the routine sampling frequency. The failed contaminant can only return to the routine sampling frequency once it is in compliance with the biosolids limits (Tables 7.1 and 7.2) from this additional verification sampling.

8.3.3 Soil sampling

Soil should be sampled *before* the application of biosolids to determine the existing contaminant concentration, and for bulk users of biosolids, every five years thereafter (see Volume 1, section 7.5.6). In this way the accumulation of contaminants in the soil can be monitored.

Regular monitoring of soil is only recommended for the application of restricted use biosolids, as contaminant limits in unrestricted use biosolids are low enough to prevent the rapid accumulation of contaminants. However, periodic monitoring of soil that has had Aa grade biosolids applied to it would be useful and good management practice. These data should be collected centrally and held as a public record.

These *Guidelines* recommend a minimum of 10 soil samples per hectare be taken as part of any monitoring programme (see Volume 1, section 7.6.2). Soil cores should be taken to a depth to which the biosolids were incorporated, up to a maximum of 200 mm. If there is no soil incorporation, the sampling depth should be 200 mm.

It is not necessary to monitor pathogen concentrations in the soil, as biosolid application will not cause a cumulative increase in pathogens.

8.3.4 Test Methods

There are a number of analytical methods for contaminants and pathogens. The following methods (or updates where available) are recommended:

Campylobacter: specific test method (see Appendix I).

Enteric viruses: specific test method, (see Appendix II).

E. coliform: Part 9221 F or Part 9223 B, *Standard Methods for the Examination of Water and Wastewater* (APHA, 1998).

Helminth ova: W.A. Yanko, *Occurrence of Pathogens in Distribution and Marketing Municipal Sludges*, EPA/600/1-87/014, 1987. PB 88-154273/AS, National Technical Information Service, Springfield, VA.

Percent volatile solids reduction calculation: *Environmental Regulations and Technology: Control of Pathogens and Vectors in Sewage Sludge* (USEPA, 1992).

Salmonella sp bacteria: Part 9260 D, *Standard Methods for Examination of Water and Wastewater*, (APHA, 1988), or Detection and enumeration of salmonella and *Pseudomonas aeruginosa* (Kenner and Clark, 1974).

Specific oxygen uptake rate: Part 2710 B, *Standard Methods for the Examination of Water and Wastewater* (APHA, 1998).

Total, fixed, and volatile solids: Part 2540 G, *Standard Methods for the Examination of Water and Wastewater* (APHA, 1998).

No test methods are specified for metals, persistent organochlorine pesticides, PCBs or dioxins, since there are a number of methods that can be used for measuring these contaminants in biosolids. However, New Zealand laboratories generally adopt USEPA methods, especially for organic contaminants (see section 9.9 for the USEPA web site where their analytical methods can be accessed).

8.4 Factors affecting biosolids application

As a general rule it is recommended that biosolids (regardless of whether they are Aa, Ab, Ba or Bb grade) should not be applied to land that is:

- frozen solid
- waterlogged
- under snow
- sloping steeply (e.g., >15%)

- in close proximity (say 20 m) to any watercourse, including a:
  - river
  - estuary
  - ocean
  - lake
  - reservoir
- measured as having a pH < 5.5.

Refer to Volume 1, section 6.2 for further discussion on the sensitivity of the receiving environment.

If users wish to apply restricted use biosolids to sites that fall into one or more of the above categories, then the consenting authority should consider each resource consent application on its merits. There is no control over the use of unrestricted use biosolids. However, any label/information sheet that accompanies unrestricted use biosolids should included information relating to land types that are not suitable for application (see Volume 1, sections 5 and 7).

9 SAMPLING PROCEDURES

9.1 Introduction

Obtaining representative samples and maintaining their integrity are critical parts of any monitoring programme. Analytical methods have been standardised, but the results are only as good as the sample collection and preservation methods.

The USEPA has estimated that 95% of the total error in environmental measurements is due to sample collection and handling, and that only 5% is due to mistakes during laboratory analysis. The 95% can be further broken down into 85% of error from sample collection and 10% from sub-sampling in the laboratory (Rosecrance and Adolfo, 1996). This gives some indication of the importance of using correct sampling procedures at all times.

In sampling, the objective is to collect a small portion of an environment that is representative of the whole body. Once the sample is taken, the constituents of the sample must stay in the same condition as when they were collected.

9.2 Sample type

9.2.1 Grab samples

A grab sample is one where the whole sample volume is collected at a particular time and place and represents the composition of the source at that time and place. Results from grab samples (also called ‘spot’ or ‘catch’ samples) can be said to represent the composition of a source product over a greater period of time *only* if the composition of the source is known to be relatively constant in space and time. Biosolids may not fall into this category, and so results from individual grab samples cannot be assumed to be representative of the sample source over time.

This is not to say that grab samples do not have their place in biosolids monitoring. Grab samples are essential if the aim of the sampling programme is to prove compliance with standards that are not related to average quality.

Grab samples should be used for determinands that deteriorate or change quickly after sampling, such as pathogens (see section 8.3.1).

9.2.2 Composite samples

Composite samples are prepared by mixing a number of grab samples. They are very useful if there is thought to be much variability in the characteristics of the source being investigated. For this reason they are particularly useful for investigating biosolids. When the composite sample is analysed, the results give the average concentration for the parameter in question over the period of time the sample was collected.

Composite samples have an advantage over grab samples in that combining the individual grab samples for analysis means the laboratory costs are much lower. However, composite samples are not suitable for parameters that degrade/alter as a result of storage. In other words, composite sampling should only be used for components that can be shown to remain unchanged under the conditions of sample collection, preservation and storage. Consequently, composite samples should only be used for the chemical contaminants (i.e., metals, organics and dioxins) covered by these *Guidelines*. Composite sampling should *not* be used for monitoring pathogens.

9.2.3 Microbiological samples

Special attention must be given to microbiological samples, because they are very susceptible to being contaminated by poor sampling technique. When taking a microbiological sample, the following must be taken into account:

- Containers and tools should be sterilised.

- The lids of sterile sampling containers should have a seal over them, which has to be broken before you take the sample. If this seal is damaged in any way, do not use the container as it may no longer be sterile.
- The container may have a use-by date on it. If it has and the date has passed, do not use the container as it may no longer be sterile.
- When taking the sample do not touch the neck of the container, or the inside of the lid. The lid must not be put down on any surfaces as this can contaminate the sample.
- When transporting microbiological samples, keep them separate from other non-sterile samples and cool with ice. Take care not to let melted ice come into contact with the container tops (this is best achieved by keeping the ice inside a plastic bag, separate from the sample container). Remember that even if the outside of the container is dirty, while it may not directly affect your sample it could contaminate the laboratory. Samples must not be exposed to direct sunlight and must reach the laboratory within the specified time limit.

If you have to take samples for microbiological analysis and you are unfamiliar with the aseptic technique, you must contact the laboratory for advice before collecting any samples. If you receive unexpected results from microbiological samples, remember how easy it is for poor sampling technique to affect the results before drawing any conclusions.

9.3 Where to sample

In general, more representative sampling occurs when the biosolids being sampled are moving rather than stationary. However, this is not always possible. The USEPA have provided advice on the best locations for taking samples related to the type of biosolids involved. This is reproduced (slightly modified) in Table 9.1.

9.4 Sample equipment

The type of sample equipment chosen is usually dependent on the type of biosolids being sampled. However the following rules apply to all situations.

- Automatic sampling equipment (such as that used at wastewater treatment plants) is not suitable for sampling biosolids. All samples need to be taken manually.
- Equipment should be easy to clean and constructed of non-corrosive materials, such as Teflon, glass or stainless steel.
- Equipment used for biosolids sampling should not be used for any other purpose.
- Equipment should be well cleaned after use and stored in a clean location. It may be advisable to wrap any equipment between uses to ensure it stays clean.

For special requirements relating to the collection of microbiological samples, refer to section 9.2.3.

9.5 Sample containers

The following factors must be considered when choosing a sample container:

- high resistance to breakage
- good sealing efficiency
- ease of reopening
- good resistance to temperature extremes
- practicable size, shape and mass

Table 9.1: Sampling points for biosolids

Biosolids type	Sampling point
Anaerobically digested	Collect sample from taps on the discharge side of positive displacement pumps.
Aerobically digested	Collect sample from taps on discharge lines from pumps. If batch digestion is used, collect sample directly from the digester. Note that when aeration is shut off, solids may settle rapidly.
Thickened	Collect sample from the taps on the discharge side of positive displacement pumps.
Heat treated	Collect sample from the taps on the side of positive displacement pumps after decanting. Be careful when sampling heat-treated biosolids because of: ■ a high tendency for solids separation ■ the high temperature of the sample can cause problems with sample containers due to cooling and subsequent contraction of entrained gases.
Dewatered, dried, composted or thermally reduced	Collect sample from material collection conveyors and bulk containers. Collect sample from many locations within the biosolids mass, and at various depths.
Dewatered by belt filter press, centrifuge, vacuum filter press	Collect sample from biosolids discharge chute or storage bin (see below).
Dewatered by biosolids press	Collect sample from the storage bin; select 4 points from within the bin, collect an equal amount from each point and combine to form one sample.
Dewatered by drying beds	Divide bed into quarters, grab equal amounts of sample from the centre of each quarter and combine to form a composite sample of the total bed. Each composite sample should include the entire depth of the biosolids material (down to the sand/drainage layer).
Compost piles	Collect sample directly from the front-end loader while biosolids are being transported or stockpiled within a few days of use.

- good potential for cleaning – this is especially important for containers used to collect samples for microbiological analysis
- availability and cost.

Sample containers are generally made out of glass or plastic. The type of determinand the sample is to be analysed for often controls the type of material the container is made from, as some containers will react with the determinands and give false results when the sample is analysed. If you require a sample to be analysed for more than one determinand, you may have to use more than one type of sample container.

If you are unsure which type of container is required, the laboratory carrying out the analysis will be able to advise you.

9.6 Sample preservation

If a sample is to remain representative of the material from which it was taken, it is usually necessary to preserve the sample to prevent changes taking place during the period prior to analysis. This is particularly important for composite samples, which are going to be collected over a period of weeks and months.

The most common way of preserving samples is to cool the sample to between 0°C and 4°C. Once collected, samples should be stored at this temperature until analysis.

Biosolid samples can take a long time to cool down, and, as general rule, the thicker they are the longer they take to cool. Similarly, the larger the sample the longer it takes to cool. It is therefore practical to minimise the sample size to ensure rapid and effective cooling. Samples less than 4 litres in size should ensure this. Note that while freezing can be used to preserve some samples, this should not be done if the sample is to be analysed for bacteria. Freezing is not normally a standard preservation technique for samples to be analysed for metals or organic contaminants. If samples are to be frozen, they should not be collected in borosilicate glass containers, which are liable to fracture.

If the sample is being collected over an extended period of time the preservation of the sample should form an integral part of the collection procedure. Keeping the samples in the dark can enhance preservation further.

9.7 Sample transportation

Samples to be analysed for microbiological parameters should be transported to the laboratory within six hours of collection. All other samples should be transported within 24 hours, unless suitably preserved. Sample transportation should be undertaken in accordance with standard quality assurance procedures, including the use of chain of custody forms.

9.8 Laboratory selection

A very important – but often overlooked – aspect of any monitoring programme is the selection of an appropriate analytical laboratory. Analysis of biosolids is a complex process because of the heterogeneous nature of the product. Therefore it is important to select a laboratory with experience in analysing this type of material.

In New Zealand the primary accreditation agency for analytical laboratories is International Accreditation New Zealand (IANZ). This agency, formerly known as TELARC, is governed by an act of Parliament. Generally laboratories with IANZ accreditation should be selected for analysis of samples because these laboratories will have quality assurance programmes in place to maintain analytical performance. Biosolids resource consents may specify that analytical measurements are to be carried out by an IANZ accredited laboratory. However, note that IANZ accreditation is test-specific and therefore not all IANZ laboratories may be accredited for the particular test you wish to use. Confirm the status of the test (i.e. analyte *and* method) accreditation with the laboratory *before* sending samples for analysis.

There are a number of New Zealand laboratories that have accreditation for the analysis of metals and organic contaminants governed by these *Guidelines*. There may, however, be microbiological tests for which no laboratory has the specific accreditation. In this case you should choose a competent laboratory and discuss the selection of an appropriate standard test method (see section 8.3.4 for recommended test methods).

9.9 Standard sampling texts

The most widely utilised text of standardised analytical procedures for wastewater and aqueous environmental samples is *Standard Methods for the Examination of Water and Wastewater* (APHA, 1998). This covers a wide range of parameters applicable to the majority of wastewater-related monitoring programmes.

The USEPA has developed a large number of standard analytical procedures, many of which parallel the APHA standard methods. The list of standard USEPA methods can be found on the website: http://www.epa.gov/epahome/index/. The USEPA also have a large number of standard methods for chemical monitoring, including the monitoring of solids and sludges such as biosolids. These are primarily found in *SW-846 Test Methods for Evaluating Solid Waste*, which is available on-line at http://www.epa.gov/epaoswer/hazwaste/test/main.htm.

International and country-specific standards (including ISO and Standards New Zealand) are also available, which cover procedures and methods for sample collection and analysis for many of the parameters covered in these *Guidelines*.

A list of recommended test methods is given in section 8.3.4.

9.10 Auditing

Any sample programme should be audited, no matter how well prepared, in order to ensure that samples are being collected, transported and analysed correctly. If any problems are identified they should be resolved immediately to prevent the reporting of erroneous results.

To audit *sampling procedures*, a chain of custody form should be used throughout the sampling process. This should record the following information:

- name and signature of person collecting sample
- date and time sample collected
- purpose of the sample
- analysis required
- location of sample point and unique reference number (if one has been assigned)
- sampling method (i.e. grab or composite)
- preservation method
- name of person receiving the sample for analysis
- date analysed
- results, and whether these have been confirmed.

The general rule is to collect as much information as possible. This includes recording times, dates and results from any sample blanks taken. On-site sampling operation audits should also be conducted to ensure that samples are being taken correctly.

Analytical procedures will be assessed as part of a laboratory's quality assurance programme for obtaining and maintaining their laboratory accreditation.

9.11 Data reporting

Analytical laboratories should provide reports that are complete, accurate and unambiguous so that clear conclusions can be drawn from the data without the need to make 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.

As a minimum, analytical reports should contain the following:

- sample identification and description
- date of receipt of the sample and conditions of storage
- date extraction of the sample commenced
- details of the sample preparation and fraction of sample analysis
- citation and summary of analytical procedure – it 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.

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, which leaves any censoring 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.
- Concentrations of analytes in biosolids or soils should be presented on an oven dry (105°C) basis, with the moisture contents of the samples presented separately if requested.
- Analyte concentrations should be corrected for the blank and for recovery.
- Use SI units e.g. mg/kg, mg/kg rather than parts-per-million (ppm) or parts-per-billion (ppb).
- If there is no observed signal for the analyte, report as ND (not detected) at the quoted Method Detection Limit (MDL).
- If the analyte signal is detectable but the concentration is less than the MDL, report the concentration but flag as < MDL and in a region of uncertainty. Terms such as “trace” should be avoided.
- If the analyte concentration is 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 quality control (QC) samples run with the field 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, 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 averages and marked in the report with the number of measurements; e.g. 0.31 mg/kg (3). Sets of laboratory replicate data should be summarised in the form of coincidence 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 (including laboratory blanks, field control samples, fortified laboratory matrix samples) run with client samples should be reported with ranges, means and confidence intervals where appropriate.

9.12 Assessment of sample results

Results from the analysis of biosolids for chemical contaminants, reported on a dry weight basis, must meet the criteria given in Tables 7.1 (metals) and 7.2 (organic chemicals) to ensure compliance with the requirements of these *Guidelines*, and for the biosolids to be beneficially used. In assessing compliance, the concentration measured for a contaminant, may exceed the limit given in the tables, providing that:

- the 95th percentile of the previous monitoring results (up to 24 months) for that contaminant are below the criteria
- the concentration does not exceed 20% of the limit value for that contaminant.

REFERENCES

ADEME (1995). Agences De L’eau; Les Différents Procédés De Stockage Des Boues D’épuration Avant Valorisation En Agriculture.

ADEME (1995a). INRA Bordeaux, FNDAE; 1995; Les Micropolluants Métalliques Dans Les Boues Ésiduaires Des Stations D’épuration Urbaines. Collection “Valorisation Agricole Des Boues D’épuration.

ANFA (1993). *The 1992 Australian Market Survey: A Total Diet Survey of Pesticides and Contaminants*. Australian National Food Authority, Canberra.

ANFA (1996). *Australian Market Basket Survey*. Australian National Food Authority, Canberra.

ANZECC (1992). *Australian and New Zealand Guidelines for the Assessment and Management of Contaminated Sites.* Australia and New Zealand Environment and Conservation Council, National Health and Medical Research Council.

APHA (1998). *Standard Methods for the Examination of Water and Wastewater,* 20th edition. American Public Health Association, Washington, DC.

Auckland Regional Council (1999). *Trace Element Concentrations in Soils and Soil Amendments from the Auckland Region.* Working Report No. 76. Auckland Regional Council, Auckland.

Backes, C.A., McLaren, R.G., Rate, A.W. and Swift, R.S. (1995). Kinetics of cadmium and cobalt desorption from iron and manganese oxides. *Soil Science Society of America Journal* 59, 778–785.

Barrow, N.J. (1986). Testing a mechanistic model. II: The effects of time and temperature on the reaction of zinc with a soil. *Journal of Soil Science* 37, 277–286.

Barrow, N.J. (1999). The four laws of chemistry: The Leeper lecture 1998. *Australian Journal of Soil Research* 37, 787–829.

Bartlett, R.J. (1997). Chromium redox mechanisms in soils: Should we worry about Cr (VI)? In: S. Canali, F. Tittarelli and P. Sequi (eds). *Chromium Environmental Issues* FrancoAngeli s.r.l., Milan.

Beckett, P.H.T. and Davis, R. D. (1982). Heavy metals in sludge: Are their toxic effects additive? *Water Pollution Control* 81, 112–119.

Benjamin, M.M. and Leckie, J.O. (1981). Multiple-site adsorption of Cd, Cu, Zn and Pb on amorphous iron oxyhydroxide. *Journal of Colloid and Interface Science* 79, 209–221.

Berrow, M.L. and Burridge, J.C. (1984). Persistence of metals in sewage sludge treated soils. In: P. L’Hermite and H. Ott (eds). *Processing and Use of Sewage Sludge.* D. Reidel Publishing Company, Dordrecht.

Blake, W. (1979). Determination of acceptable levels of heavy metals in effluents discharged do a foul water sewer. In Heavy Metals In The Environment. CEP Consultants, Edinburgh.

Bolan, N.S., Duraisamy, P., Mani, A.K. and Arulmozhiselvan, K. (2001). Biosolid compost: is it a source or sink for heavy metals in soils? *New Zealand Soil News* 49, 33–34.

Bowen, H.J.M. (1985). The natural environment and the biogeochemical cycles. In: D. Hutzinger (ed.). *Handbook of Environmental Chemistry*. Springer-Verlag, New York and Basel.

Brennan, R.F. (1990). Reaction of zinc with soil affecting its availability to subterranean clover. II: Effect of soil properties on the relative effectiveness of applied zinc. *Australian Journal of Soil Research* 28, 303–310.

Brennan, R.F., Gartrell, J.W. and Robson, A.D. (1980). Reactions of copper with soil affecting its availability to plants. I: Effect of soil type and time. *Australian Journal of Soil Research* 18, 447–459.

Bruemmer, G.W., Gerth, J. and Herms, U. (1986). Heavy metal species, mobility and availability in soils. *Zeitschrift fur Pflanzenernahrung Bodenkunde* 149, 382–398.

Bruemmer, G.W., Gerth, J. and Tiller, K.G. (1988). Reaction kinetics of the adsorption and desorption of nickel, zinc and cadmium by goethite. I: Adsorption and diffusion of metals. *Journal of Soil Science* 39, 37–52.

Buchan, M.A. (2001) Bio-availability of cadmium in soils amended with sewage sludge and composted biosolids. BSc (Hons) thesis, Lincoln University, Lincoln, NZ.

Buckland, S.J. (1999). Biosolids – the hidden issues. *Proceedings of the WasteMINZ/NZWWA workshop: Organic Waste management and Minimisation*, Christchurch.

Buckland, S.J., Ellis, H.K. and Salter, R.T. 1998a. *Organochlorines in New Zealand: Ambient Concentrations of Selected Organochlorines in Soil.* Ministry for the Environment, Wellington, New Zealand.

Buckland, S.J., Jones, P.D., Ellis, H.K. and Salter, R.T. 1998b. *Organochlorines in New Zealand: Ambient Concentrations of Selected Organochlorines in Rivers.* Ministry for the Environment, Wellington, New Zealand.

Buckland, S.J., Scobie, S. and Heslop, V. 1998c. *Concentrations of PCDDs, PCDFs and PCBs in Retail Foods and an Assessment of Dietary Intake for New Zealanders*. Ministry for the Environment, Wellington, New Zealand.

Buckland, S.J., Ellis, H.K. and Salter, RT. 1999. *Organochlorines in New Zealand: Ambient Concentrations of Selected Organochlorines in Air.* Ministry for the Environment, Wellington, New Zealand.

Buckland, S.J., Ellis, H.K. and Dyke, P. 2000. *New Zealand Inventory of Dioxin Emissions to Air, Land and Water, and Reservoir Sources.* Ministry for the Environment, Wellington, New Zealand.

Buckland, S.J., Bates, M.N., Garrett, N., Ellis, H.K. and van Maanen, T. 2001. *Concentrations of Selected Organochlorines in the Serum of the Non-occupationally Exposed New Zealand Population*. Ministry for the Environment, Wellington, New Zealand.

Cabrera, D., Young, S.D. and Rowell, D.L. (1988). The toxicity of cadmium to barley plants as affected by complex formation with humic acid. *Plant and Soil* 105, 195–204.

Cameron, K.C., McLaren, R.G. and Adams, J.A. (1994). Application of municipal sewage sludge to low fertility forest soils: The fate of nitrogen and heavy metals. *Transactions of the 15th World Congress of Soil Science*, Vol. 3a: 467–482.

Camobreco, V.J., Richards, B.K., Steenhuis, T.S., Peverly, J.H. and McBride, M.B. (1996). Movement of heavy metals through undisturbed and homogenized soil columns. *Soil Science* 161, 740–750.

Carbonell-Barrachina, A.A., Burló-Carbonell, F. and Mataix-Beneyto, J. (1997). Arsenic uptake, distribution, and accumulation in bean plants: Effect of arsenite and salinity on plant growth and yield. *Journal of Plant Nutrition* 20, 1419–1430.

Carbonell-Barrachina, A.A., Jugsujinda, A., Burlo, F., Delaune, R.D. and Patrick, W.H. Jr (2000). Arsenic chemistry in municipal sewage sludge as affected by redox potential and pH. *Water Res*. 34, 1, 216–224.

Carpi, A. and Lindberg, S.E. (1997). Sunlight-mediated emission of elemental mercury from soil amended with municipal sewage sludge. *Environmental Science and Technology* 31, 208–209.

CCME, 1999. *Canadian Environmental Quality Guidelines*. Canadian Council of Ministers of the Environment, Winnipeg.

CEC (Commission of the European Communities) (1986). Council directive (86/278/EEC) on the protection of the environment, and in particular of the soil, when sewage sludge is used in agriculture. *Official Journal of the European Communities* 181, 6–12.

Chander, K., Brookes, P.C. and Harding, S.A. (1995). Microbial biomass dynamics following addition of metal-enriched sewage sludges to a sandy loam. *Soil Biology and Biochemistry* 27, 1409–1421.

Chaney, R.L. (1980). Health risks associated with toxic metals in municipal sludge. In: G. Bitton, B.L. Damron, G.T. Edds and J.M. Davidson (eds). *Sludge: Health Risks of Land Application*. Ann Arbor Science Publications, Ann Arbor, MI.

Chaney, R.L. and Oliver, D.P. (1996). Sources, potential adverse effects and remediation of agricultural soil contaminants. In: R. Naidu, R.S. Kookana, D.P. Oliver, S. Rogers and M.J. McLaughlin (eds). *Contaminants and the Soil Environment in the Australasia-Pacific Region*. Kluwer Academic Publishers, Dordrecht.

Chaney, R.L., Ryan, J.A. and Brown, S.L. (1997). Development of the USEPA limits for chromium in land-applied biosolids and applicability of these limits to tannery by-product derived fertilizers and other Cr-rich soil amendments. In: S. Canali, F. Tittarelli and P. Sequi (eds). *Chromium Environmental Issues*. FrancoAngeli s.r.l., Milan.

Chang, A.C., Warneke, J.E., Page, A.L. and Lund, L.J. (1984). Accumulation of heavy metals in sewage sludge-treated soils. *Journal of Environmental Quality* 13, 87–91.

Chumbley, C.G. (1971). *Permissible Levels of Toxic Metals in Sewage Used on Agricultural Land.* Agricultural and Development Advisory Paper No. 10. Ministry of Agriculture, Fisheries and Food, London.

Comber, S.D.W. and Gunn, A.M. (1996). Heavy metals entering sewage treatment works from domestic sources *J. CIWEM* 10, 137–142.

Corey, R.B., King, L.D., Lue-Hing, C., Fanning, D.S., Street, J. and Walker, J.M. (1987). Effects of sludge properties on accumulation of trace elements by crops. In: A.L. Page, T.J. Logan and J.A. Ryan (eds). *Land Application of Sludge: Food Chain Implications*. Lewis Publishers Inc., Chelsea, MI.

Cornell, R.M. and Schwertmann, U. (1996). *The Iron Oxides*. VCH Publishers, Weinheim.

DeKock, P.C. and Mitchell, R.L. (1957). Uptake of chelated metals by plants. *Plant and Soil* 84, 55–62.

DEFRA (Department for Environment, Food and Rural Affairs) (1998 revision). *Code of Good Agricultural Practice: The Soil Code*. Welsh Office, Agriculture Department, UK.

DEFRA (2002a). *Soil Guideline Values for Arsenic Contamination*. Department of Environment, Food and Rural Affairs, The Environment Agency, Bristol, England.

DEFRA (2002b). *Soil Guideline Values for Cadmium Contamination*. Department of Environment, Food and Rural Affairs, The Environment Agency, Bristol, England.

DEFRA (2002c). *Soil Guideline Values for Chromium Contamination*. Department of Environment, Food and Rural Affairs, The Environment Agency, Bristol, England.

DEFRA (2002d). *Soil Guideline Values for Lead Contamination*. Department of Environment, Food and Rural Affairs, The Environment Agency, Bristol, England.

DEFRA (2002e). *Soil Guideline Values for Inorganic Mercury Contamination*. Department of Environment, Food and Rural Affairs, The Environment Agency, Bristol, England.

DEFRA (2002f). *Soil Guideline Values for Nickel Contamination*. Department of Environment, Food and Rural Affairs, The Environment Agency, Bristol, England.

Department of Health (1992). *Public Health Guidelines for the Safe Use of Sewage Effluent and Sewage Sludge on Land*. Department of Health, Wellington.

Dowdy, R.H., Latterell, J.J., Hinesly, T.D., Grossman, R.B. and Sullivan, D.L. (1991). Trace metal movement in an Aeric Ochraqualf following 14 years of annual sludge applications. *Journal of Environmental Quality* 20, 119–123.

Dowdy, R.H. and Volk, V.V. (1983). Movement of heavy metals in soils. In: D.W. Nelson, D.E. Elrick and K.K. Tanji (eds). *Chemical Mobility and Reactivity in Soil Systems*. Soil Science Society of America Inc., Madison, WI.

EC (European Commission) (1999). Revision of EC sludge directive challenges land spreading. *ENDS Report* 299, 45–46.

EC (European Commission) (2000). *Working Document on Sludge, 3rd draft*. ENV.E.3/LM, Brussels, April 2000.

EC (European Commission) (2001a). *Disposal and Recycling Routes for Sewage Sludge. Part 2: Regulatory Report, DG Environment*. Arthur Anderson, Sede.

EC (European Commission) (2001b). *Disposal and Recycling Routes for Sewage Sludge. Part 3: Scientific and Technical Sub-Component Report*. DG Environment, Arthur Anderson, Sede.

Ellis, B.G., Knezek, B.D. and Jacobs, L.W. (1983). The movement of micronutrients in soils. In: D.W. Nelson, D.E. Elrick and K.K. Tanji (eds). *Chemical Mobility and Reactivity in Soil Systems*. Soil Science Society of America Inc., Madison, WI.

Estes, G.O., Knoop, W.E. and Houghton, F.D. (1973). Soil–plant response to surface-applied mercury. *Journal of Environmental Quality* 2, 451–452.

Frankenberger, W.T. Jr and Losi, M.E. (1995). Applications of bioremediation in the cleanup of heavy metals and metalloids. In: H.D. Skipper and R.F. Turco (eds). *Bioremediation: Science and Applications*. Special Publication 43, Soil Science Society of America Inc., Madison, WI.

Fries, G.F. (1982). Potential polychlorinated biphenyl residues in animal products from application of contaminated sewage sludge to land. *Journal of Environmental Quality* 11, 14–20.

German Federal Law Gazette (1992). Part I: *Sewage Sludge Ordinance (AbfKlärV). Klärschlammverordnung (AbfKlärV).* Bundesgesetzblatt, Jahrgang 1992, Teil I, 912-934. Bonn.

Gerritse, R.G., Vriesema, J.W., Dalenberg, J.W. and De Roos, H.P. (1982). Effect of sewage sludge on trace element mobility in soils. *Journal of Environmental Quality* 11, 359–364.

Gilkes, R.J. and McKenzie, R.M. (1988). Geochemistry of manganese in soil. In: R.D. Graham, R. J. Hannam and N. C. Uren (eds). *Manganese in Soils and Plants*. Kluwer Academic Publishers, Dordrecht.

Giordano, P.M. and Mortvedt, J.J. (1976). Nitrogen effects on mobility and plant uptake of heavy metals in sewage sludge applied to soil columns. *Journal of Environmental Quality* 5, 165–168.

Graham, R.D. (1981). Absorption of copper by plants. In: J.F. Loneragan, A.D. Robson and R.D. Graham (eds). *Copper in Soils and Plants*. Academic Press, Sydney.

Gray, C.W., McLaren, R.G. and Roberts, A.H.C. (2001). Cadmium concentrations in some New Zealand wheat grain. *New Zealand Journal of Crop and Horticultural Science* 29, 125–136.

Gray, C.W., McLaren, R.G., Roberts, A.H.C. and Condron, L.M. (1998). Sorption and desorption of cadmium from some New Zealand soils: Effect of pH and contact time. *Australian Journal of Soil Research* 36, 199–216.

Gray, C.W., McLaren, R.G., Roberts, A.H.C. and Condron, L.M. (1999). The effect of long-term phosphate fertiliser applications on the amounts and forms of cadmium in soils under pastures in New Zealand. *Nutrient Cycling in Agroecosystems* 54, 267–277.

Grove, J.H. and Ellis, B.G. (1980). Extractable chromium as related to soil pH and applied chromium. *Soil Science Society of America Journal* 44, 238–242.

Hamon, R.E., McLaughlin, M.J., Naidu, R. and Correll, R. (1998). Long-term changes in cadmium bioavailability in soil. *Environmental Science and Technology* 32, 3699–3703.

Haygarth, P.M. (1994). Global importance and global cycling of selenium. In: W.T. Frankenberger, Jr and S. Benson (eds). *Selenium in the Environment*. Marcel Dekker Inc., New York.

Haygarth, P.M. and Jarvis, S.C. (2000). Transfer of phosphorus from agricultural soils. *Advances in Agronomy* 66, 196–249.

Healy, W.B. (1973). Nutritional aspects of soil ingestion by grazing animals. In: G.W. Butler and R.W. Bailey (eds). *Chemistry and Biochemistry of Herbage*, *Volume 1*. Academic Press, London.

Hill, J., Stark, B.A., Wilkinson, J.M., Curran, M.K., Lean, I.J., Hall, J.E. and Livesey, C.T. (1998a). Accumulation of metals by sheep given diets containing soil and sewage sludge. 1: Effect of ingestion of soils treated historically with sewage sludge. *Animal Science* 67, 87–96.

Hill, J., Stark, B.A., Wilkinson, J.M, Curren, M.K., Lean, I.J., Hall, J.E. and Livesey, C.T. (1998b). Accumulation of potentially toxic elements by sheep given diets containing soil and sewage sludge. 1: Effect of type of soil and level of sewage sludge in the diet. *Animal Science* 67, 73–86.

Hingston, F.J., Posner, A.M. and Quirk, J.P. (1972). Anion adsorption by goethite and gibbsite. I: The role of the proton in determining adsorption envelopes. *Journal of Soil Science* 23, 177–192.

Hogg, D.S., McLaren, R.G. and Swift, R.S. (1993). Desorption of copper from some New Zealand soils. *Soil Science Society of America Journal* 57, 361–366.

Horstmann, M. and McLachlan, M.S. (1995). Concentrations of polychlorinated dibenzo-p-dioxins (PCDD) and dibenzofurans (PCDF) in urban runoff and household wastewaters. *Chemosphere* 31, 3, 2887–2896.

Horstmann, M., McLachlan, M.S. and Reissinger, M. (1993). Investigations of the origin of PCDD/F in municipal sewage sludge. *Chemosphere* 27, 1–3.

IC Consultants (2001). *Pollutants in Urban Wastewater and Sewage Sludge*. Report for the European Commission.

ICRP (International Commission on Radiological Protection) (1974). *Report of the Task Group on Reference Man.* Pergamon Press, New York, NY.

Isaac, R.A., Gil, L., Cooperman, A.N., Hulme, K., Eddy, B., Ruiz, M., Jacobson, K, Larson, C, Pancorbo, OC. (1997). Corrosion in drinking water distribution systems: A major contributor of copper and lead to wastewaters and effluents. *Environ. Sci. Technol*. 31, 3198–3203.

Jarvis, S.C. (1981). Copper concentrations in plants and their relationship to soil properties. In: J.F. Loneragan, A.D. Robson and R.D. Graham (eds). *Copper in Soils and Plants*. Academic Press, Sydney.

Jing, J. and Logan, T.J. (1992). Effect of sewage sludge cadmium concentration on chemical extractability and plant uptake. *Journal of Environmental Quality* 21, 73–81.

John, M.K. and Laerhoven, C.J. (1976). Effects of sewage sludge composition, application rate and lime regime on plant availability of heavy metals. *Journal of Environmental Quality* 5, 246–251.

Johnson, L.R. and Hiltbold, A.E. (1969). Arsenic content of soil and crops following use of methane-arsonate herbicides. *Soil Science Society of America Proceedings* 33, 279–282.

Kabata-Pendias, A. and Pendias, H. (1984). *Trace Elements in Soils and Plants*. CRC Press Inc., Boca Raton, Florida.

Kabata-Pendias, A. and Pendias, H. (2001). *Trace Elements in Soils and Plants* (3rd edition). CRC Press Inc., Boca Raton, Florida.

Kenner, BA, Clark, HP (1974). Detection and enumeration of salmonella and *Pseudomonas aeruginosa. Journal of Water Pollution Control Federation* 46, 9, 2163–2171.

Kerndorff, H. and Schnitzer, M. (1980). Sorption of metals by humic acid*. Geochimica et Cosmochimica Acta* 44, 1701–1708.

Lester, J.N. (1981). Removal of heavy metals in conventional wastewater treatment. In: CEP Consultants Ltd: *Heavy Metals in the Environment*, 104–113, Edinburgh.

Lewis G.R. (1999). 1001 Chemicals In Everyday Products, J.Wiley & Sons, Eds.

Li, Z., Ryan, J.A., Chen, J.L. and Al-Adeb, S.R. (2001). Adsorption of cadmium on biosolids-amended soils. *Journal of Environmental Quality* 30, 903–911.

LINZ (1991). Life in New Zealand Survey. 24-Recall dietary intake analyses. University of Otago.

Logan, T.J. and Chaney, R.L. (1983). Utilization of municipal wastewater and sludge on land: Metals. In: A.L. Page, T.L. Gleeson III, J.E. Smith Jr, I.K. Iskander and L.E. Sommers (eds). *Proceedings of the 1983 Workshop on Utilization of Municipal Wastewater and Sludge on Land*. University of California, Riverside, CA.

Loganathan, P., Hedley, M.J., Gregg, P.E.H. and Currie, L.D. (1997). Effect of phosphate fertiliser type on the accumulation and plant availability of cadmium in grassland. *Nutrient Cycling in Agroecosystems* 47, 169–178.

Loganathan, P., Mackay, A.D., Lee, J. and Hedley, M.J. (1995). Cadmium distribution in hill pastures as influenced by 20 years of phosphate fertiliser application and sheep grazing. *Australian Journal of Soil Research* 33, 859–871.

MAFF/DoE (1993a). *Review of the Rules for Sewage Sludge Application to Agricultural Land: Food Safety and Relevant Animal Health Aspects of Metals*. Report of the Steering Group on Chemical Aspects of Food Surveillance, Ministry of Agriculture, Fisheries and Food/Department of the Environment. MAFF Publications, London.

MAFF/DoE (1993b). *Review of the Rules for Sewage Sludge Application to Agricultural Land: Soil Fertility Aspects of Metals*. Report of the Independent Scientific Committee, Ministry of Agriculture, Fisheries and Food/Department of the Environment. MAFF Publications, London.

McBride, M.B. (1989). Reactions controlling heavy metal solubility in soils. *Advances in Soil Science* 10, 1–56.

McBride, M.B. (1991). Processes of heavy and transition metal sorption by soil minerals. In: G.H. Bolt, M.F. De Boodt, M.H.B. Hates and M.B. McBride (eds). *Interactions at the Soil Colloid-Soil Solution Interface*. Kluwer Academic Publishers, Dordrecht.

McBride, M.B. and Blasiak, J.J. (1979). Zinc and copper solubility as a function of pH in an acid soil. *Soil Science Society of America Journal* 43, 866–870.

McBride, M.B., Richards, B.K., Steenhuis, T., Russo, J.J. and Sauvé, S. (1997). Mobility and solubility of toxic metals and nutrients in soil fifteen years after sludge application. *Soil Science* 162, 487–500.

McGrath, S.P. (1995). Chromium and nickel. In: B.J. Alloway (ed.). *Heavy Metals in Soils*, 2nd edition. Blackie Academic and Professional, Glasgow.

McGrath, S.P. and Lane, P.W. (1989). An explanation for the apparent losses of metals in a long-term experiment with sewage sludge. *Environmental Pollution* 60, 235–256.

McLaren, R.G. and Cameron, K.C. (1996). *Soil Science: Sustainable Production and Environmental Protection*, 2nd edition. Oxford University Press, Auckland.

McLaren, R.G. and Crawford, D.V. (1973a). Studies on soil copper. I: The fractionation of copper in soils. *Journal of Soil Science* 24, 172–191.

McLaren, R.G. and Crawford, D.V. (1973b). Studies on soil copper. II: The specific adsorption of copper by soils. *Journal of Soil Science* 24, 443–452.

McLaren, R.G. and Gray, C.W. (1999). *LEJV Biosolids Compost Trials: Soil and Plant Analysis and Data Interpretation*. A Report for Living Earth Joint Venture Company Ltd, Centre for Soil and Environmental Quality, Lincoln University, Lincoln, NZ.

McLaren, R.G., Lawson, D.M. and Swift, R.S. (1986). Sorption and desorption of cobalt by soils and soil components. *Journal of Soil Science* 37, 413–426.

McLaren, R.G., Naidu, R., Smith, J. and Tiller, K.G. (1998). Fractionation and distribution of arsenic in soils contaminated by cattle dip. *Journal of Environmental Quality* 27, 348–354.

McLaren, R.G. and Ritchie, G.S.P. (1993). The long-term fate of copper fertilizer applied to a lateritic sandy soil in Western Australia. *Australian Journal of Soil Research* 31, 39–50.

McLaren, R.G., Singh, D. and Cameron, K.C. (1997). Influence of pH on the desorption of native and applied zinc from soils. In: I.K. Iskander, S.E. Hardy, A.C. Chang and G.M. Pierzynski (eds). *4th International Conference on the Biogeochemistry of Trace Elements*, Berkeley, California.

McLaren, R.G., Taylor, M.D., Hendry, T. and Clucas, L. (1999). Leaching of metals and nutrients from soils treated with metal-amended sewage sludge. In: L.D. Currie, M.J. Hedley, D.J. Horne and P. Loganathan (eds). *Best Soil Management Practices for Production*. Fertiliser and Lime Research Centre, Massey University, Palmerston North.

McLaughlin, M.J., Hamon, R.E., McLaren, R.G., Speir, T.W. and Rogers, S.L. (2000). Review: A bioavailability-based rationale for controlling metal and metalloid contamination of agricultural land in Australia and New Zealand. *Australian Journal of Soil Research* 38, 1037–1086.

Merry, R.H., Tiller, K.G. and Alston, A.M. (1983). Accumulation of copper, lead and arsenic in some Australian orchard soils. *Australian Journal of Soil Research* 21, 549–561.

MHSPE (1994). *Environmental Quality Objectives in the Netherlands*. Risk Assessment and Environmental Quality Division Directorate for Chemicals, Ministry of Housing, Spatial Planning and the Environment, The Netherlands.

MHSPE (2000). *Circular on Target Values and Intervention Values for Soil Remediation*. Ministry of Housing, Spatial Planning and the Environment, The Netherlands.

MfE and MoH (1997). *Health and Environmental Guidelines for Selected Timber Treatment Chemicals*. Ministry for the Environment and Ministry of Health, Wellington.

Mitchell, M.J., Hartenstein, R., Swift, B.L., Neuhauser, E.F., Abrams, B.I., Mulligan, et al. (1978). Effects of different sewage sludges on some chemical and biological characteristics of soil. *Journal of Environmental Quality* 7, 551–559.

NEPC, 1999. *Guidelines on the Investigation Levels for Soil and Water. Schedule B(1).* National Environment Protection (Assessment of Site Contamination) Measure, National Environment Protection Council, Adelaide.

NSW EPA (1997). *Environmental Guidelines: Use and Disposal of Biosolids Products*. New South Wales Environmental Protection Authority, Sydney.

NSW EPA (1998). *Draft Guidelines for the NSW Site Auditor Scheme*. New South Wales Environmental Protection Authority, Sydney.

NRMMC (2003). National Resource Management Ministerial Council. *Guidelines for Sewage Systems: Biosolids Management.* Department of Agriculture, Fisheries and Forestry, Canberra, Australia.

Oake, R.J., Booker, C.S. and Davis, R.D. (1984). Fractionation of heavy metals in sewage sludges. *Water Science and Technology* 17, 587–598.

O'Connor, G.A., Chaney R.L. and Ryan, J.A. (1991). Bio-availability to plants of sludge-borne toxic organics. *Reviews of Environmental Contamination and Toxicology* 121, 129–155.

Ogilvie, D. (1998). *National Study of the Composition of Sewage Sludge*. NZ Water and Wastes Association, Auckland.

O'Neill. (1995). Arsenic. In: B.J. Alloway (ed.). *Heavy Metals in Soils*, 2nd edition. Blackie Academic and Professional, Glasgow.

Parker, D.R., Chaney, R.L. and Norvell, W.A. (1995). Chemical equilibrium models: Applications to plant nutrition research. In: R.H. Loeppert, A.P. Schwab and S. Goldberg (eds). *Chemical Equilibrium and Reaction Models*. Soil Science Society of America Inc., Madison, WI.

Paxéus, N., Schröder, H.F., (1996). Screening For Non-Regulated Organic Compunds In Municipal Wastewater In Göteborg, Sweden, Wat.Sci.Tech 33, 6, 9-15.

Percival, H.J., Webb, T.H. and Speir, T.W. (1996). *Assessment of Background Concentrations of Selected Determinands in Canterbury Soils*. Landcare Research Contract Report LC9596/133. Canterbury Regional Council, Christchurch.

Purves, (1985). *Trace Element Contamination of the Environment* (revised edition). Elsevier, Amsterdam.

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, R.S. Kookana, D.P. Oliver, S. Rogers and M.J. McLaughlin (eds). *Contaminants in the Soil Environment in the Australasia-Pacific Region*. Kluwer Academic Publishers, Dordrecht.

Roberts, A.H.C., Longhurst, R.D. and Brown, M.W. (1994). Cadmium status of soils, plants, and grazing animals in New Zealand. *New Zealand Journal of Agricultural Research* 37, 119–129.

Roberts, A.H.C., Longhurst, R.D. and Brown, M.W. (1995). *Cadmium Survey of South Auckland Market Gardens and Mid Canterbury Wheat Farms.* Report to the New Zealand Fertiliser Manufacturers Research Association.

Rooney, C.P. (1996). Forms and phytoavailability of lead in a soil contaminated with lead shot. BSc (Hons) dissertation, Lincoln University, Lincoln, NZ.

Rooney, C.P., McLaren, R.G. and Cresswell, R.J. (1999). Distribution and phytoavailability of lead in a soil contaminated with lead shot. *Water, Air and Soil Pollution* 116, 535–548.

SA EPA (1996). *South Australian Biosolids Guidelines for the Safe Handling, Reuse or Disposal of Biosolids*. Department of Environment and Natural Resources, South Australian Environment Protection Authority, Adelaide.

Sandberg, G.R. and Allen, I.K. (1975). A proposed arsenic cycle in an agronomic ecosystem. In: E.A. Woolson (ed.). *Arsenical Pesticides*. ACS Symposium Series No. 7, American Chemical Society, Washington, DC.

Scobie, S., Buckland, S.J., Ellis, H.K. and Salter, RT. 1998. *Organochlorines in New Zealand: Ambient Concentrations of Selected Organochlorines in Estuaries.* Ministry for the Environment, Wellington, New Zealand. ISBN 0 478 09036 6.

Shacklette, H.T. and Boerngen, J.G. (1984). *Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States*. US Geological Survey Professional Paper No. 1270. US Geological Survey, Washington, DC.

Sheppard, S.C. (1992). Summary of phytotoxic levels of soil arsenic. *Water, Air and Soil Pollution* 64, pp. 539–550.

Shuman, L.M. (1979). *Zinc, Manganese and Copper in Soil Fractions* 127, 10–17.

Sidle, R.C. and Kardos, L.T. (1977). Transport of heavy metals in a sludge-treated forest area. *Journal of Environmental Quality* 6, 431–437.

Smith, E., Naidu, R. and Alston, A.M. (1998). Arsenic in the environment: A review. *Advances in Agronomy* 64, 149–195.

Smith, E., Naidu, R. and Alston, A.M. (1999). Chemistry of arsenic in soils. I: Sorption of arsenate and arsenite by four Australian soils. *Journal of Environmental Quality* 28, 1719–1726.

Smith, S.R. (1996). *Agricultural Recycling of Sewage Sludge and the Environment*. CAB International, Wallingford, UK.

Smolders, E. and McLaughlin, M.J. (1996). Chloride increases cadmium uptake in Swiss chard in a resin-buffered nutrient solution. *Soil Science Society of America Journal* 60, 1443–1447.

Speed, (1993). Superadministerial Project Effective Emissions Reduction Diffuse Sources. Document - Heavy Metals In Surface Waters And Abatement, RIZA Report No 93012, RVIM Report Number 773003001.

Speir, T. (1999). Biosolids re-use: Towards environmental effects-based heavy metals guidelines. A paper presented at the NZWWA Annual Conference, Christchurch.

Speir, T.W., Horswell, J., van Schaik, A. and Lloyd-Jones, A. (2000). Bio-indicators to assess impacts of heavy metals in land-applied sewage sludge. In: J.A. Adams and A.K. Metherell (eds). *Soil 2000: New Horizons for a New Century, Australian and New Zealand Second Joint Soils Conference. Volume 2: Oral Papers*. New Zealand Society of Soil Science, Lincoln University, Lincoln, NZ.

Stark, B., Suttle, N. Sweet, N. and Brebner, J. (1995). *Accumulation of PTEs in Animals Fed Dried Grass Containing Sewage Sludge*. Final Report to the Department of the Environment, WRc Report No. DoE 3753/1. WRc, Medmenham, Marlow.

Stark, B.A. and Wilkinson, J.M. (1994). *Accumulation of Metals by Sheep Given Diets Containing Sewage Sludge*. OC 8910, CSA 1826. Final Report to the Ministry of Agriculture, Fisheries and Food. Report No. 7. Chalcombe Agricultural Resources, Canterbury.

Steevens, D.R., Walsh, L.M. and Keeney, D.R. (1972). Arsenic phytotoxicity on a plainfield sand as affected by ferric sulfate or aluminium sulfate. *Journal of Environmental Quality* 1, 301–303.

Steinhilbler, P. and Boswell, F.C. (1983). Fractionation and characterisation of two aerobic sewage sludges. *Journal of Environmental Quality* 12, 529–534.

Stevenson, F.J. (1982). *Humus Chemistry, Genesis, Composition, Reactions*. Wiley, New York.

Swift, R.S. and McLaren, R.G. (1991). Micronutrient adsorption by soils and soil colloids. In: G.H. Bolt, M.F. De Boodt, M.B.H. Hayes and M.B. McBride (eds). *Interactions at the Soil Colloid-Soil Solution Interface*. Kluwer Academic Publishers, Dordrecht.

Tamaki, S. and Frankenberger, W.T. Jr. (1992). Environmental biochemistry of arsenic. *Reviews of Environmental Contamination and Toxicology* 124, 79–110.

Tasmanian Department of Environment and Land Management (1999). *Tasmanian Biosolids Reuse Guidelines.* Tasmanian Department of Environment and Land Management, Hobart.

Tiller, K.G., Honeysett, J.L. and De Vries, M.P.C. (1972). Soil zinc and its uptake by plants. II: Soil chemistry in relation to prediction of availability. *Australian Journal of Soil Research* 10, 165–182.

Tiller, K.G. and Merry, R.H. (1981). Copper pollution of agricultural soils. In: J.F. Loneragan, A.D. Robson and R.D. Graham (eds). *Copper in Soils and Plants*. Academic Press, Sydney.

UK Statutory Instrument. (1989). *The Sludge (Use in Agriculture) Regulations 1989*. Statutory Instrument No. 1263. HMSO, London.

Ulmgren, L. (1999). Examples of good practices for reducing wastewater and sludge contamination: The case of Sweden. Paper presented at the International Workshop ‘Problems around Sludge’, November 1999, Italy.

Ulmgren, L. (2000a). Stockholm Water Company. Measures taken in smaller industries to avoid hazardous substances entering domestic wastewater systems. Paper presented 25 May 2000, del Instituto de Ingenierá, UNAM, Mexico

Ulmgren, L. (2000b). Stockholm Water Company. Wastewater treatment and steps taken in practice for reducing sludge contamination in Stockholm, Sweden. Paper presented 27–28 March 2000, at the conference Traitamiento de lodos de depuradora:su minimización, valorización y destino final.

US National Research Council 1996.

USEPA (1993). Part 503: Standards for the use and disposal of sewage sludge. *Federal Register* 58, 9387–9404.

USEPA (1995). *A Guide to the Biosolids Risk Assessments for the EPA Part 503 Rule*. EPA/832-B-93-005. Office of Wastewater Management, Washington, DC.

USEPA (1999). *Environmental Regulations and Technology: Control of Pathogens and Vector Attraction in Sewage Sludge (Including Domestic Septage)*. EPA/625/R-92-013. Office of Research and Development, Cincinnati, Ohio.

Van den Berg, M., Birnbaum, L., Bosveld, A.T.C., Brunstrom, B., Cook, P., Feeley, M., et al. (1998). Toxic equivalency factors (TEFs) for PCBs, PCDDs PCDFs for humans and wildlife. *Environmental Health Perspectives* 106, 775–792.

Vigerust, E. and Selmer-Olsen, A.R. (1986). Basis for metal limits relevant to sludge utilisation. In: R.D. Davis, H. Haeni and P. L'Hermite (eds). *Factors Influencing Sludge Utilization Practices in Europe*. Elsevier Applied Science Publishers Ltd., Barking.

Weggler-Beaton, K., McLaughlin, M.J. and Graham, R.D. (2000). Salinity increases cadmium uptake by wheat and Swiss chard from soil amended with biosolids. *Australian Journal of Soil Research* 38, 37–45.

Wellington City Council (1997). *Draft Proposed Management Standards for Land Application of Exceptional Quality Biosolids in the Wellington Region.*

Wells, N. (1957). Soil studies using sweet vernal to assess element availability. Part 3: Copper in New Zealand soil sequences. *New Zealand Journal of Science and Technology* B38, 884–902.

Wells, N. (1960). Total elements in topsoils from igneous rocks: An extension of geochemistry. *Journal of Soil Science* 11, 409–424.

Wells, N. (1962). 'Total' chromium in topsoils. *New Zealand Soil Bureau Single Factor Maps* Nos. 71 and 72. DSIR, Wellington.

Wilderer, P.A. and Kolb, F.R. (1997). Abwasserexfiltration und Niederschlagswasserversickerung. *Studie im Auftrag der Landeshauptstadt München* Juli.

Williams, D.E., Vlamis, J., Pukite, A.H. and Corey, J.E. (1985). Metal movement in sludge-treated soils after six years of sludge addition. 2: Nickel, cobalt, iron, manganese, chromium and mercury. *Soil Science* 140, 120–125.

WHO (1998). *Environmental Health Criteria for Copper*. World Health Organization.

WRc (1994), Diffuse Sources Of Heavy Metals To Sewers, Final Report To The Department Of The Environment. Doe 3624.

Yamada, M., Dazai, M. and Tonomura, K. (1959) Change of mercurial compounds in activated sludge. *Journal of Fermentation Technology* 47, 155.

APPENDICES

APPENDIX I: ENUMERATION OF THERMOTOLERANT CAMPYLOBACTER IN BIOSOLIDS

ENUMERATION OF THERMOTOLERANT CAMPYLOBACTER IN BIOSOLIDS

[Method developed by: A. Donnison, AgResearch Limited]

Introduction

The genus *Campylobacter* has been established as a common, if not the major, cause of human diarrhoeal disease in the developed world, with particularly high rates reported in New Zealand. Most campylobacters associated with human disease are thermotolerant and the principal species involved are *Campylobacter jejuni* and *Campylobacter coli*. Thermotolerant campylobacters are usually present in raw sewage and slaughterhouse sludges, but concentrations decrease substantially after treatment and/or storage (Jones et al., 1990; Höller & Schomakers-Revaka, 1994; Fransen et al., 1996). Sludge is apparently an unfavourable environment for campylobacters because there is a rapid decline in concentration following addition of sludge to laboratory-grown cultures (Jones et al., 1990; Höller & Schomakers-Revaka, 1994).

Sludges are non-uniform materials that require homogenisation prior to microbiological analysis. Although there is no ‘best method’ for homogenising sludges, thorough mixing is not detrimental to subsequent recovery of these oxygen-sensitive bacteria (Höller & Schomakers-Revaka, 1994).

Media and methodologies used to isolate thermotolerant campylobacters from water can be applied to sludges (e.g., Jones et al., 1990). A procedure for laboratory culture of thermotolerant Campylobacter is described below. The protocol includes a resuscitation procedure (Humphrey & Muscat, 1989) and was the method recommended to the New Zealand Ministry of Health for the analysis of food and water (Donnison, 2002).

Enumeration of thermotolerant campylobacter in biosolids

Campylobacter can be enumerated in biosolids by the most-probable-number (MPN) method (Figure A1). A three-tube per row MPN format is widely used in New Zealand and is cost effective, but laboratories that require a more precise result may wish to use a five-tube format.

The analytical procedure involves two different media: a primary enrichment broth (Exeter broth) and a secondary enrichment agar (mCCDA agar). The procedure takes a total of four days. The first step is a resuscitation procedure that consists of incubation of inoculated broths at 37°C for 4 to 6 h. The second step is transfer of broth enrichments to 42°C and incubation for 44 to 42 h (i.e., the total incubation period is 48 h). The third step involves subculture from enrichment tubes to parallel-labelled plates of mCCDA agar. Plates are incubated in a microaerophilic atmosphere at 42°C for 48 h.

Sample collection

It is important that samples of biosolids are as representative as possible. Sufficient sample needs to be collected to allow 50 g (wet weight) of treated or stored biosolids to be analysed for campylobacter. Samples should be collected using aseptic techniques (see section 9.2.3), placed on ice for transport, and analysed as soon as possible after collection. It is highly desirable that the time between collection and analysis does not exceed 24 hours.

Quality control

A suitable reference culture for quality control is *Campylobacter jejuni* (NZRM[8] 1958). If desired, *Escherichia coli* (e.g., NZRM 916) can be used as a negative control organism.

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[8] *New Zealand Reference Material*

Homogenisation of biosolids

For sewage sludge 10 ml or 10 g (wet weight) may be sufficient for campylobacter analysis, but for treated/stored biosolids 50 ml, or more usually 50 g (wet weight), is required. Although some biosolids are sufficiently liquid to be homogenised directly, most samples require the addition of liquid. The most satisfactory liquid for homogenisation is campylobacter enrichment broth. A biosolids-Exeter broth mixture (1 in 10, w/v or v/v) can be homogenised for about 1 minute either in a Waring blender or in a stomacher. Do not process longer than is necessary to obtain a homogeneous mixture.

Note: homogenisation in enrichment broth has two advantages: the oxygen- quenching agents in the medium protect against aeration during the mixing process, and volumes of 10 ml or more of homogenate can be pipetted directly into suitably sized incubation vials/tubes, allowing the use of single strength medium throughout.

Setting up an MPN test

The identity of all vials/tubes must be maintained throughout an MPN test.

MPN format

For a three-tube MPN, three tubes are inoculated with sample portions of the same size; these three tubes constitute an “MPN row”. At least three 10-fold decreasing- volume portions of homogenate (or three consecutive 10-fold dilutions) must be inoculated because results for three consecutive rows are required to obtain an MPN score.

Record the inoculum size and/or dilution for each row in the MPN series on a laboratory work sheet.

MPN inocula

Suggested typical inoculum sizes are as follows.

Sewage sludge: 3 x 1 ml, 3 x 1 ml of 10^{-1} dilution and 3 x 1 ml of 10^{-2} dilution

Dilute a homogenised sample in sterile 0.1% peptone (pH 6.8 ± 0.2). If concentrations are expected to be low, include an additional MPN row: *3 x 10 ml* (using aliquots of homogenate pipetted directly to incubation vials/tubes). If concentrations are expected to be high, an additional MPN row of *3 x 1 ml of* 10^{-3} dilution may be required.

Biosolids: *3 x 100 ml, 3 x 10 ml* and *3 x 1 ml*

If concentrations are expected to be high then the homogenate should be diluted and an additional row of 3 *x 0.1 ml* inoculated.

Vials/tubes for incubation of enrichments

Exeter broth contains oxygen-quenching agents so that broths can be incubated in air in screw-capped containers provided these are long and narrow in shape and there is only a very small head space (< 1 cm) after addition of sample. If these conditions are not met, enrichments must be incubated in a microaerophilic atmosphere.

Analytical procedure

The analytical procedure comprises three stages: resuscitation, primary selective enrichment and secondary selective enrichment. A fourth stage, confirmation, is highly recommended and should always be carried out by analysts who are unfamiliar with campylobacter isolation. Experienced analysts may confirm samples according to Good Laboratory Practice guidelines.

Primary selective enrichment

1. Pipette 100 ml or 10 ml volumes of biosolids-Exeter broth homogenate to sterile, labelled MPN tubes/vials.
2. Pipette 1 ml aliquots of homogenate or diluted homogenate to 9 ml Exeter broth in labelled MPN tubes/vials.

<u>Step 1: Pre-enrichment</u>

Place the MPN tubes/vials in an incubator at 37°C for 4 to 6 h. This step is mandatory.

Note: ensure that tubes/vials are either completely filled or placed in a microaerophilic atmosphere for incubation.

<u>Step 2: Selective enrichment</u>

Transfer tubes/vials to an incubator at 42°C and continue incubation for 44 to 42 h (i.e., the total incubation time for the primary selective enrichment procedure is 48 h).

Note: if a microaerophilic atmosphere was required for pre-enrichment it must be re-established for selective enrichment.

Secondary selective enrichment

1. Subculture from each primary selective enrichment tube by streaking a loopful of growth to a plate of mCCDA agar[9] so as to obtain single colonies. Ensure that each plate is labelled to maintain its identity in the MPN series.
2. Invert plates and place in a microaerophilic atmosphere.
3. Incubate plates at 42°C for 48 h.

Microaerophilic atmosphere: Either use an anaerobic jar and commercial kits of the required size to generate a microaerophilic atmosphere consisting of 5–6% oxygen, 10% carbon dioxide and 84–85% nitrogen, or use an incubator with an atmosphere consisting of the same gases in the same proportions.

Confirmation of thermotolerant campylobacter

Thermotolerant campylobacter colonies appear as flat, moist, grey-white colonies with irregular spreading margins on mCCDA.

To confirm isolates as thermotolerant campylobacter, select one or more typical colonies from each mCCDA plate and carry out the following tests:

- Gram stain
- microscopic examination of a wet mount
- oxidase test.

Note: Some workers prefer to use saffranin as the counterstain in the Gram stain test.

In-house oxidase test

1. Place two or three drops of reagent to the centre of a filter paper (e.g., Whatman No 1) and allow a few seconds for absorption.

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[9] Prior to innoculation, plates should be dried to remove surface moisture. Avoid over-drying and do not use a laminar flow cabinet.

2. Using a platinum or plastic loop, smear a loopful of test colony onto the filter paper over a line 3–6 mm long.
3. A positive test is the production of a dark purple colour *within 5 to 10 seconds.*

Note: Commercial preparations may be more convenient.

Score each plate as follows:

	Thermotolerant campylobacter reactions
Gram stain	Gram-negative, small gull-shaped rods
Motility (wet mount)	+ve (typically corkscrew)
Oxidase test	+ve

Reporting results

1. Sum the positive plates in each MPN row to produce an MPN score for three consecutive rows.
2. Refer the MPN score to a three-tube MPN table to obtain the concentration of thermotolerant campylobacter in the homogenate (refer Table A1).
3. Calculate the concentration of campylobacter in the original sample by multiplying the homogenate concentration by 10 (this allows for the addition of liquid prior to homogenisation).

Note: For treated or stored biosolids that are usually solid or semi-solid solid, it may be more appropriate to express results on a dry weight basis. To report concentration on a dry weight basis it is necessary to determine the water content of the biosolids sample. Procedures are available in a number of texts [e.g., *Standard Methods for the Examination of Water & Wastewater*, (APHA, 1998)].

Media: Recipes and preparation

Primary selective enrichment medium: Exeter broth

Exeter broth base

Basal medium (Nutrient Broth No. 2, Oxoid CM67)

Ingredients	g/950 ml
Lab-Lemco powder	10.0 g
Peptone	10.0 g
Sodium chloride	5.0 g
Distilled water	950 ml
	pH 7.5 ± 0.2

Prepare basal medium by dissolving 25 g of dry ingredients in 950 ml of water. Dispense 475 ml aliquots to screw-capped containers and sterilise by autoclaving at 121°C for 15 min. Basal medium can be stored for several weeks at 4°C.

Additives

(a) Growth supplement (Oxoid SR 84)

Ingredients	Vial contents (sufficient for 500 ml)
Sodium pyruvate	0.125 g
Sodium metabisulphite	0.125 g
Ferrous sulphate	0.125 g

To reconstitute, add 2 ml of sterile distilled water and mix gently to dissolve. Use immediately.

(b) Lysed/defibrinated horse blood (5% v/v)

Add 25 ml blood to 475 ml of Exeter medium.

Note: Lysed horse blood should be used within the date specified by the supplier.

(c) Antibiotics

(i) Campylobacter Selective Supplement (Oxoid SR 204E)

Ingredients	Vial contents (sufficient for 500 ml)
Polymyxin B	2,500 IU
Rifampicin	5 mg
Trimethoprim	5 mg
Amphotericin B	5 mg

To reconstitute, add 2 ml of sterile distilled water to a vial of Selective Supplement (as supplied by the manufacturer) and mix gently to dissolve. Use immediately.

(ii) Cefoperazone solution

Ingredients	mg/2ml
Cefoperazone	15 mg
Distilled water	2 ml

Stir to dissolve and add to 500 ml Exeter broth.

Note: Larger amounts can be prepared and 1 ml aliquots stored in the freezer until required.

Complete Exeter broth (500 ml)

To 475 ml of basal medium, that is at 50°C or below add: 1 vial of growth supplement, 25 ml of lysed horse blood, 1 vial of selective supplement and 1 ml of cefoperazone solution. Dispense the 500 ml of Exeter broth obtained as required volumes to sterile bottles, tubes or vials.

Note: For optimum results use Exeter broth on the day of preparation.

Secondary Enrichment Medium: Campylobacter mCCDA (Oxoid CM739)

Basal mCCDA medium

Ingredients	g/l
Lab-Lemco powder	10.0
Peptone	10.0
Sodium chloride	5.0
Bacteriological charcoal	4.0
Casein hydrolysate	3.0
Sodium desoxycholate	1.0
Ferrous sulphate	0.25
Sodium pyruvate	0.25
Agar	12.0
Distilled water	1000 ml
	pH 7.4 ± 0.2

Suspend 22.75 g of dehydrated mCCDA agar base in 500 ml of distilled water, mix well and boil to dissolve the agar. Sterilise by autoclaving at 121°C for 15 min.

mCCDA Selective Supplement (Oxoid SR 155)

Ingredients	Vial contents (sufficient for 500 ml)
Cefoperazone	16 mg
Amphotericin B	5 mg

To reconstitute, add 2 ml of sterile distilled water and mix to dissolve. Use immediately.

Complete mCCDA medium (500 ml)

Cool autoclaved basal mCCDA medium to 45 to 50°C. Add 1 vial of mCCDA selective supplement to 500 ml of mCCDA. Pour to Petri plates. Plates can be stored for up to seven days in sealed containers at 4°C.

Oxidase Reagent

Ingredients	g/100 ml
Para-aminodimethyanaline oxalate	1.0
Distilled water	100 ml

Dissolve the para-aminodimethyanaline oxalate in the distilled water by gently heating. The reagent can be stored in the refrigerator for a few days, or dispensed as small aliquots and frozen at -20°C for storage for several months. Thaw immediately before use.

Commercial preparations are available, usually as test strips or paper discs. These are both stable and convenient.

References for Appendix I

APHA (1998). *Standard Methods for the Examination of Water and Wastewater,* 20th edition. American Public Health Association, Washington, DC.

Donnison, A. (2002). *Isolation of Thermotolerant Campylobacter: Review and Methods for New Zealand Laboratories.* Ministry of Health, Wellington.

Fransen, N.G., van den Elzen, A.M.G., Urlings, B.A.P. and Bijker, P.G.H. (1996). Pathogenic micro-organisms in slaughterhouse sludge: A survey. *International Journal of Food Microbiology* 33, 245–256.

Höller, C. and Schomakers-Revaka, U. (1994). A note: Comparison of different homogenization procedures for detecting *Campylobacter* spp. in sewage sludge. *Journal of Applied Bacteriology* 77, 591–596.

Humphrey, T.J. and Muscat, I. (1989). Incubation temperature and the isolation of *Campylobacter jejuni* from food, milk or water. *Letters in Applied Microbiology* 9, 137–139.

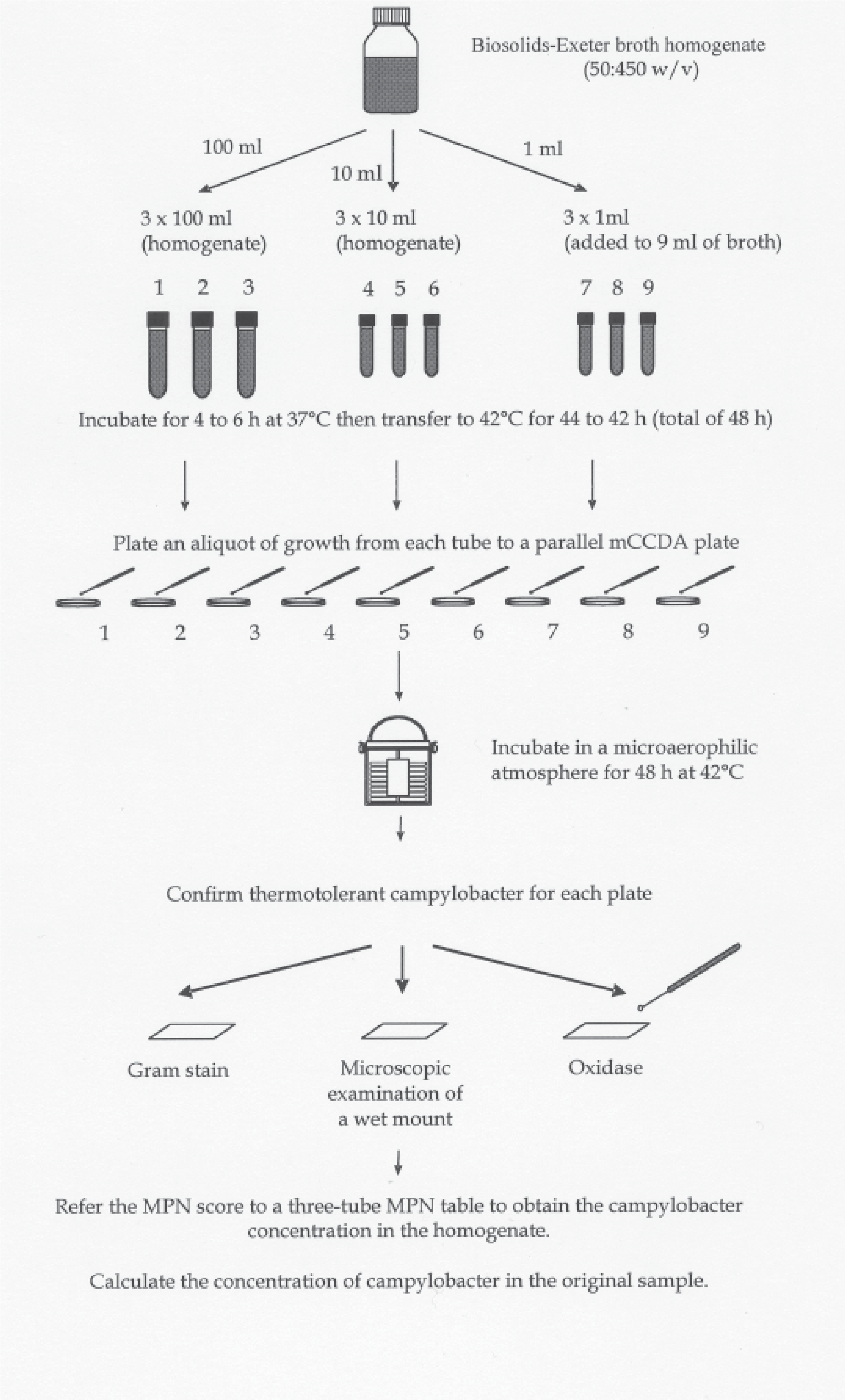
Jones, K., Betaieb, M. and Telford, D.R. (1990). Seasonal variation of thermophilic campylobacters in sewage sludge. *Journal of Applied Bacteriology* 69, 185–189.

Table A1: Three-tube Most Probable Number (MPN) table

Number of tubes giving a positive reaction			MPN per 100 ml	95% confidence limits	
3 of 10 ml each	3 of 1 ml each	3 of 0.1 ml each		Lower limit	Upper limit
0	0	1	3	<0.5	9
0	1	0	3	<0.5	13
1	0	0	4	<0.5	20
1	0	1	7	1	21
1	1	0	7	1	23
1	1	1	11	3	36
1	2	0	11	3	36
2	0	0	9	1	36
2	0	1	14	3	37
2	1	0	15	3	44
2	1	1	20	7	89
2	2	0	21	4	47
2	2	1	28	10	149
3	0	0	23	4	120
3	0	1	39	7	130
3	0	2	64	15	379
3	1	0	43	7	210
3	1	1	75	14	230
3	1	2	120	30	380
3	2	0	93	15	380
3	2	1	150	30	440
3	2	2	210	35	470
3	3	0	240	36	1300
3	3	1	460	71	2400
3	3	2	1100	150	4800

MPN and 95% confidence limits for various combinations of positive results when three 10 ml portions, three 1 ml portions and three 0.1 ml portions are used to calculate the MPN score.

Figure A1: Enumeration of thermotolerant campylobacter in biosolids by MPN



APPENDIX II: DETECTION OF ADENOVIRUSES IN BIOSOLIDS

DETECTION OF ADENOVIRUSES IN BIOSOLIDS

[Method developed by: G. Lewis, School of Biological Sciences, University of Auckland]

Adenoviruses are one of the groups of enteric viruses transmitted by the faecal–oral route. The adenovirus group is also recognised as one of the more persistent enteric viruses when subjected to environmental stresses such as temperature, visible light and UV irradiation.

Adenoviruses occur in high numbers in sewage and sludges. The adenovirus group is proposed as the sentinel virus for UV irradiation of wastewater effluent (USEPA, 2003) and virus contamination of shellfish (Pina et al., 1998), and is proposed here as an indicator of virus persistence in biosolids.

The methodology recommended here requires detection of the viable virus through culture in a susceptible cell line with detection by molecular means. The molecular methods allow detection of those adenoviruses that do not produce clearly observable cytopathic effect (CPE), avoids misinterpretation of CPE caused by other viruses or non-infectious material, and reduces overall assay time.

Recent studies have carried out comparative testing of virus detection in sewage sludges (Monpoeho et al., 2001; Mignotte et al., 1999). The recommended method is that which produced the most reliable results for virus recovery in these studies.

Method

Beef extract/sonication method for recovery of human viruses from sewage sludges (Ahmed and Sorenson, 1995; Mignotte et al., 1999; Monpoeho et al., 2001).

1. Add a sludge/biosolid volume providing 25 g (dry matter) to 225 ml of 10% beef extract (Oxoid LP029B) at pH 9.0.
2. Stir at 500 rpm for 30 minutes.
3. Sonicate the mixture on ice (100W; 0.9 s) for 5 minutes in 1 minute intervals.
4. Stir for 5 minutes as above.
5. Centrifuge 5,000 *x g* for 1 hour at 4°C.
6. Decant the supernatant and adjust the pH to 7.2. (final extract).

Decontaminate the final extract

- add one-third volume of chloroform
- mix vigorously for 30 minutes (4°C)
- centrifuge 1,500 *x g* for 10 minutes (4°C)
- decant the aqueous phase (decontaminated extract).

Adenovirus analysis

- Adenovirus should be detected by culture-PCR on BGMK cell-line.
- Methods for Adenovirus types 40, 41 are described in Chapron et al. (2000).
- Primers for PCR for other adenovirus types are as described in Pina et al. (1998) and should be used according to the protocol of Chapron et al. (2000).

References for Appendix II

Ahmed, A. and Sorenson, D. (1995). Kenetics of pathogen destruction during storage of dewatered biosolids. *Water and Environmental Research* 67, 143–150.

Chapron, C.D., Ballester, N.A., Fontaine, J.H., Frades, C.N. and Margolin, A.B. (2000). Detection of astroviruses, enteroviruses, and adenovirus types 40 and 41 in surface waters collected and evaluated by the information collection rule and an integrated cell culture-nested PCR procedure. *Applied & Environmental Microbiology* 66, 6, 2520–2525.

Mignotte, B., Maul, A.L. and Schwartzbrod, L., (1999). Comparative study of techniques used to recover viruses from residual urban sludge. *Journal of Virological Methods* 78, 71–80.

Monpoeho, S., Maul, A., Mignotte-Cadiergues, B., Swartzbrod, L., Bvillaudel, S. and Ferre, V. (2001). Best viral elution method available for quantification of enterovirus in sludge by both cell-culture and RT-PCR. *Applied and Environmental Microbiology* 67, 2484–2488.

Pina, S., Puig, M., Lucena, F., Jofre, J. and Girones, R. (1998). Viral pollution in the environment and in shellfish: Human adenovirus detection by PCR as an index of Human viruses. *Applied and Environmental Microbiology* 64, 3376–3382.

USEPA (2003). *UV Disinfection Guide*, draft.