

TECHNICAL REPORT Investigations and Monitoring Group

**Methodologies to
assess the
environmental risks
associated with non-
natural rural waste**

Methodologies to assess the environmental risks associated with non- natural rural waste

Report No. R14/58

ISBN 978-1-927299-95-1 (print)

978-1-297299-96-8 (web)

978-1-297299-97-5 (CD)

Report prepared for Environment Canterbury by

Jo-Anne E. Cavanagh and

Lynn H. Booth

Landcare Research

Contract Report LC2004

September 2014





Report No. R14/58

ISBN 978-1-927299-95-1 (print)

978-1-297299-96-8 (web)

978-1-297299-97-5 (CD)

PO Box 345
Christchurch 8140
Phone (03) 365 3828
Fax (03) 365 3194

75 Church Street
PO Box 550
Timaru 7940
Phone (03) 687 7800
Fax (03) 687 7808

Website: www.ecan.govt.nz
Customer Services Phone 0800 324 636

Executive summary

'Non-natural' rural waste is the inorganic and organic waste from manufactured products that are used on farms. It includes scrap metal, hazardous waste, construction and demolition waste, agricultural plastics, waste agrichemicals and their containers, paper/cardboard, treated and untreated timber, feed and seed bags. From farm surveys, approximately 81,598 tonnes of non-natural rural waste, along with 123,500 tonnes of organic/animal waste and 4,300 tonnes of domestic wastes, are estimated to be produced each year in the Canterbury Region. Around 92% of surveyed farmers still use burning, burying in farm pits and/or bulk storage to manage some of their waste, which may lead to detrimental impacts on the environment.

Environment Canterbury is leading a proposed national multiphase project to tackle the issue of rural waste management in New Zealand. The project will assess the risk posed by current waste management practices, and find solutions to reduce risk and improve waste minimisation and disposal. A risk assessment phase will assess the impact from current waste burning, burying, and bulk storing practices to rank and prioritise waste streams for further work.

This report reviews the effects that may potentially arise from the burning, burying in farm pits or bulk storage of the different non-natural farm wastes and recommends appropriate methodologies for use during the environmental risk assessment phase.

The main classes of non-natural waste considered here are agricultural plastics, timber (including treated timber), tyres, hazardous substances such as agrichemicals, cardboard and paper, domestic waste, scrap metal, glass, white goods and electronics.

Detrimental impacts from the burning of these wastes arise when contaminants are emitted to air and land. These include a range of volatile and semi-volatile organics such as benzene, polycyclic aromatic hydrocarbons and dioxins. Some waste may also give rise to metallic contaminants such as arsenic. Detrimental impacts from the burial and bulk storage of waste occur when contaminants are released into the soil and subsequently move into groundwater and surface waters. Some wastes (e.g. tyres) may pose a fire hazard.

In order for an environmental *risk* to exist there needs to be some form of *hazard* that impacts a *receptor* via some *pathway of exposure* (e.g. a person exposed to smoke from the open burning of farm wastes; soil biota exposed to contamination from burial of non-natural farm waste). If these three components are not present, then an environmental risk is not present.

In selecting a risk assessment approach, consideration needs to be given as to the intended use of the information and how that can aid decision-making. The objective of the wider project is to assess the risk posed by current waste management practices, and find solutions to reduce risk and improve waste minimisation and disposal. The risk assessment phase is intended to prioritise the waste streams for further work. As such, it is recommended that a two-step process is undertaken to first assess risks and then find solutions.

There are inherent uncertainties in (a) determining the hazards associated with the different waste streams and (b) identifying the assumptions that are made when providing an *absolute* quantitative or semi-quantitative assessment of the risks associated with the end-of-life options for these waste streams. Therefore, a simple risk-ranking exercise is recommended for prioritising the waste streams for further investigation. To assist with finding solutions for reducing risks it is recommended that a life-cycle approach is adopted as this provides a *relative* comparison of different options for end-of-life disposal of the different wastes.

Table of contents

Executive summary	i
1 Introduction	1
1.1 Project aims	1
2 Environmental hazards associated with different non-natural waste streams	2
2.1 Waste streams considered	2
2.2 Agricultural plastics	3
2.2.1 Open burning	3
2.2.2 Burial	5
2.2.3 Bulk storage	5
2.3 Wood, including treated timber	5
2.3.1 Open burning	5
2.3.2 Burial	6
2.3.3 Bulk storage	6
2.4 Tyres	7
2.4.1 Open burning	7
2.4.2 Burial	7
2.4.3 Bulk storage	7
2.5 Hazardous substances	7
2.5.1 Open burning	7
2.5.2 Burial	8
2.5.3 Bulk store	8
2.6 Household waste	8
2.6.1 Open burning	8
2.6.2 Burial	8
2.6.3 Bulk store	9
2.7 White goods/electronics	9
2.7.1 Open burning	9
2.7.2 Burial	9
2.7.3 Bulk storage	9
2.8 Cardboard and paper	9
2.8.1 Open burning	9
2.8.2 Burial	9
2.8.3 Bulk storage	9
2.9 Construction and demolition waste	9
2.10 Metals and glass	10
2.11 Summary	10
3 Assessing environmental risk	12
3.1 Exposure pathways and receptors associated with different waste disposal options ..	12
3.1.1 Open burning	12
3.1.2 Burial	13
3.1.3 Bulk storage	13

3.2	Environmental risk assessment approaches	13
3.2.1	Quantitative risk assessment	13
3.3	Risk-ranking	15
3.4	Risk assessment matrix	17
3.5	Life cycle analysis approaches	17
3.6	Summary	18
4	Proposed risk assessment approach.....	20
4.1	Purpose of risk assessment.....	20
4.2	Undertaking the risk assessment.....	20
4.3	Recommended methodology	20
4.3.1	Risk-ranking	21
4.3.2	Life cycle analysis	21
5	Conclusions and recommendations	22
5.1	Recommendations for next steps	23
6	References.....	24
Appendix – Summary of toxicological effects of selected contaminants associated with non-natural rural waste		27

List of Figures

Figure 2-1:	Amount of different general waste streams estimated to be produced annually by 53 surveyed farms (from GHD 2013). Note: tyres and batteries not included	3
Figure 3-1:	Exposure pathways of emissions associated with the open burning of agricultural plastics (adapted from Enviser Consulting 2013).....	13
Figure 3-2:	Generic risk matrix	17

List of Tables

Table 2-1:	Categorisation of the specific waste items identified by GHD (2013) into general waste categories used in this report.....	2
Table 2-2:	Range in emission factors for volatile organic compounds and particulate emissions from the burning of polyethylene in silviculture	4
Table 2-3:	Summary of emission factors from open burning of copper-chrome-arsenate-treated wood	6
Table 2-4:	Summary of the hazards arising from different methods of disposal of various non-natural rural waste streams	11
Table 3-1:	Summary of factors used to rank diffuse-contamination issues in the Waikato Region .	15
Table 3-2:	Summary of categories used in risk-ranking approach	16
Table 3-3:	Summary of the advantages and disadvantages of different risk assessment approaches.....	19

1 Introduction

Previous scoping studies commissioned by Environment Canterbury in 2012 (GHD 2012; SKM 2012) highlighted that little was known about rural waste management in Canterbury. Farm surveys undertaken in 2013 over a cross section of farms in Canterbury found that approximately 9 tonnes of non-natural rural wastes; approximately 14 tonnes of organic (animal/offal waste) waste and 0.5 tonnes of domestic waste are produced on average (mean) by each farm every year (GHD 2013). Across Canterbury, this equates to 81,598 tonnes of non-natural rural waste, 123,500 tonnes of organic/animal waste and 4,300 tonnes of domestic wastes being produced each year (GHD 2013). Further, the study identified that 92% of surveyed farmers still use burning, burying in farm pits and/or bulk storage to manage some of their waste. Many of the waste streams identified by the surveys have the potential to cause harm to people and the environment if managed inappropriately.

Environment Canterbury is leading a proposed national multiphase project to tackle the issue of non-natural rural waste management in New Zealand. The project will assess the risk posed by current waste management practices, and find solutions to reduce risk and improve waste minimisation and disposal. A risk assessment phase will assess the impact from current waste burning, burying, and bulk storing practices to rank and prioritise waste streams for further work.

This current study was commissioned to scope environmental risk assessment methodologies that could be used during the risk assessment phase for non-natural rural waste.

Non-natural rural waste is the term used for inorganic waste that is produced by farms. It includes scrap metal, hazardous waste, construction and demolition waste, agricultural plastics, waste agrichemicals and their containers, feed and seed bags, animal health products, etc. (Hepburn and Keeling 2013).

1.1 Project aims

The aims of this project are to:

- Review the literature to identify qualitative and quantitative research relating to the effects of burning, burial and bulk-storage of the major non-natural rural waste streams, and environmental risk assessment methodologies
- Identify potential risk assessment methodologies for considering the environmental impact from current waste burning, burial and bulk-storage practices for the different non-natural waste streams.

2 Environmental hazards associated with different non-natural waste streams

2.1 Waste streams considered

Hepburn and Keeling (2013) define non-natural rural waste (NNRW) as the inorganic waste that is produced by farms and specify the inclusion of scrap metal, hazardous waste, construction and demolition waste, agricultural plastics, waste agrichemicals and their containers, feed and seed bags, animal health products, etc. However, organic material such as timber and paper and cardboard is also included in their discussion of NNRW. Similarly, GHD (2013) also include such materials in their quantification of non-natural rural wastes in Canterbury.

A broader definition of non-natural rural wastes is therefore also proposed for this report; specifically that **non-natural rural waste** covers both inorganic and organic waste from the manufactured products used on farms. NNRW includes scrap metal, hazardous waste (e.g. oil, waste agrichemicals), construction and demolition waste, agricultural plastics, agrichemical containers, feed and seed bags, animal health products, treated and untreated timber, tyres, batteries, and paper and cardboard. It does not include “natural” wastes such as animal waste (carcasses, offal, effluent) or vegetation (e.g. tree prunings).

Further, it was not clear which specific items make up the different waste categories; this needed to be defined before we could ascertain the potential risk of the different NNRW classes. We have therefore grouped the specific waste items from the farm survey (GHD 2013) into general NNRW categories for the purposes of this report (Table 2-1).

Table 2-1: Categorisation of the specific waste items identified by GHD (2013) into general waste categories used in this report

Waste group	Description of specific waste items
Agricultural plastic	Plastic drums and containers, plastic piping, silage and bale wrap, twine and netting, weed mat, mulch film and crop cover, PE pipes, plant pots, plastic pallet wrap, polystyrene plastic, fertiliser bags, plastic seed or feed bags, greenhouse plastic sheeting, twine, animal health plastic, drench containers, trickle or irrigation tape
Timber	Treated timber, old fence posts, untreated timber offcuts, pallets, wood chip, crates
Tyres	Tyres
Paper/cardboard	Miscellaneous cardboard, paper, filter socks
Domestic waste	General household waste produced by farmers
Hazardous substances	Paints, solvents, soil, agricultural sprays, sharps, ash, oil filters
Metal	Roofing materials, wire, used vehicles/machinery, wire, spray cans, foil seed or feed bags
Glass	Glass (taken to include glass used for animal welfare products)
Construction and demolition	Building waste
Batteries	Vehicle batteries
White goods	White goods, TVs, fluorescent bulbs

The amount of the different waste groups will also contribute to the associated risk and GHD (2013) provides a summary of the amount of different NNRW estimated to be produced annually by 53 farms (Figure 2-1), although the GHD report does not state the specific items included in each of the categories. Further, the categories used in this report (Table 2-1) are different to those used by GHD;

nonetheless, the information provided can be used to indicate the relative significance of the different waste streams. Specifically, the packaging stream appears to comprise approximately 15 tonnes of cardboard packaging with the remainder being plastic (predominantly) and foil packaging (GHD 2013). Combining the amount specified for the plastic and hazardous substance containers categories with the amount of plastic packaging results in plastic being produced in the greatest amount. Wood, which includes approximately 38.5 tonnes of treated timber, was the next greatest waste stream. Batteries (mainly vehicle) and tyres are not included in Figure 2-1 although 171 batteries and approximately 183,000 tyres were produced from the 53 survey farms (GHD 2013).

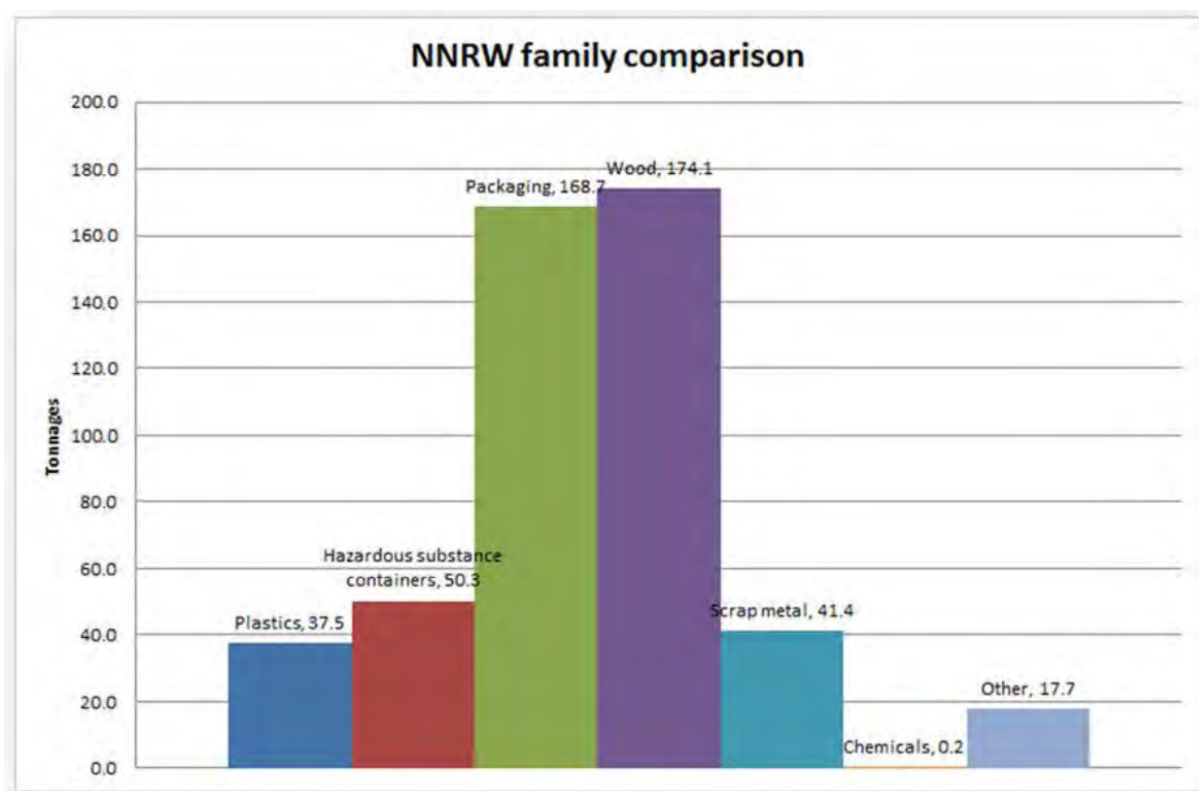


Figure 2-1: Amount of different general waste streams estimated to be produced annually by 53 surveyed farms (from GHD 2013). Note: tyres and batteries not included

The following sections provide an assessment of the hazards associated with the different waste categories specified in Table 2-1 for the different end-of-life disposal methods: open burning, burial and bulk storage.

2.2 Agricultural plastics

2.2.1 Open burning

Different plastics are used for different purposes in agriculture. Low density polyethylene (LDPE) and linear low density polyethylene (LLDPE) may be used as silage and bale wrap and greenhouse film. High density polyethylene (HDPE) is often used for agrichemical containers, while polypropylene (PP) may be used for twine and netting. Different plastics may give rise to different emissions under open-burning conditions. For example, Valavanidis *et al.* (2008) evaluated open burning of different types of plastic in a controlled laboratory setting – polyvinyl chloride (PVC), LDPE, HDPE, polystyrene (PS), PP and polyethylene terephthalate (PET). They measured polycyclic aromatic hydrocarbons (PAH), heavy metals and other elements in soot and residue solid ash from the burning of the wastes at a combustion temperature of 600–750°C. Toxic heavy metal (lead, cadmium, chromium, copper, nickel, zinc) concentrations were relatively low, although PVC produced higher concentrations of these metals than other plastics. Low molecular weight PAHs were present at higher levels in the soot than

in the ash for all plastics, while greater concentrations of higher molecular weight PAHs were found at higher concentrations in ash from PVC than from other types of plastic. Wrobel and Reinhardt (2003) also provide a good overview of the combustion of polyethylene (HDPE, LDPE) and provide a range of emission factors for volatile organic compounds and particulate emissions (Table 2-2), as well as for a wide range of individual PAHs. Emission factors for the PAHs were highly variable, for example, benzo(a)pyrene emissions ranged from 7.5 ng/kg to 1.6 mg/kg.

Table 2-2: Range in emission factors for volatile organic compounds and particulate emissions from the burning of polyethylene in silviculture (Wrobel and Reinhardt 2003)

Chemical	Emission factor			Mass emitted ¹			
	Minimum	Maximum	Units	Minimum	Maximum	Units	Reference
CO	100	175	mg/g ^a	31	55	g	Wang <i>et al.</i> (2003)
CO ₂	400	1500	mg/g	126	471	g	Wang <i>et al.</i> (2003)
Acetylene	ND	2.9	mg/g	ND	0.9	g	Wang <i>et al.</i> (2003)
Methane	0.2	7	mg/g	0.1	2.2	g	Wang <i>et al.</i> (2003)
Ethane	0.175	0.6	mg/g	0.1	0.2	g	Wang <i>et al.</i> (2003)
Ethene	1.5	12	mg/g	0.5	3.8	g	Wang <i>et al.</i> (2003)
Propene	0.4	1.5	mg/g	0.1	0.5	g	Wang <i>et al.</i> (2003)
Benzene	1.23E-02	4.78E-02	µg/g ^b	3.86E-03	1.50E-02	mg	US EPA (1992, Ch. 2, sect. 2.5)
Toluene	3.30E-03	4.60E-03	µg/g	1.04E-03	1.44E-03	mg	US EPA (1992, Ch. 2, sect. 2.5)
Ethylbenzene	6.00E-04	1.20E-03	µg/g	1.88E-04	3.77E-04	mg	US EPA (1992, Ch. 2, sect. 2.5)
1-Hexene	1.00E-03	4.30E-03	µg/g	3.14E-04	1.35E-03	mg	US EPA (1992, Ch. 2, sect. 2.5)
TSP (soot)	8	36	mg/g	2.5	11.3	g	Shemwell and Levendis (2000); Wang <i>et al.</i> (2003)

ND = not detected

¹ Mass emissions were calculated using the minimum and maximum emission factors for each chemical compound and assuming a sheet of PE with a mass of 314 g is burned.

^a Milligrams of emission per gram of polyethylene.

^b Micrograms of emission per gram of polyethylene.

Emissions and emission factors of various organic compounds, including PAHs, arising from the burning of agricultural plastics are available from a range of sources such as US EPA (1992) (<http://www.epa.gov/ttnchie1/ap42/ch02/final/c02s05.pdf>), Lemieux *et al.* (2004) and Enviser Consulting (2013), while Estrellan and Lino (2010) provide a recent general review of toxic emissions from open burning, including of agricultural plastics. The results of these and other studies indicate a high dependence on waste composition and burning conditions, as well differences in experimental and sampling methods (Lemieux 1997; Gullett *et al.* 2001; Solorzano-Ochoa *et al.* 2012).

The emission of polychlorinated dibenzo-dioxins and furans (PCDD/Fs) from agricultural plastics is a key concern in some studies, for example in Ontario PCDD/Fs are considered tier 1 pollutants by the Canada-Ontario Agreement Respecting the Great Lakes Basin Ecosystem, and all sources of the pollutant, including agricultural burning of plastics, are required to be eliminated (Sonnevera International Corporation 2011). An emission factor of 0.067 toxic equivalents (TEQ) per tonne of plastic burned is given in this study.

Gullett *et al.* (2012) also measured PCDD/Fs, PAHs, and particulate matter in emissions from simulated open burning of used agrichemical containers made from HDPE. PCDD/DF emission factors range from 0.1 to 24 ng TEQ/kg C burned, with a mean and median of 4.9 and 1.9 ng TEQ/kg C burned. While chemical residues of the pesticide 2,4-D appeared to contribute to increased PCDD/Fs emissions there was no difference in PCDD/Fs emissions for atrazine-contaminated containers – which had similar or lower levels than clean HDPE containers. Similarly, PAH emissions from combustion of the container with 2,4-D residues were higher with no difference between the rinsed

2,4-D containers and atrazine-contaminated containers. The clean HDPE containers had the highest emissions of PAHs although this was speculated to be an artefact of the combustion temperature of these containers, which differed slightly from that for the pesticide-contaminated containers. Particulate matter emissions were not affected by the pesticides (Gullett *et al.* 2012).

2.2.2 Burial

Only qualitative data on the impact of burial on plastics is available. Plastics are typically considered to be relatively inert with minimal degradation, although this is dependent on the plastic (Kyrikou and Briassoulis 2007). When some plastic degrades, it becomes brittle and breaks into smaller pieces. These “microplastic” pieces can easily be ingested by wildlife, which can lead to an artificial sensation of fullness and resulting starvation of the wildlife (Thompson *et al.* 2009). Further, plastics have also been shown to adsorb organic contaminants in the marine environment resulting in increased exposure for wildlife that consume the plastic (Teuten *et al.* 2009). A similar phenomenon has not been reported in the terrestrial environment but is plausible.

Plastics also contain many chemicals that are added to polymers to improve their performance and usability. These include plasticisers (phthalates), additives (organotin compounds and nonylphenols and monomers (Bisphenol A)), which may leach out of plastics under acidic conditions, resulting in discharge of plastic-derived contaminants into the environment (Teuten *et al.* 2009).

Any hazardous chemical residues remaining in agrichemical containers may pose an additional hazard (see Hazardous substances) through movement into the soil.

2.2.3 Bulk storage

Degradation of bulk-stored plastics may give rise to exposure of wildlife to microplastic pieces, particularly if plastics are stored in the open. Further, degraded plastic may also move to waterways through surface runoff, resulting in further contamination. If storage of plastics results in standing water, then it could be a breeding ground for mosquitos and other insects. Finally, bulk storage of plastics may also be a fire hazard, should a fire start.

2.3 Wood, including treated timber

2.3.1 Open burning

Burning of wood, treated or untreated, gives rise to gaseous and particulate matter emissions, and emissions of PAHs and PCDD/Fs (Lavric *et al.* 2004), although few studies have developed emission factors for open burning situations. Burning of treated timber will pose greater risk than burning of untreated timber due to the additional chemicals present in treated timber. A variety of preservatives may be used to treat timber, although copper-chrome-arsenate (CCA) is the only preservative approved for uses of timber and is the preferred most cost effective preservative (<http://www.nzwood.co.nz/learning-centre/treatments-wood-preservation/>) for the H3.2 to H6 hazard classes. Thus, it is anticipated to be the primary treated timber used on farms. Creosote may also be used for the treatment of timber used for external purposes, and may be used on farms, although its typical use is for railway sleepers and electricity transmission poles (<http://www.pine.net.nz/resources/faqs/userguide/7%20Preservation.pdf>).

Pentachlorophenol (PCP) was historically used to treat timber in New Zealand, although its usage ceased in 1988 (MfE 1998, p. 11), and PCP-treated timber is not anticipated to be a significant component of current NNRW.

The primary contaminant of concern in the burning of CCA-treated timber is the arsenic. In simulated open burning, Wasson *et al.* (2005) observed that between 11% and 14% of arsenic in the wood was emitted to the air, while over 99% of the chromium and copper remained in the bottom ash. The concentrations of PCDD/Fs were also determined in the study as copper is a known catalyst for PCDD/F formation, and chlorine, which is present in wood in notable concentrations, is essential for PCDD/F formation. The emission factors determined from the study, and the concentrations present in residual ash are shown in Table 2-3. With the exception of the PCDD/Fs emission factor, these emission factors are also used in a recent review of emission factors by the UK Environment Agency to enable assessment of the impact from “incident” fires (EA 2009).

Table 2-3: Summary of emission factors from open burning of copper-chrome-arsenate-treated wood (Wasson *et al.* 2005)

Contaminant	Emission factor (mg/kg wood)	Ash (mg/kg)
Arsenic	188–237	84 260
Chromium	8.4–14.9	158 740
Copper	8.7–13.4	91 620
PCDD/Fs	1.4–2.4 ng TEQ/kg wood	0.07 ng TEQ/kg wood

The concentrations of PCDD/Fs determined by Wasson *et al.* (2005) are markedly less than that (78 ng TEQ/kg treated timber) reported by Tame *et al.* (2007). Tame *et al.* (2007) also provide a good summary of PCDD/F formation from woods, including the influence of preservatives such as CCA and a newer preservative, copper-boron-azole, on PCDD/F formation.

In addition to emissions to air, the residual ash from burning treated timber also contains elevated concentrations of copper, chromium and arsenic that may also leach to underlying soil (Harden *et al.* 2013).

Creosote is a refined coal-tar oil that predominantly comprises a range of polycyclic aromatic hydrocarbons (PAH), and lesser amounts of tar acids (phenolic compounds) and bases. The burning of creosote-treated timber will primarily add to the PAH loading of the emitted particulates and residual ash (Keystone Environmental Resources 1991).

2.3.2 Burial

The environmental hazards associated with burial of treated timber arise from the leaching of chromium, copper, arsenic or components of creosote (e.g. Becker *et al.* 2001; US EPA 2003; Mercer and Frostick 2012). Many factors can affect the amount of leaching that occurs from treated wood into the soil, including how long the wood has been exposed to the environment, the size and type of wood that was treated, whether the wood is coated with a sealant, water movement, and the type of soil. The chemicals that leach from CCA-treated wood can accumulate in soils near the wood, but under certain conditions the chemicals can travel farther. In general, CCA chemicals are least mobile in organic soils, slightly more mobile in clay soils, and most mobile in sandy soils or water (US EPA 2003). Arsenic typically leaches to the greatest extent (Mercer and Frostick 2012; Harden *et al.* 2013). In creosote-treated wood, nitrogen-containing compounds were observed to leach to a greater extent than PAHs (Becker *et al.* 2001).

Burial or soil incorporation of wood chips from untreated wood, such as animal bedding, will have a beneficial influence due to the high carbon-to-nitrogen ratio of the wood, which can act to immobilise any excess nitrogen present from animal manure or urine.

2.3.3 Bulk storage

Where treated timber is stored in the open, the hazards are largely the same as that associated with burial, notably leaching of copper, chromium, arsenic, PAH or nitrogen compounds into soils.

2.4 Tyres

2.4.1 Open burning

A range of volatile, semi-volatile and metal contaminants are released from the simulated open burning of shredded tyres and tyre chunks (Lemieux and Ryan 1993, Shakya *et al.* 2008). Benzene was the dominant volatile contaminant produced (around 2200 mg/kg of tyre) with total estimated emissions of semi-volatile organics ranged from 10 to 50 g/kg of tyre material burnt. Alkyl-substituted mono- and polyaromatic hydrocarbons (PAH) were the predominant emission products identified. Airborne metals emissions were also measured in particulate matter and very low amounts (close to method detection limits) of lead were detected and higher levels of zinc.

Emission factors for metals, PAHs and organic compounds for burning tyres are available in US EPA (1992), US EPA (1997) and Lemieux and Ryan (1993).

Monocyclic aromatic hydrocarbons and PAH are also released in the smoke plume along with other combustible products (Lemieux and Ryan 1993). PAHs produced include several that are known to be carcinogenic. Benzo-a-pyrene levels are of particular concern (US EPA 1997). The emissions released from open burning will depend on the burn conditions: when tyres burn in the smouldering stage, greater emissions of toxic combustion products are produced compared with when tyre fires burn freely, producing fewer products of combustion.

In addition to air emissions, burning of tyres will produce an oily residue as a result of pyrolysis of the rubber that can contaminate soil and water, with one estimate that burning one passenger tyre will generate 2 gallons of oil (Environmental Engineering and Contracting 2002). Pyrolysis of tyres, and the subsequent production of oil, is the key motivation for energy recovery using tyres (e.g. Williams *et al.* 2001).

2.4.2 Burial

Tyres are considered to be relatively inert and there are potentially limited effects associated with burial of tyres. However, an interesting phenomenon has been observed for tyres disposed to landfill in that the tyres can move to the surface and pierce covers (Jang *et al.* 1998). There are no reports of this occurring outside of landfills although it is plausible. Limited leaching of contaminants occurs from whole tyres, although leaching of zinc from shredded or crumbed tyres may occur (Rhodes *et al.* 2012).

2.4.3 Bulk storage

The bulk storage of tyres, as is common on many farms, may provide breeding grounds for insects, rodents and other animals and some leaching of contaminants from the tyre to the soil, which may increase if tyres are damaged. However, perhaps the biggest hazard associated with bulk storage of tyres is the fire risk, primarily due to the difficulty of extinguishing a tyre fire. A whole discarded tyre presents a void space of 75%, which makes it difficult to either quench the fire with water or cut off the oxygen supply (Jang *et al.* 1998). Tyres have a high self-ignition point, and while some self-heating may occur – particularly if tyres are damaged, external sources of ignition are predominantly the cause of tyre fires. The risk of ignition in practical circumstances may be raised by contamination of the tyres (which may allow biological heating in damp conditions) or by the rusting of exposed wires (which also generates heat). As described for open burning, uncontrolled open-air burning of tyres releases pyrolytic oils and other compounds into the soil and groundwater as well as large plumes of black smoke and other contaminants into the air. In addition to this, water used to extinguish tyre fires is likely to become contaminated with tyre compounds and may lead to further contamination of soils and water.

2.5 Hazardous substances

2.5.1 Open burning

The emissions from the burning of hazardous wastes will depend on the individual substance, particularly for agrichemical wastes. Waste oil or diesel may be used to ignite fires and volatile

organics, particularly benzene, and PAHs will be the primary emissions. Lemieux *et al.* (2004) provide emission factors for the open burning of fuel and crude oil.

2.5.2 Burial

The hazards associated with burial or disposal to land of hazardous waste are very much dependent on the individual substance and where they are buried or disposed of. For example, oil may be used as a dust suppressant on gravel roads. While the impact on soil biota would be negligible due to their general absence in roads, road runoff that contains a higher PAH and metal loading may pose a greater risk to soil areas or streams receiving that runoff. Woodard-Clyde (2000) provides an assessment of the risks associated with the use of oil as a dust suppressant in a New Zealand context.

2.5.3 Bulk store

Hazards arising from the bulk storage of waste hazardous substances will depend on how well they are stored. If the containers they are stored in leak or corrode, then the substances can leak into the soil, and eventually into the groundwater. Spillage may also occur if substances are being transferred from a smaller container to a larger container.

2.6 Household waste

Domestic wastes comprise a mix of waste materials that have been used for household purposes. This may include plastics, paper and food scraps.

2.6.1 Open burning

There are numerous studies on the contribution of the open burning of domestic waste to atmospheric burden of persistent PCDD/Fs, PAHs and organic carbon bound into particulate matter (e.g. Lemieux *et al.* 2000; Wevers *et al.* 2004; Nakao *et al.* 2006), and more recently polybrominated dibenzo-dioxins and furans arising from combustion of polybrominated flame retardants (Gullett *et al.* 2010).

The focus for a number of studies is the emission of dioxins and furans, partly because open burning of household waste is estimated to comprise up to 28% of the national release of dioxins and furans in developing countries (Solorzano-Ochoa *et al.* 2012). Such studies have typically been undertaken using an emission factor of 300 ug TEQ/kg waste to air, and 600 ug TEQ/kg to land. A number of studies have also recently been undertaken to assess the relevance of the emission factors in the UNEP toolkit (UNEP 2005) for different waste streams and found this tends to overestimate air emissions (Solorzano-Ochoa *et al.* 2012). Lemieux (1997) and Lemieux *et al.* (2000) examined PCDD/F emissions from backyard burning of household waste in barrels in the United States. Emissions of total PCDD/Fs ranged between 0.0046 and 0.48 mg/kg of waste burned, with backyard burning emitting more PCDD/Fs on a mass of refuse burned basis than various types of municipal waste combustors (MWCs). The higher emissions of PCDD/Fs from household barrels was attributed (Lemieux *et al.* 2003) to generally poorer combustion conditions compared with MWCs, including low combustion conditions and oxygen-starved conditions.

Gullett *et al.* (2001) found that dioxin emissions varied considerably ranging from 10 ng TEQ/kg to over 6,000 ng TEQ/kg of waste, with emissions dependent on both waste composition and burn conditions. Shibamoto *et al.* (2007) examined the formation of PCDD/Fs in a range of wastes and found that total PCDFs comprised 70–90% of the total dioxin formed. The amount of total PCDFs formed ranged from 0.78 ng/g (newspaper) to 8,490 ng/g (PVC burned in high CO concentration) while the amount of total PCDDs formed ranged from 0.02 ng/g (newspaper) to 430 ng/g (PVC). Samples that did not contain chlorine or were not combusted with chlorides exhibited low TEQ values. In contrast, samples with high chlorine content, such as PVC (51.3%), gave high TEQ values.

2.6.2 Burial

The hazards associated with the burial of household waste depend on the nature of the wastes buried. Where organic wastes are present, decomposition of these materials may result in acidic leachate and increased leaching of contaminants. In addition to the chemical hazard posed by the waste, a physical hazard may also exist with the area being uninhabitable to soil biota if the waste layer is thick enough.

2.6.3 Bulk store

The bulk storage of household wastes will depend on how it is stored, and whether there is any potential for leaching of contaminants if the waste is open to the elements. Bulk storage may also pose a fire risk.

2.7 White goods/electronics

There is limited information on what wastes are included in this category in GHD (2013) – other than TVs and fluorescent bulbs – although it may also include obsolete whiteware such as fridges, freezers, cookers/ovens, printers and even photocopiers (I. Hepburn, Environment Canterbury, pers. comm.).

2.7.1 Open burning

In general, wastes in this category are not easily burned thus hazards associated with emissions from open burning where these wastes are present are mainly associated with any paints on the surface or plastic coverings, or the release of any hazardous substances through breakage of the white goods (e.g. foam insulation, refrigerants).

2.7.2 Burial

Hazards arising from the burial of white goods and electronics arise from the degradation of the white goods and the presence of any hazardous substances present in the particular white goods, for example refrigerant fluid. Mercury, cadmium, lead or brominated flame retardants may be present in light bulbs and electronic goods, and may leach into the soil.

2.7.3 Bulk storage

The hazards associated with bulk storage of white goods and electronics are similar to those of burial if the wastes are stored on open ground and are exposed to the elements, with leaching of hazardous contaminants to the ground the primary concern. Surface runoff of contaminants may be greater for bulk-stored items if stored in the open.

2.8 Cardboard and paper

2.8.1 Open burning

Paper and cardboard are readily combustible and will result in the emissions of particulates, and various volatile and semi-volatile compounds such as PAHs and dioxins, depending on the burn conditions – incomplete combustion will give rise to higher concentrations of organic contaminants.

2.8.2 Burial

Burial of paper and cardboard is unlikely to occur in bulk as these materials are readily combustible. If they are buried, they will readily decompose and have minimal environmental effects.

2.8.3 Bulk storage

Bulk storage of paper or cardboard may present a fire hazard as these materials are readily combustible.

2.9 Construction and demolition waste

Construction and demolition waste may include wood, although the hazards associated with treated and untreated wood has been discussed under section 3.3. Construction and demolition waste in GHD (2013) appears primarily to refer to building rubble, as opposed to additional materials. As noted in GHD (2013), the greatest potential hazard is the presence of asbestos in any building materials that are removed. The greatest risk arises where these materials are disturbed, allowing for the release of fibres that may be inhaled by people. This may include the storage of asbestos-containing materials,

for example, if stored in the open, or if these materials are disposed of into fires. Asbestos will not burn, thus any materials placed in the fire will remain in the fire residue and may allow for the release of fibres during and after a fire.

Where construction and demolition wastes are buried, the primary impact of these materials will be physical discontinuity in the soil, which may be uninhabitable for soil biota if present in a thick enough layer.

2.10 Metals and glass

Metals and glass wastes are largely inert materials, with limited potential for emissions to air and leaching to soil. The primary impact of these materials if buried will be physical discontinuity in the soil, which may be uninhabitable for soil biota if present in a thick enough layer. A further hazard may be associated with any residues of hazardous substances, such as animal welfare substances, remaining in glass vials. Where metal scrap is derived from old vehicles, hazards may arise from the release of vehicle fluids, such as brake fluids and oils, to the environment.

2.11 Summary

There is a variable amount of literature available on the hazards posed by the different waste streams, and more literature is available to provide a greater range of likely emission factors under different burn conditions, or burial conditions. However, the key factors influencing open-burning emissions are the composition of the waste stream and the burn conditions experienced in New Zealand, which will be largely unknown. Similarly, the composition of buried waste will also influence hazards associated with burial, for example, burial with organic materials such as animal carcasses or vegetation will likely result in the production of acidic leachate that in turn will result in greater leaching of any co-occurring metal contaminants. Table 2-4 provides a summary of the types of contaminants, and factors influencing their emission, for the different waste streams. As can be seen, many of the same contaminants are released from different waste streams that are burned, although the amount will differ between different wastes under the same burn conditions.

Table 2-4: Summary of the hazards arising from different methods of disposal of various non-natural rural waste streams

Waste category	Disposal method		
	Open burning ¹	Burial ²	Bulk storage ³
Plastics	Emission of pollutants to air PCDD/Fs, PAHs, particulate matter (PM)	Microplastic chips, leaching of plasticisers	Potential fire risk
Timber – untreated	Air emissions of PAHs, PM	-	-
Copper-chrome-arsenate (CCA)- treated timber	Air emissions of arsenic, PAHs (creosote), PM	Leaching of copper, chromium and arsenic, PAHs	Leaching of copper, chromium and arsenic, PAHs
Tyres	Air emissions of PAHs, metals, PM	PAHs, metals	Fire risk
Hazardous substances	Wide ranging and variable depending on individual substances	Wide ranging and variable depending on individual substances	Wide ranging and variable depending on individual substances
Domestic waste	Air emissions of PAHs, PCDD/Fs, PM	Variable depending on waste	Variable depending on waste
White goods/electronics	Release of any hazardous substances contained within the goods (e.g. refrigerants)	Leakage and leaching of any hazardous fluids or substances contained within the goods	Leakage and leaching of any hazardous fluids contained within the goods
Batteries	Explosion risk, release of acid	Leakage of acid, lead, cadmium	Leakage of acid, lead, cadmium
Cardboard/paper	Emissions of PAHs, particulates	-	Fire risk
Construction and demolition (excluding timber)	Inert (if present, asbestos fibres may be released)	Inert (if present, asbestos fibres may be released during burial of building materials)	Inert (if present, asbestos fibres may be released during movement of building materials for storage)
Glass and metal	Inert	Inert, metal scrap derived from vehicles may release hazardous substances (e.g. oil, brake fluid)	Inert, metal scrap derived from vehicles may release hazardous substances (e.g. oil, brake fluid)

¹Extent of emissions of individual pollutants will be dependent on the composition of waste burned, and burn conditions.

²Hazards relate to exposure of soil biota to contaminants and leaching; influenced by composition of buried waste, in particular if organic waste is present – for example, organic wastes such as animal carcasses will generate acidic leachate and enhance leaching of co-occurring metal contaminants.

³Hazards are similar to that associated with burial of waste if stored in the open although there is a greater potential for surface runoff. A fire hazard exists for some wastes (e.g. tyres, paper/cardboard), and bulk storage may also provide a breeding ground for insects, rodents and other animals if storage is in the open.

3 Assessing environmental risk

In order for an environmental risk to exist there needs to be some form of hazard that has an impact on a receptor via some pathway of exposure (e.g. a person exposed to smoke from the open burning of farm wastes; soil biota exposed to contamination from non-natural farm waste). If these three components are not present, then an environmental risk is not present. Section 3.1 provides a general discussion on potential receptors and exposure pathways associated with the different forms of waste disposal.

The size of the risk will depend on the severity of the hazard and the likelihood and extent of exposure:

$$\text{Risk} = \text{hazard} \times \text{exposure}.$$

The hazards associated with the different waste categories were discussed in the previous section. There are quantitative and qualitative assessment approaches that can be used to assess risk, which depend on the extent of information available and the purpose for which the risk assessment is being undertaken. Section 3.2 provides an overview of alternative approaches to assessing risk.

3.1 Exposure pathways and receptors associated with different waste disposal options

3.1.1 Open burning

For open burning there are potentially multiple exposure pathways leading to an environmental risk, as shown in Figure 3-1. While this figure was developed for assessing the risks associated with open burning of plastics, it is largely applicable to open burning of all waste streams. A point to note is the potential wider impact arising from dioxin emissions, due to the potential bioaccumulation in the food chain – more than 90% of human exposure is through food and mainly meat and dairy products, fish and shellfish. To assess the scale of risk requires knowledge of the location of the burning to determine the relevant exposure routes, and the number of potential receptors (e.g. people, livestock). A critical component in assessing risk is also the conversion of literature information on emissions, typically expressed as emission factors (i.e. mg emissions/kg waste burned), to concentrations in the air and deposition rates. This can be achieved through the use of air dispersion models, which require input of meteorological parameters – which will be dependent on the location and likely the time of year during which burning is considered to take place.

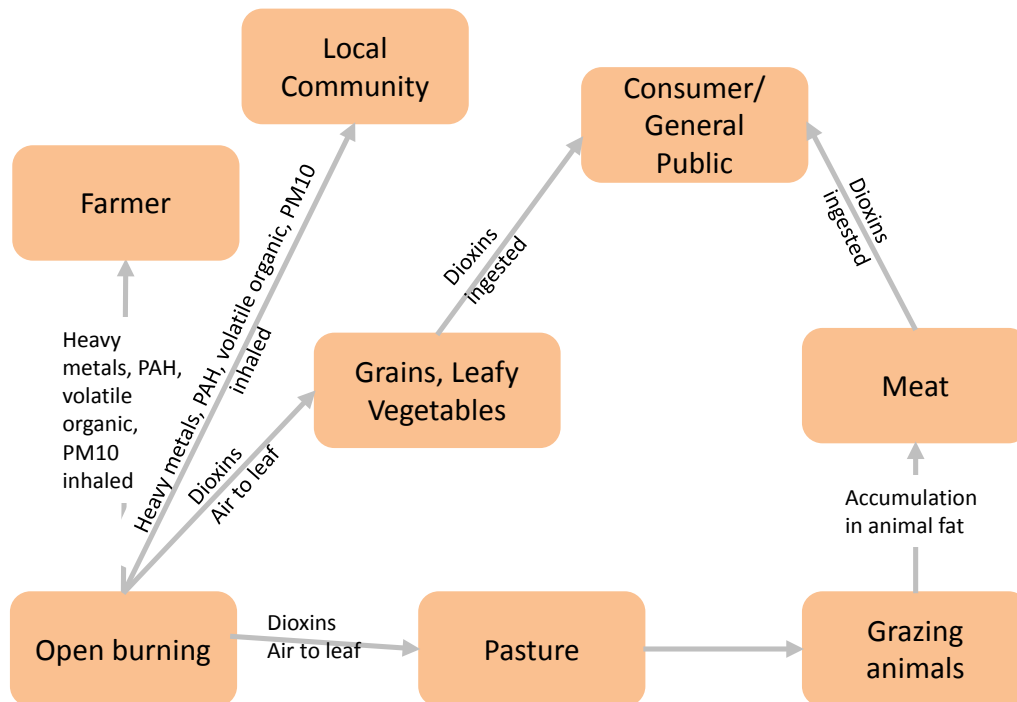


Figure 3-1: Exposure pathways of emissions associated with the open burning of agricultural plastics (adapted from Envise Consulting 2013)

3.1.2 Burial

The environmental risks associated with the burial of waste largely arise from the release of hazardous substances to the surrounding soil, and groundwater if the burial pits intercept groundwater. This may affect biota present in the soil, or may result in leaching to groundwater or to surface water.

3.1.3 Bulk storage

Beyond tyres, no literature on the environmental risks associated with bulk storage of rural wastes was found. The risks associated with bulk storage will depend on how the waste materials are stored. If the wastes are stored in the open, the environmental risks are anticipated to be similar to those for burial, although there is greater potential for contamination of surface waters through surface runoff.

3.2 Environmental risk assessment approaches

3.2.1 Quantitative risk assessment

Quantitative risk assessment (QRA) approaches have been developed over the years to assess the risks associated with environmental contamination, particularly in the context of contaminated land. A good overview of the approaches used to assess human health and environmental risks is provided at http://www.epa.gov/risk_assessment/guidance.htm. In general, QRA requires detailed knowledge about the nature of the hazard, the nature of the receptors (i.e. location-specific information) and typically includes the use of models to assess risk, for example air dispersion models or soil leaching models.

The Risk-Screening Environmental Indicators (RSEI), developed by the United States Environmental Protection Agency (US EPA) to provide community information about the risks associated with industrial emissions, is an example of how the risk associated with open burning of non-natural farm wastes could be assessed using a more quantitative approach. Essentially, this tool correlates information on the amounts of pollutants emitted from different industries and their toxicity, and draws

on air dispersion modelling to calculate air concentration and risks to hypothetical receptors located within a circle with a radius of 49 km surrounding each facility.

The limitation of quantitative risk assessment approaches is that detailed information is required, and the information needs to be site-specific. A lot of resources could be spent on providing a semi-quantitative risk assessment of the different waste streams (e.g. see Woodward-Clyde (2000) for the assessment of risks associated with waste oil). Further, the risks associated with open burning, given the uncertainties inherent in the assessment process and the value of the information provided by that risk assessment, need to be considered in the context of the ultimate aim of minimising the environmental risks posed by the different waste streams. This ultimately depends on the alternative options available for handling the end of life. It may be determined as part of the assessment. A brief description of the considerations for undertaking quantitative risk assessment for open burning and burial is given below.

Open burning

To assess the effect arising from the emissions from open burning requires conversion of the emissions to concentrations in air, determination of the number of people likely exposed and the duration of exposure, or determination of the deposition and uptake of contaminants into the food chain (plants and livestock). Concentrations in air will be dependent on the size of the fire and the dispersion of those emissions, which will depend on the meteorological conditions prevailing at the time. Potentially, assumptions could be made about the size of the fire and meteorological conditions on the basis of the typical size of fires and the timing of when open burning typically occurs. This information could be used in air quality dispersion models to indicate the likely concentrations of contaminants and the extent of and deposition of contaminants from the associated plume.

To determine risk to human health, a further step is then required to determine the number of people likely exposed, and the duration of exposure. Assumptions about the latter could be based on the standardised scenario/s used to generate the air quality data. Similarly, assumptions could be made about the number of people likely to be exposed. Alternatively, if the locations of farm fire pits are known, these could be plotted on a map, and an estimation made of the number of people likely to be present in a given radius (e.g. number of houses, proximity to farm buildings). The final step in assessing the risk is relating the exposure of the persons to some measure of effect. This could be a simple comparison with air quality guidelines, or more complex determination of the likely effect from health effects information.

Assessment of the risk associated with the deposition and uptake of contaminants into the food chain requires assessing contaminant uptake and transfer up the food chain, which could be achieved using published information.

In all cases, careful consideration and justification of the assumed exposure scenarios used to estimate risk are required.

Burial

To assess the scale of the risks requires knowledge of the hazard and likelihood of effect. The likelihood of effect depends on the scale of the activity, that is, the area affected and the number of farm dumps; the duration of activities; and the nature of the ecosystem. The geographical scale of effects associated with burial will be reasonably discrete in nature for an individual farm dump, although potentially greater if groundwater is affected. Risk to groundwater will be influenced by the depth to groundwater, or whether the pit intercepts groundwater. The nature of the ecosystem is largely unknown unless the locations of the farm pits are known. Even where the locations of farm pits are known the sensitivity of ecosystem is still largely unknown although some qualitative assessment may be able to be made. For example, a sensitive system might be located near to surface waters, or in otherwise undisturbed land. A less sensitive system might be one that has been disturbed, such as through agricultural activities.

The potential for leaching to groundwater could be assessed by using spatial databases such as S-map (<http://smap.landcareresearch.co.nz/home>), which provides a range of soil attribute information including parameters that can be used to assess vulnerability to leaching. This information could be overlain with information on the depth to groundwater to highlight areas that could be at greater risk of

leaching. The location of farm pits would then need to be overlain to provide some assessment of the risk.

Once again, careful consideration and justification of the assumed exposure scenarios used to estimate risk are required.

3.3 Risk-ranking

An alternative risk assessment approach, which could be used to identify higher-priority waste streams, is a risk-ranking process. Such a process was used by Waikato Regional Council to identify the highest priority diffuse-contamination issues in the Waikato Region (Kim *et al.* 2013). The council's approach involved identifying all possible source categories by which substances enter air, soil, groundwater, food, surface waters and sediments, and then assessing those factors using different categories of effect. The factors were chosen to reflect the purpose of the Resource Management Act 1991, which is to maintain the life-supporting capacity of air, soil, water and ecosystems while providing for the needs of existing and future generations. These factors, which were given different weightings (Table 3-1), and then ranked according to risk. Table 3-2 provides an extended description of the factors considered and the scoring system used. The different issues were assessed within environmental compartments (e.g. air, soil, water) and also across environmental compartments where they could affect more than one compartment.

Table 3-1: Summary of factors used to rank diffuse-contamination issues in the Waikato Region (Kim *et al.* 2013)

Factor	Scoring scale	Factor weighting
1. Scale (geographic or based on exposed population)	1 to 5	2
2. Accumulation capacity	1 to 15	1
3. Reversibility	0 to 5	1
4. Human health - potential for chronic harm	0 to 5	3
5. Human health - potential for serious acute harm	0, 2.5, or 5	1
6. Environmental impact	0 to 5	2
7. Impact on animal welfare and agricultural production	0 to 5	2
8. Harm to trade	0 to 5	1
9. Reduction in land use flexibility	0 to 5	1

Table 3-2: Summary of categories used in risk-ranking approach (Kim et al. 2013)

Factor 1. Scale of issue	Factor 2. Accumulation Capacity	Factor 3. Reversibility	Factors 4 and 5. Potential for harm to human health	Factor 6. Environmental impact	Factor 7. Impact on animal welfare and agricultural production	Factor 8. Harm to trade	Factor 9. Harm to land use flexibility
Considers scale in terms of both geographic area and population exposed	<p>1. The substance accumulates in the compartment of interest, or biomagnifies, or (for food) bioaccumulates</p> <p>2. Pooling of the substance occurs</p> <p>3. The most sensitive endpoint of harm caused by the substance represents the accumulation of damage through repeated exposure</p> <p>Total score for Factor 2</p>	Whether the contamination is reversible	<p>Chronic exposure: likelihood of an adverse impact on human health via direct exposure, indirect exposure or food chain (most sensitive endpoint)</p> <p>Acute exposure: potential for significant immediate harm.</p>	Ecological receptors may include soil organisms (microbes, invertebrates, plants), aquatic organisms, wildlife and higher animals, and the functioning of ecosystems as a whole.	Impact on animal welfare and production. This factor has special relevance to the Waikato Region due to the agricultural nature of its regional economy.	Trade impact. Considered as a separate factor to production because several diffuse contamination issues that are of negligible importance to production can nonetheless have a negative impact on trade	Some diffuse contamination issues have the capacity to contaminate soil to a degree that reduces its viable range of uses
1 (small) to 5 (maximum)	<p>0 Not cumulative to 5 strongly cumulative</p> <p>0 Not prospect of pooling; 2.5 some pooling; 5 extensive pooling</p> <p>0 Not cumulative; 2.5 Moderately cumulative; 5 strongly cumulative</p>	<p>0 Readily reversible once source removed; 2.5 partly reversible; 5 not easily reversible</p> <p>0 No impact on general human population to 5 Likely impact on most people</p>	<p>0 (not likely), 2.5 - not known but the possibility exists, or 5 (yes, it has occurred)</p> <p>0 Negligible; 2.5 Possible now or over the longer term; 5 Likely now or over the longer term</p>	<p>0 Not an animal health or production issue; 2.5 Some issues; 5 Likely issue</p>	<p>0 Does not impact future land uses; 2.5 may impact some future land uses; 5 will impact most future land uses</p>	<p>0 No impact possible; 2.5 Moderate impact possible; 5 Eventual impact likely</p>	

Global generation of greenhouse gases was the highest ranked issue in the air compartment and the integrated list, accumulation of cadmium in soils through phosphate fertiliser application was the key issue for soil, while discharge of nitrogen to groundwater ranked as the top priority for groundwater. A total of 132 issues were identified, with some being intentional subsets of other issues while others were inserted for purposes of calibrating the ranking system.

A risk-ranking approach could be used to prioritise different non-natural rural wastes by adapting the categories to ensure that relevant features for NNRW are captured (see section 4.3.1). A risk-ranking would need to be undertaken for each disposal category.

3.4 Risk assessment matrix

A risk matrix (Figure 3-2) is a common way to assess a variety of risks (e.g. SAEPA 2007; MBIE 2013). This typically comprises some measure of severity (e.g. environmental effect) and some measure of the likelihood of this occurring. Similar to the risk-ranking approach, a separate risk matrix would likely need to be developed for each end-of-life disposal method. A score could be in the different cells to assist in ranking. The same criteria used to develop a risk-ranking can be used to develop the categories for likelihood and severity although they would need to be “bundled” to provide the single severity or likelihood measures required to place the waste into an individual cell in the matrix. Measures of severity would consider the extent of exposure of the different receptors identified (e.g. people, soil biota, groundwater). Likelihood of the impact occurring would be a combination of the volume of waste and its intrinsic hazard (i.e. toxicity).

Severity Likelihood	←	→	Higher Lower	→	←
↑	Yellow	Yellow	Red	Red	Red
More Less	Green	Yellow	Yellow	Yellow	Red
↓	Green	Green	Green	Yellow	Yellow
	Green	Green	Green	Green	Yellow

Figure 3-2: Generic risk matrix

3.5 Life cycle analysis approaches

A final approach that is pertinent is that of life-cycle assessment (LCA) or, more specifically, life cycle impact assessment. This approach typically considers a broader range of environmental impacts and notably considers wider resource use issues such as:

- depletion of abiotic resources
- global warming
- stratospheric ozone depletion
- human toxicity
- freshwater and marine aquatic ecotoxicity
- terrestrial ecotoxicity
- photo-oxidant formation
- acidification
- eutrophication.

LCA has most often been used to assess the environmental sustainability of a given product, to assess product stewardships schemes, including in New Zealand (e.g. Garrett and Collins 2009), and to assess waste management options (e.g. Dolan 2002; URS and NZIER 2003; ICF International 2007). A recent study in Canada used LCA to identify the lost opportunity for recycling as a result of open burning or burial of agricultural plastic and paper fibre wastes generated on Ontario farms (Envise Consulting 2013). Specifically, recycling of the approximately 13,500 tonnes of plastic and paper fibre wastes generated annually on Ontario farms was estimated to yield environmental benefits equivalent to saving over 75,000 barrels of oil or avoiding the pollution from 7,919 cars driving on the road for one year.

In New Zealand, Dolan (2002) used LCA to address the management of waste farm plastics from the point at which they are discarded by the farmer. Two types of plastics were considered:

- high density polyethylene (HDPE) chemical containers; and
- low density polyethylene (LDPE) plastic films, such as hay bale and silage wrap

The study found that a programme whereby farmers drop off waste plastics at transfer stations for recycling, into products as a replacement for virgin plastic, would have the least negative effect on the environment, when compared with other scenarios and the situation at the time. This was due to avoided air acidification, human toxicity, greenhouse effect and depletion of non-renewable resources that result from the avoidance of extraction, processing and transport of virgin-plastic raw materials.

On-farm disposal (burning or burial) resulted in greater adverse effects in respect of air acidification, human toxicity and greenhouse gas effects than off-farm management scenarios, while on-farm burning had a greater negative effect than burial.

There are an array of LCA models that can be used (e.g. ICF International 2007; Hauschild *et al.* 2013), although the WISARD (Waste – Integrated Systems Assessment for Recovery and Disposal) model developed in the UK used by Dolan (2002) appears to be no longer available. Current models typically have inbuilt databases or approaches for assessing the impact, and Hauschild *et al.* (2013) compared the approaches used in different models with a view to providing recommendations for the LCA practitioner on the best model to use for specific endpoints. Hauschild *et al.* (2013) provide a good discussion of the ways in which impacts are assessed in the different models although not all models are assessed, including those that have been used in New Zealand (SimaPro, Impact 2000+). For the current purpose the choice of model is less of a consideration than defining the boundaries for assessment, and having a clear understanding of the alternative end-of-life options to be considered. Further, while life cycle assessments may be in-depth and complex, a simplified approach, focusing on the key factors influencing risk, can be used.

3.6 Summary

The approaches that can be used to assess environmental risk are largely dependent on the amount of information available and the scale of the assessment. Detailed quantitative risk assessment approaches are more suited to site-specific assessment although semi-quantitative assessments could be undertaken on a broader scale. Risk-ranking and risk assessment matrices require less data and provide a qualitative assessment of risk that can be useful to inform subsequent decision-making. Life cycle analyses can be complex and require a clear understanding of the system being addressed but provide a good approach to assessing the relative risks associated with different end-of-life disposal options. A summary of the advantages and disadvantages of the methods described above for the assessment of NNRW are shown in Table 3-3.

Table 3-3: Summary of the advantages and disadvantages of different risk assessment approaches

Approach	Advantages	Disadvantages
Quantitative risk assessment (QRA)	Provides a quantitative assessment of the risk. Can be undertaken along a continuum of complexity	Significant data requirements, in particular QRA requires knowledge or assumptions regarding the location of farm waste dumps to assess potential risk (receptors, pathways). A lot of resources can be spent obtaining the information required to develop a robust assessment, particularly at a national level.
Risk ranking	Flexible approach to provide a relative assessment of risk of the different waste streams. Can be simple or more complex, depending on the number of factors used. Less data intensive than quantitative risk assessment approaches. Allows for additional factors (e.g. impact on trade) to be included	A full assessment of the risk requires that all factors that should be considered are considered. Selection or risk factors, or weighting of risk factors, may be subjective.
Risk assessment matrix	Simplified risk-ranking approach.	Same as above although, arguably, there is a loss of information through the “bundling” of factors to provide the two key measures of severity and likelihood.
Life cycle assessment	Evaluates a broader range of environmental impacts. Valuable for providing a comparative assessment of different options.	Can be overly complex. Most suited for the assessment of the relative risk of alternative end-of-life scenarios.

4 Proposed risk assessment approach

4.1 Purpose of risk assessment

In selecting an applicable risk assessment approach, consideration needs to be given as to the intended use of the information and how that can aid decision-making. The objective of the wider project is to assess the risk posed by current waste management practices, and *find solutions to reduce risk and improve waste minimisation and disposal*. The risk assessment phase is intended to help in understanding the risks associated with existing rural waste management methods and to prioritise the waste streams for further work. The value of the information provided by the risk assessment needs to be considered in the context of finding solutions, in particular whether the alternative end-of-life options do actually reduce the overall risk.

4.2 Undertaking the risk assessment

The basis for any good risk assessment is having good base data. For the current project, this includes having a clear understanding of what is captured in each waste category and knowing the total amounts of those wastes, and the amounts of each waste stream disposed of via burning, burial or stored. For the latter, while it is assumed (GHD 2013) that 92% of all NNRW is burned, buried or bulk stored, no further breakdown into individual waste streams disposed of via the different methods is currently available. Although GHD (2013) indicates some waste streams (e.g. batteries and scrap metal) will be fully recycled, the time frame for such recycling is uncertain.

As noted above, to adequately assess risk, a clear understanding is required of what wastes are captured in each waste category. For example, whereas a total 27,000 tonnes of wood waste were estimated to be created and potentially burned or buried each year in Canterbury (GHD 2013), only 6,000 tonnes of this was CCA-treated timber and posed a greater hazard.

The next step in assessing risk requires understanding of the receptors that may be affected, and is more challenging to assess on a national basis. The receptors potentially affected are highly dependent on the environment in which burning, burying and stockpiling is undertaken and requires knowledge, or sufficient information to create representative scenarios that adequately capture the range of disposal options for the different wastes. Such information includes: number of people and livestock, amount and type of crops potentially affected from open burning, proximity to surface watercourses and groundwater resources, nature of ecosystems, and the potential for leaching (influenced by soil type and including the likelihood of co-disposal with organic wastes, which can generate acidic leachate and enhance leaching of metals). Models, such as air dispersion models or soil leaching models, are required for quantitatively assessing risk. In addition, there will be considerable uncertainty associated with the assessment of the risks of open burning, as emissions depend on burn conditions, which are likely to be highly variable and largely unknown, as well as the composition of the waste burned. Considerable resources could be used in obtaining information to enable a quantitative or even semi-quantitative assessment of the risk, and clarity is required around the value of obtaining detailed information in relation to the value of the information provided by risk assessment. For example, if the purpose of assessment is solely to establish the risk associated with given disposal practices then it may be appropriate to use the resources to determine the risk, recognising that there are inherent uncertainties in the assessment. If, however, the purpose of assessment is to inform solutions to reduce risk, then there may be more value in expending those resources on assessing the relative risks associated with alternative end-of-life options.

Given the considerable resources that could be spent in providing what at best would be a semi-quantitative risk assessment of current disposal practices, it is strongly recommended that assessing the risk associated with current waste management practices is conducted alongside considerations of the risks posed by alternative end-of-life options.

4.3 Recommended methodology

As highlighted above, there are considerable challenges in undertaking risk assessment of NNRW due both to limited knowledge of the “system” (i.e. the location and surrounding environment of the farm dumps) and the inherent variability in the hazards posed by the different wastes, in particular for the

open burning of wastes. As such, it is recommended that a two-step process is undertaken to assess risks and find solutions.

For prioritising the waste streams for further investigation, it is recommended that a simple risk-ranking exercise is undertaken. To provide a more extensive assessment and focus on assisting with finding solutions it is recommended that a life-cycle approach is adopted, although it is recognised that a more quantitative assessment of high-priority wastes may be required to justify the implementation of alternative end-of-life options.

As noted above, in order to undertake a good risk assessment, knowledge of the following is required: what items are captured in each waste category, the total amounts of those wastes, and the amount of each waste stream disposed of via burning, burial or bulk storage.

4.3.1 Risk-ranking

To undertake the risk-ranking, a series of assessment factors are required to form the basis for ranking. These should include:

- Type and volume of waste being considered
- Hazard to air (based on emission factors and toxicity of different contaminants, these could be done for individual groups of contaminants, e.g. PAHs, dioxins, or an overall measure)
- Hazard to soil (based on toxicity of different contaminants, these could be done for individual groups of contaminants, e.g. PAHs, dioxins, or an overall measure)
- Hazard to groundwater and surface water from leaching (based on mobility of contaminants); this should include consideration of whether leaching is to drinking water supplies, with greater weighting given for leaching to these supplies.
- Geographical scale of impact (greater for open burning and leaching, less for impact of burial and bulk storage on the soil compartment)
- Receptor scale of impact (effectively a measure of the number of likely receptors, most relevant for open burning; could also consider global warming potential)
- Sensitivity of the ecosystem (relevant for burial, bulk storage)
- Potential for accumulation in different environmental compartments

A scoring system, such as that shown in Table 3-1, can be used to then provide the ranking. Providing a robust ranking requires that the potential effects arising from the disposal of the different wastes by the various methods are identified and captured in the risk-ranking process. After risk-ranking it may be decided that a more quantitative approach is needed for the assessment of the high-priority wastes to justify the implementation of alternative end-of-life options. This could be undertaken alongside further assessment of the risks using life cycle analysis.

4.3.2 Life cycle analysis

To assist in evaluating potential solutions for reducing risk associated with end-of-life disposal options for the different waste streams, a life-cycle-analysis approach has been recommended. In undertaking this analysis there needs to be a clear definition of the system boundaries to be considered, and also of the different end-of-life options being considered. The clear definition of the system boundaries includes the aspects that are to be measured (e.g. emissions to air from waste transportation vehicles), as well as how much of the life cycle of a given waste is considered; for example, it is likely that the point at which a material becomes a waste is the most relevant starting point. The second critical point in being able to effectively use LCA is having a clear understanding of what alternative end-of-life options (e.g. recycling and use of the recycled material; European Commission 2007) exist.

An important point to remember is that while LCA will provide a quantitative result, the uncertainties inherent in the data used to generate the LCA, including estimates of effect, should not be forgotten. Thus, the relative impacts of the different end-of-life options are the most important point for comparison.

5 Conclusions and recommendations

There is a variable amount of literature available on the hazards posed by the different waste streams, and more literature is available to provide a greater range of likely emission factors under different burn conditions, or burial conditions in particular. Only a limited amount of information is available for the bulk storage of NNRW although the hazards can largely be surmised from the literature and common sense. This report has provided direction to some of that literature and more is available upon further searching. However, in assessing the risks associated with NNRW the key challenge is having a robust understanding of the overall system for which the risks are being assessed, to determine which literature is most relevant, or what risks are relevant to consider. This includes a clear understanding of what is captured in each waste category, the amounts of those wastes and how much of each waste is disposed of via burning or burying or is stored (and how and for how long). Following this is the need for a good understanding of the environment in which the wastes are being burned, buried or stored, and the receptors that are likely to be affected by waste disposal. This includes: number of people and livestock, type of crops potentially affected from open burning, proximity to watercourses and groundwater resources, and nature of ecosystems (more or less sensitive). Even with this understanding there remain some inherent uncertainty and variability in likely risk; for example, the key factors influencing open-burning emissions are the composition of the waste stream and the burn conditions, which are largely unknown and highly variable. Similarly, the composition of buried waste will also influence hazards associated with burial, for example, burial with organic materials such as animal carcasses or vegetation will likely result in the production of acidic leachate that in turn will result in greater leaching of any co-occurring metal contaminants.

The approaches that can be used to assess environmental risk are largely dependent on the amount of information available, and the scale of assessment. Detailed quantitative risk assessment approaches are more suited to site-specific assessment although semi-quantitative assessments could be undertaken on a broader scale. Models such as air dispersion models or leaching models are required to provide the link between emissions and effect on receptors for quantitative or semi-quantitative assessment. Risk-ranking and risk assessment matrices require less data and provide a qualitative assessment of risk that can be useful to inform subsequent decision-making. Life cycle analyses can be complex and consider a broader range of environmental impacts including resource use. However, LCA approaches can be streamlined and with a good understanding of the system being assessed provide a good approach to assess the relative risks associated with different end-of-life disposal options.

In selecting an applicable risk assessment approach, consideration also needs to be given to the intended use of the information and how that can aid decision-making. The objective of the wider project is to assess the risk posed by current waste management practices, and *find solutions to reduce risk and improve waste minimisation and disposal*. Thus, the value of the information provided by the risk assessment needs to be considered in the context of finding solutions, in particular whether alternate end-of-life options do actually reduce the overall risk.

There are considerable challenges in undertaking risk assessment of NNRW due both to limited knowledge of the “system” (i.e. the location and surrounding environment of the farm dumps) and the inherent variability in the hazards posed by the different wastes, in particular for the open burning of wastes. Given the considerable resources that could be spent in providing what at best would be a semi-quantitative risk assessment of current disposal practices, it is strongly recommended that assessing the risk associated with current waste management practices is conducted alongside considerations of the risks posed by alternative end-of-life options. As such, a two-step process has been proposed to assess risks and find solutions:

- For prioritising the waste streams for further investigation, it is recommended that a simple risk-ranking exercise is undertaken. This inherently requires that the potential effects associated with the different wastes are identified.
- To provide a more extensive assessment and focus on assist with finding solutions it is recommended that a life-cycle approach is adopted. However, it is recognised that a more quantitative assessment of high priority wastes may be required to justify the implementation of alternative end-of-life options. To undertake LCA or quantitative assessment of the risks

requires a clear definition of the system and boundaries for assessment, with quantitative assessment requiring more detailed information.

5.1 Recommendations for next steps

The following steps are recommended as those that need to be next undertaken to assess the risk associated with disposal of NNRW:

- Clearly identify what is captured in each waste category (Table 2-1 provides a starting point); waste categories need to be considered from the perspective of the hazard posed, although from the perspective of considering alternative end-of-life options, more detail on the product (e.g. type of plastic) or use of other materials (e.g. packaging) may be required.
- Establish the best possible estimates of the amount of each waste disposed of via burning, burial or storage. This may be achieved by a reworking of the data in GHD (2013) or additional data may be needed. If the assessment is on a national scale, these estimates need to be developed up to a national scale, and appropriate rationale provided.
- Further develop the risk-ranking system proposed in section 4.3.1 to ensure that all relevant factors are captured and that an appropriate weighting system is developed – this is likely to be best achieved by consensus from a number of different experts.
- Identify all alternative end-of-life options / waste minimisation strategies for the different waste streams for further assessment.
- Develop generalised scenarios that encompass the likely range of different environments and receptors affected by the different disposal methods of NNRW or that may influence alternative end-of-life options (e.g. proximity to watercourses, distance from nearest recycling point etc.). These scenarios provide the system for assessment using life cycle assessment or form the basis of any quantitative risk assessment. More information would be required to quantitate risk.
- Use the generalised scenarios and information on alternative end-of-life options to undertake a streamlined life cycle analysis to determine the best solutions to minimise the risks associated with NNRW.

6 References

- Becker L, Matuschek G, Lenoir D, Kettrup A 2001. Leaching behaviour of wood treated with creosote. *Chemosphere* 42: 301-308.
- Dolan L 2002. Life cycle assessment of management options for waste farm plastics. Wasteminz conference.
- Environment Agency 2009. Review of emission factors for incident fires. Science report: SC060037/SR3.
- Environmental Engineering and Contracting 2002 Tire pile fires prevention, response, remediation. Report for
- Enviser Consulting 2013. Ontario Agricultural Waste Management Study – Phase II: Risk assessment, collection, processing and stewardship options. 97 p. Available at http://www.cleanfarms.ca/sites/default/files/studies/OntarioStudyPhaseII_FINAL_20130721.pdf
- Estrellan CR, Lino F 2010. Toxic emissions from open burning. *Chemosphere* 80: 193–207.
- European Commission 2007. LIFE and waste recycling: Innovative waste management options in Europe.
- Garrett P, Collins M 2009. Life cycle assessment of product stewardship options for mercury-containing lamps in New Zealand. Published by Environmental Resources Management (ERM) for the Ministry for the Environment.
- GHD 2012. Canterbury non natural rural waste regional assessment and guidance note development. Report for Environment Canterbury (R.12/74). Available at <http://ecan.govt.nz/publications/Plans/CanterburyNonNaturalRuralWasteRegionalAssessment.pdf>
- GHD 2013. Non-natural rural wastes: Site survey data analysis. Report prepared for Environment Canterbury (R13/52). Available at <http://ecan.govt.nz/publications/Reports/non-rural-wastes-survey-r13-52.pdf>
- Gullett BK, Lemieux PM, Lutes CC, Winterrowd CK, Winters DL 2001. PCDD/F emissions from uncontrolled domestic waste burning. *Chemosphere* 43(4–7): 721–725.
- Gullett BK, Wyrzykowska B, Grandesso E, Touati A, Tabor DG, Ochoa GS 2010. PCDD/F, PBDD/F, and PBDE emissions from open burning of a residential waste dump. *Environmental Science and Technology* 44: 394–399.
- Gullett BK, Tabor D, Touati A, Kasi J, Fitz N 2012. Emissions from open burning of used agricultural pesticide containers. *Journal of Hazardous Materials* 221–222: 236–241.
- Harden JD, Pitt RE 2013. Effects of Soil Components and Liming Effect of CCA-Wood Ash upon Leaching of Cu, Cr, and As from CCA-Wood Ash in Ultisol Soil. *Soil & Sediment Contamination* 22:39-55.
- Hauschild MZ, Goedkoop M, Guinee J, Heijungs R, Huijbregts M, Joliet O, Margni M, De Schryver A, Humbert S, Laurent A, Sala S, Pant R 2013. Identifying best existing practice for characterization modeling in life cycle impact assessment. *International Journal of Life Cycle Assessment*. 18: 683–697.
- Hepburn I, Keeling C 2013. Non-natural rural wastes – site survey data analysis: summary report. Environment Canterbury. Available at <http://ecan.govt.nz/publications/Reports/NNRW-survey-summary-report-2013.pdf>.
- ICF International 2007. Review of life cycle assessment (LCA) tools for municipal solid waste (MSW) management. Report prepared for Environment Canada. See <http://ec.gc.ca/gdd-mw/default.asp?lang=En&n=407AB1EC-1>
- Jang J-W, Yoo T-S, Oh J-H, Iwasaki I 1998. Discarded tire recycling practices in the United States Japan and Korea. *Resources, Conservation and Recycling* 22: 1–14.

- Keystone Environmental Resources 1991. The impact of open burning of railways sleepers in British Columbia. Report for the British Columbia Ministry for the Environment. Available at: <http://www.for.gov.bc.ca/hfd/library/documents/bib65529.pdf>.
- Kim ND, Taylor MD, Rumsby A 2013. Inventory and prioritisation of diffuse contamination issues in the Waikato Region. Waikato Regional Council Technical Report 2013/04. Hamilton, Waikato Regional Council.
- Kyrikou I, Briassoulis D 2007. Biodegradation of agricultural plastic films: A critical review. *Journal of Polymers and the Environment* 15: 125–150.
- Lavric ED, Konnov AA, De Ruyck J 2004. Dioxin levels in wood combustion—a review. *Biomass and Bioenergy* 26: 115–145.
- Lemieux PM 1997. Evaluation of emissions from the open burning of household waste in barrels. EPA-600/R-97-134a (vol 1).
- Lemieux PM, Ryan JV 1993. Characterisation of air pollutants emitted from a simulated scrap tire fire. *Air & Waste* 43(8): 1106–1115.
- Lemieux PM, Lutes CC, Abbott JA, Aldous KM 2000. Emissions of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans from the open burning of household waste in barrels. *Environmental Science and Technology* 34: 377–384.
- Lemieux PM, Gullett BK, Lutes CC, Winterrowd CK, Winters DL 2003. Variables affecting emissions of PCDDs/Fs from uncontrolled combustion of household waste in barrels. *AWMA Journal* 53: 523–31.
- Lemieux PM, Lutes CC, Santoianni DA 2004. Emissions of organic air toxics from open burning: a comprehensive review. *Progress in Energy and Combustion Science* 30: 1–32.
- MBIE 2013. External moisture – a guide to using the risk matrix. Wellington, Ministry of Business Innovation and Employment.
- Mercer TG, Frostick LE 2012. Leaching characteristics of CCA-treated wood waste: a UK study. *Science of the Total Environment* 427–428: 165–174.
- MfE 1998. Ambient concentrations of selected organochlorines in soils. ME318. Wellington, Ministry for the Environment. Available at <http://www.mfe.govt.nz/publications/hazardous/organochlorine-concentration-in-soil-dec98.pdf>
- Nakao T, Aozasa O, Ohta S, Miyata H 2006. Formation of toxic chemicals including dioxin-related compounds by combustion from a small home waste incinerator. *Chemosphere* 62: 459–468.
- Rhodes E, Ren Z, Mays DC 2012. Zinc Leaching from Tire Crumb Rubber. *Environmental Science & Technology* 46: 2856–12863
- SAEPA 2007. Environmental regulation using a risk-based approach: A guide for EPA staff. Adelaide, South Australian Environmental Protection Authority.
- Shakya PR, Shrestha P, Tamraker CS, Bhattarai PK 2008. Studies on potential emissions of hazardous gases due to uncontrolled open-air burning of waste vehicle tyres and their possible impacts on the environment. *Atmospheric Environment* 42: 6555–6559.
- Shemwell BE, Levendis YA 2000. Particulates generated from combustion of polymers (plastics). *Journal of the Air & Waste Management Association* 50: 94–102.
- Shibamoto T, Yasuhara A, Katami T 2007. Dioxin formation from waste incineration. *Reviews in Environmental Contamination and Toxicology* 190: 1–41.
- SKM (Sinclair Knight Merz) 2012. Non-natural farm waste scoping study. Report prepared for Environment Canterbury (R12/51). Available at <http://ecan.govt.nz/publications/Plans/report-non-natural-farm-waste-scoping-study.pdf>
- Solorzano-Ochoa G, de la Rosa DA, Maiz-Laralde P, Gullett BK, Tabor DG, Touati A, Wyrzykowska-Ceradini B, Fiedler H, Abel T, Carroll WF Jr 2012. Open burning of household waste: Effect of experimental condition on combustion quality and emission of PCDD, PCDF and PCB. *Chemosphere* 87: 1003–1008.

- Sonnevera International Corporation 2011. Ontario Agricultural Waste Study: Environmental impacts of open burning agricultural plastics.
- Tame NW, Dlugogorski BZ, Kennedy EM 2007. Formation of dioxins and furans during combustion of treated wood. *Progress in Energy and Combustion Sciences* 33: 384–408.
- Teuten EL, Saquing JM, Knappe DRU, Barlaz MA, Jonsson S, Björn A, Rowland SJ, Thompson RC, Galloway TS, Yamashita R, Ochi D, Watanuki Y, Moore C, Viet PH, Tana TS, Prudente M, Boonyatumanond R, Zakaria MP, Akkhavong K, Ogata Y, Hirai H, Iwasa S, Mizukawa K, Hagino Y, Imamura A, Saha M, Takada H 2009. Transport and release of chemicals from plastics to the environment and to wildlife. *Philosophical Transactions of the Royal Society of London Series Biological Sciences* 364: 2027–2045
- Thompson RC, Moore CJ, vom Saal FS, Swan SH 2009. Plastics, the environment and human health: current consensus and future trends. *Philosophical Transactions of the Royal Society of London B: Biological Science* 364: 2153–2166.
- UNEP 2005. UNEP dioxin tool kit 2005. http://www.pops.int/documents/guidance/toolkit/ver2_1/Toolkit-2005_2-1_en.pdf.
- URS and NZIER 2003. Life cycle analysis for the management of waste farm plastics and economic analysis of waste farm plastic management options. In: 3R 2006. Study of the New Zealand Product Stewardship Scheme for Agrichemical Containers. Report prepared for the Ministry for the Environment. See <http://www.mfe.govt.nz/publications/waste/product-stewardship-agrecovery-may06/html/page3.html>
- US EPA 1992. Compilation of air pollutant emission factors, AP-42. 5th edn. Available at <http://www.epa.gov/ttnchie1/ap42/ch02/final/c02s05.pdf>
- US EPA 1997. Air emissions from scrap tire combustion. USEPA-600/R-97-115.
- US EPA 2003. Reregistration Eligibility Decision for Chromated Arsenicals. EPA-HQ-OPP-2003-0250.
- Valavanidis A, Fiotakis K, Vlachogianni T 2008. Airborne particulate matter and human health: toxicological assessment and importance of size and composition of particles for oxidative damage and carcinogenic mechanisms. *Journal of Environmental Science and Health, Part C* 26: 339–362.
- Wang Z, Wang J, Richter H, Howard JB, Carlson JA, Levendis Y 2003. Comparative study on polycyclic aromatic hydrocarbons, light hydrocarbons, carbon monoxide, and particulate emissions from the combustion of polyethylene, polystyrene, and poly(vinyl chloride). *Energy & Fuels* 17: 999–1013.
- Wasson SJ, Linak WP, Gullett BK, King CJ, Touati A, Huggins FE, Chen Y, Shah N, Huffman GP 2005. Emissions of chromium, copper, arsenic, and PCDDs/Fs from open burning of CCA-treated wood. *Environmental Science and Technology* 39: 8865–8876.
- Wevers M, De Fre R, Desmedt M 2004. Effect of backyard burning on dioxin deposition and air concentrations. *Chemosphere* 54: 1351–1356.
- Williams PT, Cunliffe AM, Brindle AJ 2001. Enhanced pyrolysis processing of scrap tyres. *Journal of the Institute of Energy*. 74: 100-112.
- Woodward-Clyde 2000. Assessment of the effects of combustion of waste oil, and health effects associated with the use of waste oil as a dust suppressant. Report prepared for the Ministry for the Environment. Available at <http://www.mfe.govt.nz/publications/waste/used-oil-dust-suppressant-aug00.pdf>
- Wrobel C, Reinhardt T 2003. Review of potential air emissions from burning polyethylene plastic sheeting with piled forest debris. Report by URS Corporation for USDA Forest Service, Seattle. Available at <http://www.oregon.gov/odf/fire/docs/smp/finalpereportoct2803.pdf>

Appendix – Summary of toxicological effects of selected contaminants associated with non-natural rural waste

This appendix gives information on the environmental and human health effects associated with individual contaminants or contaminant groups. More detailed information on the human health effects is available from MfE (2011) while further information on environmental effects associated with individual contaminants is available from Cavanagh and O'Halloran (2006) and Cavanagh (2006).

Inorganic compounds

Arsenic (As) – Arsenic exposure can have numerous cancerous and non-cancerous effects including dermal lesions, pigmentation, keratoses, skin cancer and internal cancers (bladder, lung, liver), peripheral vascular disease (e.g. blackfoot disease) and cardiovascular effects (US EPA 2001; WHO 2001; ATSDR 2007a). Internal cancers such as bladder and liver cancers may arise from exposure to lower arsenic concentrations than will give rise to other effects, although exposure must also occur over a long term (i.e. years).

In the environment, arsenic compounds may cause acute and chronic effects in individuals, populations and communities including lethality; inhibition of growth, photosynthesis and reproduction; and behavioural effects. The degree of toxicity of arsenic is largely dependent on the form (e.g. inorganic or organic) and the oxidation state of the arsenic compound. In general, inorganic arsenicals are more toxic than organoarsenicals, and arsenite (As III) is more toxic than arsenate (As V). The primary mechanism of arsenite toxicity is considered to result from its binding to protein sulfhydryl groups. Arsenate affects the key energy producing processes that take place in all cells – including oxidative phosphorylation by competition with phosphate, as the compounds are structurally similar. In environments containing high phosphate levels, the toxicity of arsenate to biota is generally reduced (Gomez-Camirero *et al.* 2001).

Cadmium (Cd) – Cadmium is toxic to a wide range of organs and tissues, and a variety of toxicological endpoints have been observed in experimental animals (reproductive toxicity, neurotoxicity, carcinogenicity) and subsequently investigated in human populations (WHO 1992; Jarup *et al.* 1998; ATSDR 2008, CONTAM 2009). Cadmium is primarily toxic to the kidney, especially to the proximal tubular cells where it accumulates over time and may cause renal dysfunction (CONTAM 2009). Dietary intake is the primary source of cadmium intake for non-smoking people.

Detrimental human health effects arising from the long-term accumulation of dietary cadmium is typically the primary concern about the environmental release of cadmium, as uptake into plants often occurs at concentrations lower than that giving rise to detrimental impacts on soil biota. The general symptoms of toxicity in plants are stunting and chlorosis, with legumes, spinach, radish, carrots and oats suggested to be the most sensitive crops, and reduced yields may occur at concentrations from 1 mg/kg (Das *et al.* 1997). Cadmium may also inhibit biological nitrogen fixation, such as through *Rhizobium* (Marino *et al.* 2013). Cadmium may also bioaccumulate in the food chain, and may lead to detrimental effects on birds or small mammals (US EPA 2005; EU 2007).

Chromium (Cr) – Chromium exists in two oxidation states, with the trivalent state the most stable and most commonly found in the environment. Chromium VI in the environment primarily occurs as the by-product of several industrial processes. Chromium III is an essential element, but at high concentrations, and particularly in its hexavalent state, it is toxic. In humans and animals chromium III is an essential nutrient that plays a role in glucose, fat, and protein metabolism through potentiation of the action of insulin (IOM 2001). Limited data on the toxicity of chromium III are available (ATSDR 2000). Similarly, limited data on the toxicity of ingested chromium VI are available, although toxic effects including reproductive and developmental effects and allergic contact dermatitis have been observed. Extensive data on the toxic effects of chromium VI resulting from inhalation exposure are

available. Human health effects arising from chromium VI have primarily been documented from industrial inhalation exposures and demonstrate a wide range of effects including cancer, kidney damage and blood disorders.

Chromium has not been demonstrated to be essential to plants, although in animals chromium III is considered an essential element due to its involvement in glucose and cholesterol metabolism (Kabiata-Pendias and Pendias 2001). Chromium VI is not considered to be essential and is more toxic than chromium III due to high oxidising potential and ability to penetrate cellular membranes (Environment Canada 1999a). Plant uptake of chromium III is limited although chromium VI is more readily available. Stimulatory effects of chromium on plants have been observed in soils with a low soluble-chromium content, although phytotoxicity has been reported in other soils. Symptoms of chromium toxicity are inhibition of enzymatic activities and effects on leaf and plant growth, including chlorosis of young leaves, wilting and reduced yield at soil concentrations from 60 mg/kg (Kabiata-Pendias and Pendias 2001; Shanker *et al.* 2005). Chromium may impact on soil microbial activity, including respiration, N-mineralisation and nitrification (Environment Canada 1999a; Kabiata-Pendias and Pendias 2001). Limited data on the toxicity of chromium to invertebrates are available, with most relating to earthworms. Effects of chromium on earthworm survival, growth and reproduction have been observed.

Copper (Cu) – Copper is an essential element, required for good health and proper functioning of biological processes in plants and animals. Toxic effects can arise from deficiency as well as high concentrations of copper. Liver damage is the critical endpoint for intake of high levels of copper in animal and human studies (ATSDR 2004).

Detrimental effects on soil biota will occur at much lower concentrations than those giving rise to human health concern, with plant toxicity often the primary ecological effect of copper contamination. Visible symptoms of copper toxicity include chlorotic leaves and early leaf fall, stunted growth, and reduced root development. Copper is also toxic to invertebrates and results in mortality and reduced growth and cocoon production in several earthworm species and can also result in decreased soil microbial activity (e.g. reduction in respiration, inhibition of soil dehydrogenase activity, reduction in population growth) at relatively low concentrations (4.2–300 mg/kg).

Lead (Pb) – Human health effects associated with exposure to inorganic lead and compounds include, but are not limited to, neurotoxicity, developmental delays, hypertension, impaired haemoglobin synthesis, and male reproductive impairment. The most sensitive targets for lead toxicity are the developing nervous system, the haematological and cardiovascular systems, and the kidney. However, due to the multi-modes of action of lead in biological systems (ATSDR 2007b), lead could potentially affect any system or organs in the body. The most significant critical effect of low concentrations of lead is considered to be reduced cognitive development and intellectual performance in children (FAO/WHO 1986).

Lead is not an essential element for plant growth, and can inhibit growth and photosynthesis (through inhibition of enzymes for photosynthesis) as well as interfering with cell division and respiration. However, due to the high adsorption of lead onto soil particles, lead concentrations in the order of 100–1000 mg/kg are needed to cause visible phytotoxic effects. In bacteria and plants the majority of lead is associated with the cell wall, and very little is translocated to shoot and leaf tissue (Kabiata-Pendias & Pendias 2001).

Lead may cause impaired development and reproduction in nematodes and caterpillars, although woodlice appear to be tolerant of lead exposure. Lead interferes with the synthesis of haem, which ultimately results in accumulation of non-haem iron (i.e. an increased proportion of immature red cells in the blood). In animals lead tends to accumulate in kidneys and liver, resulting in decreased organ-to-body-weight ratios (WHO 1989).

Mercury (Hg) – Mercury exists in both organic and inorganic (including elemental) forms, although primarily exists in inorganic forms in soil. Exposure to inorganic mercury can give rise to a number of health effects in animals, including gastrointestinal effects, liver damage, kidney damage and tumours.

Limited information on the ecological toxicity of mercury compounds is available (ATSDR 1999). Mercury can be toxic to plants, with symptoms of toxicity being stunting of seedling growth and root development, inhibition of photosynthesis and, as a consequence, reduction in yield (Patra & Sharma 2000). Marigomez *et al.* (1986) reported a reduction in feeding activity in slugs fed mercury-contaminated food, although a marked reduction in weight occurred only after exposure to 1000 mg/kg mercury. Hartenstein *et al.* (1981) reported inhibition of the growth of earthworms at concentrations between 480 and 4800 mg/kg, with mortality occurring from 2400 mg/kg. Effects on microbial activity typically occur at lower concentrations, with reductions in microbial activity reported at concentrations ranging from 0.06 to 200 mg/kg (Environment Canada 1999b).

Organic compounds

Benzene – Benzene is a known carcinogen, particularly via inhalation. Benzene exposure is most dangerous when it occurs over a long period of time or when the concentration of benzene to which a person is exposed is very high. The bone marrow is the target organ for benzene toxicity and all blood cells (i.e. erythrocytes, leukocytes, and platelets) may be affected to varying degrees. Chronic benzene exposure can result in bone marrow depression expressed as a variety of conditions including aplastic anaemia and leukaemia (ATSDR 2007c).

The volatile nature of benzene poses challenges with demonstrated effects on soil biota, although effects on plants, and soil invertebrates, earthworms and springtails, may occur at high concentrations (Environment Canada 1999c).

Polycyclic aromatic hydrocarbons and benzo(a)pyrene (BaP) – Polycyclic aromatic hydrocarbons (PAHs) are ubiquitous contaminants, primarily derived from incomplete combustion of organic materials such as wood, coal and fuels. Exposure to PAHs may lead to cancer and other detrimental health effects (CCME 2008). Benzo(a)pyrene is considered to be the most potent carcinogen, and representative of a range of carcinogenic PAHs. Thus the potential carcinogenicity of environmental PAH mixtures is determined using potency equivalence factors (PEF) that express the carcinogenicity of an individual PAH relative to that of benzo(a)pyrene (CONTAM 2008).

Limited data are available on the toxicity of individual PAHs, or even PAH mixtures, to soil organisms, with some studies concluding there is no evidence that PAHs in soils at hazardous waste sites resulted in any adverse effects to plants, invertebrates or wildlife (Kaputcka 2004). Available data suggest a generally low toxicity, particularly of high-molecular-weight PAHs and aged residues. For example, Henner *et al.* (1999) observed that plant germination and growth were strongly inhibited by volatile, water-soluble, low-molecular-weight hydrocarbons (<3 rings) whereas PAHs with 3–5 rings did not show any phytotoxicity. Slaski *et al.* (2002) observed greater reductions in root length and chlorophyll-a content in plants exposed to gaseous-phase creosote, which contains higher-molecular-weight hydrocarbons, compared with naphthalene. No specific mode of toxicity of PAHs in invertebrates has been put forward, other than general modes of action of PAHs, including the formation of DNA adducts (Walsh *et al.* 1997).

Dioxins – The term “dioxins” encompasses a group of 75 polychlorinated dibenzo-p-dioxin (PCDD) and 135 polychlorinated dibenzofuran (PCDF) congeners. The toxicity of individual dioxin congeners differs considerably and only 17 are of toxicological concern. Dioxin acts on the endocrine system leading to reproductive and developmental problems, as well as damaging the immune system and interfering with hormonal systems (van Leeuwen and Younes 2000; EC-SCF 2001; FAO/WHO 2002). Dioxin exposure has been linked to birth defects, inability to maintain pregnancy, decreased fertility, reduced sperm counts, endometriosis, diabetes, learning disabilities, immune system suppression, lung problems, skin disorders and lowered testosterone level. Developmental effects occur at concentrations lower than those giving rise to carcinogenic effects.

Dioxins have a common mode of toxicity through binding to a cellular protein, known as the aryl hydrocarbon (Ah) receptor. Binding to this receptor appears to be the initial step leading to biochemical, cellular, and tissue-level changes occurring in organisms exposed to dioxins (Hahn and Stegeman 1992). Ah receptors are present in animals and birds, and thus exposure to dioxins may result in reproductive and developmental effects. The observed lack of sensitivity of plants and

invertebrates is attributed to the absence of an Ah receptor in invertebrates. However, bioaccumulation in invertebrates provides a route of exposure for birds.

References

- ATSDR (Agency for Toxic Substances and Disease Registry) 1999. Toxicological profile for mercury. Atlanta, GA, United States Department of Health and Human Services.
- ATSDR (Agency for Toxic Substances and Disease Registry) 2000. Toxicological profile for chromium. Atlanta, GA, United States Department of Health and Human Services.
- ATSDR (Agency for Toxic Substances and Disease Registry) 2004. *Toxicological Profile for Copper*. United States Department of Health and Human Services: Atlanta, GA, USA.
- ATSDR (Agency for Toxic Substances and Disease Registry) 2007a. Toxicological profile for arsenic (Update). Atlanta, GA, United States Department of Health and Human Services. Retrieved from <http://www.cdc.gov/nceh/lead/spotlights/changeBLL.htm> (January 2008).
- ATSDR (Agency for Toxic Substances and Disease Registry) 2007b. Toxicological profile for lead. Atlanta, GA, United States Department of Health and Human Services.
- ATSDR (Agency for Toxic Substances and Disease Registry) 2007c. Toxicological profile for benzene (Update). Atlanta, GA, United States Department of Health and Human Services. Retrieved from <http://www.cdc.gov/nceh/lead/spotlights/changeBLL.htm> (January 2008).
- ATSDR (Agency for Toxic Substances and Disease Registry) 2008. Toxicological profile for cadmium. Draft. Atlanta, GA, United States Department of Health and Human Services.
- Cavanagh JE 2006. Development of soil guideline values protective of ecological receptors: Part 2. Landcare Research Contract Report LC0506/179 for the Auckland Regional Council.
- Cavanagh JE, O'Halloran K 2006. Development of soil guideline values protective of ecological receptors. Landcare Research Contract Report LC0506/065 for Auckland Regional Council.
- CCME (Canadian Council of Ministers for the Environment) 2008. Canada-wide Standard for Petroleum Hydrocarbons (PHC) in Soil: Scientific Rationale. Supporting Technical Document. Canadian Council of Ministers for the Environment.
- CONTAM (European Food Safety Authority Panel on Contaminants in the Food Chain) 2008. Scientific opinion of the Panel on Contaminants in the Food Chain on polycyclic aromatic hydrocarbons in Food. The EFSA Journal 724.
- CONTAM (European Food Safety Authority Panel on Contaminants in the Food Chain) 2009. Cadmium in food. The EFSA Journal 980: 1–139.
- Marino, D. 2013. Inhibition of nitrogen fixation in symbiotic *Medicago truncatula* upon Cd exposure is a local process involving leghaemoglobin. *Journal Of Experimental Botany* Volume: 64 Issue: 18 (2013-12-01) p. 5651-5660.
- Das P, Samantaray S, Rout GR 1997. Studies on plant toxicity in cadmium: a review. *Environmental Pollution* 98: 29–36.
- EC-SCF (European Commission Scientific Committee on Food) 2001. Opinion of the SCF on the risk assessment of dioxins and dioxin-like PCBs in food. Update. SCF/CS/CNTM/DIOXIN/20 Final. European Commission: Brussels, Belgium.
- Environment Canada 1999a. Canadian soil quality guidelines for arsenic. Scientific supporting document. Ottawa, National Guidelines and Standards Office, Environment Quality Branch, Environment Canada.
- Environment Canada 1999b. Canadian soil quality guidelines for lead. Scientific supporting document. Ottawa, Environment Canada.
- Environment Canada 1999c. Canadian soil quality guidelines: cadmium (environmental effects). Scientific supporting document. Ottawa, Environment Canada.
- EU 2007. European Union risk assessment report: Cadmium oxide and cadmium metal Part 1 – Environment. European Commission.

- FAO/WHO 1986. Lead (evaluation of health risks to infants and children). WHO Food Additive Series 21. Geneva, World Health Organization. Retrieved from <http://www.inchem.org/documents/jecfa/jecmono/v21je16.htm> (January 2009).
- FAO/WHO 2002. Safety evaluation of certain food additives and contaminants polychlorinated dibenzodioxins, polychlorinated dibenzofurans, and coplanar polychlorinated biphenyls. Food Additives Series 48. Geneva, World Health Organization.
- Gomez-Caminero A, Howe P, Hughes M, Kenyon E, Lewis DR, Moore M, Ng J, Aitio A, Becking G 2001. Arsenic and arsenic compounds. 2nd ed. Environmental Health Criteria 224. Geneva, World Health Organisation.
- Hahn ME, Stegeman JJ 1992. Phylogenetic distribution of the Ah receptor in nonmammalian species: implications for dioxin toxicity and Ah receptor evolution. *Chemosphere* 25: 931–937.
- Hartenstein R, Neuhauser EF, Narahara A 1981. Effects of heavy metals and other elemental additives to activated sludge on growth of *Eisenia foetida*. *Journal of Environmental Quality* 10: 372–376.
- Henner P, Schiavon M, Druelle V, Lichtfouse E 1999. Phytotoxicity of ancient gasworks soils. Effect of polycyclic aromatic hydrocarbons (PAH) on plant germination. *Organic Geochemistry* 30: 963–969.
- IOM 2001. Dietary reference intakes for vitamin A, vitamin K, arsenic, boron, chromium, copper, iodine, iron, manganese, molybdenum, nickel, silicon, vanadium and zinc. A report of the Panel on Micronutrients, Subcommittees on Upper Reference Levels of Nutrients and of Interpretation and Use of Dietary Reference Intakes, and the Standing Committee on the Scientific Evaluation of Dietary Reference Intakes, Food and Nutrition Board, Institute of Medicine. National Academy Press: Washington, DC.
- Jarup, L, Berglund, M, Elinder, C, Nordberg, G, Vahter M 1998. Health effects of cadmium exposure – a review of the literature and a risk estimate. *Scandinavian Journal of Work and Environmental Health* 24: supplement 1.
- Kabiata-Pendias A, Pendias H 2001. Trace elements in soils and plants 3rd edn. Boca Raton CRC Press.
- Kaputska LA 2004. Do PAHs pose unacceptable ecological risks to terrestrial receptors at hazardous waste sites? *Human and Ecological Risk Assessment* 10: 233–243.
- Marigomez JA, Angulo E, Saez V 1986. Feeding and growth responses to copper, zinc, mercury and lead in the terrestrial gastropod *Arion ater* (Linne). *Journal of Molluscan Studies* 52: 68–78.
- MfE 2011. Toxicological intake values for priority contaminants in soil. Wellington, Ministry for the Environment.
- Patra M, Sharma A 2000. Mercury toxicity in plants. *The Botanical Review* 66: 379–422.
- Shanker AK, Cervantes C, Loza-Tavera H, Avudainayagam S 2005. Chromium toxicity in plants. *Environment International* 31: 739–753.
- Slaski JJ, Archambault DJ, Xiaomei L 2002. Physiological tests to measure impacts of gaseous polycyclic aromatic hydrocarbons (PAHs) on cultivated plants. *Communications in Soil Science and Plant Analysis* 33: 3227–3239.
- US EPA 2001. Arsenic in drinking water: Final Rule. *Federal Register* (66 FR 6976) January 22.
- US EPA 2005. Ecological soil screening level for cadmium. Washington DC, US EPA.
- Van Leeuwen FXR, Younes MM eds 2000. Proceedings of the World Health Organization and International Programme on Chemical Safety consultation, 25–29 May 1998, Geneva, Switzerland: Assessment of the health risk of dioxins: re-evaluation of the tolerable daily intake (TDI). In: *Food Additives and Contaminants* 17 : 223–369 (whole volume).
- Walsh P, El ALdouni C, Nadeau D, Fournier M, Coderre N, Poirier GG 1997. DNA Adducts in earthworms exposed to a contaminated soil. *Soil Biology and Biochemistry* 29: 721–724.
- WHO (World Health Organisation) 1989. Lead – Environmental aspects. Environmental Health Criteria 85. Geneva, World Health Organisation.

- WHO 1992. *Environmental Health Criteria 134: Cadmium*. International Programme on Chemical Safety, World Health Organization: Geneva.
- WHO 2001. *Environmental Health Criteria 224. Arsenic and Arsenic Compounds*. International Programme on Chemical Safety, World Health Organization: Geneva.

