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Public health implications of metal ash concentrations from prescribed burns: A study of open jarrah forests

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Public health implications of metal ash concentrations from prescribed burns:

A study of open jarrah forests along the Darling escarpment adjacent to the
Perth metropolitan area.

This thesis is presented for the degree of

Doctor of Philosophy

John Laurence Edwards

Edith Cowan University

School of Medical and Health Sciences

2018

For

Sarah Edwards

23 March 1945 to 13 April 2017

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I also thank the staff and management of the Western Australia Department of Environment Regulation and the Department of Parks and Wildlife (formerly the Department of Environment and Conservation) for the assistance and support provided during this project.

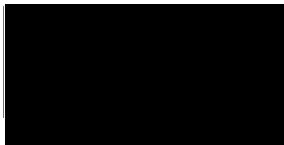
Finally, and most importantly, I would like to thank my family for their love and encouragement. To my mum (recently passed), who raised me with the ability to persevere and who has supported me in all my pursuits to get to this point. To my wife Heidi for her love and support and for taking over the duties I should have been doing during the final stages of this PhD. My appreciation is beyond words. Last and not least, to my daughter Annabelle; my favourite (she knows why). Thank you.

I would also like to thank my friends for their support and encouragement when it was needed and received (even though they will refuse to call me Doctor!).

Declaration

I certify that this thesis does not, to the best of my knowledge and belief:

- I. incorporate without acknowledgment any material previously submitted for a degree or diploma in any institution of higher education;
- II. contain any material previously published or written by another person except where due reference is made in the text of this thesis; or
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A solid black rectangular box used to redact the signature of John Edwards.

John Edwards

8 February 2018

Abstract

Ash is not a homogenous product. It is the solid residue of combustion and contains a complex mixture of chemical products. The fire that produces ash is non-discriminatory in that it will burn anything that is combustible. The ash residue resulting from fires varies and is dependent on its source, and burn characteristics such as the temperature of the fire. Ash comprises particles of carbon, soot and trace elements.

Ash presents public health risk to people and communities, through direct and indirect ingestion, inhalation and absorption. The health effects of ash exposure are not limited to symptoms affecting the eyes, throat and lungs. They can contribute to chronic disease and increase the risk of cancer.

The purpose of this study was to evaluate the public health implications of controlled burns in the Darling Escarpment, adjacent to Perth's metropolitan area in Western Australia. Concentrations of metals in unburnt vegetative litter and ash (post burn), stratified by size fraction, were determined to assess the potential mobilisation of metals caused by prescribed burns and the potential public health implications associated with burns. These data can also be used in predictive modelling to ascertain the amount of metals likely to be released per hectare when authorities plan future burns in the area.

Ash samples were collected immediately following the fire to capture fine material before it was blown away. The ash samples were separated into size fractions to investigate whether there are differences within each sub sample. Australia's National Environmental Protection Measures were used to establish whether the metal concentrations were above designated thresholds for health and

environmental investigation, these being the recognised levels above which metals are deemed to pose a risk to public (or environmental) health.

Vegetation and ash samples were collected from three sites immediately after the prescribed burn. Samples were analysed for thirteen (13) metals with nine (9) showing statistically significant increases in concentrations in vegetation as compared to ash. The percentages of metals are higher in ash than in vegetative ground litter. The metals identified are Manganese - Vegetation (158.3 ± 89.3 mg/kg), Ash (442.2 ± 462.6 mg/kg); Barium - Vegetation (19.4 ± 25.3 mg/kg); Ash (41.8 ± 62.7 mg/kg); Zinc - Vegetation (15.3 ± 9.7 mg/kg), Ash (25.6 ± 29.7 mg/kg); Vanadium - Vegetation (31.0 ± 76.3 mg/kg), Ash (32.2 ± 51.0 mg/kg); Copper - Vegetation (4.2 ± 1.5 mg/kg), Ash (10.9 ± 9.8 mg/kg); Chromium (Total) - Vegetation (8.3 ± 15.6 mg/kg), Ash (9.6 ± 12.4 mg/kg); Lead - Vegetation (6.1 ± 8.3 mg/kg), Ash (12.2 ± 8.5 mg/kg); Nickel - Vegetation (2.2 ± 2.2 mg/kg), Ash (4.7 ± 4.4 mg/kg) and Cadmium - Vegetation (0.6 ± 0.9 mg/kg), Ash (0.6 ± 0.9 mg/kg).

The ash samples were sieved through three aperture's (2-4 mm, 1-2 mm, <1 mm) to stratify the samples by size fraction and concentrations in all but three of the metals increased as the ash size decreased.

It was demonstrated that the level of metal present within the ash samples did not exceed health investigation levels and, with the exception of Manganese, and did not exceed environmental investigations levels. Metals were unlikely to pose a risk if left in-situ. However, modelling estimated the volume of metal released and it was determined that metals can pose a subsequent risk if mobilised by wind or water. The likelihood of such mobilisation is high and this finding therefore has

public health implications for surrounding communities that are subjected to increases in their exposure to metals, associated with bushfires. The findings from this study contribute to the management of prescribed burns by providing a better understanding of the composition of ash and the effects of potential distribution via aerial deposition or runoff. Data from this study can be used to do predictive modelling of heavy metal mobilisation that may result from burns of similar vegetation environments. This becomes particularly significant where burns are conducted in water catchment areas.

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Abbreviations/Definitions

ABC	Australian Broadcasting Commission
ANOVA	Analysis of Variance
ARL	Analytical Reference Laboratories
ATSDR	Agency for Toxic Substances and Disease Registry
CI	Confidence Interval
CO	Carbon Monoxide
DAF	Dermal Absorption Factor
DEC	Department of Environment and Conservation
DPAW	Department of Parks and Wildlife
ECU	Edith Cowan University
EIL	Ecological investigation Level
FMS	Fire Management Services
Ha	Hectare
HIL	Health Investigation Level
Inhalable dust	Particulate matter that meet the ISO7708 convention which are particles with a mean aerodynamic diameter of 100 µm
LoD	Limit of Detection
NATA	National Association of Testing Authorities

NEPC	National Environmental Protection Council
NEPM	National Environmental Protection Measures
PM ₁₀	Particulate matter with a mean aerodynamic diameter of 10µm
PM _{2.5}	Particulate matter with a mean aerodynamic diameter of 2.5 µm
Respirable dust	Particulate matter that meet the ISO7708 convention which are particles with a mass median aerodynamic diameter of 4 µm
ROS	Reactive Oxygen Species
SEMC	State Emergency Management Committee
Sq. Km	Square Kilometre
t/Ha	Tonnes per hectare
TRV	Toxicological Reference Value
TSP	Total Suspended Particles (50-100 µm)
USEPA	United States Environmental Protection Authority
VOC	Volatile Organic Carbons
µg	Microgram
µm	Micrometre
µg/m ³	microgram per cubic metre

Presentations out of this research project

Interim research findings were presented at state, national and international conferences.

- Environmental Health Australia National conference, 2012, *Not just particles*, Gold Coast.
- International Federation of Environmental Health 2014, *Not just particles*, Las Vegas.
- Western Australian Environmental Health Association State Conference, 2014, *Not just particles*, Perth.

Thesis Overview

This thesis overview outlines the flow of the paper and provides the reader with a “road map” to assist with understanding of the intent and highlights the major discussion points in chapters. The focus is intended to be on public health aspects of the metals generated from ash and whether this has implications for surrounding communities.

Structurally this thesis comprises of eight chapters. The introduction and research questions are presented in the first chapter, the literature review is topic specific and presented in chapters two, three & four. Methodology is discussed in chapter five. Research results are reported in chapter six, while the discussion is presented in chapter seven, followed by the conclusions and recommendations in the final chapter.

Chapter One

Prescribed burns are the main mechanism used to prevent or mitigate the effects of uncontrolled fires. Little is known about the adverse consequences following these activities. In order to provide focus, the topic is limited to concentrations of metals recognised as priority human health risks and metals typically assessed as indicators of direct or indirect toxicity to human health. The introduction acquaints the reader with the issues of prescribed burns and their potential to trigger public health concerns in the community, the project rationale and background is presented and this chapter also establishes the context and scope of the study which was limited to open jarrah forests adjacent to the Perth metropolitan area. The aims and research questions are also introduced.

Chapter Two

This chapter introduces the processes of ash generation during a fire, including how and why metals remain in the ash and become mobilised in the environment following a fire. A comprehensive literature review explores the release of specific metals during a fire and the potential public health effects of metals on humans. Specific reference is made to the mechanisms of toxicity which include competitive receptor binding and protein-enzyme interference, preferential substitution of toxic metals, disruption of cellular metabolic and regulatory processes and oxidative stress through free radical accumulation and damage.

Chapter Three

In addition to the financial loss and safety aspects associated with uncontrolled bushfires, there are also well documented acute and long term public health consequences such as a reduction in cardiopulmonary activity and increased stress and mental health consequences. Prescribed burns, which is the main control method used for the prevention of uncontrolled bushfires, aims to reduce the initial loss of life, damage to property and the disruption to the social fabric of the community.

This chapter defines prescribed burns including the evolution of the practice from traditional Aboriginal bush management and contemporary management practices, thus setting a context for the current situation. The legislative and policy framework requirements are also highlighted to allow government authorisation of contemporary prescribed burn programs.

Chapter Four

The motivation behind the use of prescribed burns as a control measure is discussed, including the history of fire in Australia, highlighting major fire events and their costs in terms of loss of life, damage to property and cost to the community. The ecological aspects of fire and the social reasons why they occur is also reviewed. This chapter explores the increasing public health risks communities are exposed to as the metropolitan area encroaches into forested areas and concludes with a discussion of public health connections of fire to land use planning, conservation and climate change.

Chapter Five

The rationale for conducting the research during a prescribed burn, as opposed to an uncontrolled and unplanned bushfire event, along with a description of the area and its vegetation is presented in this chapter.

The brief description of the pilot study that focused on sampling smoke is also presented, this is followed by the revised methods implemented after it was decided that the viability of the original study was not possible and to study the ash characteristics.

Chapter Six

Study results, including descriptive and interpretive statistical analysis of the data are presented in chapter 6. Results are presented both in aggregate form and are also stratified results based on size fractions. ANOVA and Games-Howell post hoc tests were utilised to examine whether there were any significant differences in the metal relationship between ground vegetation litter before the fire, and metal concentrations after the fire.

Chapter Seven

The discussion begins by providing an overview of the results, as well as comparing the results of this study with other similar studies conducted in Australia and overseas. The discussion centres around field observations, comparison of monitoring sites, variations based on size fractions, and how this relates to colour and the severity of the fire. The link between fire severity and metal concentrations as a result of a prescribed burn is established

The potential public health impacts of different metal types are explored in more detail with reference to Government threshold levels. The chapter concludes by modelling the ash and metal volumes anticipated from this study and the 200,000 hectares expected to be burnt annually with a discussion on what the results mean in terms of the implications of the mobilisation of metals and subsequent potential public health implications. Recommendations for the use of the study data to perform pre-fire modelling as a normal part of burn management plans are introduced.

Chapter Eight

The conclusion provides a summary of the paper outlining the main findings and the results and findings of the study are related to the original aims and research questions.

A number of recommendations for better management of public health impacts associated with the release of metals in ash, arising from prescribed burns, are made. The need for an integrated, multi-agency approach that includes a comprehensive public health component is presented. Recommendations for future research and limitations specific to with an aim to identify further research opportunities.

Chapter 1: Introduction

Bushfires not only pose a significant threat to life, property, infrastructure, and natural resources in fire-prone forest areas in Australia they also present potential long term risks to public health through the release of metals in ash. The source of this metal is intrinsic within the original structure of the forests organic material.

This risk increases in peri-urban metropolitan environments as there is an increased ignition potential from anthropocentric sources and reduced distances to residential areas (Gill and Williams, 2009). There is currently a lack of knowledge about the release of toxic metals from bushfires and as a consequence of the recent large-scale fires in south-eastern Australia there has been an increased focus on all aspects of Australian fire management including associated public health issues (Teague, McLeod and Pascoe, 2010; Gould, McCaw, Cheney, Ellis, Knight and Sullivan, 2007; Doogan, 2006).

The area chosen for this study represents a cross section of peri urban metropolitan environments located within the Darling escarpment on the outskirts of Perth, Western Australia. The sites chosen are covered by open jarrah forests (*Eucalyptus marginata*) and are managed by the Western Australia Department of Parks and Wildlife. Prescribed burns were conducted in these areas in autumn and spring to reduce the risk of uncontrolled bushfires fires during the summer months.

The aim of this public health study was to identify the links between the presence of metals in pre-fire ground litter and post-fire ash produced by a fire and to characterise resulting metal concentrations in the different ash size fractions. The mobilisation of generated ash via water movement or by aerial deposition may pose ongoing public health risks to adjacent communities and the findings of this study would also help determine whether the ash material concentrations are above health and environmental screening levels for ground contamination.

1.1 Project rationale and background

Ash materials can be either naturally occurring or produced by industry. Bushfires generate ash through pyrolysis and the size of particles can vary from (2-4 mm) to microscopic ($>1\text{ }\mu\text{m}$) in diameter. The particle size characteristic of airborne ash determines how much it can potentially be absorbed by exposed individuals through inhalation. (Dennekamp and Abramson, 2011; Beer, 2001; Jonsson, Eklund and Hakansson, 1997). Large ash particles are trapped in the mucus which lines the body's respiratory tract and are transported to the mouth via the mucociliary escalator where they can trigger a coughing reflex. Particles can be subsequently swallowed if not expectorated (Wegesser, Pinkerton and Last, 2009).

Smaller ash particles that travel deeper into the lung structure and are more difficult to remove, and therefore they represent a greater hazard. Respirable ash particles, less than $10\text{ }\mu\text{m}$ in diameter, have been associated with increased respiratory and cardiovascular mortality and morbidity (Tellez-Plaza, Jones, Dominguez-Lucas, Guallar and Navas-Acien, 2013; Hamid, Usman, Elaigwu and Zubair, 2010; Noonan, Balmes, & The Health Outcomes Workgroup, 2010;

Naeher, *et. al.*, 2007). Kunii, *et. al.*, (2002) reported that ash material associated with bushfires can contain toxic materials, including carcinogens Al-Malki (2009) concluded that more effort is needed to protect populations from the hazardous materials contained in particulate ash material.

Over centuries vegetation in much of Australia has evolved to adapt to poor soil quality. Australian trees use bio-accumulation and bio-restriction to limit the uptake of metals and to ensure optimal growth and survival. Eucalypt species can store high levels of metals and are known as hyperaccumulators (Guala, Vega and Covelo, 2011). This study aimed to determine if metal levels increased in the environment after vegetation had been burned.

This research was undertaken in conjunction with the Western Australia Department of Environment and Conservation (DEC) as a trained employee, so access to the fire ground was possible during a managed fire event. For most researchers gaining access to gather samples at the source, particularly from an active fire ground, is difficult due to authorisation issues and safety concerns. However, the collection of ash samples immediately after the fire was critical to reduce the risk of losing a portion of the ash through wind movement and to reduce the potential for sample contamination by external sources.

Only a few studies have focused on the metal composition of ash derived from biomass composition in Australia (Reisen, Hansen and Myer, 2011; Larson and Koenig, 1993). Attempts have been made to study chemical composition of ash, but these were focused on the receptor of the pollution and, as such, confounding factors could not be excluded. Spinks, Phillips, Robinson and Van Buynder (2006) conducted a study of bushfire ash settling on roofs and subsequently

contaminating rainwater tanks. However, this study found that there were many other confounding variables such as Iron and Zinc deposition from metal roofs that made it impossible to definitively identify ash as the source of metal contamination. Karthikeyan, Balasubramanian and Louri (2006), also conducted a study of metals in an ash haze in Singapore where the presence of metals was detected but the source of the metals could not be accurately determined. It is likely that, in addition to the burning of biomass, there were other sources of metal contamination, such as from industry and motor vehicle emissions.

Researchers, such as Smith, Cawson, Sheriden, and Lane, (2011a) and Jalaludin, O'Toole, Morgan and Leeder, (2004) identified the need to conduct further research to quantify the metal components of ash. The gap in the current body of knowledge of metals in ash may be due to the prominence of other contaminants within fire residue, the acute health effects of these other contaminants, and the belief that metals represented only a small component of the post fire material.

In a study of the size fraction of ash, by Pereira, Cerda, Mataix-Solera, Arncenegui and Zaval (2013) the metal concentration in source materials (ash) and its relationship to size, fire severity and mobilisation potential was quantified. Information derived from this study can assist land managers in estimating the quantity of metals at the receptor and assist in implementing mitigation strategies to reduce the impact of ash entering the surrounding environment.

1.2 Scope and limitations

This research study was conducted in a specific geographic location, with a specific forest type, and a particular type of burn characteristic. Eucalypt species are tolerant to environments with a high metal loading and it is possible that these species can accumulate higher than normal metal levels in their structure (Guala, *et. al.*, 2011).

In this study prescribed burns are defined as a subset of a controlled burn. Prescribed burns in this context relate to the application of fire to reduce a fuel load to prevent or mitigate the effects of an uncontrolled bushfire. Whereas a controlled burn is considered to be the application of fire for a range of functions, these can include stubble reduction from agricultural land or the practice of land clearing (Penman *et. al.*, 2011).

This study is contextualised to open jarrah forests in the Darling escarpment adjacent to Perth, Western Australia (WA). The parameters associated with this research provide insights for the overall body of knowledge on prescribed burns and generates assumptions that can be used in other areas of similar enquiry.

This study concentrated on the initial link between metal levels in unburnt, partially burnt, and combusted litter materials stratified into three size fractions, 2-4 millimetres, 1-2 millimetres and less than 1 millimetre. These sizes relate to coarse and fine ash material (Noske, Lane, and Sheridan, 2010; Smith, Sheridan, Lane, Noske and Heijnis, 2011b) which is the route that may present the greatest risk of metal exposure for people through inhalation of wind-blown ash, the larger fraction is also likely to be transported through runoff of rainwater into water bodies.

The Australian Broadcasting Commission (2009) reported that as a result of the Victorian bushfires in 2009, a layer of ash between two and three inches thick was deposited on the beaches and in the tidal zones at Cape Conran, hundreds of kilometres east of the fire. Foster and Charlsworth (1996) in their study of river corridors in Midland England observed that the transport and deposition of ash is controlled by the hydrological cycle. Even though that study focused on quantification of metals at the burn site, it also provides an insight into the potential for the mobilisation of metal through wind and stormwater runoff.

This study was field based and not all the environmental and burn parameters could be controlled. Sampling was dependent on when the Department conducted prescribed burns. Inter-site differences such as terrain slope, the vegetative fuel load of leaf litter, and dominant species within the micro environment impacted on the characteristics of the burn. In addition, the fire temperature varied due to differences in moisture content, which influences the variety of pollutants generated. Therefore, it is not possible to replicate within each sampling site the exact features of each prescribed burn.

The analysis was also limited to 13 metals (Arsenic, Barium, Beryllium, Cadmium, Chromium, Cobalt, Copper, Lead, Manganese, Mercury, Nickel, Vanadium, Zinc) as recommended by the National Environment Protection Council (NEPC) for the screening of a potentially contaminated site (USEPA, *Priority Pollutants List*, 2014). These metals have been identified to be hazardous and pose a potential risk to human health.

1.3 Aims and objectives

- The aim of this study was to assess the public health impacts of controlled burns through the determination of the levels of metals in ground vegetative litter and the subsequent residual ash generated during prescribed burns in the Darling escarpment adjacent to Perth's metropolitan area. Different ash size fractions were analysed to establish a link between metal concentrations, particles size, colour and fire severity. To meet this aim the project had the following objectives:
- Assess metal concentrations at three different ash size fractions (2-4 millimetres, 1-2 millimetres and less than 1 millimetre) that occur as a result of the ash generated during prescribed burns.
- Establish the relationship between ground vegetation litter and ash metal concentrations.
- Determine the potential linkage between metal concentrations in ash and public health implications.

1.4 Research questions

1. What are the concentrations of metals in ground vegetation litter within the open jarrah forests on the Darling escarpment in WA?
2. What are the concentrations of metals in ash after a prescribed burn in the open jarrah forests of the Darling escarpment in WA?
3. What is the relationship between unburnt ground vegetation litter and post-burn ash levels of metals?

4. How does the size fraction of ash, (2-4 mm, 1-2 mm, and less than 1 mm) relate to metal concentration?
5. Does metal exposure arising from prescribed burns in the Darling escarpment have the potential to affect the health of adjacent communities?

Chapter 2: The generation of metals in ash

2.1 The ash generation process

The fundamental external factors that determine whether a bushfire will occur include the presence of a fuel type, available oxygen, an ignition source, and an uninhibited chemical chain reaction (Figure 2.1). In general terms, the intensity of the fire and the speed of how a bushfire spreads will depend on the ambient temperature, available fuel load and fuel moisture, wind speed and topography (slope angle) (Drysdale, 2009; Geoscience Australia, 2016).

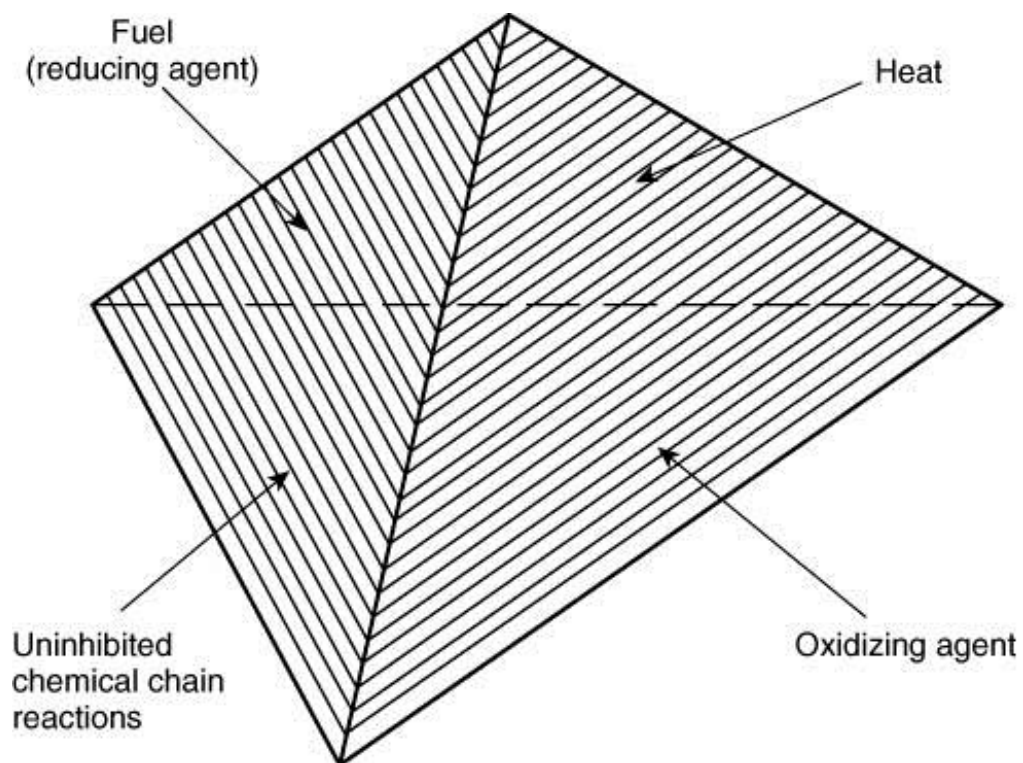


Figure 2.1: Elements essential for fire growth

(Source: NFPA, 2016)

Table 2.1 provides a more detailed description of the external factors that contribute to whether a sustained fire will occur. These factors include fuel quantity and moisture, wind speed, terrain slope, ambient temperature and relative humidity. If all these factors align catastrophic fire conditions can prevail (Geoscience Australia, 2016).

Table 2.1: External Factors Affecting the Occurrence of a Fire

(Source: Geoscience Australia, 2016).

Fuel load	The fuel load is measured in tons per hectare and higher fuel loads will result in hotter and more intense fires. Concentrated fuel with adequate spacing will burn faster than heavily compacted or scattered fuel sources but generate less heat.
Fuel moisture	Dry fuel will ignite quickly, but damp fuel may not ignite at all. The time since rainfall and the amount of rain received is an important consideration in assessing bushfire danger.
Wind speed	Winds drive a fire by blowing the flames into fresh fuel, initiating ignition and providing a continuous supply of oxygen. Wind also promotes the spread of fire by spotting, the ignition of new fires by burning embers lofted into the air by wind. Spotting can occur up to 30 kilometres downwind from the fire front. A change in wind direction can turn a long flank of the fire into a large head resulting in a larger fire front.
Ambient temperature	Higher temperatures make it more likely that a fire will start or continue to burn due to the fuel being closer to its ignition point; pre-heated fuel loads also burn faster.
Relative humidity	More moisture in the air will reduce the intensity of a fire as it requires more energy to maintain combustion. Plants become more flammable in dry environments because they can release their moisture more readily.
Slope angle	Fires pre-heat potential fuel sources through radiation and convection. Fires accelerate when travelling uphill as the potential fuels are pre-heated by the hot gases coming off the burning vegetation below. The speed of a fire front advancing will double with every 10-degree increase in slope.

Fire behaviour is the reaction of fire to factors in the environment and fire movement through a landscape; this is a self-perpetuating cycle of energy generation, release, and absorption with the visible component being a flame. Fuel absorbs heat and gives off gases until ignition occurs. The fuel itself then becomes the heating source releasing up to 100 times more energy than it is capable of

absorbing. This energy release heats adjacent fuels and the process continues. The energy stored in biomass is released as heat when materials such as leaves, grass or wood combine with oxygen to form carbon dioxide, water vapour and other substances. The combustion process can be described as reversing the photosynthesis process except that it is much faster (Blekin, 2003).

Photosynthesis: $\text{CO}_2 + \text{H}_2\text{O} + \text{Solar energy} \rightarrow (\text{C}_6\text{H}_{10}\text{O}_5)^n + \text{O}_2$

Combustion: $(\text{C}_6\text{H}_{10}\text{O}_5)^n + \text{O}_2 + \text{ignition source} \rightarrow \text{CO}_2 + \text{H}_2\text{O} + \text{heat}$

Pyrolysis refers to the chemical decomposition of biomass by fire and occurs through a number of phases. O'Byron (2005) and Blekin (2003) identify three phases in the combustion of solid fuel. These are typically:

1. the pre-heating stage, where the fuel ahead of the fire front is heated
2. the active stage, where flame combustion ignites flammable hydrocarbon gases
3. the smouldering stage, where remaining charcoal burns as a solid (Blekin, 2003).

In the pre-heating stage, a heating source (direct flame or radiation) heats the fuel to a point where volatile gases are released within a heated environment. The hotter the heat source or closer the fuel to the heat source the faster the heating process. Once the fuel reaches 100°C, water evaporates. At 200-280°C, lignin and hemicellulose material degrades into char and flammable tars (O'Byron, 2005).

The second phase is flame ignition and this occurs as a visible flame when the fuel reaches between 300-400°C depending on the fuel material. The smouldering phase occurs after the majority of fuel has been converted to energy. Coarse fuels and charcoal can continue to burn with a stationary flame or observable red glow for many hours (O'Byron, 2005).

The observable landscape indicators of the severity and speed of the energy cycle are flame height and the rate of spread of the flame moving across an area. A grass fire has a low flame height but moves across the landscape quickly. This is due to the fuel being fine, easy to ignite, and a fast burn rate. A forest fire has a larger fuel load and the speed is much slower but it generates an enormous amount of energy. A 10 t/ha forest litter bed contains 18,000 kJ/m² of fuel bed and can generate temperatures in excess of 1200°C (O'Byron, 2005).

Essentially, fuel is the most significant feature in determining the intensity and spread of a fire as it provides the energy source. The quantity and size of fuel governs the maximum energy available to a fire and is determined by the difference between the total potential energy released by the complete combustion of available fuel, minus the residual material. Very few fires cause the complete combustion of the biomass. Thus, the quantity of fuel or fuel load is positively correlated with fire intensity (Blekin, 2003).

The structure and composition of fuel is an important prerequisite for the spread of a fire, especially in the initial stages after ignition. An estimate of potential fuel burnt needs to consider the chemistry, distribution, size and configuration (e.g. compactness) of the fuel. The chemical composition of the fuel can increase flammability; eucalypt oil is an example of fuel chemistry properties that can

increase flammability. The continuous distribution of fuel will dictate the spread of the fire and the subsequent formation of burnt and unburnt areas. The size and compactness of fuel will determine speed and intensity of the fire (Blekin 2003).

Fuel moisture content is a major controlling factor that limits energy release rate of a fire. It takes more energy to evaporate the higher water content of the fuel, thus reducing its intensity. Late summer forest fires consume dry fuels at a high intensity (>7000 kW/m) while a spring prescribed burn releases less than 500 kW/m of energy. This reduces the flame height from over 15 metres to approximately 1.5 metres (O'Byron, 2005; Blekin, 2003).

Conversely, recently-burnt vegetation will not provide sufficient available fuel to support another intense fire. The fine fuel would have been removed and, until replaced with other material, the fire will not progress in the same manner as the original fire. Thus there is an historical association between fire intensity and fire frequency in any ecosystem (Blekin, 2003).

2.2 Metals in post-fire ash

Metals are ubiquitous in the environment and are found within soil, air, water, and plant matter. They are transferred naturally through ecological processes between these four mediums (ORS, 2002). Plants require essential trace elements for growth. Minerals are taken in through root systems from soil and water and through the air by absorption via leaves. The uptake of metals by trees is a complex process and its efficiency depends on a range of factors including the pH level and texture of soil, redox potential, organic matter component, soil mineral

content, and metal availability (Pundyte, Baltreinaite, Pereira and Paliulis, 2011; Brunner *et. al.*, 2007; Unterbrunner *et. al.*, 2006).

In a natural ecological setting trees use bio-accumulation and bio-restriction to maintain equilibrium by ensuring their level of metal uptake is optimal for growth and survival. Some species of plants can accumulate higher levels of metals and are known as hyperaccumulators, but once metal levels exceed the tolerable threshold limits of the plant, growth becomes retarded or the plant dies. Due to the high mineral content of Australian soils, endemic plant species have adapted to tolerate high levels of metal in soil. The Eucalypt and Acacia species are two examples of natural hyperaccumulators. (Guala *et. al.*, 2011; Maiti, 2007; Cerdà and Doff, 2005).

Anthropological inputs into an ecosystem can alter this natural balance, thus creating adverse effects through the release of high concentrations of metals into the environment. Tomasevic, Rajsic, Tasic, Krystic, and Novakovic (2004), in a study of metal accumulation in leaves from urban areas, found a positive relationship between the atmospheric deposition of metals and metal concentrations in plant material. Kahle (1993) also found increasing problems from the airborne burden of forest soils in industrialised and semi-rural regions of the world.

Excessive metal concentrations in the terrestrial environment pose a significant risk to soil quality, plants, waterways, and human health. In some geographic areas, metal contamination has become an important environmental issue because of their persistence in nature and long biological half-life for removal from the body (Pundyte *et. al.*, 2011; Siblings *et. al.*, 2005). An extreme example of

metal contamination risk is the Chernobyl Exclusion Zone in northern Ukraine. The area contains 260,000 hectares of forest and is highly contaminated with Plutonium 238, 239, 240, Caesium 137 and Strontium 90. These radio nuclei are located in the soil and vegetative litter. In the event of a fire this radioactive material will enter the atmosphere in smoke and mineral dust. A lack of forest management over the past 25 years has resulted in over 1.4 million cubic metres of dead radioactive wood being located within the exclusion zone. It is anticipated that the prevailing winds could carry radioactive material several hundred kilometres from the site should the area burn. This poses serious health risks for local and regional communities from the smoke itself and the radioactive material entering the food chain (Zibtsev *et. al.*, 2011).

2.2.1 Composition of ash

Post-fire ash deposits are defined as the residue remaining on-site after biomass burning and consist of a range of residual mineral materials and organic compounds affected by different grades of charring. An outcome of biomass fires is the production of ash. As a result of incomplete combustion a percentage of the biomass is transformed into pyrogenic organic matter (i.e. charred organic materials, charcoal, black carbon). It is estimated that the residual ash comprises less than three percent of the original material (Santin, *et. al.*, 2012). Kumar *et. al.* (2009) in their study of calorific value of Eucalyptus hybrid species provides a more accurate assessment of the ash percentage, and they found that ash percentage weight was between 0.43% and 1.09% of the original biomass. This finding is similar to that of Misra, Ragland and Baker (1993) who found ash percentages between 0.43% and 1.82% of biomass in oak and aspen species.

Based on these findings, the range utilised to estimate biomass conversion to ash for this study was between 0.4% and 2.0%.

A portion of the pyrogenic organic materials are comprised of condensed refractory compounds, which are highly resilient to degradation and likely to remain stored within the ash matrix (Santin, *et. al.*, 2012). Carbonates are generally formed at low burn temperatures when the combustion product, primarily carbon dioxide, surrounds and mixes with the burning material. High yields of carbonates are an indication of these conditions during a low temperature ashing procedure. Ash formed at high temperatures in an oxidising atmosphere, such as during uncontrolled bushfires consist primarily of metal oxides (Ubeda, Pereira, Outeiro, and Martin, 2009; Misra, *et. al.*, 1993).

Since the availability of ash nutrients is controlled by the temperatures reached during combustion, and by the type of vegetation and minerals present. Ash may derive from the combustion of vegetation (exogenous ash) and also from pre-existing soil organic matter (endogenous ash). The composition of these two types of ash differs significantly in their metal structure and solubility (Pepper, Gerba and Brusseau, 2004; Soto and Diaz-Fierros, 1993). The chemistry of the ash is a function of the type, quantity and part of vegetation burnt (leaves, twigs, litter), including soil content and structure, climatic conditions during combustion, and the collection and storage of trace elements (Pereira and Úbeda, 2010). Surface soils are considered to be "element releasers". The direction and rate of release is determined by the fire intensity through its effects on the volatilisation of the soil organic matter. As a result, certain elements (e.g. nitrogen and carbon) may be increasingly lost to the atmosphere with increasing temperature. Mineralisation

release occurs when the fire reaches peak temperatures between 350-450°C. The temperatures reached during fires differ in intensity and between the upper and lower horizons with temperatures recorded between 400 and 1200°C in the upper horizon and a corresponding temperature between 100 and 700°C in the lower horizon. The influence of fire on volatilisation and mineralisation therefore will differ accordingly (Soto and Diaz-Fierros, 1993).

In the upper horizon, the level of volatilisation is proportional to the temperature while mineralisation is dependent on the duration of the fire. Cations in exchange positions are released first, followed by cations bound to organic matter and mineral structures. At temperatures of approximately 380°C and above, the combustion of the organic matter initiates a decline in the soil's cation exchange capacity, thus increasing the release of trace elements. Leaching of the elements contained in the lower horizon is governed by the solubility of the elements principal forms while the divalent elements, occurring principally as oxides and carbonates, are mobilised more slowly (Soto and Diaz-Fierros, 1993).

The composition of the elements liberated within ash is influenced by plant material (e.g. stem, twigs and leaf) plant species and soil substrate. Ash produced from this material will occur at different stages of combustion. At temperatures near 500°C a light grey or white-coloured ash is produced with the complete combustion of these fuels. The heating of the ash changes the morphology and water repellency, as well as altering the clay and iron oxide minerals with resultant transformations in ash colour (Gill, Stephans, and Cary, 2013; Goforth, Graham, Hubbert, Zanner, and Minnich, 2005).

Pohanish (2002) identified the relevant melting/boiling points of heavy metals, as shown in Table 2.2, and the prescribed burn temperatures were well below volatilisation temperatures, with the exception of Mercury. It is therefore anticipated that the metals would remain in a carbonate or oxide form within the ash as other carbonaceous material burn off from the original vegetative material.

Table 2.2: The boiling and freezing/melting point of the heavy metals investigated in this study

(Pohanish, 2002).

	Boiling Point (°C)	Freezing/ Melting Point (°C)
Arsenic	615 (sublimation)	
Barium	1640	725
Beryllium	2970	1278
Cadmium	765	321
Chromium	2642	1900
Cobalt	2870	1495
Copper	2595	1083
Lead	1750	327
Manganese	1962	1244
Mercury	356	-39
Nickel	2837	1555
Vanadium	3380	1917
Zinc	908	420

2.3 Public health aspects of metals in the environment.

Metals are defined as elements that exhibit metallic properties and this includes a range of transitional and metalloid elements. Metals occur naturally throughout the environment, in varying concentrations, including within living organisms, such as plants and animals, at typically low or trace concentrations. Many living organisms rely on metals for normal physiological function at normal or homeostatic

concentrations, however at higher concentration, many metals and metalloids, both essential and non-essential, can be toxic to those living organisms (Garg and Singla, 2011; Kahli, 1993).

The presence of metals in ash, or as suspended airborne particulates associated with a fire pose a number of potential human health risks. The toxicity of metals is determined by a number of factors including the dose, exposure route, solubility, atomic weight or density and the species or oxidation state. While some metals have a relatively low level of toxicity, requiring chronic long term exposure to result in physical harm, other metals can pose a high toxicity risk even at low concentrations. These more toxic metals include a number of carcinogens and systemic toxicants that are associated with a range of biological effects including multiple organ damage (Tchounwou *et. al.*, 2014; Chaney, Sterrett and Mielke, (1984). The toxicity of individual metals varies as does the mechanism of their toxicity.

The severity and symptoms of metal toxicity, as a result of an exposure, can vary significantly depending on the duration and degree of exposure. The health effects associated with both acute and chronic exposure to toxic metals is well documented in the medical and scientific literature (Monisha, Tenzin, Naresh, Blessy, and Krishnamurthy, 2014; Nagajyoti, Lee, and Sreekanth, 2010; Naehler, *et. al.*, 2007; Nriagu and Jin Kim, 2000). The mechanism of metal toxicity typically occurs through one or a combination of the following biological processes including competitive receptor binding and protein-enzyme interference, preferential substitution of toxic metals, disruption of cellular metabolic and

regulatory processes and oxidative stress through free radical accumulation and damage (Fan *et. al.*, 1996).

Competitive receptor binding and protein-enzyme interference occurs in both acute and chronic exposures to metals, resulting in an increased uptake of metals by humans. Among other biological effects, this exposure results in an increased serum and cellular concentration of metals. Many essential metal ions, such as Iron²⁺ and Calcium²⁺ have important biological roles in metalloenzymes and DNA binding proteins (Ellis *et. al.*, 2006). Because essential and toxic metal ions share similar chemical characteristics such as ionic radius, where an excess of non-essential or toxic metals is present, these will naturally compete within biological systems for specific protein and DNA binding sites. For example, studies have shown that Cadmium can replace Calcium in many of its binding sites owing to an almost identical ionic size (Jacobson *et. al.*, 1980).

This competitive binding of non-essential or toxic metals into cellular ligands alters the molecular geometry and functionality of the protein and can result in the deactivation of the protein or enzyme as well as damaging DNA and nuclear proteins resulting in conformational changes that can lead to cell cycle modulation, carcinogenesis or apoptosis (Chang, Magos and Suzuki, 1996).

Metals such as Calcium, Copper and Zinc have important roles in normal cellular function. However, some toxic metals such as Lead may be preferentially substituted by direct competition for Calcium⁺ and Calcium²⁺ binding sites, particularly at hydroxyl and phosphate binding sites. This preferential substitution can result in increased absorption and tissue accumulation of toxic metals,

including through intestinal absorption, accumulation and absorption into nano-crystalline bone structure as well as other functional organs (Ellis *et. al.*, 2006).

Disruption of cellular metabolic and regulatory processes also results from competitive binding and absorption processes where non-essential or toxic metals bond to form cellular ligands and may be substituted into protein sulfhydryl, carboxyl or ammonia groups and enzymes. This substitution has also been reported to adversely affect the functionality of cellular organelles such as cellular membranes, mitochondria, endoplasmic reticulum and nuclei involved in cellular metabolism, energy and ATP production cycles as well as a number of regulatory and detoxification mechanisms (Wang, *et. al.*, 2001). Arsenic as a specific example which is a protoplasmic poison is known to preferentially bond with sulfhydryl groups resulting in the malfunctioning of cell respiration, cell enzymes and mitosis (Gordon, *et. al.*, 1948).

Oxidative stress occurs in cells as a result of an imbalance between the production of free radicals and the generation of antioxidants to detoxify cells through the removal of reactive oxygen species (ROS) (Monisha, *et. al.*, 2014). Essential metals such as Iron, Copper and Chromium are considered redox-active metals and readily undergo redox cycling, whereas many other transition or heavy metals, such as Lead, Cadmium and Mercury, are redox-inactive. Where acute or chronic exposures result in an increased cellular uptake of these redox-inactive metals, the result is an increased production of ROS including hydroxyl radicals (OH \cdot), superoxide radicals (O $^{2\cdot-}$) or hydrogen peroxide (H 2 O 2) (Ercal, Gurer-Orhan, and Aykin-Burns, 2001). This increase in ROS can overwhelm the normal antioxidant buffering capacity of a cell resulting in ROS or free radical damage to cell

organelles including lipids, proteins, DNA and mitochondria and induce general cell dysfunction and even cell death.

The presence of metals within ash and suspended particulates as a result of the combustion of plant material poses a public health risk that must be considered, when planning a prescribed burn. During the combustion process, naturally occurring metals within plant material will predominantly be reduced through combustion to form metal oxides, which will be present in ash and as airborne suspended particulates. Given the established toxicity of metals and the potential for human exposure through various potential pathways, prescribed burning is deemed to be a high-risk environment for the formation of toxic metals that provides a vector for human exposure through thermally facilitated distribution within the environment.

The analysis of ash samples, undertaken as part of this study, included testing for the National Environmental Protection Measure (NEPM) 13 metal suite which included Arsenic (As), Barium (Ba), Beryllium (Be), Cadmium (Cd), Chromium (Cr), Cobalt (Co), Copper (Cu), Lead (Pb), Manganese (Mn), Mercury (Hg), Nickel (Ni), Vanadium (V) and Zinc (Zn). This analytical suite was selected based on the broad inclusion of metals recognised as priority human health risks (USEPA, *Priority Pollutants List*, 2014), and metals typically assessed as indicators of direct toxicity or indirect toxicity to human health.

The following discussion provides information on the prevalence of the selected metals in the environment. Issues related to mechanisms of toxicity and public health risk are highlighted.

2.3.1 Arsenic

Arsenic is a metalloid that forms a steel-grey solid in elemental form and usually combines with other elements in the environment. Inorganic Arsenic is found in soil and rocks, especially in ore bodies that contain Copper and Lead. It is uniformly distributed in major rock formations and distributed in the environment through wind-blown dust. Arsenic may enter water bodies through runoff and leaching (ATSDR, 2015a; NEPC, 2014; Wang and Mulligan, 2013; Kabata-Pendias, 2010; Bradl, 2005).

Arsenic is considered a Group 1 carcinogen (IARC, 2017) in that it is carcinogenic to humans by oral ingestion and inhalation routes. Ingestion of Arsenic through soil, food and contaminated water is considered the most substantial exposure pathway for inorganic Arsenic, particularly for children. Arsenic present in dust can lead to increased risk of pulmonary carcinoma, irritation of the respiratory system, and neurological effects. In addition to cardio-pulmonary effects, the ingestion of inorganic Arsenic can result in adverse effects on other systemic organs. Long-term, low level exposure has been shown to result in the development of peripheral neuropathy. Chronic Arsenic levels in contaminated drinking water have been associated with an increased incidence of adverse birth defects particularly in developing countries and in highly industrialised areas (ATSDR, 2015a).

The toxicity of Arsenic is highly influenced by its oxidation state and solubility as well as a number of other factors including dose, frequency and duration of exposure as well as age and health of exposed individual and genetic and nutritional factors (Abernathy, *et. al.*, 1999). The predominant mechanisms of Arsenic toxicity are through competitive receptor binding and protein-enzyme

interference where Arsenic will preferentially form ligand complexes with sulfhydryl groups in proteins and enzymes, resulting in functional impairment or deactivation. Arsenic is also known for the disruption of cellular metabolic and regulatory processes, particularly through impairment of cellular respiration, the inhibition of a number of mitochondrial enzymes and the uncoupling of oxidative phosphorylation (Tchounwou, *et. al.*, 2012).

While Arsenic is generally considered to be only weakly mutagenic, the genotoxicity of Arsenic is well documented with Arsenic exposure known to result in inhibition of DNA repair, chromosomal aberrations, micronuclei formation and sister-chromatid exchanges leading to significant gene mutation (Patlolla, *et. al.*, 2005).

The Toxicological Reference Values (TRV) for Arsenic via ingestion and dermal routes is 0.002 mg/kg/day and it has an Oral Bioavailability (BAO) of 25 to 70%. The dermal absorption factor is 0.005 (or 0.5%) and the inhalation TRV is 0.001 mg/m³ (NEPC, 2014). Table 2.3 shows the Health Investigation Levels (HILs) classified according to land use and the potential for human access to the soil as either; low residential densities (Residential A), high residential properties (Residential B), Parks and Reserves (Recreational) or industrial land (Commercial). The exposure pathway for Arsenic is predominantly through direct ingestion of dust. This increases the risk factor for the airborne dispersion of dust and the ingestion of ash through contaminated saliva (NEPC, 2014).

Table 2.3: Health investigation levels for Arsenic based on land use types

(DEC, 2010; NEPC, 2014)

Land Use Types	HIL (mg/kg) DEC, 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	100	100	86	9	5	<1
Residential (B)	400	500	81	-	19	<1
Recreational (C)	200	300	90	-	10	<0.1
Commercial (D)	500	3000	72	-	28	<1

2.3.2 Barium

Barium is a silver-coloured metal that is present as a trace element in both igneous and sedimentary rocks. It is not found in its elemental form but combines with various compounds in nature. The level of Barium in the environment is low. Some foods, such as nuts, seaweed, fish, and several plants species could contain higher than normal levels of Barium. Fish and other fresh water and marine life can be subject to bio-accumulation of Barium (Kabata-Pendias, 2010; ATSDR, 2007a).

There is no evidence to suggest that Barium is a carcinogen (IARC, 2017). Inadvertent exposure to Barium is typically through inhalation and the digestion of food and contaminated water, there is also a potential for exposure through skin contact. Short-term exposure to Barium in food and water, normally as a Barium salt, above background levels may cause stomach problems, changes in blood pressure and muscle weakness (ATSDR, 2007a). There is limited information on the TRV for multiple routes for Barium but the Canadian Council of Ministers of the environment (2013) recommends an overall intake of 0.2 mg/kg/day.

Systemic Barium poisoning can result in profound hypokalaemia progressing to respiratory and cardiac arrest, while respiratory exposure is reported as resulting in baritosis, a benign pneumoconiosis. However, the predominant biological mechanism of toxicity for Barium as a metal is via preferential or competitive substitution, which results in Barium being preferentially stored in bone and slowly excreted via faeces (Olson, *et. al.*, 2004).

2.3.3 Beryllium

Beryllium is a naturally-occurring steel-grey metal that does not originate in a pure metallic form in nature. It is the lightest of the alkaline elements and fine Beryllium dust particles can be dispersed by air and settle over land and water. It exists in relatively small quantities, comprising <10 mg/kg of the major rock types (Kabata-Pendias, 2010). Acute and chronic lung diseases have been associated with Beryllium exposure and dermatological effects can arise as a result of contact with skin (ATSDR, 2015b).

Beryllium compounds are classified as a Group 1 human carcinogen (IARC 2017), although this relates to inhalation exposure only. There is limited data available on the potential for the uptake of Beryllium in fruit and vegetable crops. It is believed that Beryllium is restricted to the root system, with limited movement to the above-ground aspects of the plant. Thus, the potential for significant plant uptake is considered low and only root crops should be considered as a potential source of exposure (ATSDR, 2015b; NEPC, 2014; Gebrekidan, Weldegebriel, Hadera, and Van der Bruggen, 2013).

Beryllium is generally considered to pose a relatively low level of toxicity by comparison to other metals. However, cellular toxicity generally occurs through increased production of reactive oxygen species resulting in oxidative stress, and potentially competitive exclusion as a result of competition for protein and enzyme binding sites. Comparatively, Beryllium oxides pose a significant health risk via chronic and long-term inhalation, being demonstrated to result in pulmonary carcinomas (ATSDR, 2015). Additionally, chronic exposure via inhalation is also noted as resulting in a condition known as berylliosis, being a fibrotic lung disease similar to asbestosis and silicosis (Olson, *et. al.*, 2004).

The TRV for Beryllium through ingestion is 0.002 mg/kg/day and the dermal TRV is 0.000014 mg/kg/day. The dermal absorption factor (DAF) is 0.001 (or 0.1%) and the inhalation TRV is 0.0002 mg/m³ (NEPC, 2014). Table 2.4 provides the HILs for the various Beryllium scenarios and the likely exposure pathway percentages. For Beryllium the exposure pathway is predominantly through dermal absorption (between 56 and 88%) with ingestion accounting for between 8 to 30%. This poses an exposure risk but exposure time would not be as fast as ingestion and would require long-term exposure, thus the dermal risk would be lower (NEPC, 2014).

Table 2.4: Health investigation levels for Beryllium based on land use types
(DEC, 2010; NEPC, 2014)

Land use types	HIL (mg/kg) DEC, 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	20	60	30	12	56	2
Residential (B)	80	90	11	-	86	3
Recreational (C)	40	90	20	-	79	1
Commercial (D)	100	500	8	-	88	4

2.3.4 Cadmium

Cadmium is a silver-white soft metal and it co-exists in nature in Zinc ores, Zinc-Lead compounds, and Copper-Lead-Zinc compounds. Its concentration in the environment is dependent on the parent rock. The boiling point for Cadmium is 765°C and surface deposition and mobilisation is influenced by fire. Cadmium and Cadmium oxides are insoluble in water while some Cadmium salts are soluble in water (NEPC, 2014; ATSDR, 2012a; Kabata-Pendias, 2010; Bradl, 2005; Pohanish, 2002).

Cadmium and Cadmium compounds are classified as Group 1 carcinogens (IARC, 2017) in that they are carcinogenic to humans and inhalation of Cadmium has been specifically linked to pulmonary carcinomas. In a non-occupational environment, the low volatility of Cadmium restricts its exposure routes to inhalation of dust and ingestion of contaminated soil or water. Ingestion is considered the most significant exposure pathway for the general population. Cadmium is present in fruit and vegetable crops as well as occurring naturally in leafy vegetables, root vegetables, shrub and tree fruit (NEPC, 2014, Gebrekidan *et. al.*, 2013; ATSDR, 2012a; Pohanish, 2002). Cadmium has a long half-life (10-30 years) in the human body and it is estimated that in western countries approximately 10-12 µg of Cadmium is ingested daily (Pohanish, 2002).

Cadmium is a severe gastrointestinal and pulmonary irritant which is documented as being potentially lethal if ingested or inhaled in doses ranging from as low as 350 mg through ingestion and 5 mg/m³ via inhalation (Olson, *et. al.*, 2004). Acute inhalation exposure can result in chemical pneumonitis and noncardiogenic pulmonary oedema, normally within 12 to 24 hours after exposure. Acute ingestion

will result in nausea, vomiting, abdominal pain and often bloody diarrhoea within minutes of exposure. Deaths from ingestion typically result from associated shock and acute renal failure (Olson, *et. al.*, 2004).

Notwithstanding the highly toxic and potentially lethal nature of acute Cadmium exposure, the mechanism of its toxicity in low levels and chronic exposures is not well understood. As with many metals, Cadmium exposure is known to result in the increased production of reactive oxygen species, resulting in single-strand DNA damage which disrupts the synthesis of nucleic acids and proteins (Mitra, *et. al.*, 1984).

Studies have also demonstrated that Cadmium's toxicity may also be attributable to its accumulative toxicity based on its effect of multiple cellular and biological processes. This is supported by studies which show several stress response systems are expressed as a result of low level and chronic Cadmium exposures including oxidative stress, heat and cold shock and stringent response, as well as affecting signal transduction pathways, increasing cytosolic free Cadmium levels, reducing the rate of DNA repair and blocking calcium channels (Tchounwou, *et. al.*, 2012).

The TRV for Cadmium is 0.0008 mg/kg/day through the ingestion of produce and direct ingestion of soil. The dermal route of exposure is negligible and the inhalation TRV for Cadmium is 0.000005 mg/m³ mainly in commercial and open residential areas. Table 2.5 provides the HIL for the various Cadmium scenarios and the likely exposure pathway percentages. For Cadmium the direct ingestion of soil contributes between 31 and 78% of exposure in residential (A) and residential (B) scenarios respectively. The ingestion of produce accounted for 67% in

residential (A) areas. This poses an increased exposure risk for land areas adjacent to large forested areas (NEPC, 2014).

Table 2.5: Health investigation levels for Cadmium based on land use types
(DEC, 2101; NEPC, 2014)

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	20	20	31	67	-	2
Residential (B)	80	150	78	-	-	22
Recreational (C)	40	90	97	-	-	3
Commercial (D)	100	900	65	-	-	35

2.3.5 Chromium

Chromium is the seventh most abundant element on earth and can be detected in groundwater and surface water as well as in soil and rock. The three main forms of Chromium are Chromium (0), Chromium (III), and Chromium (VI), with Chromium (VI) being the most toxic of the Chromium metals. In order to accurately assess the toxicity of Chromium it is necessary to ascertain the form in which it occurs. In soil, Chromium is predominantly present as Cr (III) but it can change from one form to another depending on environmental conditions e.g. Redox potential, pH and the incidence of oxidising compounds (NEPC, 2014; ATSDR, 2012b; Kabata-Pendias, 2010; Bradl, 2005).

The public are most likely to be exposed to Chromium through dietary consumption. Chromium is present in many foods including varieties of vegetables, nuts, beverages and meats. Plant uptake of Chromium is limited and concentrations of Chromium in a form that can be dangerous are low in most soil types (NEPC, 2014; ATSDR, 2012a).

Chromium is classified as a Group 3 carcinogen but hexavalent Chromium or Chromium VI compounds are classified as Group 1 carcinogens (carcinogenic to humans) and are characterised as highly mutagenic (IARC, 2017; Stohs, *et. al.*, 1995). Exposure to Chromium VI can occur through ingestion of contaminated soils or dust, but this exposure scenario is rare in a natural setting. However, within an urban or industrialised environment, exposure to Chromium VI through consumption of contaminated food and groundwater is an issue, particularly in developing countries.

Chromium is most commonly present in the environment in several oxidative forms, with the most prevalent being Chromium III which is considered relatively benign due to its weak membrane permeability. Chromium VI is considerably more active in passing through cell walls, where it is able to enter a cell through anion channels in the phospholipid bilayer (Monisha, *et. al.*, 2014). Chromium VI is a strong oxidising agent and once it has entered a cell it will be reduced by biological reductants such as thiols and ascorbate resulting in the formation of reactive oxygen species including superoxide ions, hydrogen peroxide and hydroxyl radicals. The result is the induction of oxidative stress causing damage to cellular organelles, DNA and proteins (Stohs, *et. al.*, 1995). Chronic exposures to

Chromium VI are associated with chronic bronchitis, dermatitis and lung cancer (Olson, *et. al.*, 2004).

This study analysed total Chromium in ash and the 2010 guidelines only provide levels for Chromium III and Chromium VI. The published threshold level for Chromium III for the residential (A) scenario is 120,000 mg/kg and for Chromium VI it is 100 mg/kg).

The TRV for Chromium VI is 0.001 mg/kg/day through the ingestion of produce and direct ingestion of soil. The dermal route of exposure is negligible and the inhalation TRV for Chromium is 0.0001 mg/m³. Table 2.6 provides the HIL for the various Chromium VI scenarios and the likely exposure pathway percentages. For Chromium, the direct ingestion of soil contributes between 58 and 99% of the exposure pathway and the ingestion of produce accounts for 41% in residential (A) areas (NEPC, 2014).

Even though the potential for Chromium VI to be present in ash exists, contamination is more likely to be in the form of the less reactive Chromium III, which is unlikely to pose an increased exposure risk for residential land areas adjacent to large forested areas.

Table 2.6: Health investigation levels for Chromium VI based on land use types
(DEC, 2010; NEPM, 201C)

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	100	100	58	41	-	1
Residential (B)	400	500	97	-	-	3
Recreational (C)	200	300	99	-	-	1
Commercial (D)	500	3600	94	-	-	6

2.3.6 Cobalt

Cobalt is a hard, silver-grey element with similar properties to iron and Nickel. It is strongly associated with clay soils and organic matter. Cobalt is essential for biological activity as a component of vitamin B12. It is naturally found in small amounts in rocks, soil, water, plants and animals (NEPC, 2014; Kabata-Pendias, 2010; Bradl, 2005).

Cobalt is classified as a Group 2B (IARC, 2017) possible human carcinogen, but it is also considered an essential dietary element (B12). Excessive amounts of Cobalt can produce adverse effects in humans. Inhaled Cobalt dust may be absorbed via the lungs or ingested. Soil ingestion is believed to be the most important exposure pathway with inhalation considered to have a negligible effect (NEPC, 2014; Kabata-Pendias, 2005; ATSDR, 2004a).

While Cobalt is an essential metal in a number of biological processes, like many essential metals, in a condition of homeostasis, it is present in ultra-trace concentrations within cells. However, when present in high concentrations, such as in an acute or chronic exposure unmediated cellular accumulation occurs, even essential metals such as Cobalt will become toxic. The mechanism of this toxic effect is primarily through the induction of oxidative stress, where excess Cobalt results in an increased production of reactive oxygen species including superoxide and hydroxyl radicals, which in the case of Cobalt can have specific effects on mitochondria, microsomes and peroxisomes (Valko, Morris, and Cronin, 2005).

The TRV for Cobalt is 0.0014 mg/kg/day through ingestion. The DAF is 0.001 (or 0.1%) and the inhalation TRV for Cobalt is 0.0001 mg/m³. Table 2.7 provides the HILs for the various Cobalt scenarios and the likely exposure pathway percentages. For Cobalt the exposure pathway is predominantly through direct ingestion of soil (65-97%) accounting for 33% in residential (A) areas. This poses an increased public health risk, particularly for children (NEPC, 2014).

Table 2.7: Health investigation levels for Cobalt based on land use types
(DEC, 2010; NEPC, 2014).

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	100	100	65	33	1	1
Residential (B)	400	600	91	-	5	4
Recreational (C)	200	300	97	-	3	<1
Commercial (D)	500	4000	87	-	7	6

2.3.7 Copper

Copper is a malleable reddish metal that can occur naturally in its elemental form or in various other mineral forms found in rock, soil, water and sediment. Copper interacts with the mineral and organic components of the soil and it is an essential trace element necessary, however, it can have detrimental effects when present in high concentrations. (NEPC, 2014; Wang and Mulligan, 2013; Kabata-Pendias, 2010; Bradl, 2005).

Copper is required for normal cellular function and is a cofactor in many metalloenzymes and DNA-binding proteins and it is an activator of many regulatory proteins (Fan, *et. al.*, 1996). However, like most essential metals, when Copper is present in excess within a cell it can have a toxic effect. The mechanism of Copper toxicity, also known as Copperiosis, is through the induction of oxidative stress. When present in excess, Copper can act as a catalyst in the formation of reactive oxygen species, such as hydroxyl radicals resulting in oxidative damage to cells (Gaetke, *et. al.*, 2003).

Soil ingestion is considered the most significant pathway of exposure for Copper with inhalation exposures considered to be of less significance. Acute Copper dust exposure can cause irritation of the nose, mouth, and eyes, and cause headaches, nausea, and diarrhoea. Plant growth is affected by Copper deficiency as well as exposure to excessive levels of Copper. Fortunately, vegetables containing excessive concentrations of Copper present a stunted, deformed growth and harvesting would be unlikely (NEPC, 2014; Gebreidan *et. al.*, 2013; ATSDR, 2004b).

The TRV for Copper is 0.14 mg/kg/day. Table 2.8 provides the HIL for the various Copper scenarios and the likely exposure pathway percentages. Direct ingestion of soil is considered the only route of exposure with 100% attributed to this pathway (NEPC, 2014). This poses an increased exposure risk for sensitive residential areas, particularly children.

Table 2.8: Health investigation levels for Copper based on land use types

(DEC, 2010; NEPC, 2014)

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	1000	6000	100	-	-	<1
Residential (B)	4000	30000	100	-	-	<1
Recreational (C)	2000	17000	100	-	-	<1
Commercial (D)	5000	240000	100	-	-	<1

2.3.8 Lead

Lead is a grey, soft, dense, corrosion-resistant element that is rarely found as a natural metal but, once in the soil, it binds to the upper layers of the soil particles. The natural Lead content is inherited from the parent rock but, due to high levels of Lead pollution, elevated levels are likely to be found in the top soil horizons (NEPC, 2014; Wang and Mulligan, 2013; Kabata-Pendias, 2010; ATSDR, 2007b; Bradl, 2005).

Lead is classified as likely to be carcinogenic in humans - Group 2B (IARC, 2017). Exposure to Lead occurs primarily through inhalation of contaminated dust, aerosols or particulates and the ingestion of contaminated foods or water. Lead is the least available metal to plants and soil uptake is limited due to low bioavailability and poor translocation to the roots and the shoots of plants (NEPC, 2014; Bunner, Luster, Gunthardt-Goerg and Frey, 2007; ATSDR, 2007b).

Lead is considered a significant human health risk due to its high toxicity to a number of biological systems and processes within the human body. This multisystem toxicity occurs through several biological processes including preferential substitution, where Lead will preferentially replace calcium within nanocrystalline bone structures. Lead toxicity also results from competitive exclusion, where Lead will compete with other essential metals for binding sites, altering the cellular transport of calcium and generally inhibiting enzyme activity. Additionally, Lead will bind to a number of biological molecules including sulfhydryl and amide groups, proteins and DNA binding sites, alternating or deactivating the functionality of the molecules or proteins (Tchounwou, *et. al.*, 2012).

While acute exposures, other than by ingestions of gram quantities of Lead, are rare, these will result in abdominal pain and can progress to toxic hepatitis and encephalopathy (Olson, *et. al.*, 2004). Chronic exposures however can have significant systemic health effects. Lead is absorbed readily through the lungs and is characterised by rapid pulmonary absorption directly into the blood stream. Once in the blood stream it will readily cross the placental barrier where it has been attributed to neurological impairment and other birth defects in developing fetuses. Similarly, Lead will also cross the blood-brain barrier in adults and children resulting in significant damage and impairment of the central nervous system resulting in impaired concentration, reduced visual and motor coordination, peripheral neuropathy and a range of other serious health effects (Olson, *et. al.*, 2004).

Table 2.9 provides the HIL for the various Lead scenarios and the likely exposure pathway percentages. For Lead the direct ingestion of soil is considered the only route of exposure (NEPC, 2014). This poses an increased risk for sensitive residential areas, particularly for children.

Table 2.9: Health investigation levels for Lead based on land use types
(DEC, 2010; NEPC, 2014).

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	300	300	100	-	-	<1
Residential (B)	1200	1200	100	-	-	<1
Recreational (C)	600	600	100	-	-	<1
Commercial (D)	1500	1500	100	-	-	<1

2.3.9 Manganese

Manganese is a steel-grey metal found in a wide range of rocks and soil. Elemental Manganese is a silver-coloured metal but does not exist in a natural form; it combines with other substances such as oxygen, sulphur, and chlorine to form oxides, carbonate and silicates. Manganese is an important soil constituent essential for plant nutrition. It is one of the most abundant trace elements with a range between 350 to 4250 mg/kg (NEPC, 2014; ATSDR, 2012c; Kabata-Pendias, 2010).

Manganese is a dietary essential metal required for bone mineralisation, energy metabolism and metabolic regulation. Health issues can occur as a result of Manganese deficiency as well as excessive intake. Manganese is classified as a Group D Carcinogen in that it is not classifiable (IARC, 2017; NEPC, 2014; ATSDR, 2012c).

While acute high level inhalation of Manganese may result in irritant pneumonitis, the precise mechanism of Manganese toxicity is not known. However, the principal location of toxic effect is the central nervous system. Toxicity is typically associated with low level exposure over a period of months or years, which can present as psychiatric disorders similar in appearance to schizophrenia or atypical psychosis combined with other neurotoxicity indicators such as extrapyramidal movement disorders (Olson *et. al.*, 2004).

Despite the fact that inhaled Manganese compounds are more toxic, soil ingestion is considered the most important exposure pathway for this metal due to the absorption rates within the target organs; the dominant exposure route remains ingestion. Manganese does not readily penetrate the skin. Plant growth is affected by Manganese, yet its translocation into above-ground crops is considered negligible and the metal tends to concentrate in the root systems of plants (NEPC, 2014; ATSDR, 2012c).

The TRV for Manganese is 0.14 mg/kg/day through direct ingestion of ash or the ingestion of produce. The inhalation TRV for Manganese is = 0.00015 mg/m³. Table 2.10 provides the HIL for the various Manganese scenarios and the likely exposure pathway percentages. For Manganese the direct ingestion of soil exposure pathway in residential (A) and Residential (B) contributes between 32%

and 29% of exposure respectively. The ingestion of produce accounts for 49% in residential (A) areas. The inhalation of dust in residential (A) and residential (B) scenarios accounts for 19% and 71% respectively (NEPC, 2014). This poses an increased exposure risk because of the predominance of direct ingestion and inhalation being a contamination pathway for residential land areas.

Table 2.10: Health investigation levels for Manganese based on land use types

(DEC, 2010; NEPC, 2014).

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	1500	3800	32	49	-	19
Residential (B)	6000	14000	29	-	-	71
Recreational (C)	3000	19000	80	-	-	20
Commercial (D)	7500	60000	18	-	-	82

2.3.10 Mercury

Mercury is a shiny, silvery metal that presents as a liquid at ambient temperature. The breakdown of minerals in rocks and soil releases inorganic Mercury, which is persistent in the environment, yet it is limited in its migration. Land and surface water deposition results in subsequent Mercury absorption into soil and sediments. The toxicity of Mercury is dependent on the form in which it occurs (NEPC, 2014; ATSDR, 2013; Kabata-Pendias, 2010; Bradl, 2005).

Soil ingestion is considered the predominant exposure pathway and inhalation exposures associated with dusts are expected to be less important than direct or indirect soil ingestion. Inorganic Mercury is biotransformed into methyl Mercury in the course of environmental degradation. It bioconcentrates as it progresses up

the food chain, with increasing concentrations in the higher order animals (ATSDR, 2013; Oosthuizen and Ehrlich, 2001).

The potential for dermal absorption of Mercury can occur as Mercury reacts with skin proteins. Inorganic Mercury is classified as a Group 3 carcinogen (IARC, 2017) in that it is not classifiable. There is a potential for Mercury uptake into green, root and tuber vegetables, shrub fruit and tree fruit. The exposure levels vary depending on the type of flora; green vegetables 0.0038 mg/kg, root vegetables 0.0069 mg/kg, tuber vegetables 0.0042 mg/kg, and tree fruit 0.001 mg/kg (NEPC, 2014; ATSDR, 2013; Gebrekidan *et. al.*, 2013).

High level acute exposures to Mercury can result in immediate and potentially lethal effects, depending on the form of Mercury and route of exposure, and it is known to be a severe pulmonary irritant, CNS poison and highly nephrotoxic (Olson, *et. al.*, 2004). The mechanism of Mercury toxicity in chronic and low-level exposures is no less significant and is considerably more relevant in the context of this research. The molecular mechanism of Mercury toxicity occurs primarily through oxidative stress. Mercury has a strong and preferential bonding affinity with sulfhydryl groups. Once Mercury has entered a cell it will form a covalent bond with the amino acid cysteine, while present as a protein residue. This will deplete cellular antioxidants which act as a primary defensive mechanism against not only Mercury but also other metals and reactive oxygen species (Valko, Rhodes, Monocol and Izakovic-Mazur, 2006). This reduction in cellular antioxidants results in an accumulation of reactive oxygen species which would normally be eliminated, resulting in oxidative damage (Tchounwou, *et. al.*, 2012).

The result of oxidative stress or oxidative damage as a result of Mercury toxicity, as for most toxic metals, can affect a variety of biological functions and express as a range of symptoms affecting a number of systemic organs. The accumulation of Mercury has been shown to correlate with the severity of hepatotoxicity and nephrotoxicity (Lund, Miller and Woods, 1991).

The TRV for inorganic Mercury is 0.0006 mg/kg/day through ingestion and the dermal routes of exposure. The DAF is 0.001 (or 0.1%), gastrointestinal absorption factor is 0.07 and the inhalation TRV for Mercury is 0.0002 mg/m³ for elemental Mercury only. Table 2.11 provides the HILs for the various Mercury scenarios and the likely exposure pathway percentages. For Mercury the direct ingestion of soil exposure pathway contributes between 56 and 72% of exposure pathway and dermal absorption accounts for between 13 and 52%. The ingestion of produce was 21% in residential (A) areas (NEPC, 2014). There is a potential increased exposure risk because direct ingestion is the dominant contamination pathway. In the Swan and Canning rivers, the levels of Mercury, Cadmium and Lead are quite low in fish stock and a person would have to eat more than one fish per day for it to be of concern (Oosthuizen and Boyce, 2002).

Table 2.11: Health investigation levels for Inorganic Mercury based on land use types

(DEC, 2010; NEPC, 2014)

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM, 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	15	40	66	21	13	<1
Residential (B)	60	120	56	-	44	<1
Recreational (C)	30	80	72	-	28	<1
Commercial (D)	75	730	48	-	52	1

2.3.11 Nickel

Elemental Nickel is a hard, silvery metal found in all soils. In the environment it is primarily found as oxides or sulphides but is easily mobilised and attaches to organic-bound forms where soils can contain higher levels (NEPC, 2014; Kabata-Pendias, 2010; Bradl, 2005; ATSDR, 2005a).

Nickel is classified as a Group 2B carcinogen (IARC, 2017) in that it is known to cause cancer in humans. Non-occupational exposure to Nickel is most commonly associated with ingestion of contaminated food or water. Within an occupational setting, exposure to Nickel dust or fumes via inhalation represents a significant exposure risk. Dermal exposure can cause hypersensitivity skin reactions with between 1 and 4% of men and between 8 and 20% of women being Nickel-sensitive (NEPC, 2014). Nickel can be absorbed by, and accumulate in, green, root and tuber vegetables and tree fruit. The TRV for Nickel is 0.012 mg/kg/day through the ingestion of produce and recommended levels in green vegetables are 0.0038 mg/kg, root vegetables 0.0043 mg/kg, tuber vegetables 0.0019 mg/kg, and tree fruit 0.0034 mg/kg (NEPC, 2014; Gebrekidan *et. al.*, 2013; ATSDR, 2005a). Nickel also has a dermal route of exposure with the DAF being 0.5%. The inhalation TRV for Nickel is 0.00002 mg/m³.

As for many metals, the toxicity of Nickel is dependent on its form, solubility and duration of exposure (ATSDR, 1988). While ingestion is considered the most likely route of exposure for the general population there is no evidence of an increased cancer risk via this pathway (Olson, *et. al.*, 2004). Ingestion is known to result in toxicity to the kidneys and also has some adverse impacts on the gastrointestinal tract (ATSDR, 1998). There is substantial evidence (Demir *et. al.*, 2005;

Grandjean, 1992; WHO, 1991) that demonstrates the carcinogenicity of inhaled Nickel, of the issue of exposures to airborne nickel contaminated particulates during a prescribed burn is therefore relevant from a public health perspective (Enterline, *et. al.*, 1982). Following combustion, Nickel will be present in a number of oxidised forms, including Nickel oxide. Recent data has identified Nickel oxide, subsulfate and soluble Nickel as likely candidates for initiating carcinogenic responses in humans (Roberts, Julian, Jadon and Muir, 1994).

Table 2.12 provides the HIL for the various Nickel scenarios and the likely exposure pathway percentages. Direct ingestion of soil contributes between 42 and 80% of the exposure pathway, while the ingestion of produce accounts for 26% in residential (A) areas. Dermal absorption and the inhalation of dust in residential (A) scenario accounts for 4 and 15% respectively (NEPC, 2014). Nickel poses an increased exposure risk because direct ingestion is a contamination pathway relevant for residential land areas.

Table 2.12: Health investigation levels for Inorganic Nickel based on land use types

(DEC, 2010; NEPC, 2014).

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	600	400	55	26	4	15
Residential (B)	2400	1200	42	-	11	47
Recreational (C)	600	1200	80	-	11	9
Commercial (D)	3000	6000	30	-	11	59

2.3.12 Vanadium

Vanadium is a greyish ductile metal that is found bonded to other compounds in nature. Vanadium concentrations in soil relate to those of the parent rocks and the mobility of Vanadium is dependent on the host minerals and is often absorbed into the mineral structures of clay soils (ATSDR, 2012d; Kabata-Pendias, 2010).

Vanadium is present in small concentrations in all plants with the main exposure route of Vanadium being ingestion with food. Marine species, especially invertebrates, have higher concentrations of Vanadium than mammals. Natural sources of Vanadium, such as continental dust and marine aerosols, only result in low natural background levels in air.

The toxicity of Vanadium is also related to its chemical species. For examples, elemental Vanadium in its base electronic state is significantly less toxic than oxidised forms, with Vanadium pentoxide recognised as the most toxic form of Vanadium (Olson, *et. al.*, 2004). The 2010 DEC guidelines provide limited information for Vanadium and the HIL for Residential (A) is 550 mg/kg and Commercial is 7200 mg/kg. There is not much information available on the TRV via multiple routes of exposure. The USEPA (2005) set an overall TRV of 0.344 mg/kg/day. Chronic exposure to Vanadium compounds, particularly Vanadium pentoxide produces a range of respiratory effects, that range from coughing to more serious effects such as pneumonia and bronchitis, at levels above 1 mg/m³ (ATSDR, 2012d).

However, aside from inadvertent industrial exposures, ingestion remains the dominant route of exposure for the general population. While ingestion of Vanadium is known to cause gastrointestinal dysfunction, including cramping, diarrhoea or black stools, the weight of medical research indicates no significant long-term adverse impacts from ingestion (Olson, *et. al.*, 2004). At a cellular level, Vanadium oxides have been shown to result in oxidative stress through the depletion of cellular antioxidants and the accumulation of reactive oxygen species.

2.3.13 Zinc

Zinc is ever-present and uniformly distributed throughout the environment. In its metallic form Zinc is a light blue metal but in the environment it is only found in the oxidative state. Clay and organic matter soils hold Zinc quite strongly and accumulation of Zinc in these soil types is often observed. In Australia Zinc occurs at concentrations of between 29-86 mg/kg (NEPC, 2014; Wang and Mulligan, 2013; Kabata-Pendias, 2010; ATSDR, 2005b; Bradl, 2005).

The ingestion of soil is considered the most important exposure pathway, with an oral TRV of 0.5mg/kg/day. The bioavailability of Zinc from food ranges from 10 to 40%. Dermal exposure is considered to be negligible as Zinc is used in many skin creams, such as sunscreen and the DAF is 1% (NEPC, 2014). Zinc is not a volatile metal and inhalation exposures are expected to be of less importance than soil ingestion. Zinc is not considered a human carcinogen (NEPC, 2014; ATSDR, 2005b).

Zinc is an essential metal and plays an important role in many cellular functions and metalloenzyme reactions, and is present in biological cells at trace levels. However, while Zinc is normally considered non-toxic, recent studies have confirmed that the presence of Zinc in excess can saturate protein, enzyme and DNA binding sites and result in the suppression of Copper and Iron uptake, (Olson, *et. al.*, 2004). Resulting deficiencies in Copper and Iron may result in symptoms of anaemia and neutropenia as well as impaired immune function (Fosmire, *et. al.*, 1990).

An important factor for plants to absorb Zinc is its solubility and mobility in soil and water. Zinc toxicity in plants affects general physiological processes where the adverse effects on plant development occur between 200 and 300 mg/kg (ATSDR, 2005a).

Table 2.13 provides the HIL for the various Zinc scenarios and the likely exposure pathway percentages. For Zinc the direct ingestion of soil contributes between 93% and 98% of the total exposure pathway. The ingestion of produce and inhalation are considered negligible pathways. Dermal absorption of dust is low and it contributes between 1 and 7% of the exposure pathway (NEPC, 2014). Zinc poses an increased exposure risk because of the predominance of direct ingestion being a contamination pathway for residential land areas.

Table 2.13: Permissible exposure levels for Zinc

(DEC, 2010; NEPC, 2014).

Land use types	HIL (mg/kg) DEC 2010	HIL (mg/kg) NEPM 2014	Percentage contribution from exposure pathways			
			Ingestion of soil/dust	Ingestion of produce	Dermal absorption of Soil / Dust	Inhalation of dust
Residential (A)	7000	7400	98	-	1	<1
Residential (B)	28000	60000	95	-	5	<1
Recreational (C)	14000	30000	98	-	2	<1
Commercial (D)	35000	400000	93	-	7	<1

2.4 The mobilisation of ash.

Fire management techniques such as prescribed burning and active bushfire suppression have been used to mitigate the long-term extreme watershed sediment yield into reservoirs and to protect urban environments on the interface with peri-urban forests. Where prescribed burning has been deferred in extensive forest areas it has resulted in an excessive accumulation of forest litter. The ash from this accumulated material can become transported by bushfires resulting in high-intensity fires and spotting situations. During a fire, spatial temperature distribution can be heterogeneous, leading to a mosaic of areas burnt at different severities and as a consequence there is generally a complex pattern of differences in fire temperatures and fuel loads that leads to significant variation in the type and amount of metal released. (Gill and Williams, 2009; Goforth, *et. al.*, 2005).

Fine ash is highly-mobile and can easily become airborne and is subsequently transported by wind, especially during severe bushfire events. Laboratory experiments have shown that fire severity and the associated variation in combustion temperature is a key factor impacting on the physical and chemical properties of ash. Furthermore, the species of vegetation burnt and duration of the fire also influence ash characteristics. The aerodynamics of fine ash makes it stable in wind currents allowing the material to travel over long distances (Pereira and Úbeda, 2010).

Bushfires can also directly affect the soil's physicochemical properties through the input of ashes and changes to soil structure due to the high temperatures attained within the soil matrix. These changes include structural and textural degradation and the release of cations. Overall these changes, together with the increase in nutrient concentrations resulting from the input of plant ashes and the ongoing diminished nutrient uptake by plants, can lead to an escalation in nutrient availability and this can increase soil fertility. Nevertheless, there is also a risk of water pollution if these nutrients and metals are mobilised through rain water run-off. Contaminants may leach from the soil despite some retention by underlying horizons unaffected by thermal shock (Pepper *et. al.*, 2004; Soto and Diaz-Fierros, 1993)

The decomposition of humidified organic matter in the upper soil layers generates water-repellent compounds in unburnt surface soil horizons. These organic compounds volatise in soils subjected to heating intensities of approximately 250°C over a period of several minutes to form a tier with diminished wettability. The composition of this water-repellent layer is dependent on antecedent

properties that affect the heat penetration of the soil (e.g. soil texture and moisture conditions), as well as the surface combustion intensity and duration of soil heating (Goforth et. al., 2005).

Rapidly extinguishing bushfires limits the combustion process thus mitigating sediment yield into surface water. Conversely, where burning is deferred, there is a build-up of forest detritus that has more volume and burns hotter, the combustion process is completed faster and produces more fine ash (Goforth, *et. al.*, 2005). When the plant material is burnt the metals in the fine ash become mobilised by wind. Airborne ash particles are subsequently deposited elsewhere on leaf material, soil and water and the remaining ash is retained within the soil, from where metals can enter the waterways through run-off (Lyon and O'Connor, 2008; Ferreira, Coelho, Boulet, and Lopes, 2005; Radojevic, 2003; Berinato, 1996; Kahle, 1993).

In the immediate period after a fire, wind and water mobilises surface ash that can increase the vulnerability of soil to erosion in some areas while reducing it in others. Localised climatic conditions after a fire determines how the ash will interact with the environment. Immediately after a bushfire, ash and the remaining vegetation cover are the principal protections against erosion. There is a short window of time where runoff and surface water is controlled by high infiltration rates which are initially high due to the burning of the surface organic matter and the effect of ash acting as mulch. Water moves through the soil gaps but, once saturated, the flowing water mobilises the remaining ash (Pérez-Cabello, 2012; Pereira and Úbeda, 2010).

Flume experiments have demonstrated that fine ash can initially inhibit the infiltration of runoff, thus leading to increased volumes of overland flow. Even a thin layer of fine ash can significantly reduce the infiltration capacity of a coarse-grained soil. The swelling of wet ash might intensify its sealing effect by as much as 13% when moistened due to the absorption of water and the creation of calcium silicate hydrate, a compound with swelling properties. Conversely, a layer of coarse ash can significantly delay the generation of overland flow until saturation is reached and mobilisation commences (Gabet and Bookter, 2011).

Regardless of the specific run-off mechanism, fires producing vast quantities of ash on hill slopes with steep gradients will generate run-off containing high ash concentrations. Vegetative ash, present in mountainous terrain, could be a significant factor in the initiation of progressively bulked ash slurries. For example, assuming an ash layer with a porosity of 83% and a runoff depth of 10 mm would only need to mobilise 7 mm of ash to attain a solid fraction concentration that is within the range considered to be debris flow (Gabet and Bookter, 2011).

Smaller ash particles incorporated into runoff will increase the density of the flow and its viscosity. Where ash particles are of a similar size and packaged in a close hexagonal format, ash aggregates as large as 1 mm could behave as part of the flow because of their low density. The increase in viscosity decreases the settling velocity of coarser material entrained in the flow. The thick layers of post-fire ash, therefore, provide an ample supply of finely grained material that promotes debris flow behaviour in run-off (Gabet and Bookter, 2011).

Cerdà and Pereira (2013) demonstrated that ash run-off was enhanced after rain from intense thunderstorms exceeded infiltration rates and eroded the ash layer. They found that low-intensity rainfall contributed to higher ash infiltration rates that secured the metals to soil allowing them to be used by the remaining plant material. As a consequence of the transient infiltration characteristics of the ash cover surface run-off will reach its peak at approximately four months and can continue for a period up to two years after the fire.

The post-fire leaching-induced nutrient losses and water contamination risks may be largely counteracted by nutrient retention in the lower soil horizons. However, on steeply-inclining sites or in hydrophobic soils, often the case with bushfire-affected soils, this compensating effect may be lacking or minor. Water and nutrient loss, in either case, may be considerable. Organic horizons untouched by thermal shock may become contributors of hydrogen ions and organic matter to surface and subsurface waters, and therefore contribute to water contamination (Soto and Diaz-Fierros, 1993).

Layers of vegetative ash can blanket the burnt terrain to depths of many centimetres. Given this sudden addition of large amounts of finely-grained material it is reasonable to assume that this should have an effect on the soil and subsequent ecosystems. The presence of fine ash may decrease the soil infiltration capability by congesting the pores at the soil surface. The effect is also intensified by the swelling properties of wet ash in the upper soil horizons. The ash layer may also absorb and accumulate rainfall thus delaying the runoff onset and delaying the hydrological response to rainstorms. However, in heavy rainfall situations ash particles are likely to be instrumental in the development of the

increasingly bulked debris flows that have been observed in burnt landscapes (Gabet and Bookter, 2011; Pereira, Úbeda and Martin, 2012).

Pereira and Úbeda (2010) also ascertained that during the period immediately following a fire, ash cover strongly influenced soil hydrological properties, such as water retention and wettability. Ash is also a valuable source of nutrients essential for plant recuperation (Pereira *et. al.*, 2012), but can act as a source of contamination where the soil-ash profile is abundant with metals (Pereira and Úbeda, 2010).

2.5 Consequences of the mobilisation of ash

Metals in ash are mobilised via water run-off that could potentially contaminate public drinking water supplies or water for irrigation of crops. This has public health implications for the Australian population as affected communities are subjected to increases in their exposure to metals, associated with bushfires. Health impacts of metal exposure can take decades to manifest since most of the impacts would be chronic health effects associated with long term low level exposure. Metal contamination has the potential to adversely impact on people with suppressed immune systems who are not as resilient and able to cope with the increased environmental pollutant burden (Nriagu, 2000).

Climate change will continue to affect the world's fresh water ecosystems resulting in adverse social and economic consequences. Changing rainfall patterns will result in precipitation increases in some areas and reductions in others. Since 1950, the Australian average annual and seasonal temperatures have increased, leading to severe droughts and increased fire risk. These increased temperatures

also cause the drying of waterways and, as Australia is the driest inhabited continent, it will be severely affected by the changing climate and impact on its water resources. Metals washed into drying river beds can accumulate. When a flushing event occurs, drinking water catchment areas can be polluted by an increased accumulated volume of metal contaminated water (Jenkins, 2011).

Generally, the composition of metals in a water body is low due to the removal of the contaminants by hydrodynamic processes. However, Australian waterways are already some of the most variable in the world, with long periods of drought followed by extensive flooding. If a fire is included in the equation, post-fire flooding can cause ash to enter a waterway due to increased erosion; a result of the removal of vegetation that absorbs the ash or dams the flow of surface materials. Once the metal contaminated ash enters a water body the dissolution of metals in the water will depend on the pH, redox conditions, chemical composition of the soil and total sediment input (Smith, *et. al.*, 2011a; Lyon and O'Connor, 2008; Chiew, Midway, Duncan, and McMahon, 1997).

The ash deposits can settle and accumulate in temporary pools and other slow-moving areas and can reach a thickness of two to three centimetres. When the influx of water exceeds the holding capacity of these areas the sediment ash plume can move as a floating body causing immediate fish deaths through the removal of oxygen. It can also lead to the accumulation of metals in the remaining fish stocks. Once the fire contaminant plume has passed the streams usually return to pre-fire conditions within five to 20 years (Bond, Clarke, Lake, and MacNelly, 2010; Nelson and Milligan, 2003; Kilby and Batley, 1993).

In Australia there have been numerous examples of water quality reduction due to sediment flow. In 2003, as a result of the bushfires around Canberra, a sediment plume travelled 200 kilometres from upland streams and killed 95% of fish stocks. It took 24 months for fish stocks to return to normal levels. Iron and Manganese levels exceeded guideline levels by factors of three and four respectively. As a result of the increases in metal levels, water restrictions were introduced in Canberra (Smith, *et. al.*, 2011a; Lyon and O'Connor, 2008). Lake Glenmore and Lake Buffalo in north-eastern Victoria also suffered a reduction of water quality as a result of sedimentary flow post-bushfires (Smith, *et. al.*, 2011a).

Ash mobilisation through wind transport is also a public health concern. Fine ash (<1 mm) (Noske, Lane and Sheridan, 2010) can be transported long distances and settle in various potable water systems including roofs and rain water collection tanks. This material can subsequently be washed into the tanks where it can settle causing significant contamination. Multiple fire events can cause a build-up of material, including metals in the potable water systems that can provide exposure at a later date (Spinks *et. al.* 2006; Karthikeyan *et al.* 2006).

Chapter 3: What is a prescribed burn?

Fire creates a fear in the community that is unmatched by any other natural disaster. The initial consequences of fire can be dire, including loss of life, property and a disruption to the social fabric of the community (Pawson, 2011). Bushfires also pose a long term public health risk. There can be direct effects, particularly on the cardiopulmonary system (Vicedo-Cabrera, Esplugues, Iniguez, Estarlich, Ballester, 2016; Denekamp and Abramson, 2011) as well as indirect effects that could include mental health consequences (Reifels, et. al., 2015).

It is becoming apparent that people play an important role in the frequency and effects of fire. Prescribed burning is one method of risk reduction and without management tools to reduce the risks associated with fire there is an increased likelihood that the more acute effects of fire will be felt by more community members.

Bushfires have the capacity to significantly impact the natural ecological regime. A natural fire interval pattern would occur if it was not influenced through human intervention. In the rural-urban interface this natural pattern is not possible as human activity alters the fire intervals. A fire-altered landscape can occur through intentional, illegal activities such as arson, or unintentional, indirect activities like the littering of discarding cigarette butts and hot work activities (Certini 2014, Keelty, 2011; Gill and Williams, 2009; Mysterud, Mysterud, and Blekin, 2003).

The main aim of controlled burns globally is to reduce hazardous fuel loads, prepare sites for agricultural purposes, maximise water run-off in drinking water catchment areas, improved appearance and access. Despite the benefits of controlled burns, fire can also be damaging to the environment depending on its intensity, seasonal timing and age of the forest. Fire can lead to the removal of humus or other organic matter required for plant growth and cause nutrients, changes in habitat that often increases the likelihood of weed invasion, and soil erosion from wind and water (Antunes, Curado, Castro and Goncalves, 2009; Alexander and Thomas, 2006). In Australia the objective of a prescribed burn is generally to prevent uncontrolled fires.

Many researchers, including Gould, McCaw and Cheney (2011), Penman and York (2010) and Mysterud *et. al.* (2003), consider the fine litter component of the fuel load to be the most important parameter that determines the severity of bushfires and, in conjunction with weather conditions, increases the speed of the fire spread. The frequency of hazard-reduction fires therefore has to be regular enough to reduce the fuel load below the level that can lead to uncontrolled fires. The frequency of controlled fire is dependent on the vegetative species, topography, weather conditions and growth rates within the defined area.

The focus of land management through prescribed fire during mild weather conditions has been to prevent large uncontrolled fires during summer periods where the consequences cannot be managed or mitigated. The development of a long-term fuel reduction strategy is to reduce the risk for large-scale high-intensity fires, by burning larger or smaller sections of the landscape in a planned rotational pattern (Keelty, 2012; Mysterud *et. al.*, 2003).

Vines (1998) described a model of a prescribed burn as the use of fire under specific weather conditions in a defined location to reduce the fuel load. This viewpoint is idealistic in that it describes the process without the ambiguity associated with the fire event. In terms of ignition, fire is viewed as either opportunistic or planned. Naturally-ignited fires can produce beneficial results in terms of attaining land management objectives, when aligned with anthropocentric values and they are sometimes allowed to burn with limited intervention on the periphery to achieve fuel reduction.

Mysterud (2003) built on Vines model and further refined the definition of prescribed burning to include the use of low intensity fire as a risk management tool to manage the intensity of uncontrolled fires should they occur. Alexander (2006) modified Mysterud definition to include a control aspect to a prescribed burn.

A fire regime is viewed as the specific fire characteristics that occur at a location or in a region. The regional fire pattern is characterised by a range of features including fire intensity, severity, fire intervals (frequency), fire commencement, extent and form of the burn profiles, and the type of fire that occurs. The fire regime implemented in the eucalyptus forest of south-eastern Australia consists of an average burn area of 165 hectares on a 43 year cycle. This gives an indication of a potential accumulated fuel load in the area of concern (Keeley, 2009; Burrows, 2008; Blekin, 2003).

The main argument for hazard-reduction through prescribed burning is the fear that the alternative could expose communities to catastrophic uncontrolled fire risks that could result in loss of life, property damage, and large-scale release of toxic pollutants. The Western Australia Government has adopted the position that prescribed burning is the most effective method to manage fuel loads and mitigate the impact of uncontrolled bushfires. Prescribed burning emulates a low-intensity fire through the manipulation of the fuel load across the landscape for the purpose of minimising the size and intensity of future bushfires, should they occur. Controlled burns utilise fire under controlled environmental conditions in a defined area at a predetermined time, intensity, and rate of spread. The reduced intensity and fuel load of a controlled burn allows firefighters to effectively manage the fire and to limit its impact (McCaw, Gould, and Cheney, 2008; Gould *et. al.*, 2007).

The Department of Environment, Parks and Wildlife (DPAW), formerly DEC is responsible for the management of fuel loads in over 26 million hectares, across Western Australia. The land that DPAW manages includes national and regional parks, state forests and timber plantations. DPAW operates a burning program to reduce fuel loads in DPAW-managed estates in the south-western region of Australia by approximately 200,000 hectares per year. To manage these estates DPAW has established procedures for the management of prescribed burns based on traditional bush management concepts.

The DPAW uses a landscape-scale management approach to its prescribed-burning program to ensure that a “mosaic” of fuel-reduced areas is created. Prescribed burns require approval from senior management and other agencies, such as the Department of Fire Emergency Services and the Office of Bushfire

Risk Management, to reduce the risk of a smoke haze to nearby communities. However, despite careful planning it will be inevitable that at some point ash and smoke will impact an urban area. This practice ensures that major bushfires do not develop and progress continuously through high-fuel-load areas towards the rural-urban interface. Research has shown that landscape-scale mosaic burns have slowed the momentum of bushfires prior to arriving at the urban interface (Gill and Williams, 2009). This provides firefighters a greater opportunity to allocate resources to control or extinguish the fire before it impacts on life or property (Keelty, 2011).

3.1 Traditional bush management

Australia is a fire-prone continent that emerged during the Pleistocene era (1.8 million to 10,000 years ago) where the drying climate on the Australian continent changed the biota structure. Australian habitats evolved to more sclerophyllous and pyrolytic habitats where fire became more frequent and subsequently accelerated the flora change process (Burrows, 2008; Oliveras and Bell, 2008).

Eucalyptus species became the dominant vegetative species found in forests and woodlands in Australia. It is possible to study the chronology of fire frequency by studying the number and characteristics of the tree rings, with single trees being up to 400 years old. Eucalypts often survive fire and many reflect distinct ring growth patterns that highlight seasonal variations in temperature and precipitation (Von Platen, Kirkpatrick and Allen, 2011 Attiwell, 1994).

Fire regimes have evolved through three distinct phases:

- unmanaged landscapes – large late season fires
- Aboriginal – small fires in the late dry season
- contemporary management.

It is predicted that initially large natural fires often occurred late in the dry season and burnt large tracts of land. Control of the fire was through natural mechanisms of rain or the fact that no combustible vegetative material remained (Bowman *et al.*, 2007).

The introduction of Aboriginal people dramatically changed the natural fire regime of Australia. The wave of anthropogenic burning altered the distributions of sclerophyll forests while assuring the dominance of *Eucalyptus* and *Acacia*. Although Aboriginal “fire-stick” farming practices varied across the continent the landscape often reflected modifications caused by Aboriginal use of fire. Aboriginal people used fire to clear land for hunting and large fires were used occasionally to generate rain through the formation of pyro cumulus clouds. These rain events increased new plant growth on deliberately burnt areas encouraging fauna for increased hunting prospects. Uncertainty remains as to the extent of these impacts and how they changed the Australian landscape. It is estimated that Aboriginal colonists caused a massive variation from the natural background fire frequency rate in that it triggered a phase shift in the biogeography and climate of the Australian continent (Oliveras and Bell, 2008; Bowman, Dingle, Johnston, Parry, and Foley, 2007).

There is a clear distinction from when Aboriginal people were in control of their country to the introduction of European settlement, especially when agricultural pursuits and mineral prospecting commenced. Fire was seen as the easiest method to clear land. White settlement saw an increase in deliberately ignited fire frequency and severity following the abolition of the Aboriginal socio-economy. Between 1740 and 1850, during indigenous occupation of the Tasmanian eucalypt forest, the fire frequency ranged from an average of 0.4 to 0.7 per 10 years. Between 1850 and 1909 fire frequency increased to 0.8 and 1.2 per ten years then sharply increased again. Between 1910 and 1989 it varied between 1.3 and 1.7 per ten years. Tree ring counts in south-western, Western Australia, indicate that the mean interval between injurious fires in *E. marginate* forests decreased from 80 years to around 17 years after European occupation (Enright and Fontaine, 2014, Von Platen *et. al.*, 2011).

Graziers used fire to clear pastures of unwanted growth. The introduction of vehicles further increased the frequency of ignition as more people gained access to the bush. During the majority of the 20th Century the lighting of fires was seen as an acceptable and legitimate activity. Fire interval studies demonstrate that changes in the human use of fire became more important in determining fire frequency than climatic changes (Von Platen *et. al.*, 2011).

3.2 Contemporary bush management

Historically, fire is an ecological broad-spectrum disturbance of a landscape. Fire influences the physical-chemical environment through the volatilisation of some nutrients, the release of inorganic minerals in ash, and reducing plant cover. Preventive firefighting risk reduction involves biomass manipulation as the fuel

load of litter is the most important aspect of the intensity and severity of a fire. Fire also controls plants, animal species and communities and influences macro-level ecosystem processes and characteristics such as nutrient cycles, diversity, productivity and stability (Gill and Williams, 2009; Alexander, 2006; Gill, 1979).

The arrival of Europeans created a new fire regime, with European-style burning practices. Early European settlement focused on the clearing of land for agricultural and grazing purposes. Fires escaping from pastures and farms combined with high-fuel loads in the surrounding forests often were catalysts for a series of extensive bushfires. The fear of fire also meant continual suppression around the fringes of populated areas (Oliveras and Bell, 2008).

Since European settlement there have been large-scale bushfire events recorded in southern Australia every 2-3 years. As a result of these large bushfires state agencies were created to oversee management of public forests and to impose some degree of regulation. In 1939, a Royal Commission of Inquiry recommended the use of low-intensity fire to mimic past natural or Aboriginal burning practices and to conduct scientific and technical research into the use of fire for land management practices. This fire-based research focused specifically on land management and the use of fire for fuel reduction. It was soon discovered that one of the issues associated with prescribed burns was the fast rate of accumulation of fine fuels, which dictates burning frequency (Penman and York, 2010; Oliveras and Bell, 2008). The issue is compounded by the infrequency of suitable environmental conditions for prescribed burns. This ultimately results in high-risk burns being undertaken with the potential escape of the fire from containment lines (Keelty, 2012).

Current bushfire management practices evolved after the 1960s based on initial research that supported the holistic focus on forestry and other primary industries. Contemporary fire management has become more coordinated in an attempt to balance conservation and health values with fire protection measures. The accessibility of satellite GIS systems showing the pattern of fire scars, increased availability of specialised resources, and a greater understanding of fire in recent years has allowed a more strategic focus to achieve conservation and health outcomes with the protection of economic assets (Bowman *et. al.*, 2007; Gill and Williams, 1996).

Fires are dependent on fuel, heat and oxygen. The first two involve the use of fire, and can be manipulated by prescribed burning. It is recommended that prescribed burns be undertaken in weather conditions where the fire does not burn too much litter, or burn down to the mineral soil, which can lead to other ecosystem disturbances such as wind and water erosion. Optimal conditions for a prescribed burn are just after rainfall periods or early spring so that the deeper humus layer is protected from burning by dampness, while the surface litter is dry enough to be ignited (Blekin, Mysterud, and Mysterud, 2003).

In contemporary fire management prescribed burns are the preferred method of fuel reduction. It is considered the most cost-effective method used to treat large fire-prone landscapes and is less expensive and quicker than the use of machinery. Typically, prescribed burns cost \$25 to \$125 per acre (\$62-309/ha) while machinery typically costs \$250 (\$1,112/ha) and can be as high as \$2,700 per acre (\$6,669/ha) when dozers and aircraft are involved (Alexander, 2006). Often, in contemporary prescribed burns, fire and machinery are used together to

control the fire process. Fire reduces the biomass while machinery is used to make the mineral earth fire breaks that limit the spread of the fire to other areas (Keelty, 2012). Prescribed burns have been defined by Alexander (2006) as following a sequence:

Step 1: Identifying the objectives relating to the site being managed.

Step 2: Developing burning process that acknowledge fire danger ratings, weather conditions, season, time of day, ignition pattern.

Step 3: Timing of the burn to ensure that it is within the desired weather parameters.

Step 4: Inclusion of smoke management considerations.

Step 5: Preparation and execution of the operational requirements of the plan.

Step 6: Review the prescribed burn to determine whether the objectives were achieved, issues arisen and control measures adapted. A pre- and post-burn fuel analysis of photos with quantitative measurements provides a beneficial tool for future planning and burning prescription formulation.

The temperature of the burn is of critical importance. Prescribed burns are low-intensity fires ($<400^{\circ}\text{C}$) which are used to reduce the build-up of leaves and twigs on the forest floor while leaving the larger trees intact. This procedure has proven to be an effective way of reducing the future risks of destructive, high-intensity ($>500^{\circ}\text{C}$) bushfires (Debano and Conrad, 1978). The frequency of prescribed burns in a given area is variable, with intervals between burns ranging from five to 15 years. The preferred seasons are usually during spring and autumn months. The method used to burn depends on the size of the area to be burnt, prevailing weather, objectives for each burn, and the availability of resources. Control lines are established before burns are conducted. The most common method of ignition

is a hand-held drip torch, which is a canister of flammable liquid fitted with a wand and a burner head (De Vos, Reisen, Cook, Devine and Weinstein, 2009).

There are two general strategies for prescribed-burn ignition; random and planned ignition. Random ignition involves managing the effects of a bushfire and allowing the fire to progress to prevent future fires. It is reactive in nature and carries considerable risks in practical terms. Planned ignition of fires includes ignition through preventive measures, such as prescribed burning, and management of bushfires through practices of back burning to limit the spread of the fire. The conflict between ecological considerations and socio-political conditions relating to the management and connection to economic resources prevents the widespread application of planned prescribed burning in the off season. Politically it is easier to prescribe burn during bushfires than to obtain approval to ignite prescribed fires for preventive measures (Mysterud *et. al.*, 2003).

The practical organisation of fire programs are of great importance for the intended results of a prescribed burn. There are numerous examples showing that the combination of overgrazing and burning have resulted in unforeseen circumstances for the forest area. More knowledge must be gained about anthropogenic ignition sources and the overall influence on the ecosystem (Mysterud *et. al.*, 2003).

3.3 Legislative Parameters

The *Environmental Protection Act 1986* (WA) provides the mechanism to undertake prescribed-burn activities through reducing the risk of uncontrolled bushfires. However, the Act also prescribes restrictions to prevent the discharge of

waste in circumstances likely to cause pollution. Table 3.1 details the relevant provisions that relate to the practice of prescribed burns in the *Environmental Protection Act 1986*.

Table 3.1: Description of the relevant provisions of the *Environmental Protection Act, 1986*.

Section	Description of the relevant provisions
3	<p>Beneficial use is defined as the “use of the environment, or of any portion thereof, which is “conducive to public benefit, public amenity, public safety, public health or aesthetic enjoyment and which requires protection from the effects of emissions”</p> <p>Discharge is defined “in relation to waste or other matter, includes deposit it or allow it to escape, or cause or permit it to be, or fail to prevent it from being, discharged, deposited or allowed to escape”</p> <p>Emission is defined as “discharge of waste”</p> <p>Environmental value is defined as “a beneficial use” or “an ecosystem health condition”.</p> <p>Waste is defined to includes matter “whether liquid, solid, gaseous or radioactive and whether useful or useless, which is discharged into the environment”</p>
3A	Pollution means “direct or indirect alteration of the environment to its detriment or degradation that involves an emission”.
4	Binds the Crown.
4A	<p>The object of this Act is to protect the environment of the State, having regard to the following principles:</p> <ul style="list-style-type: none"> • Precautionary principle - when there is a risk of serious or permanent environmental damage, scientific uncertainty should not be the basis for delaying measures to prevent environmental degradation. The decisions must still be guided by an appraisal to avoid environmental damage and the risk assessment must be weighed against the effects of other considered options. There is little argument of the preventive value of a prescribed burn in relation to the consequences of an uncontrolled bushfire. • Principle of intergenerational equity - the status of the natural environment is to be maintained or improved by the current generation for the future generations benefit. • Principle of biological diversity conservation and ecological integrity - the primary consideration of any activity should be the maintenance of ecological integrity and the conservation of biological diversity. • Principles relating to improved valuation, pricing and incentive mechanisms - environmental factors should be built into assets and services valuations. Waste and pollution generators should accept the containment, avoidance or abatement costs. Environmental goals should be followed in a way to take advantage of benefits while minimising costs. • The principle of waste minimisation - operational methods should reduce waste generation and environmental discharges.
5	Where there are inconsistent laws and, the provision of the <i>Environmental Protection Act 1986</i> is inconsistent with a provision contained in another statute law, the provision of the <i>Environmental Protection Act 1986</i> prevails.
6	An exemption can be granted with the Governor’s approval that all or any of the provisions of <i>Environmental Protection Act 1986</i> do not apply in respect to an activity specified in the order.
49	A person who causes or allows pollution commits an offence.
50	A person who deposits waste that causes pollution where it has the potential to gain access to the environment commits an offence. The seriousness of the offence is within Section 50A and 50B of the <i>Environmental Protection Act 1986</i> .
74	A defence mechanism in respect of an emission if the action was to prevent danger to humans or permanent damage to the environment.
74B	A defence for causing pollution or in respect of an emission was a result from an authorised act that does not contravene another statute law and was carried out in accordance with an authorisation or exemption given under other statute laws.

Figure 3.1 shows the links between the legislative definitions to establish the elements of the offence for these provisions and their relationship to public health. The Environmental Protection Act does not provide a definition of public health and this is subsequently defined in the *Public Health Act 2016* (WA).

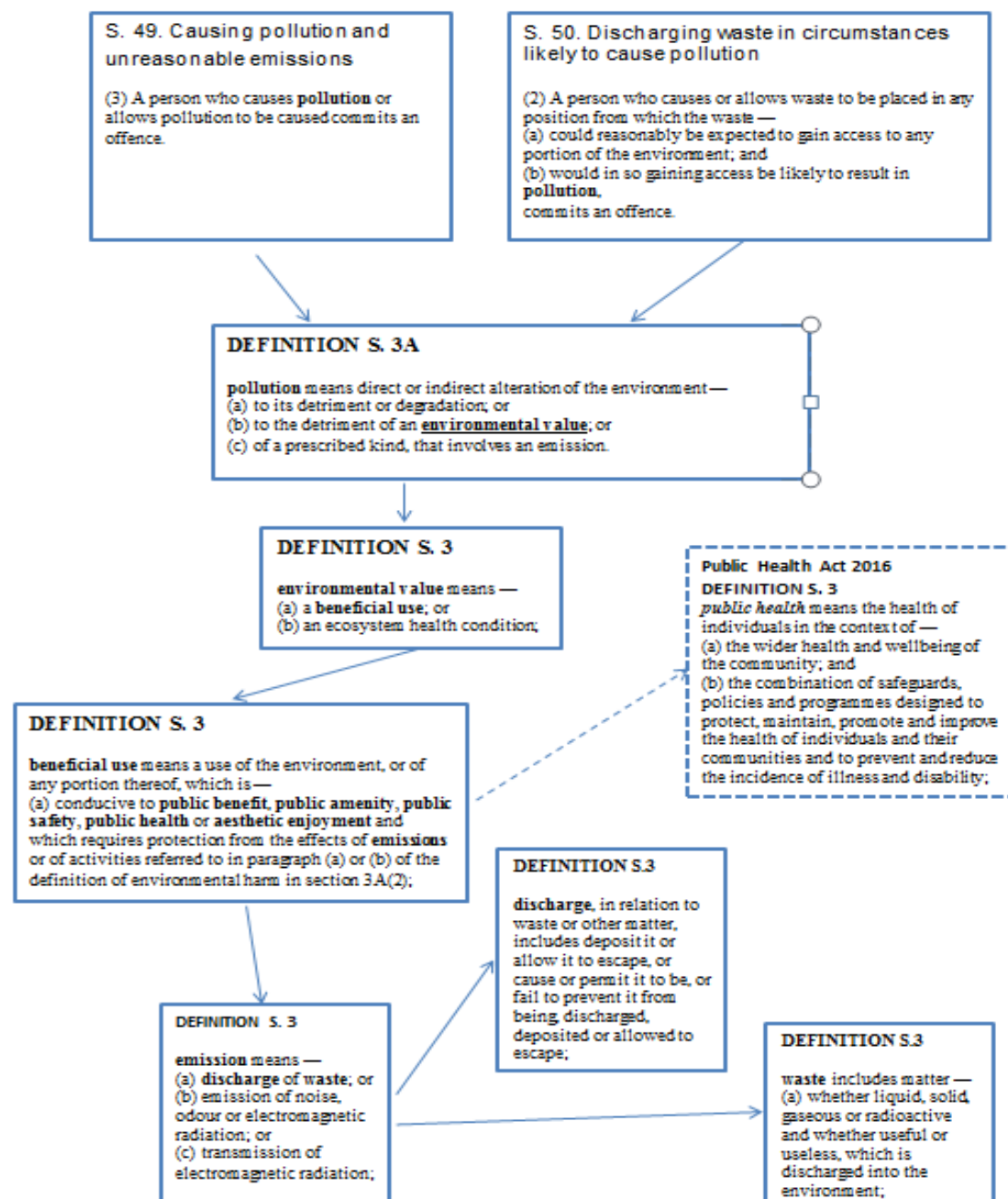


Figure 3.1: Legislative definitions for pollution and discharging waste

(Source: Environmental Protection Act, 1986)

While the *Environmental Protection Act 1986* can be seen as conflicting with the *Emergency Management Act 2005* (WA) [a prescribed burn could be viewed as causing pollution (Section 49) or discharging a waste (Section 50)], there are defence provisions in Section 74 and 74B of the *Environmental Protection Act, 1986* that could permit prescribe burns if the activities meet certain criteria.

Section 74 could apply if it is argued that a prescribed burn is mitigating the adverse environmental effects of an uncontrolled bushfire as mentioned in Section 4A of the *Environmental Protection Act 1986*. Section 74B can apply if the activity is conducted in accordance with the authorisation or approval under another written law.

The *Emergency Management Act 2005* provides the central legislative tool for authorised government agencies to perform emergency management functions in relation to specific technological or natural hazards. The *Emergency Management Act 2005* links with other fire-related Acts to meet its purpose “... *to provide for prompt and coordinated organisation of emergency management in the State, and for related purposes.*” (*Emergency Management Act 1986*, p. 1). The sections that relate to the approval to conduct a prescribed burn are described in Table 3.2.

The *Emergency Management Act 2005* establishes authority through the provisions of the Act and through the creation of various policies and plans. The main state plan for fire is WESTPLAN-Fire. This plan requires the development of sub-management plans that provide specific detail for each prescribed burn.

Table 3.2: Description of the relevant provisions of the *Emergency Management Act 2005 (WA)*

Section	Description of the relevant provisions.
3	<p>Specifies “emergency management” as the management of emergencies unfavourable effects. These management functions focus on four areas and include:</p> <ul style="list-style-type: none"> • Prevention - the reduction of the probability of the occurrence and potential adverse effects of an emergency. • Preparedness - the planning required to respond to an emergency. • Response - combating the effects of an emergency, limiting the damage, and ensuring an effective recovery. • Recovery - the support of affected communities in the rebuilding and reinstatement of societal assets (infrastructure, environmental, psychosocial and economic).
4	<p>Provides for a government agency or person to be authorised as the hazard management agency for a particular hazard because of that agency’s functions under a written law or specialised knowledge, expertise and resources relating to the hazard for which it is prescribed. Regulation 17(h) of the <i>Emergency Management Regulations 2006</i> prescribes the Fire and Emergency Services Commissioner to be the hazardous management agency of structural and bushfires within the state. This allows for the Department of Fire and Emergency Services to have overall authority over the emergency management functions for urban and rural bushfires.</p>
6	<p>Provides a mechanism for a department to be delegated a specific role or “combat agency” authorisation due to its specialised knowledge, expertise and resources to perform specific activities. The combat agency works under that authority of the hazard management agency. Regulation 30 of the <i>Emergency Management Regulations 2005</i> designates Department of Parks and Wildlife as a combat agency responsible for the emergency management activity of bushfire suppression.</p>
7	<p>Binds the Crown in complying with the Act.</p>
8	<p>Provides the power to override other Acts where there are inconsistencies with the <i>Emergency Management Act 2005</i> provisions. The only exemption to this provision is the <i>Fuel, Energy and Power Resources Act 1972 (WA)</i>.</p>
14	<p>Provides a mechanism for the formation of the State Emergency Management Committee (SEMC) to have overall responsibility for emergency management within the state. The committee’s main functions include providing ministerial advice on state preparedness and emergency management to combat emergencies and to develop and manage risk management strategies for community emergency vulnerabilities. This function includes making arrangements for the preparing state emergency management policies (Section 17) and state emergency management plans (Section 18).</p>
20	<p>Allows the SEMC to direct public agencies to prepare and maintain a State Emergency Management Plan for their area of responsibility. In accordance with State Emergency Management Policy 2.2 (2009), the FES Commissioner, as the prescribed hazard management agency for fire, is responsible for the management of the preparation and maintenance of the State Emergency Management Plan for fire. WESTPLAN-fire is the state’s plan for structural and bushfires and details the emergency management arrangements for fire throughout Western Australia. The objective of WESTPLAN-Fire is to assign emergency management functions (preventive, preparedness, response and recovery) that apply to stakeholders who may have a role with fire emergencies.</p>

A requirement of WESTPLAN-Fire is for the generation of an integrated Bush Fire Risk Management Plan (BRMP), which details the bushfire prevention and mitigation measures. The risk management must be underpinned by processes based on AS/NZS ISO 31000:2009 that require risks to be identified, analysed, assessed, treated and reviewed. These plans document the outcome of an application of the risk management process by the involved agencies.

The *Conservation and Land Management Act 1984* (WA) also delegates the authority for an authorised person to light fires for the purpose of preventing or suppressing fires. This authority is provided by Section 104 and Section 120. This authority is expanded through Section 45 that provides an authorised officer with the functions of the *Bushfire Act 1954* (WA). The Act applies to the whole of Western Australia and is used for the prevention, preparedness and response elements of bushfires across all landholder types. This function also links to the designated emergency management functions as identified in the *Emergency Management Act 2005*. The *Bushfires Act 1954* also places a responsibility on “Occupier of land” and/or “owner or occupiers”, including government agencies with a vested interest in land to have statutory obligations to prepare for, prevent, and respond to bushfires on their land.

There is also ambiguity in relation to the *Emergency Management Act 2005* (Section 7) and the *Environmental Protection Act 1986* (Section 4) binding the Crown. Both Acts state that they take precedence where there is a conflict with other Acts (Section 8 of the *Emergency Management Act, 2005* and Section 5 of the *Environmental Protection Act 1986*).

The definition used for “emergency management” is all encompassing in the prevention, preparedness, response and recovery functions and provides a mechanism for precedence. If the Emergency Management Act 2005 takes precedence this means that the provisions of the *Environmental Protection Act 1986* can be seen as secondary in consideration. The Department of Parks and Wildlife (DPAW) would still be required to consider the provisions of other Acts and incorporate management controls to mitigate any potential impacts of their actions. Where a conflict arises DPAW would still need to demonstrate that it incorporated control aspects that consider the provisions of the other Acts when planning or undertaking a prescribed burn.

3.4 Prescribed burn issues

Fire, as a management tool and a process, is generally unappreciated and the benefits of prescribed burns are still debated. The low-intensity burn generates more smoke and ash due to incomplete combustion. This settles in urban areas and water courses and may result in an increase in health-related complaints to government agencies. Pressure is brought to bear on fire management agencies to limit their prescribed burns to weather conditions conducive where wind directs the discharge away from urban areas. This is particularly difficult when urban areas are on the fringe of cities and towns surrounded by the forest (Alexander, 2006; Gill and Williams, 1996).

The overall benefits of prescribed fires are controversial and there are many uncertainties that could impact on the intended final outcome. On the one hand, fires in areas that are “over-saturated” with fuel can develop into “catastrophic fires”. On the other hand, the overuse of preventive fires has exposed problems

including the localised extinction of obligatory seed setters through the elimination of seed banks and the development of areas dominated by weeds. These can be more combustible than the previous species (Mysterud *et. al.*, 2003).

Bowman *et. al.*, (2007) argues that prescribed-burn practices have mistakenly been understood to imitate Aboriginal fire management practices that were modified to accommodate the changes in land management caused by European settlement. This was proven incorrect as the sudden nature of the transition from Aboriginal to European fire management confused the rationale behind the management of prescribed burns, as the initial reasoning for the burns was lost. The imprecise nature of historical records initiated a number of contrasting interpretations about Aboriginal landscape management through the use of fire. Focus centred on the seasonal patterns of burning rather than the long-term frequency of burns. One practice was to clear the land while the other was to effect the phonological and breeding cycles of the flora and fauna. Other indirect management practices, such as the control of feral buffalo grass in the 1980s, caused a subsequent increase in the biomass of flammable grasses and changed the pattern of fuel loads.

The main management issue with prescribed burns is the prediction of the intensity and the overall amount of organic matter burnt. In ecosystems with a mixture of vegetation types there will still be seeds of many plant species remaining dormant in the soil. Low-intensity fires will assist the germination of seeds from the upper layer of the soil, while high-intensity fires may kill the surface seeds and stimulate germination of those buried deeper. Species population can rapidly decline if the fire frequency is not managed correctly or if another fire

commences before there is opportunity for new generations of seedlings to develop (Cheney, Gould, McCaw, and Anderson, 2012, Mysterud *et. al.*, 2003).

Balancing the benefits against the consequences of prescribed burns is also important. Burning outside or on the fringe of mild burn seasons has the potential for devastating ecological and community effects. Uncontrolled prescribed burns can also alter the habitats for wild animal species. The distribution of burnt and unburnt regions after a fire is an important characteristic that fire management authorities must consider. Instead of removing all the fuel from an area, the fire objective may be to burn only 40% of the area to produce a desired mosaic of burnt and unburnt vegetative landscapes. This would allow the creation of unburnt refuges that provide animals with the opportunities for movement and escape from the prescribed fire front (Mysterud *et. al.*, 2003; Sherwin *et. al.*, 2003).

Fires outside the high-fire season, when the litter is still moist and/or the climatic conditions are more moderate, will have a lower intensity and also develop more as a mosaic pattern than fires during time periods where the vegetation is at its driest. The fire mosaic will depend on the history of earlier fires but the litter beneath some tree species will differ in flammability, size, degree of aeration and moisture content producing local variations in fire intensity. Burning depth of the litter varies with the fire intensity but largely relies on the depth of the ground vegetative litter layer (Blekin *et. al.*, 2003).

The incorrect assessment of burnt environments can result in changes to the growth patterns of trees, shrubs and grasses. Seedlings can be destroyed prior to the development of their fire resistant characteristics. Incorrect burning can also limit the non-fire organic nutrient replenishment process as there is insufficient

time between the fires to produce sufficient biomass for degradation (Blekin *et. al.*, 2003).

3.5 Summary of literature relating to prescribed burns

Prescribed burning does not substitute other effective fire suppression practices, but it is seen as the most resource effective method to control large areas of land. The risks associated with prescribed burning are inherent within the burning process. Consultation with the community is essential for a successful prescribed-burning program. Regulatory constraints, such as those associated with pollution and conservation concerns, need to be discussed and community concerns must be alleviated. This may include identifying the risks associated with not burning an area and comparing the comparative costs.

Chapter 4: Motivation for prescribed burns

It is becoming apparent, through increased media attention and changing lifestyles, that bushfires pose an extreme risk to Australian communities. An increasing issue for communities is when bushfires occur in the rural-urban interface and areas of economic importance, causing loss of life, damage to property, and livestock loss (Gill and Williams, 2009). Australia experiences at least one major bushfire annually, often occurring in the same location on a generational cycle (Keelty, 2011).

4.1 Bushfires in Australia and Western Australia

Some fires were so significant that they become embedded in Australia's history. On 16 February 1983, the Ash Wednesday series of 180 bushfires occurred in south-eastern Australia, causing widespread destruction in Victoria and South Australia. Seventy-five people died in these fires including 17 firefighters, 8000 people were evacuated, and South Australia declared a state of disaster; for the first time in the State's history (Emergency Management Australia, 2012a). These fires were the deadliest bushfires in Australian history until the Black Saturday bushfires of 2009 that caused the death of 173 people. At the time, Victoria was experiencing one of its most severe and prolonged heatwaves. Temperatures in Melbourne exceeded 43°C for three consecutive days, which was the first time this had occurred since recordkeeping started. On the 7th of February 2009 temperatures were forecast to reach the low 40's combined with strong winds. The landscape was described as 'tinder dry' and the forests and grasslands were the driest they had been since the Ash Wednesday fires in 1983. On the day,

temperatures reached to the mid-40s with Melbourne recording a high of 46.4°C. A total of 316 fires were recorded in Victoria on that day with the most serious consequence being the deaths of 173 people. Approximately 450,000 hectares were burnt, over 2 000 houses were destroyed, and 12 000 head of livestock were lost. There were also substantial environmental impacts. It is estimated that the cost exceeded \$4 billion. This was Australia's worst bushfire disaster and one of Australia's worst natural disasters on record (Teague *et. al.*, 2010).

The cost attributed to the Ash Wednesday fires made it Australia's costliest natural disaster. Over 310 000 hectares was burnt, 3 700 buildings were destroyed or damaged, and 340 000 sheep and 18 000 cattle were lost. Over 4 540 insurance claims were paid totalling \$176 million with a total estimated cost of well over \$400 million in 1983 monetary values (Emergency Management Australia, 2012a).

It would be a mistake to think of Ash Wednesday and Black Saturday as 'one-off' events. Other fires such as the fires in Canberra in 2003 destroyed over 500 homes on the outskirts of Australia's capital city (Emergency Management Australia, 2012b). There are numerous recordings of major fires throughout Australia's colonial history. With increasing populations at the rural-urban interface and the impact of climate change the risks associated with bushfires are likely to increase (Gill and Williams, 2009; Drollette, 2005).

Western Australia also has a long history of large bushfires. In 1961 the Dwellingup fire destroyed 160 homes and burnt through 1.8 million hectares of land with the township almost destroyed. In 1978, Cyclone Alby caused hundreds of fires through lightning strikes resulting in two lives lost and 114,000 hectares burnt (Collins, 2006). In 1997 the Wooroloo fire destroyed 16 homes and burnt

through 10,500 hectares of bushland. The Toodyay fire of 2009 destroyed 38 houses and caused more than \$50 million in damages. In 2011 the Margaret River fires destroyed 32 homes and disrupted the community (Keelty, 2012). The 2014 Parkerville fire burnt down 57 homes and 1 386 people were evacuated. The Northcliffe and Esperance fires of 2015 resulted in four deaths and over 300,000 hectares of farmland burnt (Emerson and Mercer, 2016). The most recent catastrophic fire occurred on 4 January 2016. A bushfire in the Shire of Harvey killed two people, destroyed 180 properties and 70,000 hectares of farmland (ABC, 2016).

4.2 Fire ecology

Fire is an important natural component of the Australian ecosystem. Fire behaviour in Australia is closely associated with its natural and subsequent human histories. Southern Australia's dry climate (Figure 1) and meteorological conditions consist of high temperatures, low humidity, and strong summer wind patterns. This, combined with flammable fuel types and ignition likelihood, creates risks of large fires throughout the Australian landscape (Oliveras and Bell, 2008).

The external elements influencing fires are inter-connect in complicated ways. Fire severity is a measurement of the total available consumption of organic material by the fire and this correlates to the devastation of fauna and flora as the fire severity increases. The progress of a fire will be modified through interactions with fuel (quantity, vertical and horizontal spatial distribution, flammability and energy content), weather conditions, topography and burn history. Due to these factors, it is difficult to accurately estimate how much fuel is going to be consumed (Blekin *et. al.*, 2003; Perez, 1998).

Lightning accounts for 50% of fires that have ignited ecosystems and, historically, has played an important part in their shaping. More recently, the majority of fires have been caused through anthropogenic ignition sources to the point where human activity has changed various aspects of the “natural” fire regime. In the USA, only 9% of fires are considered to be by caused natural mechanisms. Activities connected with change to agrarian-based settlers have increased the anthropogenic ignition frequency significantly (Blekin *et. al.*, 2003).

Fires from anthropocentric sources account for the majority of fire-ignition sources and are classified as accidental or deliberate. Deliberately-lit fires can result from arson or be designed to achieve a beneficial agricultural outcome or through prescribed burns. The risks are that changes in weather conditions could result in uncontrollable bushfires. Regrettably anthropocentrically lit fires are more common near populated areas and pose a much higher risk of damage to infrastructure (Keelty, 2012).

In the short-term a fire’s impact on the environment can seem severe. There is vegetation loss and shrub and ground cover species may be completely removed. The foliage of tree species may be scorched or removed and trees may be killed or severely scarred. Some animals may be killed by the fire or subsequently die due to the lack of food or from increased exposure to predators. The soil may become exposed and subject to erosion and waterways may become silted. Provided the vegetation is fire-adaptable these effects are usually transitory and of no long-term ecological significance (Commonwealth of Australia, 1984). Fire-adapted vegetation relies on fire to regenerate and maintain its particular species composition, structure and function. The adaptive processes vary in intensity, burn

duration, frequency and distribution of existing vegetation types. Where the fire frequency alters from the normal cycle, as with anthropocentric sources, this can result in the destruction of the fire-adapted vegetation in that area and the introduction of new species (Blekin *et. al.*, 2003).

4.3 Land use factors

Land use planning does not directly deal with the public health risks of uncontrolled fires and prescribed burns but there is an indirect connection through increasing populations in high risk peri urban areas. The area (2188 sq. km.) of this study consists of three adjacent peri urban Local governments with a total population of approximately 105,000 people. The majority of the 39,000 dwellings in this area are located adjacent to or on the escarpment or within the hills area (West Australian Local Government Association, 2016).

The intent of the Western Australian State Planning Policy 3.7- Planning in bushfire prone areas is to implement development that preserves life and reduces the impact of bushfire on property and infrastructure (WAPC, 2015). This focus on development in high risk areas may be due to historical and financial constraints to deal with property rights that were previously sold to the public and a reluctance of governments to purchase back properties or the consideration of other competing social factors. These peri-urban localities are often considered desirable places to live, characterised by lower-density development and scenic landscapes. Despite the increasing appeal to be near nature many people remain unaware of the risks and ongoing obligations associated with living in these fire prone environments (Bond and Mercer, 2014). Figure 4.1 highlights the interaction with development (in yellow) and high risk fire prone areas (in pink).

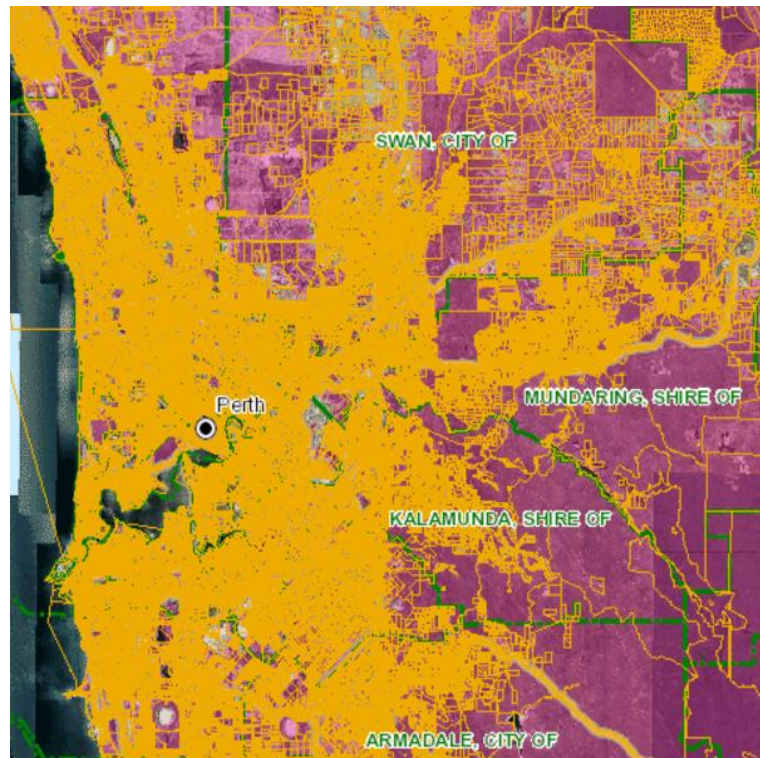


Figure 4.1: Map of bush high risk fire prone area in the Perth metropolitan area.

(Source: Landgate, 2016)

As the Perth metropolitan area grows, more people will locate in its peri-urban region; both on the metropolitan fringe, in nearby towns, and interspersed with a variety of rural residential lot sizes in this transitional area. The land use characteristics are neither fully urban nor entirely rural and have a unique character and function. The rural aesthetics and proximity to the city attract settlement in these areas. Quality of life factors are key drivers of peri-urban popularity with the amenity value enhanced by availability to a major city, enabling increased work and social links (Buxton, Haynes, Mercer, and Butt, 2010).

Urban development has led to a potentially dangerous mix of an expanded population in areas susceptible to fires which could lead to loss of life and property through fire. Development has occurred where little was understood of the ecology of the area. Many assumptions about the nature of the landscape are inaccurate

due to a lack of knowledge and information available as well as cultural notions learnt in other environments (Buxton *et. al.*, 2010).

Land use planning can play a central role in reducing bushfire risks through locational decisions for housing and increased construction standards in areas of medium and high fire hazard (Buxton *et. al.*, 2010). The “Black Saturday” royal Commission identified that subdivision design was a major component that contributed to loss of life and property in Victorian fires of 2009 (Bond and Mercer, 2014).

However, current planning controls in peri-urban areas are not suited to the land characteristics. Often development includes a high proportion of properties with sizes between 0–2 hectares. The small lot sizes increase the possibility of exposure to fire. Increasing migration to these areas where people build on existing lots and newly subdivided lots will increase the number of dwellings exposed to serious bushfire danger (Buxton *et. al.*, 2010). Policies that mitigate bushfire risk through subdivision design are weak with little prescription on setbacks; proximity to vegetation, risk based urban design, ridge-top development, and buffer zones (Bond and Mercer, 2014).

Land use planning and public policy are critical for mitigation against natural hazards and the risks they pose from future disaster losses. Impacts between key factors potentially increasing regional vulnerability to adverse risks are not being addressed. In particular, the connections between population increases, natural resource use, biodiversity loss, water security, human health, emergency management and governance. Even though these measures have been recognised through statutory planning controls for over 20 years. In practice

dwellings are still constructed in close proximity to forests through the approval of subdivisions on land containing significant vegetation. It is seen that policy fragmentation between Government institutions is obstructing the ability of governments at all levels to develop suitable policies and processes to adapt to changing circumstances. This fragmentation is characterised by ad-hoc discretionary land use planning systems (Bond and Mercer, 2014; Buxton *et. al.*, 2010).

A more holistic approach to risk based planning is required that includes a prescriptive element as well as a review of the capacity of the relevant controls to manage risk. This will require a more systematic and scientific approach which mandates uses of the precautionary principle of anticipated change and uncertainty is essential to minimise risks and avoid property damage and loss of life from bushfire (Buxton *et. al.*, 2010).

Unfortunately, the use of planning statutes to segregate dwellings from combustible landscapes does not have political support and is at odds with the amenity values that the public places on living in peri urban landscapes. This cultural attitude has created the public perception that the risks associated with natural landscapes must be controlled which hampers debate about options for preventing people for residing in these areas. Short term mitigation strategies such as prescribed burns are then used to reduce the perceived risk of an impact from an uncontrolled fire (Bowman and Boggs, 2006).

4.4 Climate change and fire severity

4.4.1 Fire severity

The Fire Danger Index (FFDI) offers the foremost assessment of potential rate of spread and fire intensity in Australia. It originates from the studies of rates of fire spread utilising a mixture of wind speed, temperature, relative humidity and a drought index. The FFDI ranges from 0 to 100 but can be categorised beyond that limit and values >100 are conceivable during catastrophic weather conditions as seen with the 2009 Victorian fires recording a value greater than 150 (Enright and Fontaine, 2014). Figure 4.2 illustrates the Australian fire seasons with the South West and South East of the country recoding the highest risk to fire events.

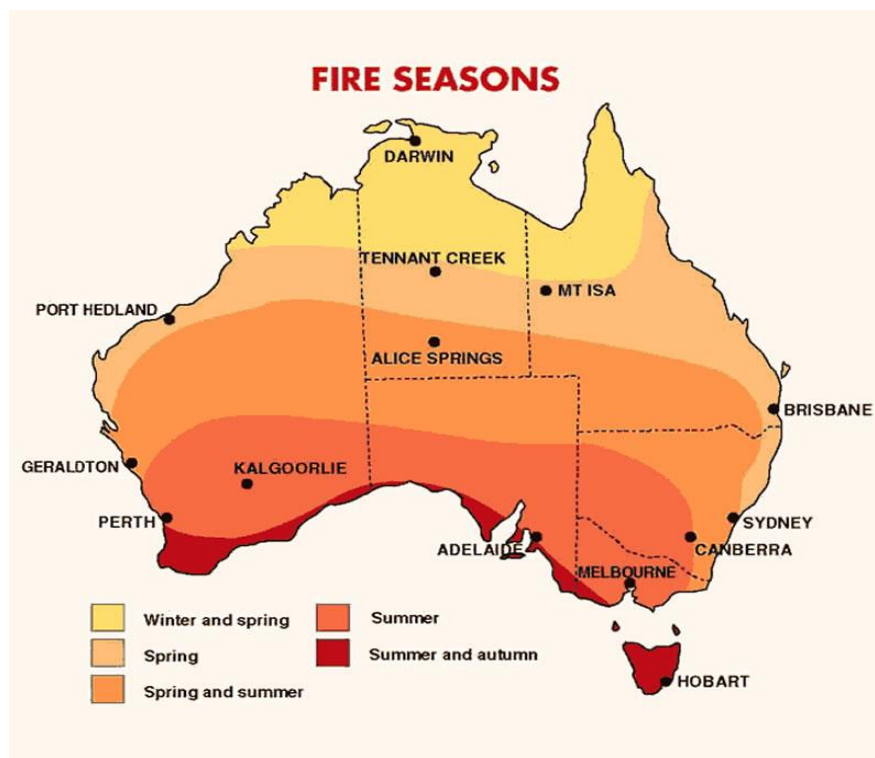


Figure 4.2: Australia's fire seasons highlighting summer and autumn extreme periods in Southern Australia

(Source: Geoscience Australia, 2016).

McCarthy and Tolhurst (2001) studied over 100 bushfires in south eastern forests for the 1990s period and determined as the fire danger increases, the gains from mitigation and suppression strategies reduces. Weather conditions become more significant than fuel parameters in terms of successful risk reduction operations. While uncontrolled fires differ between regions, resources are often concentrated near urban areas and not in natural forests resulting in reliance towards preventative measure such as fuel load management. The current premise is that response mitigation strategies should be directed to protecting human life and assets but these high-risk environments necessitate improved preventative interventions. The Roleystone and Margaret River fires of 2011 have raised uncertainties about the effectiveness of fire mitigation strategies in the southern areas of Western Australia, and highlight this complex relationship between fire, people and the environment (Enright and Fontaine, 2014).

4.4.2 Climate change

Australia is currently experiencing consequences of climate change. Since 1910 Australia has warmed 0.9°C, greater than the international average of 0.7°C for the equivalent period. The bulk of this warming has occurred after 1950, with each decade being warmer than the former; the amount of record-breaking hot days has also risen every decade since the 1950s (Head, Adams, McGregor, and Toole, 2014). This increased warmth has altered climate zones for 10.5–29.5°S south by over 100km along the west coast of Australia. The southwest of Western Australia has also experienced a steady regression in rainfall since the mid-1970s with a 15% decrease in seasonal rainfall. This has resulted in a 60% decline in river flows (Enright and Fontaine, 2014; Head *et. al.*, 2014).

Historically climate trends have been warming, with rainfall increases in northern areas and rainfall decreasing in southern areas. For the forty-year period to 2000, 30% of years were warmer and drier than the previous overall average resulting in the mean fire rate of spread being 66% higher in the December warm-dry years (Matthews, Sullivan, Watson, and Williams, 2012). Forecasted climate changes will have major impacts on Australia's agriculture, flora and fauna. In southern areas life threatening weather events such as bushfires are likely to increase in intensity and frequency, mostly clustered in summer (Head *et. al.*, 2014). Many people believe that the prospect of preventing climate change entirely has lapsed and increases in the intensity and frequency of climate related events will create new demands and vulnerabilities (Enright and Fontaine, 2014; Head *et. al.*, 2014)

Current climate change predictions for southern Australia include even higher temperatures with reduced rainfall periods resulting in longer dry spells. This is likely to increase the fire risk in areas that are already susceptible to high-intensity bushfires (Hollis *et. al.*, 2011; Keelty, 2011; Drollette, 2005).

Future climate change influences will intensify many of the current trends as it is expected that by 2020 Australia's climate is will dry further by 10% and 32% by 2050 with significant increases occurring in the southern summer fire season (Enright and Fontaine, 2014; Matthews *et. al.*, 2012). Fire behaviour is controlled by topography, fuel and weather. Of these, fuel and weather are both vulnerable to climate change (Matthews *et. al.*, 2012). The increased occurrence of fire signifies shorter fire intervals and is likely to have major impacts on the age-distribution, composition and structure of forests (Enright and Fontaine, 2014; Williams, Karoly, and Tapper, 2001).

Society is confronted with the challenge of increased risk of uncontrolled fires due to increased fire conducive weather, and asset development in fire prone areas (Enright and Fontaine, 2014; Tonmoy *et. al.*, 2014). There will be more high risk fire days and the moisture content on those days will be lower, with corresponding adverse implications for fire behaviour (Matthews *et. al.*, 2012).

The overall effects on fuel load are uncertain, a warmer and drier climate in southern Australia could lead to reduced overall plant growth and slower rates of litter accumulation (Matthews *et al.*, 2012). Increasing atmospheric CO₂ improves plants water use efficiency which may lead to increased fuel loads due to reduced organic breakdown as a result of a drier climate (Enright and Fontaine, 2014).

Climate change is anticipated to produce modifications in both the extent and composition of vegetation profiles. Fuel load patterns are likely to remain similar to current levels, so that weather will increasingly become the significant factor influencing fire behaviour (Enright and Fontaine, 2014). A reduced winter period associated with lower fuel moisture levels and greater number of fire days will result in an escalation of the rate of spread thus having an adverse impact on fire behaviour (Matthews *et. al.*, 2012).

4.4.3 Prevention measures

Prescribed burning is premised on the theory that fuel age and accumulation rate is important in deciding the fire danger. Fuel loads normally grow with the time since the previous fire, increasing quickly in the first few years until reaching equilibrium where gains from litter production are counterbalanced by losses through decomposition. Because of this, the effectiveness of prescribed burns in

relation to bushfire control decreases as the time since last fire increases (Enright and Fontaine, 2014)

There are a numerous restrictions placed on the areas that can be managed by prescribed burns. These include the planning and endorsement procedures, weather conditions permitting burn activity, potential smoke pollution, and level of resourcing. Weather conditions are important as prescribed burns cannot commence under elevated fire risk weather conditions due to the risk of fire escaping, so the number of burning days available may be low relative to the required area that needs to be burnt (Enright and Fontaine, 2014).

In the forested areas of south west Australia, DPAW plans to burn around approximately 8% or 200 000 Ha of the jarrah (*Eucalyptus marginata*) forests annually to keep fine litter fuel load below 8 t Ha⁻¹. Increasing the required burn areas to achieve specified management objectives may not be. Recurrent fire intervals of less than 5 years in forests in the south west of Australia are likely to lead to biodiversity losses and vegetation structure alterations (Enright and Fontaine, 2014).

Climate change is predicted to increase the number of catastrophic fire days per year resulting in higher fire intensities compounding the perceived necessity for further prescribed burn activities with shorter intervals. Fire intervals of less than 10 years are liable to cause vegetation losses, especially after drought years. With increased burning, the understorey of the forest is substituted by shrubs and grasses so that fine-fuel accumulation rates are likely to increase rapidly increasing the risk of high-intensity fires that could occur every 2 to 4 years (Enright and Fontaine, 2014).

Chapter 5: Methodology

This chapter outlines the methodology of the study of ground leaf litter and ash generated from the fire. It includes study design, the area under study, approval requirements and the study methodology.

5.1 Study design

This cross-sectional study concentrates on the potential public health impacts from metals that may be found in fine ash following a prescribed burn. The literature suggests that fine ash is likely to be mobilised by either wind or water runoff affecting nearby communities or water supplies with potentially cumulative metal exposure to receptor populations. NEPM health impact levels in soil provides the most relevant toxicity thresholds for metal contamination in ash and are commonly used as a measure of contamination of a specific area.

This study relied on the collection and analysis of samples of pre-fire ground leaf litter and post-fire ash collected within a prescribed burn on the fringe of the Perth metropolitan area. This provided an assessment of the metal content of vegetation and ash with limited contamination from other sources. Ash was collected as soon as possible after the fire had swept past the area in order to collect the fine ash particles before they were blown away. It was noted through the literature that even one day after the fire there was a reduction in the amount of fine particles (Carroll, *et. al.*, 2007; Zavala, Jordan, Gil, Bellinfante and Pain, 2009; Debano and Conrad, 1978).

The ash study was to allow an estimate of how much metal might be released by a specific burn by comparing the difference between metals in the original ground litter material and that found in the final ash. Also the study can provide an indication of how metal concentrations may differ with changing size parameters. These results can then provide a link between the temperature of the fire (by ash colour), particle sizes and metal concentrations.

The study area provided a unique opportunity to assess the impact of a fire on Western Australian vegetation near a metropolitan area. Samples were collected from the site during an active prescribed burn to reduce the influence of other contaminant sources. Prescribed burns were chosen for this study for a number of reasons:

- It is possible to prepare the equipment in advance, as it is known when and where the burns will commence and be undertaken.
- The unpredictable nature of bushfires (less predictable than prescribed burns) means it is too dangerous to be located at the head of the fire during bushfire fighting operations.

Prescribed burns are manageable events and the results of the study can provide additional information that can contribute to the overall management and effectiveness of the burn.

The original aim of this study was to determine the concentrations of a variety of air pollutants, including particulate matter Total Suspended Particles (TSP (50 - 100 μm) (EEA, 2008), Particulate Matter (PM_{10}), Particulate Matter ($\text{PM}_{2.5}$), Particulate Matter (PM_1) and metals in smoke generated during prescribed burns

in the Darling escarpment area surrounding Perth's metropolitan area. This information gained from the original study of particulate matter and metal concentrations was intended to be correlated with volatile organic carbon and carbon monoxide levels and establish a relationship between the levels of particulate matter and gases generated during a fire. These parameters are easily monitored using field based scientific equipment.

5.1.1 Size fraction determination

This study has been designed to establish whether there is a relationship between metal concentrations and size fraction in ash. The selection of the sieve sizes (1 mm, 2 mm and 4 mm) in this study was based on the procedures used in previous similar studies (Smith, *et. al.*, 2011a; Zavala *et. al.*, 2009; Carroll *et. al.*, 2007; Johnson, Murphy, Walker, Glass, and Miller, 2007; Neill, Patterson, and Crary, 2007; Earl and Blinn, 2003; Debano and Conrad, 1978). In their studies of erosion events for analysis as it was considered that the 2 mm size fraction best related to wind and water erosion. Noske *et. al.* (2010) in the study of phosphorus in stream sediment used a 1 mm sieve and a 5 mm sieve to separate samples, <1 mm samples were defined as fine material and samples between 1 mm and 5 mm were classified as coarse samples. Materials greater than 5 mm were discarded.

This study selected three sieve sizes (1 mm, 2 mm and 4 mm sieves) to separate the samples. This selection covers a spectrum of coarse and fine materials from less than 1 mm, 1-2 mm and 2-4 mm samples, material larger than 4 mm was discarded as this was considered outside the scope of the study and little relevance to air and water mobility.

5.2 Ash study area

The Darling escarpment is an escarpment parallel to the Western Australian coastline east of the Swan Coastal Plain and Perth metropolitan area. The escarpment extends north from Bindoon to south of Manjimup and towards Toodyay and Boddington. A network of reserves and regional parks in this area protect Perth's water supply and the natural component of the escarpment. The Swan Coastal Plain, which contains the Perth metropolitan area, is a low-lying, relatively flat region 30 kilometres wide stretching from the Indian Ocean directly eastwards to the Darling escarpment. It runs from Cape Naturaliste in the south to the north of Perth. The plain consists of sandy soils, river estuaries, and a number of wetlands. A number of rivers cross the plain from the Darling escarpment towards the sea including the Swan and Canning rivers. Rainfall ranges between 600 and 1000 mm annually. The climate is Mediterranean (Mitchell, Williams, and Desmond, 2002). Figure 5.1 identifies with study samples sites along the escarpment.

Open eucalypt forests are the predominant type of forest in the Darling Escarpment area with jarrah (*Eucalyptus marginata*) being the dominant species. In the early 2000s, Greenmount National Park and John Forrest National Park were repeatedly burnt by bushfires. Arson was suspected as the major cause of these fires. In 2004, the largest bushfire in 100 years burnt through these eucalypt forests and created substantial issues for forest management, tourism, and the surrounding communities. To mitigate the fire risk, the Western Australian Government increased the number of prescribed burns in the area to reduce the fuel load (Department of Planning and Urban Development, 1993).

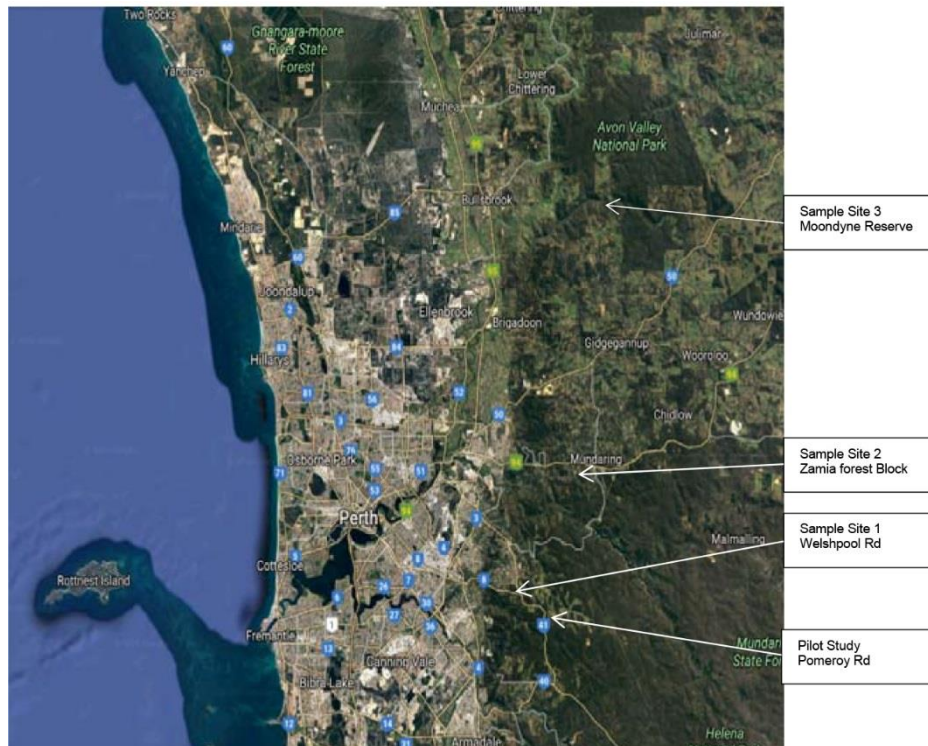


Figure 5.1: Map of Perth Darling escarpment with sample site locations

Three jarrah sampling areas were chosen for this study to represent the diverse influences of populated areas adjacent to open jarrah forests across the Darling Escarpment. Two of the sites border the outer Perth metropolitan areas of Lesmurdie and Mundaring and one site was located in a predominantly rural area 16 kilometres north-east of Bullsbrook. The details of each sampling site and general burn characteristics are presented in Table 5.1.

Table 5.1: A summary of site details and prescribed burn characteristics

	Pilot Study	Sample Area 1	Sample Area 2	Sample Area 3
Site Name	Pomeroy Rd	Welshpool Rd	Zamia	Moondyne NR
Location	Bickley	Lesmurdie	Mundaring	Bullsbrook
Local government	Kalamunda	Kalamunda	Mundaring	Toodyay
Burn ID	PHS 806	PHS 807	PHS 204	PHS 202
Area	20 Ha	50 Ha	1100 Ha	2170 Ha
Perimeter	2.1 km	3.5 km	20 km	23 km
Year last burnt	1941	1961	1997	1997
Fuel Type	Northern Jarrah	Northern Jarrah	Northern Jarrah	Jarrah/ Wandoo
Fuel Load (Ave)	14.7 t/Ha	19.3 t/Ha	12.5 t/Ha	9 t/Ha
Fuel Load (Max)	25 t/Ha	51 t/Ha	15.6 t/Ha	11 t/Ha
Litter depth (Ave)	25.1 mm	39.2 mm	19.1 t/Ha	11.2 mm
Rate of Spread	38-44 m/hr	38-44 m/hr	28-34 m/hr	38-46 m/hr
Flame Height Max	8	8	6	8

5.2.1 Pilot Study: PHS 807 Pomeroy Road, Bickley

The pilot study prescribed burn was located on Pomeroy Road within a number of freehold and crown reserve parcels within the Bickley townsite (Perth Hills Site 806). The site consists of jarrah forest and is 20 hectares in size with a perimeter of 2.1 kilometres. The last prescribed burn was in 1961. The risk management objectives of the prescribed burn were to minimise the potential size and intensity of uncontrolled bushfires and reduce the risk of damage to the Bickley, Carmel and Walliston town sites. The aim was to burn over 80% of the forest under prescribed burn conditions.

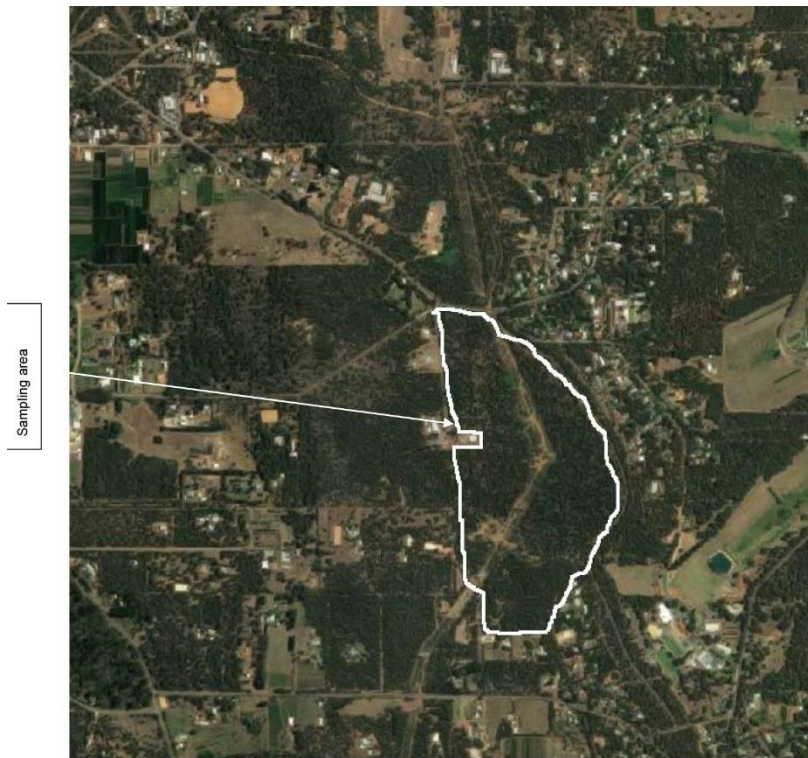


Figure 5.2: Map of PHS 806 Pomeroy Road, Bickley burn area and sample site location.

5.2.2 Sampling area 1: PHS 807 Welshpool Road, Lesmurdie

The Welshpool Road prescribed burn was located at the corner of Canning Road and Welshpool Road in Lesmurdie (Perth Hills Site 807). The site is managed by the Water Corporation and consists of jarrah forest. As seen in Figure 5.3, the burn site is approximately 50 hectares in size with a perimeter of 3.5 kilometres and was last burnt under prescribed fire conditions in 1961. The risk management objectives of the prescribed burn were to minimise the potential size and intensity of uncontrolled bushfires and reduce the risk of damage to key public assets in the area (Channel 10 tower, water pipelines), nearby private properties, and the

community. The aim was to burn over 70% of the forest under prescribed burn conditions.

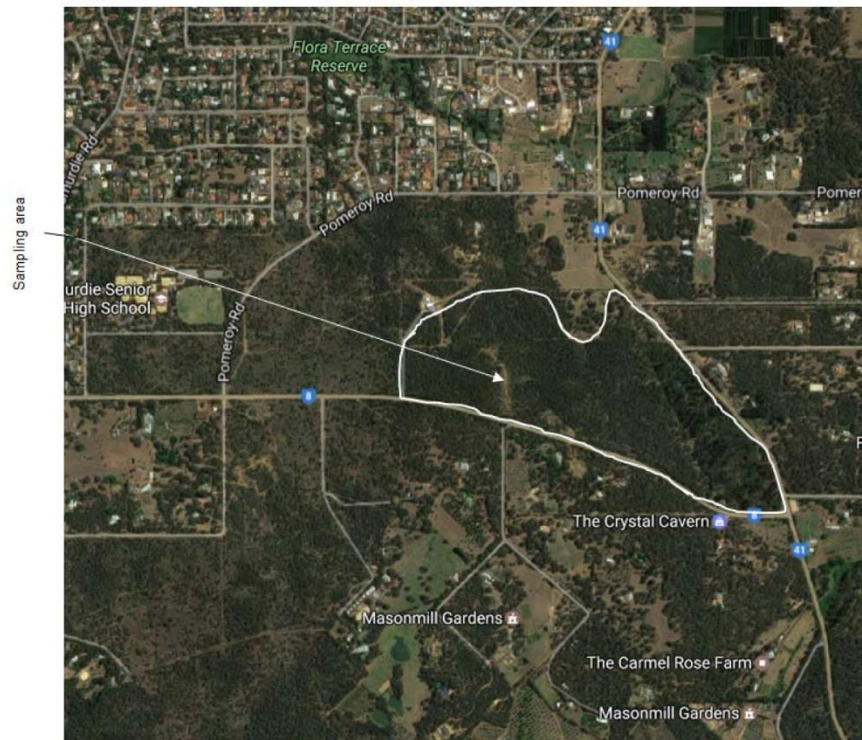


Figure 5.3: Location of PHS 807 Welshpool Road, Lesmurdie burn area and sample site location.

5.2.3 Sampling area 2: PHS 204 Zamia Forest block, Mundaring

The second sampling area was located within the Zamia Forest block (Perth Hills Site 204); part of the Beelu National Park. It is located adjacent to the south side of Mundaring and is near both the Darlington and Pauls Valley settlements. As seen in Figure 5.4 the area is situated within the Helena Valley on the north side of the Helena River. The area consists primarily of northern jarrah forest and is approximately 1100 hectares in size with a perimeter of 20 kilometres. The site

was last burnt under prescribed burn conditions in 2005 and parts of the property were burnt in 1997.

The risk management objective of the prescribed burn was to minimise the potential size and intensity of unplanned bushfires and to reduce the risk of damage to the Zamia Forest block and Beelu National Park, adjoining private properties, electrical infrastructure, and nearby town sites. These objectives were to be achieved through the application of fire under controlled conditions to reduce the quantity of combustible materials and to create a mosaic of fire intensities of burnt and unburnt areas at both landscape and local scales.

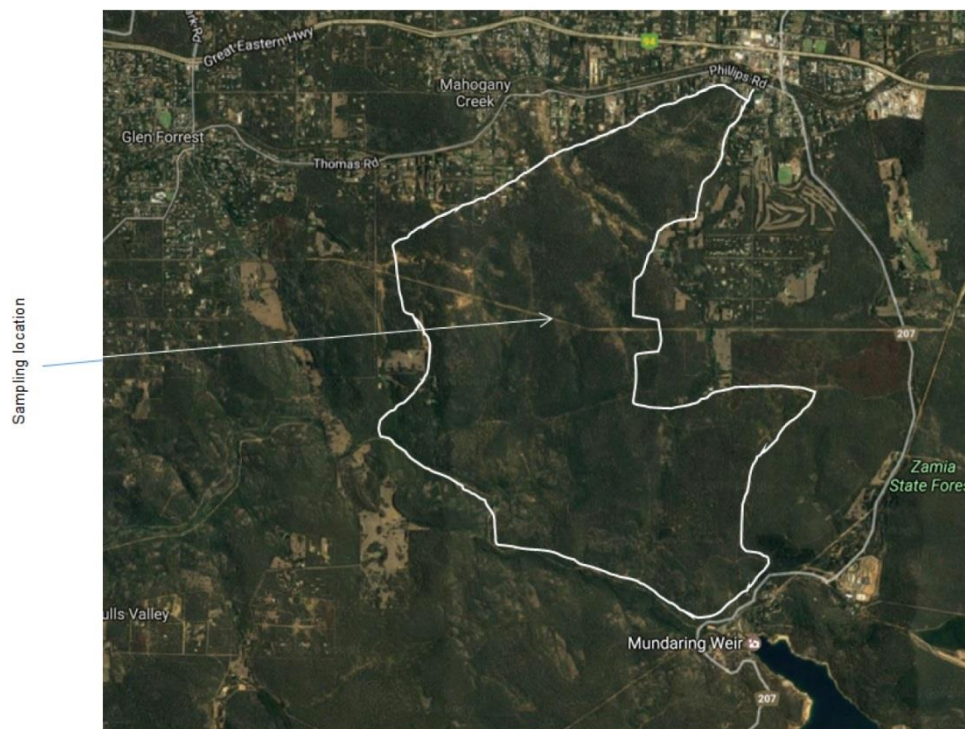


Figure 5.4: Location of PHS 204 Zamia site, Mundaring burn area and sampling location

5.2.4 Sampling area 3: PHS 202 Moondyne Nature Reserve, Bullsbrook

The third sampling area was the Moondyne Nature Reserve located 16 kilometres north of the Bullsbrook town site (Perth Hills Site 202). As seen in Figure 5.5 the site is approximately 2170 hectares in size with a perimeter of 23 kilometres. The vegetation type is primarily northern jarrah, marri and wandoo woodland and was last burnt under prescribed burn conditions in 1997. Sappers Road, Coffees Road, Keating Road and firebreaks adjoining private property surround the burn area. Due to the size of the property, the burn was ignited by aircraft incendiary ignition.

The objectives of the prescribed burn were to protect, maintain and enhance biodiversity within the precinct of the Avon Valley National Park and to reduce the potential damage to other properties by bushfires originating from or passing through the Moondyne National Reserve. These objectives were achieved through burning under controlled conditions to reduce the quantity of combustible materials, thus creating a mosaic of vegetation landscapes.

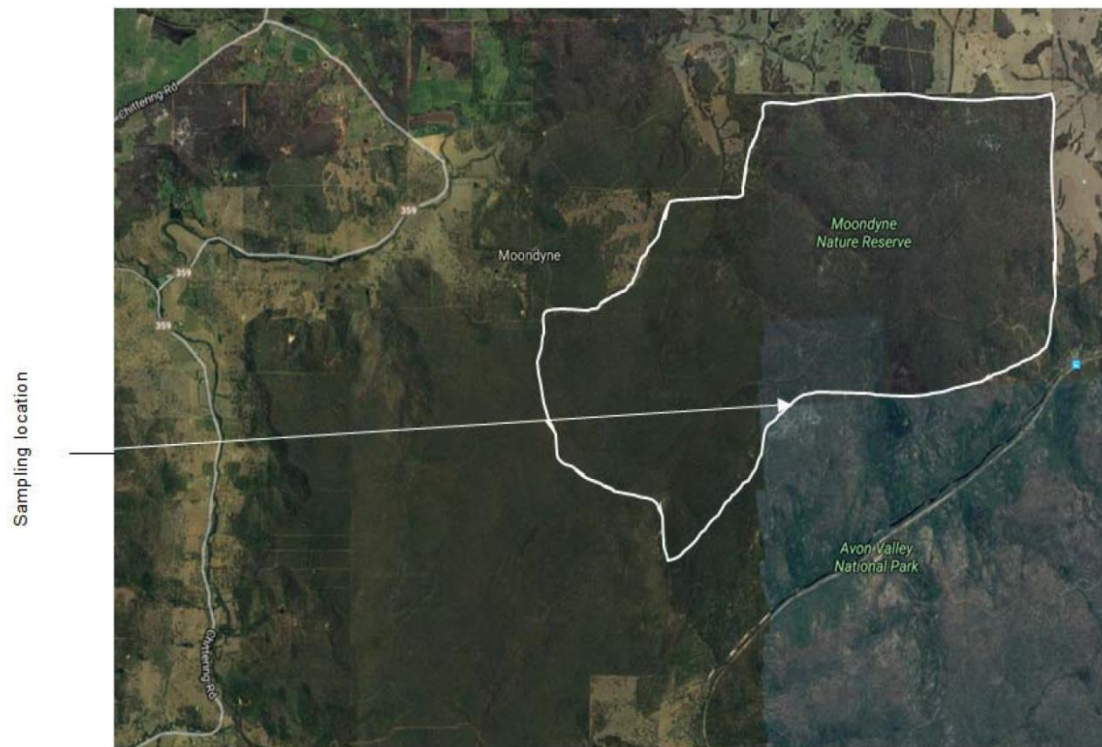


Figure 5.5: Location of PHS 202 Moondyne Site, Bullsbrook burn area and sampling location

5.3 Study approvals

There were a number of approvals that were required prior to the study commencing. These included ethics, risk management, agency authorisation and personal protective equipment requirements.

5.3.1 Ethics approval

An Ethics Declaration form was completed and submitted on 13 August 2012 to the Edith Cowan University Human Research Ethics Committee. It was determined that no ethical considerations were needed as the study did not involve interviews.

5.3.2 Risk management

Collecting ash samples within an active fire ground presents significant risk in terms of exposure to heat, falling branches or trees, and exposure to smoke. These risks were managed through the completion of DEC fire management training and the use of personal protective equipment including boots, fire jacket, respirator and goggles. The main risk management controls were based on the principles of time, distance and shielding but a fire extinguisher was also in the vehicle in the event of unexpected fire characteristics.

On the day of sampling a communication briefing was held with the prescribed-burn controller to discuss specific hazards, communication channels to be used while on the fire ground, and planned-burn activity. All fire ground personnel are required to sign in and out of the fire ground area using an attendance “T” card system. A job safety analysis was completed prior to any sampling to identify any site specific risks and signed off by the incident controller. Once the sampling had been completed a debriefing session was held with the incident controller to discuss the activities of the day and any issues arising. These measures ensure that the strategies in place reduce the potential for injury to people on the fire ground.

5.3.3 Study authorisation

The project required four levels of authorisation from the DEC. This involved the Fire Management Services, Fire Research Branch, Fire Management Services, Environmental Hazards, and Swan Hills District Office. Under normal circumstances approval would be difficult if not impossible for an external

researcher to enter an active fire ground to conduct a sample program as safety concerns, fire training and operational protocols would all have to be signed off by the respective branch managers. However, the researcher was an employee of the DEC in the Fire Management Services section and had received training and worked as a member of department's pre-formed fire teams.

Authorisation for the project was first granted by the Fire Management Services (FMS) research branch. Principle researchers reviewed the study and made suggestions to enhance it and to enable the study to be used for practical application. Once their approval was obtained discussions were held with senior operation staff within FMS to gain access to the fire ground. When approval was granted by FMS, the Swan Hills region operational staff had to also authorise the project. Their main concern was the management of risks associated with sampling on an active fire ground and how this would work in with normal operational procedures. Approval was required from the Regional Manager, the District Manager and by the Swan Hills Fire Operations Manager. Once all authorisations were obtained from FMS the Environmental Hazards Branch approved the project with signoff by the Director of Environmental Regulations Division and Manager Environmental Hazards. The final authorisation from the Director General was for the use of DEC equipment and research time during work hours. Generally, all managers were supportive of the study and the department allocated a scholarship to assist with the sampling costs.

Each prescribed burn required a series of authorisations before the operations could commence. The approval for the burn at site 1 PHS 807 Welshpool Road, Lesmurdie commenced on 3 April 2013. It was acknowledged that the burn might

have smoke impact on businesses and properties in the Lesmurdie area. On the day of the prescribed burn (1 July 2013) the State Duty Officer checked the validity and currency of the plan and authorised the burn to proceed.

The implementation plan for the prescribed burn at the second site, PHS 204 Zamia Forest, Mundaring was endorsed on 12 August 2013. On the day of the prescribed burn (9 September 2013) the State Duty Officer authorised the burn to be conducted. Site 3 PHS 202 Moondyne Nature Reserve, Bullsbrook was initially endorsed on the same day as the second sampling site on 12 August 2013. The State Duty Officer authorised the burn on 7 October 2013.

5.3.4 Safety and personal protective equipment

Specific work health and safety equipment required included a 4WD vehicle fitted with a Western Australian Emergency Radio Network, emergency lights and a first aid kit. Personal protective equipment included a high visibility safety vest, Proban-treated fire resistant overalls, safety boots, safety helmets, safety goggles and an air purifying respirator.

Prior to sample collection the researcher was required to attend a briefing meeting where specific site hazards are addressed. An audit of personal protective equipment was conducted by fire operations staff to ensure suitability for fire operations. The researcher was briefed on communication protocols including radio and mobile phone arrangements for each particular site. Procedures for entering, sampling and leaving the fire ground were established to ensure that fire operations staff were aware of the researcher's presence throughout the operations.

5.4 Study methodology

The methods used to collect samples in the study involved the researcher obtaining specific training to be able to enter a controlled fire ground, conducting the pilot study and sampling and analysis of the main study.

5.4.1 Authorisation and Training

The Department of Parks and Wildlife required specific fire training to be completed before access was granted to enter a fire ground. This training focused on identifying the risks associated with fire and the use of team protocols to enable a shared understanding of the dangers associated with this activity. The researcher had received specific, specialised training as part of his employment as a Senior Environmental Officer within the Environmental Hazards Branch of the Department of Environment Regulation, specifically the Pollution Response Unit. This training, in conjunction with fire operations training, was required to allow access to an active fire ground.

5.4.2 Pilot study

A pilot study (Appendix 1) was conducted to test the methodology of the study which was intended to measure a range of air pollutants (Total Suspended Particles (TSP), Particulate Matter 10 micrometres (PM₁₀) or less in diameter, particulate matter 2.5 micrometres (PM_{2.5}) or less in diameter, carbon monoxide and volatile organic compounds) and metals generated during the prescribed burn. The pilot study was conducted in the Swan Coastal Plain area within the Bickley town site. The total area for the prescribed burn was 20 hectares with a 2.1

kilometre perimeter. Sampling was undertaken 400 metres from the south-western edge of the burn site and approximately 5 metres from the edge of the burn area.

Local fire front weather data (temperature, wind speed and direction, and relative humidity) were collected during the sampling phase with the aid of a Davis Vantage Pro two mobile weather station (Davis Instruments Corp., Hayward, CA, USA). Regional weather data were also sourced from the Bureau of Meteorology and compared to the local fire front data.

The concentrations of airborne particulate matter were measured continuously during the sampling period using a Turnkey DustMate (Turnkey Instruments Ltd. Northwich, Cheshire, UK) that measures TSP, PM₁₀ and PM_{2.5} concentrations. Sampling was conducted as close to the fire front as was possible, (approximately 5 metres from the area to be burnt) while ensuring the safety of the researcher and the equipment. The DustMate was operated continuously during the sampling period.

TSP particulate samples were collected on 47 mm, 0.8 µm mixed cellulose ester filter papers using three MicroVol 1100 Low Volume Air Samplers (Ecotech Pty Ltd., Knoxfield, Victoria, Australia) internally calibrated at a flow rate of 3 L/min for 90 minutes. A total of six samples were collected from two different sampling points over a 1.5-hour period. The samples were analysed by Analytical Reference Laboratories in Western Australia using method reference ARL No. 401: metals in soil and sediment by ICP-OES.

A VRAE Hand-Held 5-Gas Surveyor (RAE Systems Inc., San Jose, CA, USA) was used to sample carbon monoxide and volatile organic compounds (VOCs) as combustible gases (%VOL). This instrument was located with the other equipment, 5 metres from the fire front.

A bulk ash sample was also collected from the burn area with the aid of a small plastic shovel. Approximately 300 grams of ash per sample was collected. The sample was stored in a sealable bag for laboratory analysis of metals. Thirteen metals were chosen for analysis as identified by the NEPM as the screening measure for possible metal contamination. The 13 metals included Arsenic, Barium, Beryllium, Cadmium, Chromium, Cobalt, Copper, Lead, Manganese, Mercury, Vanadium, Nickel, and Zinc. The ash was analysed using NEPM method number 201 (US EPA method 3120) by Analytical Reference Laboratories in Western Australia.

5.4.3 Main Study

The results of the pilot study did not provide sufficient results for the particulate analysis of the smoke and subsequently this part of the study was discontinued. The ash samples provided sufficient results for further exploration and formed the main line of exploration for the remainder of the study.

5.4.4 Main study operation protocol

The parameters for conducting a prescribed burn depend on a range of factors, in particular, the suitability of local weather conditions. Once it was confirmed that a prescribed burn would be undertaken details were provided on the location of the control point. A meeting time was set to discuss the most suitable location to

conduct the sampling based on prevailing winds and staff operations. A procedural protocol for the prescribed burn was established at the briefing meeting prior to commencing the sampling. On arrival at the sampling area the Incident Controller was contacted and directions were given relating to areas to be ignited or that has just been lit, the direction of the fire front, and specific risks of the area including:

- likelihood of falling branches and underground burning roots
- heavy machinery movements
- fire ground conditions
- upcoming changes in weather
- the staging of the burn activity.

Patrolling officers were directed to check on the researcher's welfare during the sampling period and to provide up-to-date information on the fire's characteristics. When the sampling was completed contact was made with the Incident Controller to advise them that the researcher would be leaving the fire ground.

At the end of the day a debrief meeting was conducted with all staff to advise on the progress of the burn, specific risks that were identified, and any observations about the characteristics of the burn. Ongoing post-burn monitoring was discussed during these meetings.

5.4.5 Main study sampling methodology

Once the sampling location was determined a sampling grid of approximately 50 metres by 50 metres (2500 m²) was selected and a random sampling pattern commenced. Each sample was taken from a point within a 10 metre by 10 metre area. The sample location and direction to the next sample was determined by safety factors such as flame activity within the grid area, risk of underground root burn activity, and the overhead branch drop activity in the area. Figures 5.6, 5.7 and 5.8 provide a graphical representation of the sequence and where the samples were taken within the 50 metre by 50 metre grid for each of the sampling sites.

	0-10 m	10-20 m	20-30 m	30-40 m	40-50 m
40-50 m	9		8	7	6
30-40 m	10	4		5	
20-30 m		3	12		11
10-20 m	2			13	
0-10 m	1		15	14	

Figure 5.6: PHS 807 Welshpool Rd, Lesmurdie Sampling Sequence Grid

	0-10 m	10-20 m	20-30 m	30-40 m	40-50 m
40-50 m	7	8		9	
30-40 m		5			13
20-30 m	1		6	10	
10-20 m		4		14	12
0-10 m		2	3	15	11

Figure 5.7: PHS 204 Zamia Forest, Mundaring sampling sequence grid

	0-10 m	10-20 m	20-30 m	30-40 m	40-50 m
40-50 m				10	8
30-40 m		11	12	6	7
20-30 m		4	5		9
10-20 m	2	3		13	
0-10 m	1	1	14	15	

Figure 5.8: PHS 202 Moondyne NR, Bullsbrook sampling sequence grid

Due to the low intensity of the burn it was possible to collect both ash and vegetation litter samples within the same 10 metre by 10 metre area (100 m²) as the fire activity was sporadic and only partially burnt the area. A total of 15 ash and 15 ground vegetation litter samples were collected from each prescribed burn area.

Ash samples were collected from each grid using a plastic trowel and placed into a plastic bag for subsequent size separation. The samples were then divided based on size fractions using three Endecott's 200 mm diameter test sieves and a receiver. The apertures of the sieves were 4 mm, 2 mm and 1 mm that could be interconnected. The material was carefully deposited on the top largest sieve and gently agitated so that the finer particles would not blow away. The partially burnt material remaining in the top sieve consisted of twigs and stones etc. and was discarded. As the smaller particles passed through each sieve the larger ash samples were captured by each aperture until the very fine particles were collected in the receiver. This provided sample material that was between 2 and 4 mm, 1 and 2 mm and less than 1 mm. All material collected was placed in sealed plastic bags and labelled for laboratory analysis.

The vegetation samples were collected from the same grid as close as possible to the area where the ash samples were collected and placed in plastic bags for analysis. These samples consisted of leaf material and small twigs.

5.4.6 Laboratory Analysis

The analysis of samples was undertaken at Analytical Reference Laboratory (WA), which is accredited by National Association of Testing Authorities (NATA), which analysed the vegetative litter and ash in accordance with the American Society for Testing and Materials (ASTM) standards. The analysis included testing for the National Environmental Protection Measures (NEPM) suite of 13 metals (Arsenic, Barium, Beryllium, Cadmium, Chromium, Cobalt, Copper, Lead, Manganese, Mercury, Nickel, Vanadium and Zinc). The analytical methodology followed was in accordance with the methods specified in Table 5.2. The vegetative litter was prepared by acid digestion of Nitric and Hydrochloric acid (Ratio 3:1) at 95°C for two hours and analysed in accordance with biological samples for metals methodology. The ash was prepared using the same methodology to ensure consistency in sample preparation and was analysed using Metals in soil ICP-OES methodology. Mercury was analysed by Cold Vapour AA methodology (ARL, 2013).

Table 5.2: Analytical methods used for analysis by Analytical Reference

Laboratories

(Source: ARL (2013) Kim Rodgers, General Manager. Analytical Reference Laboratories (WA))

Method Description	Matrix	Method No.	NATA Accr.	Preparation	Analysis	Method References
Biological samples for Metals	Biological	26	√	Acid Digestion	ARL No 027, 030, 031, 401, 039, 040, 065, 066	ASTM D 4638-95, APHA 3030E
Mercury by cold vapour AA	Water/Soil	39	√	Acid Digestion	AAS-VGA	ASTM D 3223-95, APHA 3112 B, EPA SW-846 Method 7471A, Varian VGA-77, Publication No 85 101047 00
pH in Soil/Biosolid	Soil/Bio	138	√	Extraction	Probe	EPA SW846 Method 9045D, NEPM, NATA Technical Note 21
Metals in Soil by ICP-OES	Soil	401	√	Acid Digestion	ICP-OES	NEPM Method No 201, EPA Method 3120

5.4.7 Statistical analysis of results

All data tests with numerical output were reported as mean \pm standard error. A skew and kurtosis test was performed to test for parametric data, it was considered that the sample size (>15 samples) satisfied the requirement to use the ANOVA parametric test (Frost, 2015). An Analysis of Variance (ANOVA) to test for statistically significant differences in measurement means was conducted on samples. Pairwise comparisons between sample sizes and vegetation were completed using a Levine's test and a Games-Howell post-hoc test at $p < 0.05$ significance level. All statistical analysis was conducted using SPSS V.23 software. Where a Limit of Detection (LoD) recording was not obtained, a

reference value of 70% of the LoD was used as a conservative figure for statistical analysis. Ogden *et. al.*, (2011) states that the use of a value for the LoD is debatable and no single method can be recommended as being the preferred methodology. As such, the LoD value of 70% was chosen as this option was previously used in other studies, thus making comparison of results valid (CJCU, 2015).

Chapter 6: Study Results

6.1 Field observations

On the day of sampling an initial briefing was conducted with the Incident Controller to discuss the anticipated direction of the fire and the most suitable place to sample. The objective of the briefing was to become familiar with the day's activities and the resources that would be operating in the area. These resources included earth moving equipment, fire appliances and 4WDs patrolling for potential break outs. A safety component was included so authorised people on the fire ground knew the area where the study was being conducted. They could include that area in their patrols in the event any unforeseen safety issues might occur.

Safety was a major consideration in the selection of sample sites. The sites were selected based on the predicted path of the fire in relation to topography and wind speed as these factors can influence the fire's rate of spread. The area chosen provided extended egress options in case the fire conditions changed. Table 6.1 provides information about the characteristics of each site that was provided during the initial operational briefing.

Table 6.1: Site characteristics provided during the initial operational briefing on the day of sampling

	Pilot Study: PHS 806 Pomeroy Road, Bickley	Site 1: PHS 807 Welshpool Road, Lesmurdie	Site 2: PHS 204 Zamia Forest, Mundaring	Site 3: PHS Moondyne Nature Reserve Bullsbrook
Sample date	30 April 2013	1 July 2013	9 September 2013	7 October 2013
Temperature (°C)	24	16	20	20
Wind Direction	North North West	North West	Variable	North West
Wind Speed (km/h)	15	15	5	20
Wind Gust (km/h)	20	25	10	35
Fuel Load (t/Ha)	14.7	19.3	12.5	9
Litter Depth (mm)	25	39	19	11
Flame Height (m)	8	10	6	8
Rate of flame spread (m/hr)	38 - 44	34 - 44	28 - 34	65 - 72

6.2 Observations of monitoring sites

Due to the sites being similar in topography, forest type and climatic conditions, field observations were made for all three sites. Figure 6.1 shows the typical forest in the burn areas. It is defined as open jarrah forest with shrubbery interspersed between trees. The soil surface is characterised by loams, gravels, clay-gravel, sands, quartzite sands and exposed granites. The leaf litter is at its heaviest at the base of the trees or shrubs and the site becomes bare in some areas. There is an accumulation of organic material in places, like logs and other obstructions where wind pushes the litter against the structure.



Figure 6.1: Forest environment prior to the prescribed burn

Once the fire approached, the smoke density increased dramatically and some spotting occurred ahead of the fire. The noise level also increased and this made radio communications difficult. It was observed that the fire would creep along the ground at a slow pace where leaf litter material could be consumed. This fire behaviour was consistent with the description provided by O'Bryan (2005) that a low flame height represented a low intensity fire. It was quite easy to become complacent about the risks at this stage of the fire and care was taken to be attuned to sudden increases in flame height. Figure 6.2 shows the creeping process and the associated wall of smoke that moves with the fire.



Figure 6.2: Examples of an approaching fire creeping along the ground

When the fire front comes into contact with a large deposit of leaf litter or a volatile plant (e.g. *Xanthorrhoea Sp.* (Bradstock, Williams, and Gill, 2012), the fire intensity suddenly increases in that area, as observed in Figure 6.3. An increase in radiant heat was felt, the flame height also increased and, where a tree was in the vicinity, the upper foliage ignited. The ground fire continued to move forward while the larger branches and logs burn.



Figure 6.3: Variations in fire intensity associated accumulated litter or volatile plants

Once the fire has passed, the landscape was burnt in some areas while other areas remained untouched as the fire itself provided its own micro-mosaic burning effect. It was seen that white ash residue occurred where the intensity of the burn increased due to a greater fuel load and the leaf material was effectively combusted. The white ash was observed predominantly at the base of trees and volatile plants as well as where fallen branches had been consumed. Large logs can remain smouldering for months after a burn and must be extinguished. Figure 6.4 displays the smouldering effect and the white ash residue at the base of plants and fallen branches.



Figure 6.4: Aftermath of a prescribed burn

The final task of the DEC “mop-up” crew was to extinguish any log fires (including turning the logs over as the red glow often occurs underneath), extinguish areas at the edge of the burn, and fell any damaged trees that could pose a fall risk in the coming months. Figure 6.5 demonstrates the mop-up process.



Figure 6.5: DEC staff extinguishing fires on the edge of the burn

6.3 Ash sample observations

6.3.1 Pilot Study results

The ash sampled during the pilot study was collected as a whole and consisted of a mixture in terms of size fractions and colour that was consistent with the ash found in the main study. There was no separation of the particles at this stage of the study as the initial focus was on air sampling. Three ash samples (Table 6.2) were analysed and the results provided an indication that the collection of ash was a viable alternative sampling medium.

Table 6.2: Litter and Ash samples results of the pilot study

Metals in soil and sediment	LoD	Units	Ash Site 1A 13-3223-1	Ash site 1b 13-3223-2	Ash Site 1c 13-3223-3
Arsenic	5	mg/kg	<5	<5	5
Barium	1	mg/kg	90	91	100
Beryllium	0.1	mg/kg	<0.1	<0.1	<0.1
Cadmium	0.1	mg/kg	3.7	3.2	4.2
Chromium	1	mg/kg	28	21	36
Cobalt	1	Total µg	1	2	2
Copper	1	mg/kg	25	43	30
Lead	1	mg/kg	21	24	26
Manganese	1	mg/kg	430	460	510
Mercury	0.02	mg/kg	<0.02	<0.02	<0.02
Nickel	1	mg/kg	3	4	5
Vanadium	2	mg/kg	200	180	220
Zinc	1	mg/kg	83	91	97

6.3.2 Study sample observations

The ash was separated into the various size fractions and was distinctive in their appearance with the larger diameter samples displaying a characteristic similar to charcoal. Within the matrix of the 2-4 mm size fraction ash there was partially burnt carbonaceous material from small stems and twigs. There was no evidence of unburnt leaf litter in the ash samples. The 1-2 mm size fraction samples displayed similar characteristics to the 2-4 mm size fraction, however there were distinctive white ash particles that were wafer-like in appearance and could easily be broken up if disturbed. The <1 mm size fraction samples presented a completely different appearance to that of the larger sized samples. The <1 mm samples appeared to be like a fine grey particle, uniform in size. The charcoal aspect was generally not visible except for small dark specks within the sample. Figure 6.6 provides a visual representation of the characteristics of the different sized samples.

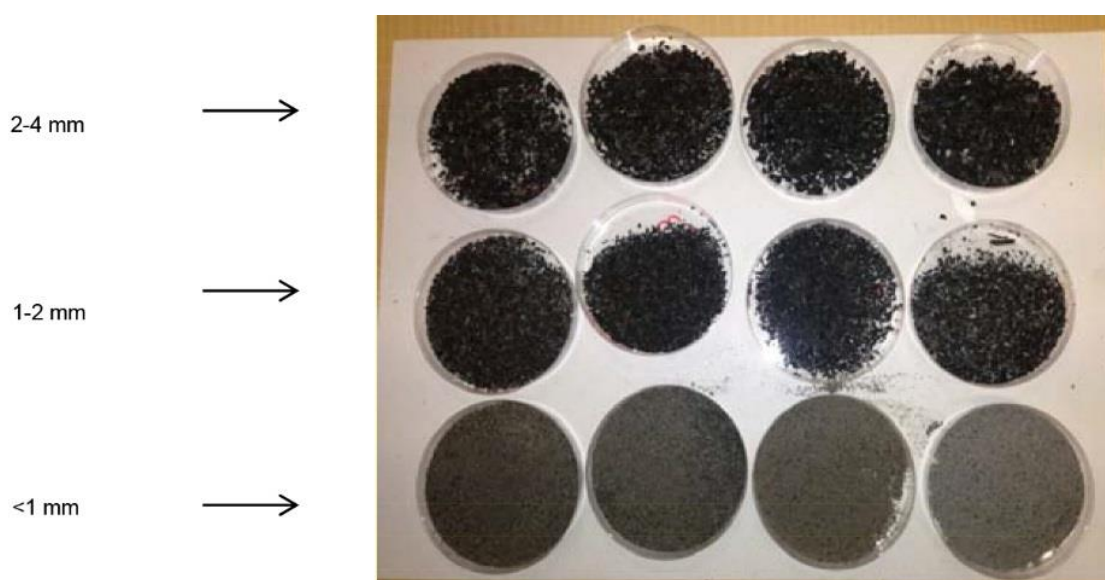


Figure 6.6: Variations in colour and texture of ash as a result of the separation process shown in 90 mm petri dishes.

6.4 Analytical results per metal type

The results from the data on levels of metals are presented in order of average concentration of metals, from highest to lowest (Manganese, Barium, Zinc, Vanadium, Copper, Chromium, Lead, Nickel, Cadmium) and then the metals (Arsenic, Beryllium, Cobalt and Mercury) that did not provide sufficient evidence to warrant further investigation.

6.4.1 Manganese

6.4.1.1 Results from the sampling program

All the Manganese ash and litter samples (n=180) were above the LoD (1 mg/kg). The concentrations on manganese in the ash samples ranged from 21 mg/kg to 3300 mg/kg and the distribution was highly-skewed (+3.099) and leptokurtic (+13.340). The majority of the results (98%) were less than 200 mg/kg, presented in Table 6.3, which indicates that the presence of Manganese reduces with increasing ash size.

Table 6.3: Manganese concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
<500	73	100	49	69	73
500-1000	19	0	22	31	27
1000-2000	6	0	22	0	0
2000-3000	~1	0	4	0	0
> 3000	1	0	2	0	0

Table 6.4 shows the descriptive statistical analysis for each size fraction and the overall analysis for Manganese. The 95% CI range for the <1 mm size fraction was between 592.7 and 1023.3 mg/kg, the 1-2 mm 95% CI range was between 349.1 and 504.2 mg/kg, the 2-4 mm ranged from 302.9 to 448.2 mg/kg, and the 95% CI range for the vegetation samples was between 131.7 and 185.3 mg/kg.

Table 6.4: Descriptive statistical analysis, results for Manganese (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Manganese	<1 mm	45	808.0	510.0	716.8	592.7	1023.3
	1-2 mm	45	426.7	340.0	258.0	349.1	504.2
	2-4 mm	45	375.6	310.0	241.7	302.9	448.2
	Vegetation	45	158.4	130.0	89.3	131.7	185.3
	Overall	180	442.1	295.0	462.6	374.1	510.2

Follow-up tests were conducted to evaluate differences between each size particle group. A Levine's test of equality of error variances demonstrated that the error variance was not equal across groups $F(3,176)=28.146$, $p<0.05$. The post-hoc statistical tests (Appendix 2) demonstrates that there were statistically significant differences between the majority of the size fraction groups as determined by one-way ANOVA $F(3,176)=20.316$, $p<0.05$. The Games-Howell post-hoc test revealed that size fraction of <1 mm (808.0 ± 716.8 mg/kg) was significantly different to the 1-2 mm size fraction (462.7 ± 258.0 mg/kg, $p=0.008$), the 2-4 mm size fraction (375.6 ± 241.7 mg/kg, $p=0.002$) and to the vegetation samples (158.4 ± 89.3 mg/kg, $p=0.000$). The 1-2 mm size fraction was not significantly different to the 2-4 mm size fraction ($p=0.767$) but was significantly different to the <1 mm size fraction and the vegetation sample. The vegetation samples were significantly different to all the ash size fractions.

6.4.2 Barium

6.4.2.1 Results from the sampling program

All 180 samples were above the LoD for Barium of 1 mg/kg. The samples ranged from 3 mg/kg to 400 mg/kg and the distribution was highly-skewed (+2.510) and leptokurtic (+8.067) with the vast majority of samples 96% (n=173) being less than 200mg/kg. The individual sample results recorded are represented in the Table 6.5 and it can be seen that there is a difference in presence of Barium from the ash to the vegetation samples

Table 6.5: Barium concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
<50	67	89	49	64	67
50-100	15	9	36	13	11
100-200	14	2	11	13	20
200-300	2	0	2	4	2
> 300	~2	0	2	4	0

As demonstrated in Table 6.6 the quantity of the 95% CI range for Barium in the <1 mm size fraction was between 47.7 and 88.3 mg/kg, the 1-2 mm 95% CI range was between 41.0 and 88.7 mg/kg, the 2-4 mm range was between 35.6 and 68.3 mg/kg, and the 95% CI range for the vegetation samples was between 11.8 and 26.9 mg/kg.

Table 6.6: Descriptive statistical analysis, results for Barium (mg/kg)

	% of Total	N	Mean	Median	SD	95% Lower CI	95% Upper CI
Barium	<1 mm	45	68.0	58.0	67.5	47.7	88.3
	1-2 mm	45	64.9	22.0	79.4	41.0	88.7
	2-4 mm	45	52.0	21.0	54.5	35.6	68.3
	Vegetation	45	19.4	10.0	25.3	11.8	27.0
	Overall	180	41.8	21.0	62.7	41.8	60.3

Follow-up tests were conducted to evaluate differences between each size particle group. A Levine's test of equality of error variances demonstrated that the error variance was not equal across groups $F(3,176)=0.8632$, $p<0.05$.

The statistical tests demonstrate that there were statistically significant differences between the sample groups as determined by one-way ANOVA $F(3,176)=15.560$, $p<0.05$. The Games-Howell post-hoc test (Appendix 2) revealed that the size fraction of <1 mm (68.0 ± 67.5 mg/kg) was similar to the 1-2 mm size fraction (64.9 ± 79.4 mg/kg), and the 2-4 mm size fraction (52.0 ± 54.5 mg/kg). All ash samples were significantly different to the vegetation samples (19.4 ± 25.3 , $p<0.05$). It can be seen that the vegetation samples were significantly different to the ash sample results.

6.4.3 Zinc

6.4.3.1 Results from the sampling program

Of the samples analysed for Zinc 97% ($n=174$) had concentrations that exceeded the LoD measure of 1 mg/kg. The LoD results were located in the ash samples

obtained from site two, Mundaring. Two LoD results were recorded in the <1 mm size fraction, one in the 1-2 mm size fraction and three in the 2-4 mm size fraction.

The samples ranged from 0.7 to 190 mg/kg and the distribution was highly-skewed (+3.241) and leptokurtic (+12.602) with 87% (n=157) of samples being between 1 and 50 mg/kg. The individual sample results recorded are represented in the Table 6.7.

Table 6.7: Zinc concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
<1 (LoD)	3	0	49	2	7
1-50	87	100	22	93	89
50-100	6	0	22	4	4
>150	4	0	4	0	0

Table 6.8 summarises the descriptive statistical analysis for each size fraction and the overall analysis for Zinc. As indicated the quantity of the 95% CI range for Zinc in the <1 mm size fraction was between 32.4 and 61.7 mg/kg, the 1-2 mm 95% CI range was between 15.3 and 24.9 mg/kg, the 2-4 mm range was between 15.4 and 24.4 mg/kg, and the 95% CI range for the vegetation samples was between 12.4 and 18.3 mg/kg. This highlights a general decreasing concentration trend with increasing size fractions.

Table 6.8: Descriptive statistical analysis, for Zinc concentrations (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Zinc	<1 mm	45	47.0	33.0	48.8	32.4	61.7
	1-2 mm	45	20.1	18.0	15.9	15.3	24.9
	2-4 mm	45	19.9	18.0	15.0	15.4	24.4
	Vegetation	45	15.3	14.0	9.7	12.4	18.2
	Overall	180	25.6	18.0	29.7	21.2	29.9

Follow-up tests were conducted to evaluate difference between each particle size group. A Levine's test of equality of error variances demonstrated that the error variance was not equal across groups $F(3176)=20.568$, $p<0.05$. The post-hoc statistical tests (Appendix 2) demonstrates that there were statistically significant differences between the majority of the size fraction groups as determined by one-way ANOVA $F(3176)=12.737$, $p<0.05$. The Games-Howell post-hoc test revealed that there was no statistically significant difference between the 1-2 mm, 2-4 mm size fractions and vegetation. The <1 mm samples were significantly different to all other samples.

6.4.4 Vanadium

6.4.4.1 Results from the sampling program

The analytical sample rate for Vanadium was 88% with 22 of the 180 samples below the LoD of 2 mg/kg. The bulk of the LoD results were in the vegetation samples. Of these 15 were located at sample site one, Lesmurdie, with all being in vegetation samples. Site two, Mundaring, recorded three LoD results with one being recorded in the 2-4 mm size fraction and two in the vegetation samples. Site

three, Bullsbrook, presented four LoD results all of which were vegetation samples.

The samples ranged from 1.4 to 410 mg/kg and the distribution was highly-skewed (+3.916) and leptokurtic (+20.319) with the 72% (n=130) of samples being between 2 and 50 mg/kg. The individual sample results recorded are represented in the Table 6.9.

Table 6.9: Vanadium concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
LoD (<2)	12	47	0	0	2
2-50	72	38	98	80	78
50-100	7	7	2	13	7
100-200	6	4	0	4	11
200-300	2	2	0	2	2
>300	~1	2	0	0	0

As indicated by Table 6.10 the outliers increased the overall average (32.2 ± 51.0) and when these were removed the average dropped considerably. The 95% CI range for the <1 mm size fraction was between 17.1 and 23.1 mg/kg, the 1-2 mm 95% CI range was between 23.9 and 50.3 mg/kg, the 2-4 mm ranged from 25.5 to 55.6 mg/kg, and the 95% CI range for the vegetation samples was between 8.1 and 53.9 mg/kg.

Table 6.10: Descriptive statistical analysis, for Vanadium concentrations (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Vanadium	<1 mm	45	20.1	19.0	10.0	17.1	23.1
	1-2 mm	45	37.1	25.0	43.8	24.0	50.3
	2-4 mm	45	40.5	17.0	50.1	25.5	55.6
	Vegetation	45	31.0	2.0	76.3	8.1	53.9
	Overall	180	32.2	17.0	51.0	24.7	39.7

Follow-up tests were conducted to evaluate differences between each size particle group. A Levine's test of equality of error variances demonstrated that the error variance was not equal across groups $F(3,176)=7.162$, $p<0.05$. There were no statistically significant differences between the majority of the size fraction groups as determined by one-way ANOVA $F(3,176)=1.406$, $p=0.243$. The Games-Howell post-hoc test revealed that there was no statistically significant difference between the samples with the exception between the 1-2 mm size fraction and the 2-4 mm size fraction ($p=0.047$).

6.4.5 Copper

6.4.5.1 Results from the sampling program

All Copper ash and vegetation samples ($n=180$) were above the LoD (1 mg/kg). The samples ranged from 3 mg/kg to 57 mg/kg and the distribution was highly-skewed (+2.616) and leptokurtic (+7.933) with 87% ($n=158$) of samples less than 20 mg/kg. The individual sample results recorded are represented in the Table 6.11.

Table 6.11: Copper concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
< 10	59	100	18	58	62
10-20	28	0	42	38	33
20-30	7	0	20	4	4
30-40	2	0	7	0	0
40-50	~1	0	2	0	0
>50	3	0	11	0	0

Table 6.12 presents the results of the descriptive statistical analysis for each size fraction and the overall analysis for Copper. This highlights that the 95% CI range for Copper in the <1 mm size fraction was between 16.7 and 25.2 mg/kg, the 1-2 mm 95% CI range was between 8.3 and 10.9 mg/kg, the 2-4 mm ranged from 7.4 to 9.9 mg/kg, and the 95% CI range for the vegetation samples was between 3.7 and 4.6 mg/kg.

Table 6.12: Descriptive statistical analysis, results for Copper (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Copper	<1 mm	45	20.9	17.0	14.1	16.7	25.2
	1-2 mm	45	9.6	9.0	4.2	8.3	10.9
	2-4 mm	45	8.7	8.0	4.3	7.4	9.9
	Vegetation	45	4.2	4.0	1.5	3.7	4.6
	Overall	180	10.8	8.0	9.8	9.4	12.3

Follow-up tests were conducted to evaluate the differences between each particle size group. A Levine's test demonstrated that the error variance was not equal across groups $F(3,176)=32.746$, $p<0.05$. The post-statistical tests demonstrates that statistically significant differences exist between the majority of the size

fraction groups as determined by one-way ANOVA $F(3176)=38.564$, $p<0.05$. A Games-Howell post-hoc test revealed that a size fraction of <1 mm (20.9 ± 14.1 mg/kg, $p<0.05$) was significantly different to all other size fractions. The 1-2 mm size fraction (9.6 ± 4.2 , mg/kg, $p=0.760$) was not significantly different to the 2-4 mm size fraction (8.7 ± 4.3 , $p=0.760$) but was significantly different to all other size fractions. The vegetation samples (4.2 ± 1.5 mg/kg, $p=0.000$) was significantly different to all the other size fractions.

6.4.6 Chromium

6.4.6.1 Results from the sampling program

Of the samples analysed for Chromium 88% ($n=168$) had concentrations that exceeded the LoD measure of 1 mg/kg. The majority of these were in the vegetation samples. The samples ranged from 0.7 mg/kg to 72 mg/kg and the distribution was highly-skewed ($+2.785$) and leptokurtic ($+9.319$) as 75% ($n=136$) of the samples were between 1 and 20 mg/kg. The individual sample results recorded are presented in Table 6.13.

Table 6.13: Chromium concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
LoD (<1)	12	38	0	2	9
1-10	58	42	73	62	56
10-20	17	7	22	24	16
20-30	5	2	4	4	7
> 30	8	11	0	7	13

Of the samples recorded, 8 LoD results were located at sample site one, Lesmurdie, with all being located in the vegetation samples. Site two, Mundaring, recorded four LoD results with one being recorded in the 2-4 mm size fraction and three in the vegetation samples. Site 3, Bullsbrook, presented 10 LoD results with one in the 1-2 mm size fraction, three in the 2-4 mm size fraction and the remaining 6 in the vegetation samples.

As shown in Table 6.14 the 95% confidence interval range for Chromium in the <1 mm size fraction was from 6.2 to 9.3 mg/kg, the 1-2 mm 95% CI range was between 7.2 and 14.2 mg/kg, the 2-4 mm range was between 7.6 and 16.1 mg/kg and the 95% CI range for the vegetation samples was between 3.6 and 12.9 mg/kg.

Table 6.14: Descriptive statistical analysis, results for total Chromium (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Chromium	<1 mm	45	7.7	6.0	5.2	6.2	9.3
	1-2 mm	45	10.7	8.0	11.7	7.2	14.2
	2-4 mm	45	11.8	6.0	14.3	7.5	16.1
	Vegetation	45	8.3	1.0	15.6	3.6	12.9
	Overall	180	9.6	6.0	12.4	7.8	11.5

Follow-up tests were conducted to evaluate the difference between particle size parameters. A Levine's test of equality of error variances demonstrated that the error variance was not equal across groups $F(3,176)=5.402$, $p<0.05$. The post-hoc statistical test demonstrated that there were no statistically significant differences between the size fraction groups as determined by one-way ANOVA.

F (3176)=1.126, p=0.340. The Games-Howell post-hoc test (Appendix 2) revealed that there was no significant difference between any of the size fractions and the vegetation samples.

6.4.7 Lead

6.4.7.1 Results from the sampling program

Of the samples analysed for Lead 96% (n=172) have concentrations that exceeded the LoD measure of 1 mg/kg, all of these were in the vegetation samples, two from site one, Lesmurdie, three from site two, Mundaring, and three from site three, Bullsbrook.

The sample mean was 12.2 mg/kg with a range between 1 and 38 mg/kg. The distribution was highly-skewed (+2.785) and leptokurtic (+9.319) with 83% (n=149) of samples between 1 and 20 mg/kg. The individual sample results recorded are represented in the Table 6.15.

Table 6.15: Lead concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
LoD (<1)	4	18	0	0	0
1-10	36	60	18	27	38
10-20	47	11	71	60	42
20-30	7	2	2	7	16
> 30	6	4	9	7	4

Table 6.16 provides a summary of the descriptive statistical analysis for each size fraction and the overall analysis for Lead. As highlighted the 95% CI range for Lead in the <1 mm size fraction was between 13.0 and 17.5 mg/kg, the 1-2 mm 95% CI range was between 11.9 and 16.2 mg/kg, the 2-4 mm range was between 11.0 and 15.9 mg/kg, and the 95% CI range for the vegetation samples was between 3.6 and 8.6 mg/kg.

Table 6.16: Descriptive statistical analysis, results for Lead (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Lead	<1 mm	45	15.2	14.0	7.4	13.0	17.5
	1-2 mm	45	14.1	14.0	7.1	11.9	16.2
	2-4 mm	45	13.5	11.0	8.2	11.0	15.7
	Vegetation	45	6.1	2.0	8.3	3.6	8.6
	Overall	180	12.2	11.0	8.5	10.9	13.5

Follow-up tests were conducted to evaluate differences between each size particle. A Levine's test of equality of error variances demonstrated that the error variance was equal across groups $F(3,176)=0.892$, $p=0.447$. The post-statistical tests demonstrates that there were statistically significant differences between the majority of the size fraction groups as determined by one-way ANOVA $F(3,176)=12.739$, $p<0.05$. A Games-Howell post-hoc test revealed that the vegetation samples (6.1 ± 8.3 mg/kg, $p<0.05$) were significantly different to all the ash size fractions.

6.4.8 Nickel

6.4.8.1 Results from the sampling program

Of the samples analysed for Nickel 94% (n=159) had concentrations that exceeded the LoD of 1 mg/kg. These were predominantly in the vegetation samples and 13 LoD results were from sample site one, Lesmurdie, with only one LoD result in the 2-4 mm size fraction. Site two, Mundaring, recorded eight LoD results with five being in the vegetation samples, two in the 2-4 mm size fraction and one in the 1-2 mm size fraction. All of site three, Bullsbrook, presented a result above the LoD.

The results ranged from 0.7 mg/kg to 18.3 mg/kg with a highly-skewed distribution (+1.395) which was platykurtic (+1.247) with the majority of samples (73% (n=131)) between 1 and 10 mg/kg. The individual sample results are represented in the Table 6.17.

Table 6.17: Nickel concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
LoD (<1)	12	38	2	0	38
1-5	51	51	38	58	51
5-10	22	4	18	27	4
10-15	9	2	22	13	2
15-20	6	0	20	2	0

As indicated by Table 6.18 the 95% CI range for the <1 mm size fraction was between 5.9 and 9.5 mg/kg, the 1-2 mm 95% CI range was from 3.8 to 5.9 mg/kg, the 2-4 mm ranged from 3.2 to 5.2 mg/kg, and the 95% CI range for the vegetation samples was between 1.5 and 2.8 mg/kg.

Table 6.18: Descriptive statistical analysis, results for Nickel (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Nickel	<1 mm	45	7.7	6.0	5.8	5.9	9.5
	1-2 mm	45	4.9	3.0	3.6	3.8	5.9
	2-4 mm	45	4.2	3.0	3.3	3.2	5.2
	Vegetation	45	2.2	1.0	2.2	1.5	2.8
	Overall	180	4.7	3.0	4.4	4.1	5.4

Follow-up tests were conducted to evaluate the difference between each size particle group. A Levine's test of equality of error variances demonstrated that the error variance was not equal across groups $F(3,176)=23.8444$, $p<0.05$. There were statistically significant differences between the majority of the size fraction groups as determined by one-way ANOVA $F(3,176)=15.560$, $p<0.05$. The Games-Howell post-hoc test (Appendix 2) revealed that size fraction of <1 mm (7.7 ± 5.8 mg/kg) was significantly different to the 1-2 mm size fraction (4.9 ± 3.6 mg/kg, $p=0.032$), the 2-4 mm size fraction (4.2 ± 3.3 mg/kg, $p=0.003$) and to the vegetation samples (2.2 ± 2.2 mg/kg, $p<0.05$). The 1-2 mm size fraction was not significantly different to the 2-4 mm size fraction ($p=0.761$) but was significantly different to all other size fractions. The vegetation samples were significantly different to all other size fractions.

6.4.9 Cadmium

6.4.9.1 Results from the sampling program

Of the 180 samples analysed 81% (n=146) had concentrations above the LoD of 0.1 mg/kg. The Cadmium concentrations ranged from 0.07 mg/kg to 5.7 mg/kg with the distribution being highly-skewed (+3.218) and leptokurtic (+12.624) with 78% (n=141) of samples between 0.1 and 3 mg/kg. The summary of the number of samples in each concentration range is shown in Table 6.19.

Table 6.19: Cadmium concentrations in ash stratified by size fraction

Concentration Range (mg/kg)	% of Total	Unburnt	Post Burn		
		% of litter samples	% < 1 mm	% 1 to 2 mm	% 2 to 4 mm
LoD (<0.1)	19	51	2	7	16
0.1-1.0	66	38	93	71	62
1.0-2.0	6	4	2	11	7
2.0-3.0	6	4	2	9	11
> 3.0	3	4	0	2	4

Of the samples below LoD, 17 were from sample site one, Lesmurdie, with two in the 1-2 mm size fraction, four in the 2-4 mm size fraction and the majority (11) not recording a result in the vegetation samples. Site two, Mundaring, had seven samples with results below the LoD with one each being recorded in the <1 mm and 1-2 mm size fraction and two in the 2-4 mm size fraction. The remaining four results were obtained from vegetation samples. Site three, Bullsbrook, had 10 samples below the LoD with only one in the 2-4 mm size fraction. The remaining nine LoD results were recorded in the vegetation samples.

Table 6.20 shows the concentrations of Cadmium measured was relatively low, the 95% confidence interval (CI) range for the <1 mm size fraction was from 0.3 to 0.5 mg/kg, the 1-2 mm 95% CI range was from 0.4 to 1.0 mg/kg, the 2-4 mm ranged from 0.5 to 1.1 mg/kg, and the 95% CI range for the vegetation samples was from 0.2 to 0.8 mg/kg.

Table 6.20: Descriptive statistical analysis, results for Cadmium (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Cadmium	<1 mm	45	0.4	0.3	0.3	0.3	0.5
	1-2 mm	45	0.7	0.4	0.9	0.4	1.0
	2-4 mm	45	0.8	0.3	0.9	0.5	1.1
	Vegetation	45	0.5	0.7	1.0	0.2	0.8
	Overall	180	0.6	0.3	0.9	0.5	0.7

Follow-up tests were conducted to evaluate the difference between each particle size parameter and group. A Levine's test of equality of error variances demonstrated that the error variance was not equal across groups $F(3,176)=8.632$, $p<0.05$. The post-statistical tests demonstrate that there were no significant differences between the size fraction groups as determined by one-way ANOVA $F(3,176)=2.356$, $p=0.074$. A Games-Howell post-hoc test (Appendix 2) revealed that size fraction of <1 mm (0.4 ± 0.3 mg/kg, $p=0.87$) was not significantly different to the 1-2 mm size fraction (0.7 ± 0.9 , mg/kg, $p=0.87$) and the vegetation samples (0.5 ± 1.0 mg/kg, $p=0.924$). However there was a significant difference with the 2-4 mm size fraction (0.8 ± 0.9 , $p=0.045$). The 1-2 mm size fraction had no statistically significant differences with any of the other samples as results from the post-hoc test presented a result that was greater than $p=0.05$ in all instances.

6.5 Analytical results of metals below the level of detection

There were four metals (Arsenic, Beryllium, Cobalt and Mercury) where the majority of the results were below the limit of detectable results and could not be analysed statistically.

6.5.1 Arsenic

6.5.1.1 Results from the sampling program

The sampling results for Arsenic were very low (5%) with only 10 of the 180 samples providing a result higher than the limit of detection, referred to in this thesis as the limit of detection measure (LoD), of 5 mg/kg. Five of these results were recorded from sample site one, Lesmurdie, with two in the 1-2 mm size fraction and three being in the 2-4 mm size fraction with a range between 5 to 22 mg/kg. Site two, Mundaring, recorded the remaining five results with only one in the 1-2 mm size fraction and the remaining four classified in the 2-4 mm size fraction (range between 6 to 10 mg/kg). No further analysis could be conducted on the Arsenic data.

Table 6.21 shows there is a bias in the descriptive statistical summary of results with a mean value of 3.8, which is heavily influenced by the LoD default figure of 3.5 mg/kg, used when no result could be obtained. This is further characterised by the median value of 3.5 mg/kg being the LoD default value used in this study.

Table 6.21: Descriptive statistical analysis, results for Arsenic mg/kg.

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Arsenic	<1 mm	45	NR	NR	NR	NR	NR
	1-2 mm	45	3.8	3.5	1.2	3.5	4.2
	2-4 mm	45	4.4	3.5	2.9	3.5	5.3
	Vegetation	45	NR	NR	NR	NR	NR
	Overall	180	3.8	3.5	1.6	3.6	4.1

NR = no result

The individual recordings were not close to the HIL of 100 mg/kg for residential areas as the highest level recorded was 22 mg/kg. Even though this exceeds the EIL of 20 mg/kg (DEC, 2010) it only registered on one analyte sample. Since there was insufficient data for an accurate statistical analysis it is concluded that Arsenic was not present in sufficient quantity in these sample areas be a concern to health and the ecosystem.

6.5.2 Beryllium

6.5.2.1 Results from the sampling program

The sampling results for Beryllium were low (24%) with only 44 of the 180 samples above the LoD of 0.1 mg/kg. Of these, one was recorded at sample site one, Lesmurdie, in the 2-4 mm size fraction with a concentration of 0.4 mg/kg. Site two, Mundaring, recorded five results with one being recorded in the 1-2 mm size fraction (0.1 mg/kg) and vegetation recording the remaining four results with a range from 0.1 to 0.3 mg/kg. Site three, Bullsbrook, presented the bulk of the positive samples with 38 results, all of which were in the soil matrix. Thirteen results were recorded in both the <1 mm and 1-2 mm size fractions and 12 were obtained in the 2-4 mm size fraction. The range was between 0.1 and 0.3 mg/kg.

Table 6.22 verifies the bias present in the results that have a mean value of 0.095 mg/kg, which is heavily influenced by the LoD default figures of 0.07 mg/kg. This is further characterised by the median figure, which is the default value of 0.07 mg/kg.

Table 6.22: Descriptive statistical analysis, results for Beryllium (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Beryllium	<1 mm	45	0.1	0.07	0.44	0.08	0.11
	1-2 mm	45	0.1	0.07	0.64	0.09	0.12
	2-4 mm	45	0.1	0.07	0.07	0.08	0.12
	Vegetation	45	0.1	0.07	0.04	0.07	0.10
	Overall	180	0.1	0.07	0.06	0.07	0.10

6.5.3 Cobalt

6.5.3.1 Results from the sampling program

The sampling results for Cobalt were low (37%) with 66 of the 180 samples above the LoD of 1 mg/kg. Twenty-three results were recorded at sample site one, Lesmurdie, with 11 being in the <1 mm size fraction with a range from 1 to 3 mg/kg. The 1-2 mm size fraction provided seven results ranging from 1-4 mg/kg. The remaining five were in the 2-4 mg/kg (range from 1 to 2 mg/kg) size fraction while there were no Cobalt deposits found in the vegetation samples.

Site two, Mundaring, did not record any results while at site three, Bullsbrook, 43 results were recorded across all size matrices. In the <1 mm size fraction there were 15 results ranging from 2 to 4 mg/kg, the 1-2 mm size fraction recorded 13 results ranging from 1 to 4 mg/kg, the 2-4 mm size fraction recorded 10 results ranging from 1 to 3 mg/kg and vegetation samples providing five results ranging

from 1 to 2 mg/kg. The plot graph (Figure 6.18) highlights the fact that statistical analysis was not viable due to the low number of results.

Table 6.23 provides evidence of the bias in the results. The mean value of 1.2 mg/kg was influenced by the large number of LoD default figures of 0.7 mg/kg. This is further characterised by the median figure of 0.7 mg/kg, which is equal to the LoD default figure.

Table 6.23: Descriptive statistical analysis, results for Cobalt (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Cobalt	<1 mm	45	1.6	1.0	0.9	1.3	1.9
	1-2 mm	45	1.3	0.7	0.9	1.0	1.6
	2-4 mm	45	1.3	0.7	0.6	0.9	1.2
	Vegetation	45	0.1	0.7	0.3	0.7	0.9
	Overall	180	1.2	0.7	0.8	1.1	1.3

The individual recordings were not close to the HIL of 100 mg/kg for residential (A) areas nor the EIL of 50 mg/kg as the highest recorded value was 4 mg/kg. This is well below either HIL threshold. It was therefore concluded that Cobalt was not present in sufficient quantities to be of concern for health and ecological limits.

6.5.4 Mercury

6.5.4.1 Results from the sampling program

The sampling results for Mercury were low (18%) with only 32 of the 180 samples above the LoD of 0.02 mg/kg. Of these, 25 were recorded at sample site one, Lesmurdie, with five falling in the <1 mm size fraction with a range from 0.02 to 0.07 mg/kg. The 1-2 mm size fraction provided six results ranging from 0.03 to 0.05 mg/kg. The remaining six results were located in the 2-4 mm size fraction

with a range from 1 to 2 mg/kg and eight recorded in the vegetation sample with a range from 0.02 to 0.93 mg/kg.

Site 2, Mundaring, provided 7 results with one in the 2-4 mm size fraction (0.03 mg/kg) and six in the vegetation samples, ranging from 0.01 to 0.03 mg/kg. Site three, Bullsbrook, did not provide any results. There was insufficient data for significant statistical analysis.

Table 6.24 provides further evidence of the bias in the statistical figures as the mean of 0.022 mg/kg is heavily influenced by the LoD default figure of 0.01 mg/kg. This is further characterised by the median value and the lower 95% confidence interval of 0.01 mg/kg, which is equal to the LoD default value.

Table 6.24: Descriptive statistical analysis, results for Mercury (mg/kg)

		N	Mean	Median	SD	95% Lower CI	95% Upper CI
Mercury	<1 mm	45	0.01	0.01	0.01	0.01	0.02
	1-2 mm	45	0.01	0.01	0.01	0.01	0.02
	2-4 mm	45	0.02	0.01	0.07	0.01	0.05
	Vegetation	45	0.36	0.01	0.14	0.01	0.08
	Overall	180	0.02	0.01	0.08	0.01	0.03

The individual results were not close to the HIL of 15 mg/kg for residential (A) areas as the highest level was 0.93 mg/kg, well below the HIL threshold. Only one sample analyte was close to the EIL of 1 mg/kg (DEC, 2010). It was considered there was insufficient data for an accurate statistical analysis and it was concluded that Mercury was not present in sufficient quantities in the sampling areas for it to be of concern to health and the ecosystem.

Chapter 7: Discussion

This study demonstrates the relationship between metals in vegetative matter prior to, and in ash after a prescribed burn. The average concentrations of metals in ash and unburnt vegetation litter are presented in Figure 7.1.

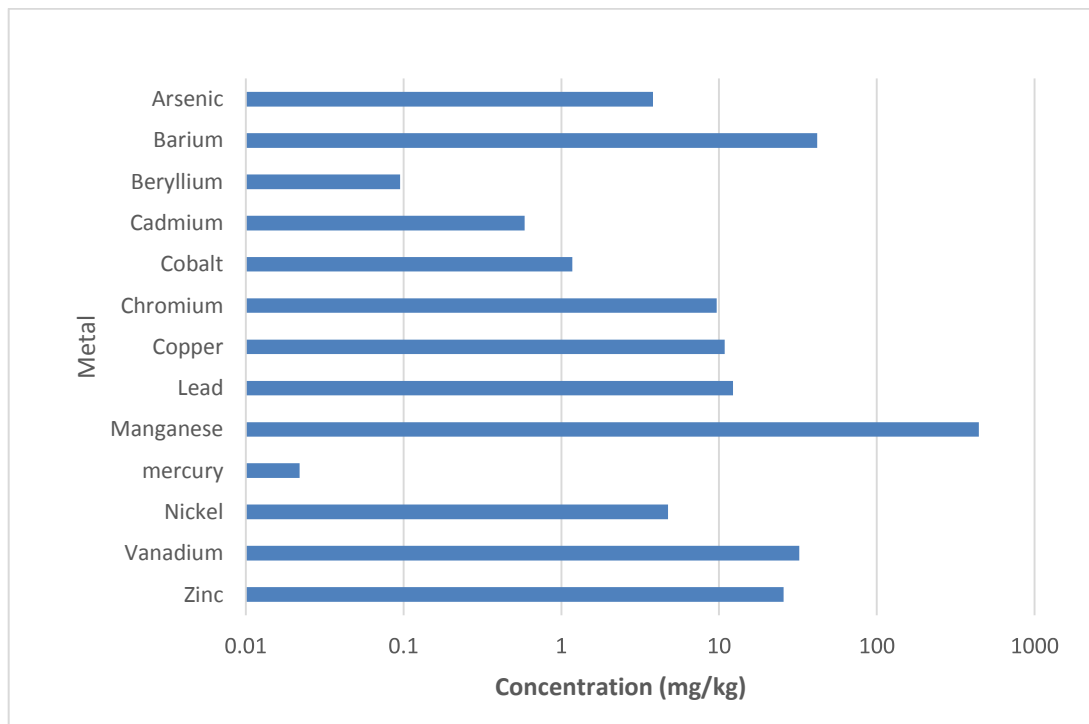


Figure 7.1: Average metal concentrations found in ash at the sample sites on the Darling Escarpment in WA

The average concentrations of the metals varied and ranged from 0.02 mg/kg for Mercury to 452 mg/kg for Manganese. A total of 180 ash and vegetation samples were collected and analysed for speciation of 13 metals thus giving a total of 2 340 results, of which 71% were above the LoD. This increased to 93% when the four metals (Arsenic, Beryllium, Cobalt, and Mercury) were removed from the analysis due to the low level (below LoD) of results from these samples. The results above LoD from all the vegetation samples was 58% but once the lowest four metals

were removed from the analysis this increased the vegetation sampling rate to 78% of samples above LoD.

7.1 Comparison of the findings of this study with other similar studies

The metals found in the ash and vegetation of this study were compared with the results of other studies (Table 7.1) conducted in Australia, Europe, USA, Jordan and other multiple jurisdictions. Only the two United States studies (Etiegni and Campbell, 1991; Plumlee, 2007) investigated Barium levels and the results of this study were well below those of studies conducted elsewhere. This could be attributed to the higher mineral substrate of the other areas or the fact that the samples were obtained from an uncontrolled burn that would release more ash.

Cadmium was reported on in a number of studies (Kiikkilä, 2003; Gomez-Rey, Madeira, and Coutinho, 2012; Tomasevic, 2004; Etiegni and Campbell, 1991; Plumlee, 2007; Massadeh, Jaradat, Momani, and Saleem, 2009; Nagajyoti *et. al.*, 2010; Augusto, Bakker, and Meredieu, 2008), and it was found that levels of Cadmium in this study were comparable to those reported elsewhere.

In the comparison studies Chromium was investigated as either total Chromium (Gomez-Rey *et. al.*, 2012; Etiegni and Campbell, 1991; Plumlee, 2007; Nagajyoti *et. al.*, 2010; Augusto *et. al.*, 2008) or as Chromium VI (Tomasevic, 2004). The levels found in this study were lower than the majority of other studies with only the Gomez-Rey (2012) study being within a similar range. Copper concentrations were comparable with the majority of the other studies (Khanna, Raison and Falkiner, 1994; Gomez-Rey *et. al.*, 2012; Tomasevic, *et. al.*, 2004; Massadeh *et.*

al., 2009; Nagajyoti *et. al.*, 2010; Augusto *et. al.*, 2008) but were considerably lower than the levels found in the United States (Etiegni and Campbell, 1991; Plumlee, 2007).

Lead comparisons were also similar to that of Copper with similarities with other jurisdictions (Khanna *et. al.*, 1994; Gomez-Rey *et. al.*, 2012; Tomasevic, 2004; Massadeh *et. al.*, 2009; Nagajyoti *et. al.*, 2010; Augusto *et. al.*, 2008) except the United States (Etiegni and Campbell, 1991; Plumlee, 2007).

Manganese was often the highest recorded metal concentration in all studies (Khanna *et. al.*, 1994; Gomez-Rey *et. al.*, 2012; Tomasevic, 2004; Nagajyoti *et. al.*, 2010; Augusto *et. al.*, 2008) only the Plumlee (2007) and Tomasevic (2004) studies recorded lower levels of Manganese. Despite the high concentration recorded in this study it was generally below the levels found in elsewhere. The levels of Nickel found in this study was below the levels found in all other studies (Khanna *et. al.*, 1994; Gomez-Rey *et. al.*, 2012; Tomasevic, 2004; Ferreira *et. al.*, 2005; Nagajyoti *et. al.*, 2010; Augusto *et. al.*, 2008; Etiegni and Campbell, 1991; Plumlee, 2007) except in the lower range identified by the Kiikkilä (2003) study.

Vanadium was not well researched with only the two studies conducted in the United States (Etiegni and Campbell, 1991; Plumlee, 2007) providing results. The results were comparable with the Etiegni and Campbell (1991) study but were only at 30% of the results found in the Plumlee (2007) study. Zinc levels were comparable with the all studies (Khanna *et. al.*, 1994; Gomez-Rey *et. al.*, 2012; Tomasevic, 2004; Massadeh *et. al.*, 2009; Nagajyoti *et. al.*, 2010; Augusto *et. al.*, 2008) except those conducted in the United States (Etiegni and Campbell, 1991; Plumlee, 2007).

Generally, the findings from this study were that metal concentrations in Western Australian ash were lower than those reported from the USA. This can be attributed to the higher soil metal levels in California and that the majority of studies were from hotter uncontrolled bushfires. These other studies did not consider the public health risks associated with the metal concentrations.

Table 7.1: A comparison of the results (mg/kg) from this study with other similar studies (USA, Europe, Jordan and Australia).

	This Study	Khanna <i>et al.</i> , (1994) ¹	Kiikkilä (2003) ²	Gomez-Rey <i>et al.</i> (2012) ³	Tomasevic (2004) ⁴	Ferreira <i>et al.</i> (2005) ⁵	Etegni & Campbell (1991) ⁶	Plumlee (2007) ⁷	Massadeh <i>et al.</i> , (2009) ⁸	Nagajyoti <i>et al.</i> , (2010) ⁹	Augusto <i>et al.</i> (2008) ¹⁰
		Australia	Europe			United States			Jordan	Multiple countries	
Barium	41.8	-	-	-	-	-	1301	541 – 858	-	-	-
Cadmium	0.56	-	0.7 - 0.9	0.21	1.34-1.73.	-	<2	<0.02 - 0.06	0.12 - 2.69	0.01 - 0.7	0 – 25
Chromium	9.6	-	-	15.4	83-117 (Cr VI)	-	52	91 - 354 Total Cr	-	5 – 3000	10 - 250
Copper	10.8	12 - 340	-	31	10.3 - 110.2	-	345	1170 – 1370	4.2 - 9.4	2 – 100	15 - 300
Lead	6.1	4 - 152	-	18.9	1.9 - 20.3	-	51	192 - 344	0.19 - 2.28	2 – 200	15 - 650
Manganese	442.2	60 - 100000	-	5400	46 – 112	2570	10549	396 - 774	-	10 - 4000	1000 - 30000
Nickel	4.7	10 - 200	3 - 50	21.9	62.4 - 73.6	-	59	36.6 - 41.6	-	10 – 100	-
Vanadium	20.1	-	-	-	-	-	13	54 - 64.8	-	-	-
Zinc	25.6	20 - 370	16 - 50	64.7	15.2 - 47.1	201	2678	211 - 458	30.6 - 81.2	10 - 300	15 - 650

Notes

- 1 A study of the element concentration in ash derived from combustion of components (leaves, bark and twigs) of eucalyptus litter
- 2 Metal concentrations from reference soil samples collected in SW Finland. Common median values for metals. Figures sourced from multiple areas.
- 3 A study of wood ash effects on nutrient dynamics and soil properties under Mediterranean climate. The study was conducted in PortugalMetal concentrations from Pine sawdust collected from Idaho and combusted at 538°C
- 4 A study of the metals accumulation in soil from urban areas around Belgrade. A study of the chemical composition of ash and soil after the 2007 Southern Californian wildfires
- 5 The Ash chemical composition observed under pine stands from a wildfire in Portugal. Metal concentrations from reference soil samples collected in SW Finland.
- 6 Metal concentrations from Pine sawdust collected from Idaho and combusted at 538°C A study of wood ash effects on nutrient dynamics and soil properties under Mediterranean climate. The study was conducted in Portugal
- 7 A study of the chemical composition of ash and soil after the 2007 Southern Californian wildfires. A study of the metals accumulation in soil from urban areas around Belgrade
- 8 A study of metals in tree leaves along the main road in an agricultural area. Figures sourced in this study are from Jordan. The Ash chemical composition observed under pine stands from a wildfire in Portugal.
- 9 A review of studies of the metal composition of uncontaminated soils. Figures sourced from multiple countries. A study of metals in tree leaves along the main road in an agricultural area. Figures sourced in this study are from Jordan.
- 10 Common median values for metals. Figures sourced from multiple areas. A study of the element concentration in ash derived from combustion of components (leaves, bark and twigs) of eucalyptus litter

7.2 Particle size, metal concentrations and combustion

There has been very little research into any variations within samples based on size fraction but there are studies that have reported on temperature ranges, which this study has demonstrated relates to size fractions. Araya, (2014), Nguyen (2011) and Pereira, Úbeda, and Martin, (2011) have studied changes in chemical composition in soils at various temperatures. Their research identified that temperature affects the metal levels presented for analysis. Temperatures below 200°C have little effect on metals in the organic matter properties and notable changes occur between 250°C and 450°C.

At temperatures above 500°C the majority of the chemical properties have already been lost or transformed into a pyrogenic product. The findings of these studies demonstrated that most the significant changes in the organic matter's physical and chemical properties occur at around 350°C.

The importance of the 250 - 450°C range for bushfires relates to the fact they are classified as low, medium or high intensity. Low-intensity fires reach surface temperatures of up to 250°C, medium intensity fires reach surface temperatures of 400°C, and high-intensity fires reach surface temperatures above 675°C (Janzen and Tobin-Janzen, 2008). Prescribed burns are commonly referred to as low-intensity burns but Figure 6.3, 6.4 and 6.6 highlights that there are instances in the fire where heat can exceed this low intensity in a localised area. It can be seen where white ash is visible at the base of trees or around highly combustible trees and the changes in colour of the ash samples, indicates periods of a higher burn temperature.

The temperatures reached within the fire provide a relationship with the size fractions used in this study. This study has demonstrated the relationship between size fraction, colour and stages of combustion as shown in Figure 6.6. Nguyen (2011) found that increases in burn temperatures significantly reduced carbon, nitrogen and potassium levels in ash but did not change concentrations of Phosphorus, Calcium, Magnesium, Iron, Manganese and Zinc. Nguyen (2011) attributed this to the relatively high volatilisation temperatures of the metals that remained. This could also account for the low Mercury levels (356°C boiling point) found in the samples in this study but does not account for the drop in concentration levels in the other metals where a statistically representative sample could not be obtained (Pohanish, 2002).

The variations in size fractions highlighted a consistent range of metal concentration differences during this study for Barium, Manganese, Zinc, Copper, Lead and Nickel. The metal concentration increased as the size fraction reduced towards the smaller-sized samples. As seen in the sample pictures (Figure 6.6) the visual aspects of the ash change as the size decreases. The colour changes from black/ brown to grey white as the combustion process is more complete and carbonaceous material is burnt off; this is consistent with the findings of Gill *et. al.*, (2013) and Goforth *et. al.* (1995).

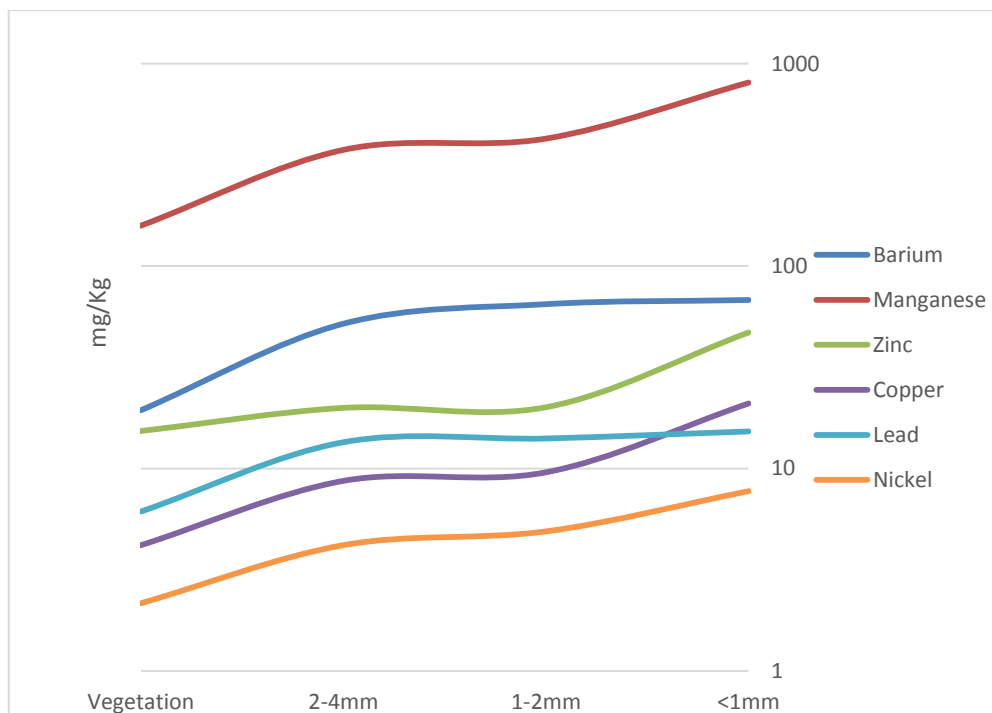


Figure 7.2: The influence of size fraction on metal concentration in ash

Figure 7.2 highlights that the vegetation averages are lower than the ash across the element spectrum for these metals. These results provided evidence that metal concentration increase as the carbonaceous material burns off and those metals remain attached to the ash residue. For instance, Barium, Copper, Lead, Nickel, Manganese and Zinc have a higher concentration attributed to varying particle sizes. However, three metals (Cadmium, Chromium and Vanadium) did not display the same increases in metal concentrations as depicted in Figure 7.2. There was a noticeable reduction in concentration levels from the 1-2 mm to the <1 mm size fractions as shown in Figure 7.3.

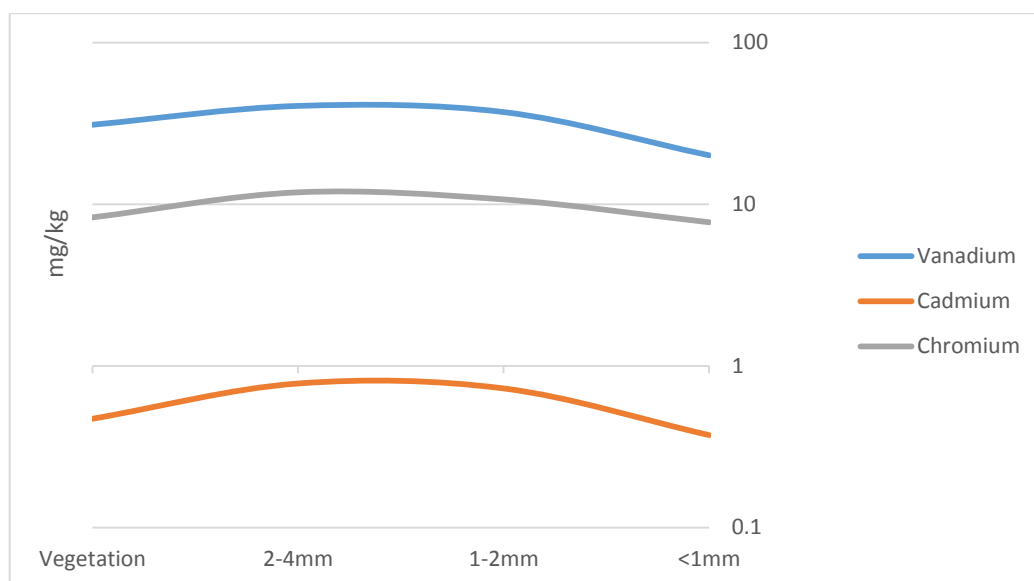


Figure 7.3: The convex relationship of Vanadium, Cadmium and Chromium to size fraction in ash

Even though it is thought that this trend was likely to be caused by differences in the formation of metal carbonates or oxides during the removal of carbonaceous matter in the combustion process. Another explanation of a drop in some of the metal samples is possible. An explanation for this occurrence is presented by Pereira *et. al.*, (2011) in his study of metals released from leaf litter when exposed to different fire temperatures. They found that the level of calcium carbonate in the ash increases at around 400°C and some metal concentrations are higher than when calcium carbonate is present while others displayed reduced concentration levels. In their studies, Iron and Zinc concentrations initially increased in low to medium temperature fires but decreased with exposure to higher temperatures, even at low pH. They concluded that at higher temperatures, the effect of pH and Calcium Carbonate inhibits the presence of metals when analysis is conducted.

It is also possible that these three metals reached their equilibrium and started to decrease. These metals could have reached their equilibrium and start to decrease if they were exposed to localised higher temperatures. This is unlikely for Chromium and Vanadium due to the high melting points but may be possible for Cadmium with a melting point of 321°C (Pohanish, 2002). It is beyond the scope of this study to explore this relational aspect but it could be a potential further study area.

7.2 Discussion by metal type

Only nine metals of the thirteen provided a statistically representative sample. Cobalt, Mercury, Arsenic and Beryllium provided insufficient results and will not be discussed further. Of the nine samples that provided sufficient results the average metal concentrations ranged from 0.56 mg/kg for Cadmium to 442.2 mg/kg for Manganese and the order based on quantity of material was Manganese > Barium > Zinc > Vanadium > Copper > Chromium > Lead > Nickel > Cadmium.

Comparisons are made between the concentrations identified in this study, the HIL levels and the potential risk to public health in situ. The discussion does not measure the effects of mobilisation of the ash offsite either through wind or water runoff, even though this is a distinct possibility. Also comparisons with EILs are mentioned as this not only presents a risk to the environment but could also provide low level impacts for humans

7.2.1 Manganese

Manganese was the most prevalent metal analysed in this study with concentrations ranging from 21 mg/kg to 3300 mg/kg, however 98% of samples exhibited concentrations below 200 mg/kg. The average results of this study were below the HIL threshold of 1500 mg/kg for Manganese in residential (A) areas but seven individual sample results exceeded the HIL threshold, with a maximum value of 3300 mg/kg recorded. Even though these individual samples exceeded threshold limits, the average level of Manganese found in the ash was not considered an acute risk to human health or the environment, based on consideration of overall average concentrations. This view was taken in light of the inherently heterogeneous nature of ash deposition, and areas of ash exhibiting above average concentrations of Manganese being isolated and not representative of the majority of samples collected or locations tested.

It should be noted that the 2010 DEC guidelines set the Environmental Impact level threshold level for Manganese at 500 mg/kg. The average Manganese levels in this study were approaching this threshold level with the overall average being 442.2 ± 462.6 mg/kg and 27% of individual samples were in excess of 500 mg/kg. Also, the average <1 mm size fraction (808.0 ± 716.8) exceeded the EIL threshold

Manganese levels found in this study were the closest to the relevant HIL or EIL screening criteria of all of the metals analysed. This is due to the prevalence of Manganese specifically within green leaf material as a result of its structural and activation role within chlorophyll protein complexes. Given that the majority of plant matter subject to combustion during a prescribed burn is leaf material, this would

account for high concentrations of Manganese identified in this study. It would be presumed that future studies will also record high levels of Manganese when assessing ash from combustion of leaf and other plant matter.

The release of Manganese in ash still poses a cumulative public health risk through the mobilisation of ash into water catchment areas and the potential concentration of metals in a plume that can cause a periodic exceedance of water quality parameters.

7.2.2 Barium

Barium was detected in 100% of samples analysed, with concentrations ranging from 3 mg/kg up to 400 mg/kg, with 96% of samples having concentrations below 200 mg/kg. The results of this study were well below the HIL of 15 000 mg/kg for residential (A) areas as well as being predominantly below the EIL of 300 mg/kg. Based on the consistently low levels of Barium detected within samples, in comparison to the HIL, it was concluded that Barium is unlikely to pose a serious risk to public health.

7.2.3 Zinc

Zinc was detected in 97% of samples in concentrations ranging from 0.7 mg/kg to 190 mg/kg with 87% of samples reporting concentrations between 1 mg/kg and 50 mg/kg. The average results of this study were significantly below the HIL threshold of 7000 mg/kg for Zinc in residential (A) areas.

The 2010 guidelines set the EIL threshold level for Zinc at 200 mg/kg. The average Zinc concentrations determined for ash samples analysed in this study were significantly below this threshold with the overall average being 25.6 ± 29.7 mg/kg, while no individual samples were recorded in excess of the EIL, four samples recorded concentrations above 150 mg/kg.

The overall Zinc levels were significantly below both the HIL and EIL criteria indicating that despite the presence of Zinc within 97% of samples, it is unlikely to pose a serious risk to human health or the environment.

7.2.4 Vanadium

Vanadium was detected in 88% of ash samples analysed with concentration ranges from 1.4 to 140 mg/kg, and 87% of samples recording concentrations between only 2 mg/kg and 5 mg/kg. The average results Vanadium in this study were significantly below the HIL threshold of 550 mg/kg for Vanadium in residential (A) areas and only one sample recorded a maximum concentration of 410mg/kg. Overall the level of Vanadium found in this study was not considered to be of consequence to human health or the environment, based on relatively low reported concentrations and considering the level and oxidised form of Vanadium being unlikely to represent either a public health exposure or toxicity risk.

Even though the current NEPM guidelines do not have a level set for Vanadium the 2010 guidelines set the EIL threshold level for Vanadium at 50 mg/kg. The average Vanadium levels in this study were below this threshold level with the overall average being 32.2 ± 51.0 mg/kg and only 16% of individual samples were in excess of 50 mg/kg.

7.2.5 Copper

Copper was detected within 100% of samples analysed, recording average concentrations between 3 mg/kg and 57 mg/kg, with 87% of samples reporting concentrations below 20 mg/kg. These results were all significantly below the HIL criteria of 1 000 mg/kg for residential (A) areas and the level of Copper found in the ash was not considered to be of consequence to human health or the environment considering both the concentrations detected and the relatively high concentrations of Copper required to represent a risk of harm or to result in Copper-induced.

The 2010 guidelines set the EIL threshold for Copper at 100 mg/kg. The average Copper levels in ash samples analysed within this study were on average 50% below the EIL criteria, with the maximum concentration recorded being 57mg/kg and the overall average only 10.8 mg/kg.

7.2.6 Chromium

Chromium was detected in 88% of samples analysed in this study with average concentrations ranging from 0.7 mg/kg to 72 mg/kg, and 75% of samples yielding concentrations between 1.0 mg/kg and 20 mg/kg. Despite analysis being conducted for total Chromium, with speciation for Chromium III and VI, it is considered reasonable to compare the results obtained for total Chromium to HIL and EIL values for Chromium III, due to the fact that given the sample matrix was composed of ash material, Chromium VI is unlikely to be present based on the oxidative conditions which occur during combustion.

The highest total Chromium level recorded was 72 mg/kg (mean 9.64) and this was lower than the HIL of 120, 000 mg/kg for Chromium III Residential (A) or 100 mg/hg for Chromium VI Residential (A) (DEC, 2010). The 2010 guidelines set the EIL threshold level for Chromium (III) at 400 mg/kg.

Overall the level of Chromium detected within samples in this study was not considered likely to represent either a health or environmental risk. This assertion is supported by the fact that the with combustion conditions favouring the formation of Chromium II and Chromium III species, these species are generally considered benign based on their weak membrane permeability, and are unlikely to pose risk, regardless of the concentrations reported.

7.2.7 Lead

The analysis for Lead undertaken in this study identified Lead in 86% of samples analysed. However, the average concentrations only ranged between 1 mg/kg and 38 mg/kg, with 83% of samples returning results between 1 mg/kg and 20 mg/kg. The results of this study were significantly below the HIL of 1000 mg/kg for residential (A) areas and the level of Lead found in the ash was considered to be unlikely to pose a risk to human health, with overall Lead levels in this study below the HIL threshold. Notwithstanding the relative toxicity of Lead generally, considering likely exposure routes of ingestion, toxic effects would still require consumption of gram quantities of Lead for acute exposure, which are considered unlikely.

The 2010 guideline set the EIL threshold for Lead at 600 mg/kg. The average Lead concentration identified in this study were significantly below these threshold criteria with the maximum concentration recorded being only 38 mg/kg and the overall average being 12.2 mg/kg.

7.2.8 Nickel

The study identified Nickel in 94% of samples analysed, with average concentrations ranging from 0.7 mg/kg to 18 mg/kg, and 73% of samples concentrations ranging between only 1.0 mg/kg and 10 mg/kg. The results of this study were also significantly below the HIL of 600 mg/kg for residential (A) areas. Given the primary toxicity and exposure risk associated with Nickel being an increased risk of pulmonary and nasal carcinogenesis, typically associated with long-term chronic exposures, the concentrations of Nickel reported in ash samples from this study are not considered likely to pose a public health or environmental risk.

The 2010 guidelines set the EIL threshold level for Nickel at 60 mg/kg. The average Nickel levels in this study were significantly below this figure with the maximum concentration recorded being 19 mg/kg, and an overall average concentration of 4.7 mg/kg.

7.2.9 Cadmium

The study identified Cadmium in 81% of samples analysed, reporting concentrations from 0.07 mg/kg to 5.7 mg/kg, and 78% of samples between 0.1 mg/kg and 3 mg/kg. All samples analysed for Cadmium were below the HIL criteria of 20 mg/kg for residential (A) areas, with the maximum concentration recorded at 5.7 mg/kg and the overall average only 0.6 mg/kg.

The 2010 guidelines set the EIL threshold criteria for Cadmium at 3 mg/kg. The average Cadmium levels in the ash were below this criterion with the maximum concentration reported being 0.8 mg/kg in the 2-4 mm size fraction sample. However, there were two individual samples collected that exceeded the EIL of 3 mg/kg. One of the samples from the Lesmurdie area recorded a result of 5.3 mg/kg and one sample from Mundaring returned a result of 5.7 mg/kg. However, considering the heterogeneous nature of ash deposition and overall average concentrations and sample distribution, these results are not considered statistically significant.

The results of this study demonstrate that Cadmium concentrations in the ash are consistent with background levels reported in the 2010 DEC technical document. Overall, it was found that Cadmium concentrations measured in this study do not pose a health or environmental exposure risk.

7.3 Public health implications

The metal levels in this study do not present a health or environmental risk if left in-situ but they may pose problems if they become mobilised by either wind or water. There is the potential for metals to accumulate in rainwater tanks from aerial deposition of ash (Spinks *et. al.*, 2006). Studies into water quality have also shown that metal levels can increase significantly from pre-fire conditions due to wind deposition, increased runoff and potential accumulation in temporal or dry waterways (Ebel, Moody, and Martin, 2012; Bowman and Bloggs, 2006). The period between May and October where the majority of prescribed burns are conducted is associated with high rainfall levels thus increasing the likelihood of mobility of the ash through runoff (BOM, 2017).

It was possible to estimate the total amount of metals in the ash released during each burn and over the total of the three burns (Table 7.2). The total area burnt during the prescribed burns was approximately 3 320 hectares having the potential to generate between 30.8 and 153.7 tonnes of ash. This ash release estimation is based on the percentage of ash generated during the prescribed burns using 0.4% lower limit and 2% upper limit as identified in the literature (Misra *et. al.*, 1993; Santin, *et. al.*, 2012; Kumar *et. al.*, 2009).

Table 7.2: Estimated total release of metals during each of the burns

	Lesmurdie	Mundaring	Bullsbrook	Total
Area (Ha)	50	1100	2170	3320
Fuel Load (t/Ha)	19.3	12.5	9	10.3
Total fuel load (t)	965	13750	19530	34245
Reduction to 8 t/Ha	11.3	4.5	1	2.3
Total reduction (t)	565	4950	2170	7636
Ash content - Lower 0.4% (t)	2.3	19.8	8.7	30.8
Ash content - Upper 2.0% (t)	11.3	99	43.4	153.7

As seen in Table 7.3 there were nine metals that provided statistical results with Cadmium having the lowest release quantity with between 117 and 585 grams being released from all burns while Manganese had the highest quantity released between approximately 19.2 and 461.9 kilograms.

Table 7.3: Estimation of total release of metals during prescribed burns during this study

Metal	Concentration mg/kg	Lesmurdie		Mundaring		Bullsbrook		Total	
		0.4%	2%	0.4%	2%	0.4%	2%	0.4%	2.0%
Barium	41.8	96.1 g	480.5 g	827.6 g	4.1 kg	363.7 g	1.8 kg	1.3 kg	6.5 kg
Cadmium	0.56	1.3 g	6.5 g	11.1 g	55.5 g	4.9 g	24.5 g	17.2 g	86.1 g
Chromium	9.6	22.1 g	110.5 g	190.1 g	950.5 g	83.5 g	417.5	295.7 g	1.5 kg
Copper	10.8	24.8 g	124.0 g	213.8 g	1.1 kg	94.0 g	470.0 g	332.6 g	1.7 kg
Lead	6.1	14.0 g	70.0 g	120.8 g	604 g	53.1 g	265.5 g	187.9 g	939.5 g
Manganese	442.2	1017.1 g	5.1 kg	8.8 kg	44.0 kg	3.8 kg	19.0 kg	13.6 kg	68.0 kg
Nickel	4.7	10.8 g	54.0 g	93.1 g	465.5 g	40.9 g	204.5 g	144.8 g	724.0 g
Vanadium	20.1	46.2 g	231.0 g	397.9 g	1.9 kg	174.9 g	874.5 g	619.1 g	3.1 kg
Zinc	25.6	58.9 g	294.5 g	506.9 g	2.5 kg	222.7 g	1.1 kg	788.5 g	3.9 kg

The WA Government aims to burn 200 000 hectares of forest each year; it is therefore possible to estimate the total potential release of metals from prescribed burns. Using the average fuel reduction from this study of 2.3 tonnes per hectare, this provides a total of 460,000 tonnes consumed to generate between 1 840 (0.4%) tonnes and 9 200 (2.0%) tonnes of ash.

Table 7.4: Estimation of total release of metals during the yearly goal of 200,000 hectares burnt through prescribed burns

Metal	Concentration mg/kg	0.4 % ash content	2.0 % Ash content
Barium	41.8	76.9 kg	384.5 kg
Cadmium	0.56	1.0 kg	5.0 kg
Chromium	9.6	17.7 kg	88.5 kg
Copper	10.8	19.9 kg	99.5 kg
Lead	6.1	11.2 kg	56.0 kg
Manganese	442.2	813.7 kg	4.1 t
Nickel	4.7	8.7 kg	43.5 kg
Vanadium	20.1	37.0 kg	185.0 kg
Zinc	25.6	47.1 kg	235.5 kg

The potential quantities of metals released in WA from prescribed burns become quite large (Table 7.4) with 4.1 tonnes of Manganese being mobilised in the ash. When considering the quantity release it is also important to note the TRV for ingestion of each metal. These values are outlined in Chapter 2.3 with Cadmium being the most toxic (ingestion) with a TRV of 0.0008 mg/kg/day while Zinc being the least toxic with a TRV of 0.15 mg/kg/day.

This modelling quantifies the possible public health risk posed by the release and mobilisation of metals in ash. The volumes released across WA and especially the south west of the state, with a high population density and fuel source, can potentially cause significant long term and accumulative health issues for nearby

population water supplies. The factors associated with mobilisation are complex (described in Chapter 2.4) but as Canberra and other areas have experienced with the right conditions can become a distinct possibility.

Over the long term it would be difficult to identify a direct causal effect relationship of metal toxicity and prescribed burns due to the low level cumulative nature of the exposure. However due to an increasing population age profile and the likely increased occurrence due to population encroachment and climate change this exposure cannot be discounted.

7.3.1 Management Implications

The results obtained from this study provide important information for the State Government. In the future, leaf litter analysis can be utilised to identify background levels of metals in areas identified for fire management to determine what levels of post fire contamination might be contaminated from other sources (P. Newell, Manager Contaminated Sites, Department of Environment Regulation, personal communication, May 15, 2017).

The findings from this study are useful for comparison of ash from the source with metals found in sediment. (F D'Souza, Principal Scientist, Department of Water, personal communication, March 7, 2017).

In terms of management application prescribed burns are seen as a preventive method of reducing the potential of uncontrolled fires but the potential for metals in sediment flow through runoff cannot be ignored. Elevated Cadmium levels in Canberra's water supply after the 2003 fires (Smith, *et. al*, 2011a) are an example of this occurrence. Where areas are burnt adjacent to water edges, prescribed

burns can contribute to sediment flow and the introduction of metals into waterways and consideration should be made to restrict the movement of ash into these water ways.

Chapter 8: Study Conclusions

This study investigated the potential public health effects from the release of metals from a prescribed burn. The overall aim of this study was to determine the public health implications from concentration of metals in ash. The focus of the study was specifically on 13 metals (Arsenic, Barium, Beryllium, Cadmium, Chromium, Cobalt, Copper, Lead, Manganese, Mercury, Nickel, Vanadium and Zinc), which are used for screening purposes in contamination investigations, often referred to as the NEPM 13. To satisfy the aim of the study a number of objectives were established.

The first objective involved the measurement of metals in unburnt vegetation in the location of a prescribed burn. This was achieved through the collection of forest ground vegetative litter within the sample area. The results of the analysis identified metals were present in the vegetative material.

The second objective was to collect and analyse resulting ash. A measurement of metal concentrations was conducted using three ash size fractions (<1 mm, 1-2 mm, 2-4 mm) as a result of the ash generated during prescribed burns. Ash from three sites was collected to provide an overall profile of the metal characteristics found in the Darling escarpment prescribes burn sites.

The third objective was to determine the relationship between vegetation and ash metals. The ash and vegetation results of the study were compared using statistical analysis, specifically using ANOVA methodology. The ANOVA test was used to determine the impact of independent variables have on the dependent variable in a regression analysis. It was shown that two main patterns emerged based on size fractions. A general increase in concentrations in the majority of

metals and in some instances a decrease in concentration as the particle size reduced.

The final and main objective was to explore the relationship between metal concentrations of the ash pollutants studied and potential adverse health and environmental effects. At the time of the research the Department of Environment and Conservation (2010) *Assessment levels for soil, sediment and water* guidelines was used to determine whether the metal concentrations exceeded HIL and EIL thresholds. The 2014 NEPM standards were also used for comparisons. In all but one of the cases, the 2010 guidelines were more conservative and, therefore, this document used for comparison purposes. The results from this study were below the HILs and the results, and with the exception of Manganese in the <1 mm size fraction, were below EILs.

To address particular aspects identified in the objectives, four specific research questions were generated to fulfil the aim and objectives of the study. The first question focused on the metal ash concentrations of open jarrah forests in the Darling escarpment. The results identified the presence of metals in all ash samples. With the exception of Arsenic, metal results were recorded in the unburnt vegetation samples.

Due to the fact that the majority of Arsenic, (3.821 +/- 1.614 mg/kg), Beryllium (0.095 +/- 0.056 mg/kg), Cobalt (1.177 +/- 0.787 mg/kg) and Mercury (0.022 +/- 0.077 mg/kg) samples were below the detection level of the analytical laboratory method a detailed statistical analysis for these metals could not be conducted. Of the remaining metals analysed the order of concentration levels are as follows;

Manganese > Barium > Zinc > Vanadium > Copper > Chromium > Lead > Nickel > Cadmium.

The second question asked if there was a difference in the metal composition associated with the size fractions of ash, namely 2-4 mm, 1-2 mm and <1 mm sizes. The study demonstrated a relationship between the size fractions and levels of heavy metals. A distinct pattern emerged as the combustion process became more complete; metals concentrated in the smaller ash fractions due to the carbon component being burnt off. In some of the samples this increase tapered off. A review of other studies provides a possible explanation in that the formation of Calcium Carbonate at around 400°C reduced the metal levels available for analysis. Even though this aspect was not explored in this study it does provide a possible explanation. Prescribed burns are considered low to medium in their heat generation and this corresponds with the 400°C temperature threshold.

The third question explored the relationship between the presence of metals in unburnt ground vegetation prior to a prescribed burn and the ash generated after the burn. The results of this study established that there was indeed a relationship. Metals in vegetation samples were lower than those found in the ash. This supports the first relationship and it is expected that the removal of carbonaceous material is the main contributor to the increased metal composition found in the ash.

The final question asked whether the concentrations of metals in ash are within the acceptable HIL and EIL as established by NEPM and other government standards. When compared to the 2010 guidelines, the concentrations of metals in the ash were within the acceptable HILs and EILs. Only Manganese in the <1 mm

size fraction exceeded the EIL but, overall, Manganese was below the level of 500 mg/kg.

The results of this study highlighted that metals were present in the post-fire ash and that the concentrations varied with the ash size fraction. The study also established a relationship between the ash generated from the prescribed burn and the unburnt vegetation in the area prior to the burn. It was also shown that the metal concentration levels identified did not exceed HILs and, with the exception of Manganese, did not exceed EILs.

An issue of concern is that the ash could potentially be mobilised into waterways through runoff or rainwater systems by the airborne deposition, though this is dependent on the weather characteristics at each event. This mobilised ash can potentially concentrate to levels that make the water non-potable, as was the case after the Canberra bushfires in 2003. Further research is required to determine mechanisms associated with this process.

The findings of this study have a public health application. They contribute to estimating the amount and concentration of metals that can be expected in the ash after a fire, and to approximate any metal loading if the ash is mobilised by wind or water into drinking water catchment areas. Agencies and landowners can use this information to conduct remedial erosion work to prevent the mobilisation of the ash. They can also advise on-site drinking consumers about methods for testing and preventing ash contaminating their drinking water supplies.

8.1 Recommendations

Studies can only capture certain aspects of the overall phenomenon being explored. It became apparent that there were limitations in the study design and the available resources. It also became obvious that further studies could be conducted to improve the overall knowledge base in this research area. Despite these limitations, the knowledge gained from this study has management applications that will assist in reducing potential ash exposure to nearby populations during prescribed burn events.

8.1.1 Implications for practical use

This study has demonstrated the presence of metals in ash from open jarrah forests. In the planning of prescribed burns, or the post-fire analysis of bushfires, agencies should give due consideration to the possibility of contamination of waterways or likely airborne deposition of ash on water collection areas. Introducing an item for consideration within the BRMP to identify areas of potential water contamination will reduce possible contamination.

Management considerations should include measures that mitigate the movement of ash through runoff controls or advise communities downwind of a fire about measures to prevent the ash entering drinking water systems.

By ensuring that the issue is identified and mitigation measures are in place, the DPAW can provide the evidence that it is compliant with the various legislative requirements including the provisions of the *Environmental Protection Act, 1986*.

8.1.2 Future studies

This study provides a connection between metals and the size fraction of ash and recognises that samples are not all uniform across the size spectrum. It became apparent during the study that significant gaps remained and there are many areas that require further exploration. The following suggestions will not only test the results gained from this study but will provide connections to other fields of study.

- This study can be replicated in other similar forest types such as karri, mallee and wandoo forests, as well as in other habitats such as grasslands and savannah landscapes. It is expected that this will confirm the results of this study as well as provide variation associated with particular vegetation and soil substrates.
- This study could be repeated in a bushfire scenario, which would provide a comparison between a low and high intensity fire. Safety consideration would be a limiting factor.
- Research could be conducted to analyse the source, pathway and receptor of metals. Other studies did not demonstrate a relationship from this viewpoint. Samples could be taken from the ash on-site, the ash plume in the waterway, or via air samples and from the receiving deposition area. The tin roof (Spinks, *et. al.*, 2006) experiment could not discount the fact that metals in water tanks did not come from other sources. This type of study can provide these links if it is designed correctly.
- The aim of this study was to provide a link between vegetation and ash and, as such, it focused on the larger ash size fractions. This methodology

allows for a sieve of much smaller diameters than an Endecott sieve of 20 microns. One of the outcomes was that metals can increase in concentration up to a certain temperature. Would this hypothesis be tested by examining finer ash sizes?

- The initial proposal for this research to study smoke is still considered as valid. If equipment problems can be overcome that would add another dimension to the area of knowledge. It could also provide a link to the smaller size fractions used in other proposed studies.

These study proposals are dependent on funding, interest and industry-based opportunities. If they can be explored they may add to the understanding of processes involved in this field of research.

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Appendix 1: Air pollutants in bushfire smoke in WA: a pilot study

Introduction to initial pilot study on wood smoke

The aim of this pilot study was to ascertain the capacity of available equipment to determine concentrations of a range of air pollutants in smoke generated at source during prescribed burns in the Darling escarpment, east of Perth's metropolitan area. To meet this aim the pilot study had the following objectives:

- Measure concentrations of a range of particulate of sizes (TSP, PM₁₀, PM_{2.5},) and metals that occur as a result of smoke generated during prescribed burns.
- Determine the relationship between fuel load, prescribed-burn characteristics and concentrations of TSP, PM₁₀, PM_{2.5} and metals.
- Establish the relationship between the concentrations of the air pollutants studied and potential adverse human health effects.

Methodology

A range of air pollutants (TSP, PM₁₀, PM_{2.5}, carbon monoxide and volatile organic compounds) and metals generated during the prescribed burn were measured on 30 April 2013. This was conducted during a prescribed burn in the Swan Coastal Plain area within the Bickley town site. The total area for the prescribed burn was 20 hectares with a 2.1 kilometre perimeter. Sampling was undertaken 400 metres from the south-western edge and approximately 5 metres from the edge of the burn area.

Local fire front weather data (temperature, wind speed and direction, and relative humidity) were collected during the sampling phase with the aid of a Davis Vantage Pro two mobile weather station (Davis Instruments Corp., Hayward, CA, USA). Regional weather data were also sourced from the Bureau of Meteorology and compared to the local fire front data.

The concentrations of airborne particulate matter were measured continuously during the sampling period using a Turnkey DustMate (Turnkey Instruments Ltd. Northwich, Cheshire, UK) that measures TSP, PM₁₀ and PM_{2.5} concentrations. Sampling was conducted as close to the fire front as was possible, (approximately 5 metres from the area to be burnt) while ensuring the safety of the researcher and the equipment. The DustMate was operated continuously during the sampling period.

Six TSP particulate samples were collected on 47 mm, 0.8 µm mixed cellulose ester filter papers using three MicroVol 1100 Low Volume Air Samplers (Ecotech Pty Ltd., Knoxfield, Victoria, Australia) internally calibrated at of flow rate of 3 L/min for a period of 90 minutes. A total of six samples were collected from two different sampling points over a 1.5-hour period. The samples were analysed by Analytical Reference Laboratories in Western Australia using method reference ARL No. 401: metals in soil and sediment by ICP-OES.

A VRAE Hand-Held 5-Gas Surveyor (RAE Systems Inc., San Jose, CA, USA) was used to sample carbon monoxide and volatile organic compounds (VOCs) as combustible gases (%VOL). This instrument was located with the other equipment, 5 metres from the fire front.

Bulk ash samples were collected from the burn area with the aid of a small plastic shovel. Approximately 300 grams of ash per sample was collected. The sample was stored in a sealable bag for laboratory analysis of metals. Thirteen metals were chosen for analysis as identified by the NEPM as the screening measure for possible metal contamination. The 13 metals included Arsenic, Barium, Beryllium, Cadmium, Chromium, Cobalt, Copper, Lead, Manganese, Mercury, Vanadium, Nickel, and Zinc. The ash was analysed using NEPM method number 201 (US EPA method 3120) by Analytical Reference Laboratories in Western Australia.

Routine site data were collected from DEC field officers during the planning phase of a prescribed burn that included predicted wind speeds, wind direction, temperature, and relative humidity as well as fuel loads and fuel types.

Results

Weather conditions

During the sampling period (12:30pm to 15:35pm) the wind direction was predominantly north-to-north east. The weather conditions were considered fine with an average temperature of 27.7°C that ranged from 26-29°C. Humidity ranged between 38-69% with the lowest levels during the middle of the burn as shown in Figure A1.

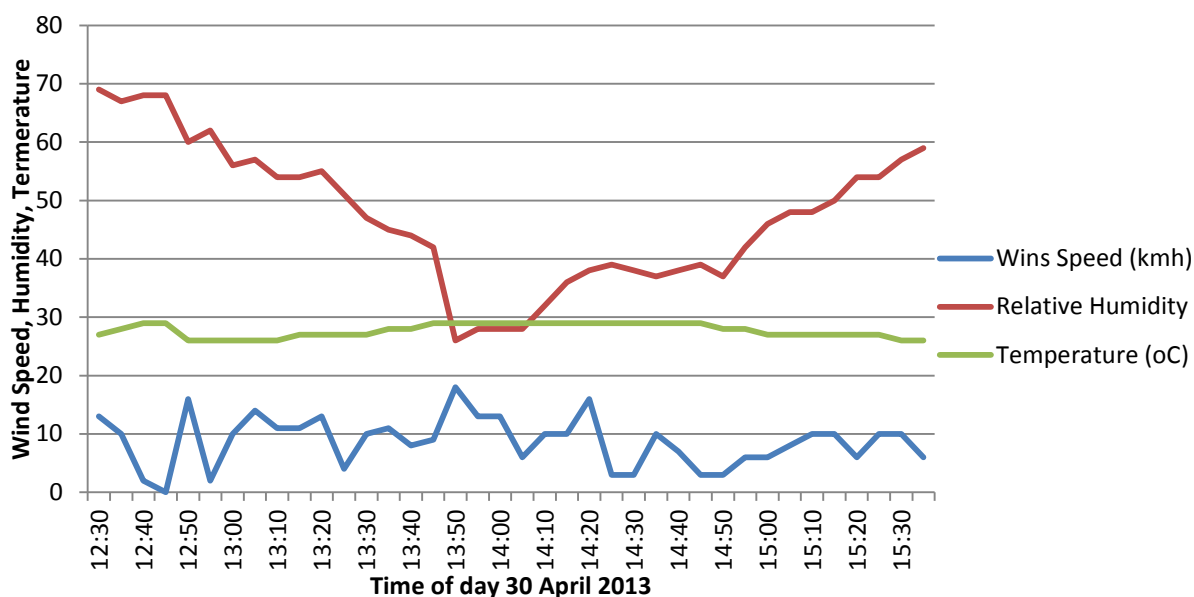


Figure A1: Temperature, Relative Humidity and Wind speed during Sampling
Period

Particulate concentrations in the smoke plume

The concentrations measured showed that the levels of TSP and PM₁₀ (Figure A2) exceeded levels of 6500 µg/m³ for the majority of the sampling period. This is considerably above the 24 hour NEPM for PM₁₀ of 50 µg/m³ and also exceeds the maximum level of detection of the instrument. The only time when the measured concentrations of TSP and PM₁₀ were below 6000 µg/m³ was during a period when a sudden gust of wind briefly cleared smoke around the sampling equipment. The PM_{2.5} concentrations averaged 579.73 µg/m³, which is also above the 24 hour NEPM guideline for PM_{2.5} of 25 µg/m³ (Franklin and Goetzmann, 2012).

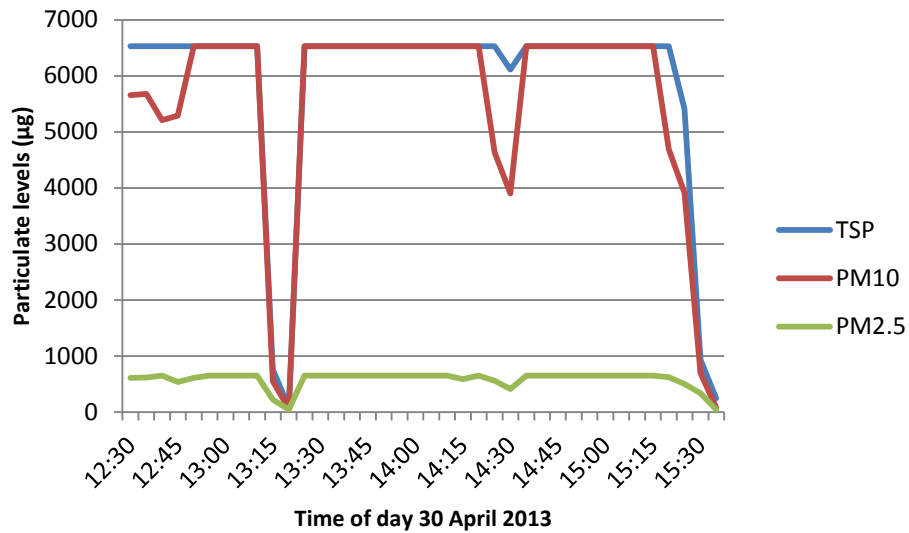


Figure A2: TSP, PM₁₀ and PM_{2.5} Measured During Sampling Period

Carbon monoxide and volatile organic compound levels

Carbon monoxide concentrations (Figure A3) averaged 10.8 part per million (ppm) with a peak of 38 ppm at 13:50pm. VOC levels had an average of 0.2 ppm and a peak level of 0.9 ppm at 13:50pm, which are well below the advisory standard for BTEX (benzene, toluene, ethylene benzene and xylene) of 5 ppm for benzene, the most toxic of the BTEX chemicals (Department of Health and Human Services, 2007; Department of Environment and Heritage, 2004).

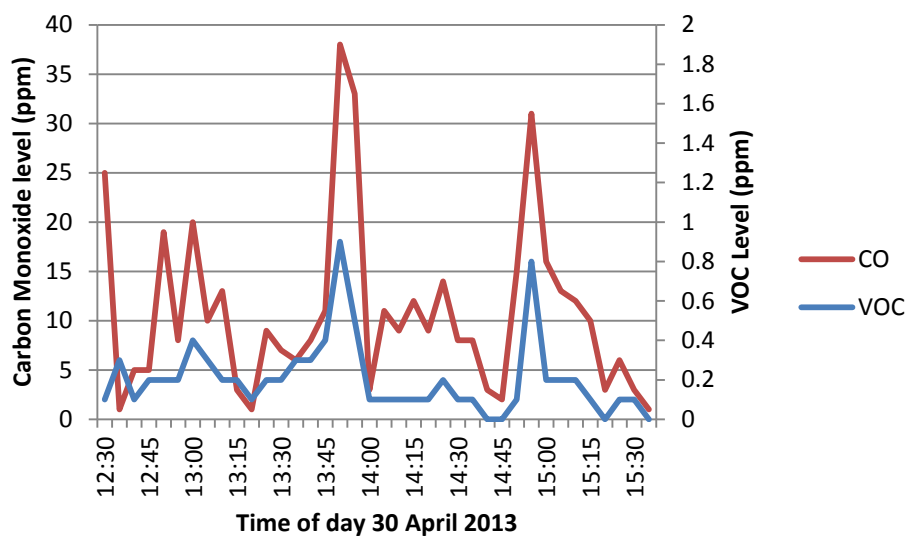


Figure A3: Carbon Monoxide and VOC Concentrations

Smoke sample

All six air samples analysed for metals were below the laboratory limit of reporting. Samples 1, 2 and 3 were collected during the first 90 minutes of the sampling period and samples 4, 5 and 6 were collected during the second 90 minutes of the sampling period.

Site 1: PHS 807 Welshpool Road, Lesmurdie

The weather characteristics for 1 July 2013 were a temperature of 16°C with a north-westerly wind at 15 kilometres per hour with gusts up to 25 kilometres per hour. The fuel load of the site was between 15.2 tonnes per hectare and 48 tonnes per hectare with an average litter depth of 30 mm. It was anticipated that this would produce a flame height up to 10 metres with a rate of spread between 20 and 50 m/hr.

Ash and litter sample

Three ash samples, collected from within the burn area, contained detectable levels of metals (Table A1). The only metals not detected in the samples were Mercury and Beryllium. Arsenic was identified at one site just above the level of detection.

Table A1: Litter and Ash samples results of the pilot study

Metals in soil and sediment	LoD	Units	Ash Site 1A 13-3223-1	Ash site 1b 13-3223-2	Ash Site 1c 13-3223-3
Arsenic	5	mg/kg	<5	<5	5
Barium	1	mg/kg	90	91	100
Beryllium	0.1	mg/kg	<0.1	<0.1	<0.1
Cadmium	0.1	mg/kg	3.7	3.2	4.2
Chromium	1	mg/kg	28	21	36
Cobalt	1	Total µg	1	2	2
Copper	1	mg/kg	25	43	30
Lead	1	mg/kg	21	24	26
Manganese	1	mg/kg	430	460	510
Mercury	0.02	mg/kg	<0.02	<0.02	<0.02
Nickel	1	mg/kg	3	4	5
Vanadium	2	mg/kg	200	180	220
Zinc	1	mg/kg	83	91	97

Discussion of pilot study

The weather conditions on the day of sampling were considered optimal for sample collection. Winds were constant in both speed and direction, allowing for a stable sampling platform in the path of the smoke plume. The humidity dropped during the middle of the sampling period and rose towards the end. It is concluded that this drop was associated with the flaming stages of the fire.

Occasional squalls highlighted the influence of wind on particulate levels. In addition, the velocity of the wind was also related to the dust particle sizes, indicating that wind conditions strongly influence the mobilisation of small particle materials. This study did not identify at what particle size threshold this influence is reduced.

Carbon monoxide and VOC samples showed a strong relationship to each other as was noted when there was a peak in carbon monoxide levels with a corresponding peak in VOC levels.

Pereira (2012) found that ash could still be mobilised by wind for a period up to 15 days after a fire. Air entrainment of fine particles is highest immediately after the fire, while the coarser particles tend to be entrained in the weeks after the burn. Ash can be deposited metres to kilometres away depending on the severity of the burn (complete combustion), wind speed, and topography.

Despite recording high-smoke levels throughout the experimental period and visible dust on the filter paper, there was insufficient particulate matter to allow reliable measurement of metals. This was due to the MicroVol 1100 only collecting

some 270 litres during the 90-minute sampling period, which was insufficient for meaningful analysis.

It was concluded that if this instrument was to be used in the project the sampling period would have to be extended significantly to gather more material. Alternatively, a different method of sample collection would have to be considered (such as high volume sampling) to ensure sufficient amounts of material could be collected to enable reliable analysis.

The ash and litter grab samples demonstrated that metals were present in the ash. These samples were a conglomerate of particle sizes, but it was concluded that similar concentrations might be present in the smoke plume.

Conclusion relating to pilot study

The pilot study identified that high levels of TSP, PM₁₀ and PM_{2.5} were present at the source of a prescribed burn and that a relationship existed between levels of carbon monoxide and VOC generation. Furthermore, it was demonstrated that the field sampling equipment used was not suitable for this type of study.

The MicroVols could not collect enough sample material for the laboratory to obtain a valid result for metal analysis. Only TSP was analysed and the flow rate of three litres per minute could not be increased, as this would have caused the filters to tear. Simply increasing the sample period was not deemed to be a viable option as the equipment would have to be moved with changes in wind conditions and proximity to the fire front. It was decided that an alternative sample collection method would need to be employed, which involved collecting a larger volume of air over the same period.

The DustMate was operating at its limit for the majority of the experiment for all of its parameters. Even though this highlighted that smoke levels were well beyond the concentration of $500 \mu\text{g}/\text{m}^3$ established by the Health Department of Western Australia, it did not indicate the true level of smoke present in the plume. Due to the upper limit of the DustMate's parameters, a relationship could not be established between carbon monoxide and VOC and dust-level increases. The TSI Dusttrak sampling instrument has a higher upper limit of $15,000 \mu\text{g}/\text{m}^3$ (rather than $6500 \mu\text{g}/\text{m}^3$). This would have provided an accurate representation of the upper limits of the plume characteristics in relation to particle size levels. Such an instrument was not available at the time of the experiment.

The ash analysis for metals indicated that metals might be present in the smoke plume. Due to the limitations of field sampling equipment it was decided to explore the metal concentration levels from ash instead of smoke at prescribed burns. Even though this approach changed the original focus of the study, the information gained can support further studies investigating metal concentrations in smoke plumes.

In conclusion, the research questions could not be fully answered using the initial methods and equipment. The study proposal has been amended to focus on an analysis of leaf litter and various size fractions of ash.

Appendix 2: ANOVA post-hoc test results for Metals conducted in this study

Table B1: ANOVA post-hoc test results for Manganese

Particle Size	Comparison	M difference	Level of Significance
<1 mm	1-2 mm	381.333	0.008
	2-4 mm	432.444	0.002
	Vegetation	649.511	0.000
1-2 mm	<1 mm	-381.333	0.008
	2-4 mm	51.111	0.767
	Vegetation	268.178	0.000
2-4 mm	<1 mm	-432.444	0.002
	1-2 mm	-51.111	0.767
	Vegetation	217.067	0.000
Vegetation	<1 mm	-649.511	0.000
	1-2 mm	-268.178	0.000
	2-4 mm	-217.067	0.000

Table B2: ANOVA Post-Hoc test results for Barium

Particle Size	Comparison	M difference	Level of Significance
<1 mm	1-2 mm	3.156	0.997
	2-4 mm	16.067	0.602
	Vegetation	48.622	0.000
1-2 mm	<1 mm	-3.156	1.000
	2-4 mm	12.911	0.935
	Vegetation	45.467	0.003
2-4 mm	<1 mm	-16.067	0.602
	1-2 mm	-12.911	0.805
	Vegetation	32.556	0.003
Vegetation	<1 mm	-48.622	0.000
	1-2 mm	-45.467	0.003
	2-4 mm	-32.556	0.003

Table B3: ANOVA Post-Hoc test results for Zinc

Particle Size	Comparison	M difference	Level of Significance
<1 mm	1-2 mm	26.927	0.005
	2-4 mm	27.073	0.004
	Vegetation	31.698	0.001
1-2 mm	<1 mm	-26.927	0.005
	2-4 mm	0.146	1.000
	Vegetation	4.771	0.324
2-4 mm	<1 mm	-27.073	0.004
	1-2 mm	-0.146	1.000
	Vegetation	4.625	0.313
Vegetation	<1 mm	-31.699	0.001
	1-2 mm	-4.771	0.324
	2-4 mm	-4.625	0.313

Table B4: ANOVA Post-Hoc test results for Vanadium

Particle Size	Comparison	M difference	Level of Significance
<1 mm	1-2 mm	-17.067	0.065
	2-4 mm	-20.476	0.047
	Vegetation	-10.944	0.776
1-2 mm	<1 mm	17.067	0.065
	2-4 mm	-3.409	0.986
	Vegetation	6.123	0.966
2-4 mm	<1 mm	20.476	0.047
	1-2 mm	3.409	0.986
	Vegetation	9.532	0.897
Vegetation	<1 mm	10.944	0.776
	1-2 mm	-6.123	0.966
	2-4 mm	-9.532	0.897

Table B5: ANOVA post-hoc test results for Copper

Particle Size	Comparison	M difference	Level of Significance
<1 mm	1-2 mm	11.378	0.000
	2-4 mm	12.267	0.000
	Vegetation	16.778	0.000
1-2 mm	<1 mm	-11.378	0.000
	2-4 mm	0.889	0.760
	Vegetation	5.400	0.000
2-4 mm	<1 mm	-12.267	0.000
	1-2 mm	-0.889	0.760
	Vegetation	4.511	0.000
Vegetation	<1 mm	-16.778	0.000
	1-2 mm	-5.400	0.000
	2-4 mm	-4.511	0.000

Table B6: ANOVA post-hoc test results for Chromium

Particle Size	Comparison	M difference	Level of Significance.
<1 mm	1-2 mm	-2.971	0.407
	2-4 mm	-4.108	0.279
	Vegetation	-0.557	0.996
1-2 mm	<1 mm	2.971	0.407
	2-4 mm	-1.136	0.976
	Vegetation	2.414	0.839
2-4 mm	<1 mm	4.108	0.279
	1-2 mm	1.136	0.976
	Vegetation	3.550	0.674
Vegetation	<1 mm	0.557	0.996
	1-2 mm	-2.414	0.609
	2-4 mm	-3.550	0.462

Table B7: ANOVA post-hoc test results for Lead

Particle Size	Comparison	M difference	Level of Significance
<1 mm	1-2 mm	1.178	0.890
	2-4 mm	1.756	0.707
	Vegetation	9.118	0.000
1-2 mm	<1 mm	-1.178	0.890
	2-4 mm	5.778	0.985
	Vegetation	7.940	0.000
2-4 mm	<1 mm	-1.756	0.707
	1-2 mm	-0.578	0.985
	Vegetation	4.511	0.000
Vegetation	<1 mm	-9.118	0.000
	1-2 mm	-7.940	0.000
	2-4 mm	-7.363	0.000

Table B8: ANOVA Post-Hoc test results for Nickel

Particle Size	Comparison	M difference	Level of Significance
<1 mm	1-2 mm	2.838	0.032
	2-4 mm	3.546	0.003
	Vegetation	5.570	0.000
1-2 mm	<1 mm	-2.838	0.032
	2-4 mm	0.708	0.761
	Vegetation	2.732	0.000
2-4 mm	<1 mm	-3.546	0.003
	1-2 mm	-0.708	0.761
	Vegetation	2.024	0.005
Vegetation	<1 mm	-5.570	0.000
	1-2 mm	-2.732	0.000
	2-4 mm	-2.024	0.005

Table B9: ANOVA post-hoc test results for Cadmium

Particle Size	Comparison	M difference	Level of Significance
<1 mm	1-2 mm	-0.352	0.087
	2-4 mm	-.405	0.045
	Vegetation	-0.099	0.924
1-2 mm	<1 mm	0.352	0.087
	2-4 mm	-0.053	0.993
	Vegetation	0.253	0.609
2-4 mm	<1 mm	0.405	0.045
	1-2 mm	0.054	0.993
	Vegetation	0.306	0.462
Vegetation	<1 mm	0.099	0.924
	1-2 mm	-0.253	0.609
	2-4 mm	-0.306	0.462