Fire and Soils

A review of the potential impacts of different fire regimes on soil erosion and sedimentation, nutrient and carbon cycling, and water quantity and quality

Tulau, M.J. (2015). Fire and Soils. A review of the potential impacts of different fire regimes on soil erosion and sedimentation, nutrient and carbon cycling, and impacts on water quantity and quality.

Cover photo: bare soil surface following fire. Photo: Sally McInnes-Clarke

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Summary

NSW National Parks and Wildlife Service (NPWS) is implementing an Enhanced Bushfire Management Program (EBMP), which will provide for an increase in hazard reduction (HR) activities, including both mechanical works and prescribed fire. However, the possible impacts of prescribed HR fire on soil-related issues such as nutrient cycling, soil erosion and sedimentation, and effects on water quantity and quality, are not fully understood.

NPWS therefore requested the Science Division of the Office of Environment and Heritage (OEH) to prepare a literature review summarising the current state of knowledge on the impacts of fire on soils, focusing on Australian research and knowledge relevant to NSW landscapes.

A key theme of this review is the relative risks and effects of wildfire compared to HR burning. Specifically, NPWS asked whether increasing HR targets and more frequent burning could result in adverse impacts on soils and the hydrologic responses of catchments.

Despite the enormous body of research that has examined soil-fire interactions in relation to nutrient supply and cycling, soil erosion and sedimentation, and impacts on water quantity and quality, the vast majority of this research has overwhelmingly concentrated on the effects of wildfire rather than on the effects of low severity burns such as HR burns. This is because wildfires can have catastrophic impacts on nutrient pools, post-fire soil erosion, sedimentation and water quality, and long-term impacts on the quantity of water discharging from affected catchments. In contrast, relatively little research has been carried out into the impacts of low severity burns, where the effects are generally more subtle and variable. Furthermore, much of the research comes from overseas, mainly countries with a Mediterranean climate and especially prone to wildfire. Much of this overseas literature is not particularly relevant to Australia due the generally thin, nutrient-poor soils, and the different landscapes and climatic patterns in this country.

Fire typically results in the reduction of fuel and soil organic nutrient pools by oxidation, volatilisation, convection, leaching, and erosion of nutrients from a site. Different nutrients may respond differently to each of these processes, but certain nutrients including carbon (C), sulfur (S) and nitrogen (N) are particularly susceptible to fire-related losses, due to their lower temperatures of volatilisation. Compared to wildfire-related losses, individual HR burns typically result in lower nutrient losses. However, nutrient losses following HR burns may still be significant in the Australian context, especially in nutrient-poor sandy soils. Furthermore, there is concern that frequently repeated burning may progressively deplete sites of nutrients and significantly reduce rates of nutrient cycling.

The magnitude of changes to catchment hydrological responses depends on the severity of a fire and a range of other factors. Following intense bushfires, runoff becomes more responsive to rainfall events, with runoff/infiltration ratios and stream flows generally increasing. As a result, there is an increased potential for sediment movement, by up to several orders of magnitude, especially if a severe fire is followed by rainfall events of high intensity and erosivity. Increased sediment movement from catchments is also generally correlated with an increase in nutrients and other pollutants delivered to streams and reservoirs, because many nutrients and pollutants are adsorbed to fine-grained organic and inorganic sediments. This period of increased catchment runoff and lower water use typically lasts ~2-10 years. In the medium to longer terms however, catchments affected by severe wildfires generally experience a reduction in water yields below pre-fire discharge levels as the vegetation enters a phase of rapid growth.

Low severity fires are generally considered to have a lesser impact on soil erosion and catchment hydrologic responses than intense bushfires. This is largely due to the greater retention of ground cover and soil organic matter. However, severe erosive events following low severity prescribed burns have been recorded in Australia. The few studies specifically focusing on the effects of low severity fire on stream water chemistry and water quality suggest that individual low severity fires offer a much lower risk of adverse effects to water supplies than

higher severity wildfires. Stream flow responses to low severity fires have been noted, but are generally smaller in magnitude compared to wildfire responses. Although prescribed HR fires are generally understood to have a lesser environmental impact than wildfires there is concern that frequent burning may cause cumulative degeneration of soil and catchment hydrological characteristics. There is also concern that relatively minor medium to long-term reductions in flows caused by increased prescribed fire regimes may be compounded by climate change-related reductions in precipitation, particularly in south-east Australian catchments.

Soils with greater thermal conductivity, including drier soils, soils with a lower bulk density, and soils with a lower specific heat (sandy soils) will be affected by fire to a greater extent than moist soils, and soil types with high specific heat (clayey soils). For any given burn severity, the soils most susceptible to erosion are fragile sandy soils or soils with otherwise high erodibility, that are thin, stony, and massive with high bulk density. The moisture and nutrient-holding capacity of soils tends to decrease rapidly with depth, therefore even small increases in excess of natural and historical sheet or rill erosion rates can have disproportionately large impacts on issues such as soil productivity and catchment hydrology.

The long-term significance of historically recent and proposed fire regimes can be inferred by comparing rates of soil formation and erosion over geomorphic time scales. It is clear that recent rates of soil erosion generally exceed prehistoric rates, sometimes several orders of magnitude greater.

This may have major implications for our understanding of the impacts of historically recent and proposed fire regimes. It is reasonable to propose that prescribed HR burning in landscapes that did not evolve with repeated low-severity fire will increase the risk of unacceptable soil-related impacts. This may particularly be the case in higher rainfall regions where, due to high cover factors, rates of soil erosion and sedimentation are generally lower than those in drier regions. Conversely, it is reasonable to propose that prescribed HR burning in landscapes that have evolved with repeated episodes of low-severity fire will not increase the risk of unacceptable soil-related impacts. Appropriate and geomorphically sustainable fire regimes need to be devised for each soil-landscape type, rather than applying general prescriptions.

Glossary

BAAT	Burnt Area Assessment Team
BFEA Code	Bush Fire Environmental Assessment Code
BKDI	Byram-Keetch Drought Index
BRIMS	Bushfire Risk Information Management System
EBMP	Enhanced Bushfire Management Program
EC	electrical conductivity
g/cm^3	grams per centimetre cubed
GL/yr	gigalitres per year
HR	Hazard Reduction
kW/m	kilowatts per metre
n.d.	no date
NPWS NSW	National Parks and Wildlife Service
OEH	Office of Environment and Heritage
RFS	Rural Fire Service
RUSLE	Revised Universal Soil Loss Equation
SFAZ	Strategic Fire Advantage Zone
t/ha/yr	tonnes per hectare per year
US	United States
USDA	United States Department of Agriculture
USLE	Universal Soil Loss Equation

Nutrients, elements and compounds

Al	Aluminium
B	Boron
Ba	Barium
C	Carbon
Ca	Calcium
Cl	Chlorine
Cr	Chromium
Cu	Copper
Fe	Iron
K	Potassium
Mg	Magnesium

Mn	Manganese
Mo	Molybdenum
N	Nitrogen
Na	Sodium
NH_4^+	Ammonium
NO_3^-	Nitrate
P	Phosphorus
Pb	Lead
S	Sulfur
SO_4^{2-}	Sulfate
Zn	Zinc

Background

Following the release of recommendations of the 2009 Victorian Bushfires Royal Commission (Parliament of Victoria 2010), the NSW Government considered a range of actions that could be implemented by fire authorities and land management agencies to mitigate fires of similar magnitude impacting on NSW. NPWS is both a statutory fire authority and a conservation land manager, managing 9% of NSW lands and 25% of the total fire prone lands across the state, and currently undertakes a large proportion of the HR activities in NSW. In order to deliver against revised mitigation targets and existing long-term fire strategies, NPWS recognised the need for an EBMP. This program provides additional resources to increase fuel reduction activities, including both mechanical works and prescribed HR fire.

Risk management requires assessment not only of risk to life and property, but to ecosystem services including soils and water (Bush Fire Cooperative Research Centre [CRC] n.d.). It is therefore important to understand and manage the potential impacts of an increase in HR burning on a range of ecosystem services, including biodiversity (Gill 2012)), water supply (Smith *et al.* 2011a), and C sequestration (Bradstock & Williams 2009).

The EBMP Monitoring Plan prepared by NPWS recognised that the potential impacts of fire on soils is an important issue and that incorporation of soils knowledge into the EBMP would help minimise soil-related environmental impacts of the program.

The potential impacts of the EBMP is especially important in the context of possible climatic variation (Hennessy *et al.* 2005; Karoly & Braganza 2005; Gergis *et al.* 2011), which may include possible changes in the frequency and duration of regional temperature extremes (Chambers & Griffiths 2008; Alexander & Arblaster 2009) and possible narrowing of optimal HR operational windows (Williams *et al.* 2001; Hennessy *et al.* 2005).

NPWS therefore requested the Science Division of OEH to prepare a well-referenced literature review summarising the current state of knowledge on the impacts of fire on soils, focussing on the Australian context (i.e. Australian research, and general knowledge relevant to Australian and NSW landscapes).

Priority Research Questions

The relevant Priority Research Questions posed by the NPWS Bushfire Research Statement (NPWS 2013) are:

- What is the impact of bushfire regimes on soil formations and on erosion and sedimentation rates, especially in drinking water catchments [i.e. what soil types are vulnerable to what fire conditions]?
- What is the effect of bushfire regimes on water flows (quantity and quality), especially in drinking water catchments?
- What is the impact of bushfire regimes on nutrient cycling and the carbon cycle?

The NPWS requested that the review should:

- summarise what information is already known in relation to the Priority Research Questions
- identify knowledge gaps.

A key theme of this review is the relative risks and impacts of HR burning compared to wildfire. Specifically, NPWS asked whether increasing HR targets and burning at reduced intervals could result in adverse impacts on soils and the hydrologic responses of catchments.

Introduction

Overview of Relevant Soil Research

Soil is an unconsolidated, thin, variable layer of mineral and organic matter on the Earth's surface that forms as a result of physical, chemical and biological processes operating over long periods of time (Singer & Munns 1996). Soil provides a range of ecosystem services – it supplies nutrients, water and support for plants; it absorbs water by infiltration, stores some as groundwater and interstitial water, and releases it via throughflow, overland and channelled flow to streams. However, soil materials can also be eroded and become sources of sediment, nutrients and other pollutants in streams (DeBano *et al*. 1998).

Agriculture and the Universal Soil Loss Equation

Research into agricultural soil erosion by the United States Department of Agriculture (USDA) in the early 20th century established the basic processes and factors involved. These included: slope steepness and slope length; cover factors, including the type of crop and crop rotations; and conservation practices such as contouring. The USDA later devised the Universal Soil Loss Equation (USLE) (Wischmeier & Smith 1965, 1978), modified in 1992 as the Revised Universal Soil Loss Equation (RUSLE) (Renard *et al*. 1997) to assist in understanding and mitigating soil erosion. The equation is:

A = R x K x L x S x C x P, where

        A = soil loss in tonnes per area

        R = rainfall erosivity

        K = soil erodibility factor

        L = slope length factor

        S = slope steepness factor

        C = cover and management factor

        P = soil conservation practice factor.

The equation remains the most commonly used method to predict the average soil loss rate through sheet and rill erosion around the world, including in Australia (Lu & Yu 2002). The equation includes factors that are also relevant to predicting soil erosion from burnt areas under native vegetation.

Agricultural concerns have also been key drivers of research into a range of other soil-related issues, including wind erosion (Tozer & Leys 2013), the mechanics of gully erosion (Blong *et al*. 1982a; Prosser & Winchester 1996), soil structural decline (Packer *et al*. 1984; Murphy *et al*. 1993), and soil acidification (Helyar *et al*. 1990).

Forestry-related Research

The imperatives of production forestry drove much of the soil- and fire-related research from the 1920s, especially in the USA. Much of this research examined the relationships between both silvicultural fire and wildfire, and nutrient availability (e.g. Chapman 1926; Demmon 1929; Fowells & Stephenson 1934; Heywood & Barnette 1934; Demmon 1935; Wahlenberg 1935; Chapman 1936; Isaac & Hopkins 1937; Wahlenberg *et al*. 1939; Finn 1943; Burns 1952; Fuller *et al*. 1955; Tarrant 1956; Weaver 1957; Cotter 1963; Knight 1966; Brender & Cooper 1968; DeBell & Ralston 1970; Wells 1971; Lewis 1974; Wells & Jorgensen 1975; Christensen 1977;

Wells *et al*. 1979; Kodama & VanLear 1980; McKee 1982; Covington & Sackett 1984; Schoch & Binkley 1986; Boyer & Miller 1994).

In Australia too, the impact of fire on nutrients has been studied in the context of both plantation and native forestry (e.g. Stephens & Bond 1957; Pryor 1963; FCNSW 1966; Humphreys 1966; Flinn *et al*. 1979; Harwood & Jackson 1975; Raison 1980; Attiwill & Leeper 1987).

From the 1970s, much of the forestry-related research effort in Australia also concentrated on soil erosion and sedimentation, and the impacts on hydrological processes after fire, and the need to address soil pollution of streams downstream of logging operations (including post-logging burns) (e.g. Burgess *et al*. 1980, 1981; Mackay *et al*. 1980; Mackay & Cornish 1982; Cornish & Binns 1987; Mackay & Robinson 1987).

Catchment Management and Water Research

The importance of catchment protection and the impacts of soil erosion and sedimentation on water supply reservoirs became a major issue following the construction of large dams in the latter part of the 20th century (Australian National Committee on Large Dams). These works inevitably placed attention on activities within the catchment areas, and the link between pressures including fire, grazing and sedimentation was established by a large body of research conducted in the Snowy Mountains (Byles 1932; Costin 1954; Australian Academy of Science 1957; Costin 1958; Costin *et al*. 1960; Clayton 1967).

The relationships between fire and sedimentation have been examined recently in relation to a number of large fires in Australian water supply reservoir catchment areas (Wilkinson *et al*. 2004; White *et al*. 2006; Rustomji & Hairsine 2006), and the issues and processes involved have been reviewed in Australian and overseas literature (Anderson *et al*. 1976; Blake *et al*. 2009; Smith *et al*. 2011a).

Hazard Reduction

The bushfires that culminated in the catastrophic 'Black Friday' of 13 January 1939 in Victoria resulted in almost two million hectares being burnt, with 71 lives lost. The subsequent Royal Commission presided over by Judge Leonard E.B. Stretton inquired into the causes of the fires, and examined measures taken to prevent the fires and to protect life and property. Stretton recommended a strategic program of what is now known as HR burning to reduce the risk of catastrophic fire. The Commission was also critical of fire management over the border in NSW, and HR burning was subsequently adopted as policy in NSW. The HR burning effort has varied over the years, but by 2012–13, in the wake of the 2010 Royal Commission, the NPWS achieved record levels of hazard reduction burning. In the 12 months to 30 June 2013, fire crews carried out more than 330 hazard reduction burns covering 205,890 ha (NPWS 2013). During 2014–15 NPWS completed 189 planned hazard reduction burns covering over 113,000 hectares (NPWS 2015).

Despite the fact that HR burning has been a key component of fire management strategies for many decades, the vast majority of research has continued to overwhelmingly concentrate on the effects of wildfire, rather than on the effects of low severity burns such as HR burns. Furthermore, much of the research comes from countries with Mediterranean climates, especially Spain, Portugal, Italy, Greece, Israel, South Africa, and parts of the US (Dunn & DeBano 1977; Hibbert 1984; Sevink *et al*. 1989; Imeson *et al*. 1992; Lavabre *et al*. 1993; Christensen 1994; Boix Fayos 1997; Inglesias *et al*. 1997; Inbar *et al*. 1998; Thomas *et al*. 1999; Mayor *et al*. 2007; Terefe *et al*. 2008; Shakesby 2011; Pereira *et al*. 2012).

Review papers that have examined various aspects of the effects of fire on soils have been prepared by: Raison (1979); Tiedemann *et al*. (1979); Wells *et al*. (1979); Christensen (1994); Neary *et al*. (1999); DeBano (2000); Gonza´lez-Pe´rez *et al*. (2004); Certini (2005); Rustomji & Hairsine (2006); Shakesby & Doerr (2006); Shakesby *et al*. (2007); Mataix-Solera *et al*. (2011);

Shakesby (2011); Smith *et al.* (2011a); and, most recently, a major review by Cawson *et al.* (2012).

Of the literature that specifically addresses the effects of prescribed low severity burning (e.g. Biswell & Schultz 1957; Weaver 1957; Hodgkins 1957; Henry 1961; Brender & Cooper 1968; van Loon 1969; Wells 1971; van Loon & Love 1973; Springett 1976; Kodama & VanLear 1980; McKee 1982; Richter *et al.* 1982; Raison *et al.* 1983b; Covington & Sackett 1984; Raison *et al.* 1985b; Raison *et al.* 1986b; Schoch & Binkley 1986; Davis 1989; Covington & Sackett 1992; Keith & Raison 1992; Williams & Melack 1997; Morales *et al.* 2000; Robichaud 2000; Arocena & Opio 2003; Woods *et al.* 1983; Stephens *et al.* 2004; Bêche *et al.* 2005; Elliot & Vose 2005; Raison 2005; Murphy *et al.* 2006; Moffet *et al.* 2007; Outeiro *et al.* 2008; Vadilonga *et al.* 2008; Arkle & Pilliod 2010; Galang *et al.* 2010; Smith *et al.* 2010), very little of it has been generated in Australia, although there are exceptions (van Loon 1969; van Loon & Love 1973; Springett 1976; Raison *et al.* 1983b; Woods *et al.* 1983; Raison *et al.* 1985b; Raison *et al.* 1986; Keith & Raison 1992; Raison 2005; Morris *et al.* 2010; Smith *et al.* 2010; and Cawson *et al.* 2012). In this review, overseas literature has been cited where relevant.

Carbon Research

Research into carbon dynamics in soils and vegetation has been prompted by increasing evidence of the role of carbon dioxide and other greenhouse gases in affecting climate change. A great deal of the research has examined the forms, sequestration and pathways of C, particularly in agricultural systems. In native vegetation systems, much of the research has again been carried out in the context of production forestry (Grierson *et al.* 1992; Grierson *et al.* 1993; Roxburgh *et al.* 2006; Mackey *et al.* 2008; Ranatunga *et al.* 2008; Keith *et al.* 2009; Dean *et al.* 2003; Dean & Wardell-Johnson 2010), and the role of various fire regimes in affecting C stores (Fellows & Goulden 2008; Hurteau *et al.* 2008).

Soil Formation and Erosion Rates

In order to formulate policies and practices aimed at protecting the soil resource, it is necessary to understand what are considered to be 'natural' rates of soil formation and soil loss (Edwards 1991), and to place current rates of soil erosion within a context of long-term landscape stability.

Soil Formation Rates

Soil formation includes two main processes – the rate of mineral soil formation by weathering of bedrock or regolith, and the rate of topsoil formation, which involves biogenic processes and the incorporation of organic matter (McCormack & Young 1981). The rate at which bedrock or regolith is weathered is extremely variable and depends on climate, vegetation, relief and the properties of the parent material itself (Jenny 1941), but of the two soil-forming processes, the weathering of bedrock is the limiting (slower) process.

Measured rates of substrate weathering vary through at least two orders of magnitude, from close to zero to 70 millimetre per 1000 years (Selby 1982; Saunders & Young 1983; Edwards 1991; Pillans 1997; Wilkinson & Humphreys 2005; Fifield **et al**. 2010). Stockmann *et al.* (2013) reviewed data from a range of sites in southeastern NSW (Heimsath *et al.* 2000; Heimsath *et al.* 2001; Wilkinson *et al.* 2005a, 2005b; Heimsath 2006; Stockmann 2010), and found potential soil production rates of between 10 and 62 millimetre per 1000 years. Fire may also be a significant factor this process, through the breakdown of rock through the process of rock 'spalling' (Selkirk & Adamson 1981). In terms of the formation of topsoil, Walker & Coventry's (1976) research is indicative, finding that an organic profile in alluvium was established within ~1000 years.

Soil Erosion Rates

García-Ruiz *et al.* (2015) carried out an analysis of erosion rates around the world, and found extremely high variability in reported rates, with much of the variability related to: the size of the study area involved; differing measurement methods; and the duration of the experiment. However, positive relationships were found between erosion rates and slope and annual precipitation; there was also a significant land use effect, with agricultural lands yielding the highest erosion rates, and forest and shrublands yielding the lowest.

Long-term soil erosion rates have been measured at various sites in Australia. Humphreys & Mitchell (1983) for example, determined average soil erosion rates in sandstone landscapes in the Sydney Basin at 2.9-61 millimetre per 1000 years (or 0.04-0.86 t/ha/yr, assuming an average bulk density of 1.4 g/cm^3). Bierman & Caffee (2002) sampled granitic landscapes along a transect in central Australia and found erosion rates of only 0.3-5.7 millimetre per 1000 years (or 0.0042-0.08 t/ha/yr). Heimsath *et al.* (2010) reported erosion rates determined from **in situ** cosmogenic beryllium-10 (^{10}Be) across a spectrum of Australian climatic zones and found that climate has a major effect on erosion rates: the highest rates, averaging 35 millimetre per 1000 years (0.490 t/ha/yr), were from sites in south-eastern NSW.

Modern soil erosion rates have been calculated from a range of agricultural and cropping landscapes in NSW (Edwards 1991) and indicate losses sometimes several orders of magnitude greater than natural rates. Lu *et al.* (2001) used these field data to model sheet and rill erosion on a monthly basis across Australia, and to determine the ratio between current and pre-European hillslope soil erosion. The results indicated that modern erosion rates are several orders of magnitude in excess of geologic rates. The greatest increases were found to occur in agricultural land uses, although rates from many forested catchments were also elevated.

Tolerable Soil Erosion Rate

The concept of the 'tolerable soil erosion rate' is defined as '*any actual soil erosion rate at which a deterioration or loss of one or more soil functions does not occur*' (Verheijen *et al.* 2009); it is considered that the upper limit of tolerable soil erosion is equal to the soil formation rate. Soil formation rates are generally far less than rates of human-induced erosion (Beckmann and Coventry 1987; Edwards 1988; Fifield *et al.* 2010) and land management practices such as fire may further exacerbate these deficits. Furthermore, even small increases in excess of natural and historical soil production rates can have disproportionately large consequences, as the moisture and nutrient holding capacity of soils tends to decrease rapidly with depth, so that even low levels of exacerbated sheet or rill erosion may have disproportionately large impacts on issues such as soil productivity and catchment hydrology.

Fire History and Soil Erosion Rates

In forests and other areas under native vegetation, soil erosion rates tend to be highly correlated with the incidence of ground cover and rainfall. Therefore, medium- to long-term erosion rates will be affected by fire regimes, and this relationship has been the basis for much stratigraphic research (e.g. Hughes & Sullivan 1981; Singh *et al.* 1981; Singh & Geissler 1985; Banks 1988; Kershaw *et al.* 2002; Black & Mooney 2006; Lynch *et al.* 2007; Mooney *et al.* 2012).

Mooney *et al.* (2012) conducted a major study into rates of charcoal and sediment deposition in order to better appreciate the impact of Aboriginal use of fire. They found that the dominant signal in the record was a climatic signal – until about 200 years ago, when fire severity and frequency dramatically increased (see also Soil Conservation Service of NSW 1983; Banks 1988). They concluded that Aboriginal people used fire selectively, concentrating on the more potentially productive landscape types (see also Kohen 1995; Benson & Redpath 1997).

This suggests that appropriate and geomorphically sustainable fire regimes need to be devised for each landscape type, rather than applying general prescriptions. In particular, it is reasonable to propose that prescribed burning in landscapes that evolved with naturally low

severity fire will not increase the risk of unacceptable soil-related impacts. Conversely, it is also reasonable to propose that prescribed burning in landscapes that did not evolve with low-severity fire will increase the risk of unacceptable soil-related impacts.

These conclusions may have major implications for our understanding of the impacts of more recent fire regimes, especially in higher rainfall regions, where rates of soil erosion and sedimentation are generally contained, due to high percentages of ground cover (Brooks *et al*. 1997). Once ground cover is lost, areas of high rainfall erosivity become highly susceptible to soil erosion.

Definitions

Fire Regime

The term 'fire regime' encompasses:

- fire severity
- fire frequency
- the burn season
- the spatial pattern of the burn, including patchiness.

The fire regime of a site, whether a rare, severe, summer wildfire that consumes vegetation and soil organic matter over large areas, or a semi-regular, low severity, prescribed HR burn that results in a patchy burn pattern with retention of most soil organic matter, is generally acknowledged as an important factor determining erosion risk and water quality impacts (Arkle & Pilliod 2010; Benavides-Solorio & MacDonald 2005; Richter *et al*. 1982; Cawson *et al*. 2012).

Fire Severity and Intensity

Of the four elements of a fire regime, fire severity impacts on a number of factors that are relevant to the determination of soil erosion under the RUSLE. Fire severity comprises two key factors:

- peak temperature (or intensity of the fire)
- duration of the fire at a site.

Intensity is related to the energy generated in the combustion process, measured in kilowatts per metre (kW/m) of fire front. Fire severity takes into account additional factors that affect the combustion process, including the:

- amount, nature and moisture of live and dead fuel
- air temperature and humidity
- wind speed
- topography of the site (Certini 2005).

A mild fire, such as a properly conducted HR burn, may produce up to ~350 kW/m (McArthur 1967), and attain temperatures within the range 90-400 $^{\circ}$C (Humphreys & Lambert 1965; Craig 1968; van Loon 1969; Watson 1977). The crown layer is generally retained, as is some of the understorey and litter layer, and much if not most of the organic matter in the Organic (O) and Aluminium (A1) horizons (topsoil) in the soil, leaving a layer of ash and coarse debris protecting the soil (Underwood *et al*. 2008).

A medium intensity fire may produce 1700-3500 kW/m, and a high intensity fire 3500->7000 kW/m, as a result of which soil surface temperatures may reach in excess of 400-800 $^{\circ}$C (Humphreys & Lambert 1965; Roberts 1965; Cromer & Vines 1966; Craig 1968; Tunstall *et al*. 1976). Under these conditions, there may be complete crown and understorey loss, complete combustion of litter on the soil surface, and combustion of soil organic matter down to depths of greater than several centimetres.

An extreme fire may generate 20,000-60,000+ kW/m (Bushfire Front Inc. n.d.), with temperatures in excess of 1500 °C (Neary *et al.* 1999).

Fire severity is therefore strongly correlated with the crucial RUSLE cover factor. The more severe fires may also induce changes in various physical properties of the soil that affect soil erodibility and infiltration-runoff ratios (see below).

Fire Frequency

Fire frequency is relevant to the chance of recently burnt areas with low percentages of ground cover receiving erosion-inducing rainfall soon after a fire event. The greater the proportion of time that the soil surface is recovering from fire-induced loss of cover, the greater is the chance of such an event occurring. A frequent fire regime may also produce significant long-term cumulative impacts, as opposed to acute event-specific impacts, including changes to vegetation and litter dynamics, and changes to soil structure and other properties.

Burn Season

The chance of receiving erosion-inducing rainfall soon after a fire event may be reduced or increased to some extent by the seasonality of the burn. The most erosive rainfall events are generally associated with intense convective storms, as a result of which rainfall erosivities are often particularly high in north-eastern NSW. The seasonality of fire also affects fire severity – late dry season fire for example, can be more difficult to manage than a cool winter burn, due to drier fuels, higher temperatures and lower humidity levels.

Spatial Pattern

The spatial pattern, including patchiness, of a fire is relevant to the effect of surface runoff on erosion, due to the slope length factor. After the RUSLE cover C-factor, slope and slope length are the variables that generally have the greatest impact on soil erosion predictions. A HR burn that leaves unburnt patches on hillslopes and/or as filter strips in drainage lines, and retains large obstacles such as logs, will reduce both the effective slope length and cover factors. Cawson *et al.* (2011) found that unburnt patches on a burnt hillslope are highly effective at reducing runoff and sediment from burnt areas above –for rainfall events with an ARI < 1 year, sediment loads were reduced by 92-99%.

Priority Research Questions

Impacts of Different Bushfire Regimes on Nutrient and Carbon Cycling

Impact of Fire on Nutrients Generally

Soil fertility and nutrient supply are fundamental factors in controlling the functioning and distribution of plant, and therefore, faunal communities (Raison 1980). Nutrients enter the plant-soil ecosystem in a limited number of ways, by:

- precipitation
- dust fall
- N fixation
- the geochemical weathering of rocks (DeBano *et al.* 1998).

In geomorphically depositional sites, including floodplains, nutrients may also enter the system via overland flow, adsorbed to clay, silt and coarser particles. The nutrients may then be incorporated into living organic matter. The relative proportions of the available nutrients that are incorporated into living organic matter and standing biomass at any one time varies between different ecosystem types. In many Australian ecosystems, a significant proportion of total site nutrients are stored above ground (Raison 1980). That proportion is typically high in systems such as rainforests and forests on Quaternary sands; conversely, it is relatively low in grasslands and woodlands (Swift *et al.* 1979). Nutrients may be returned to the soil via litter fall and decomposition, and may be lost from the local system due to leaching, erosion and, under certain circumstances, denitrification (Figure 1).
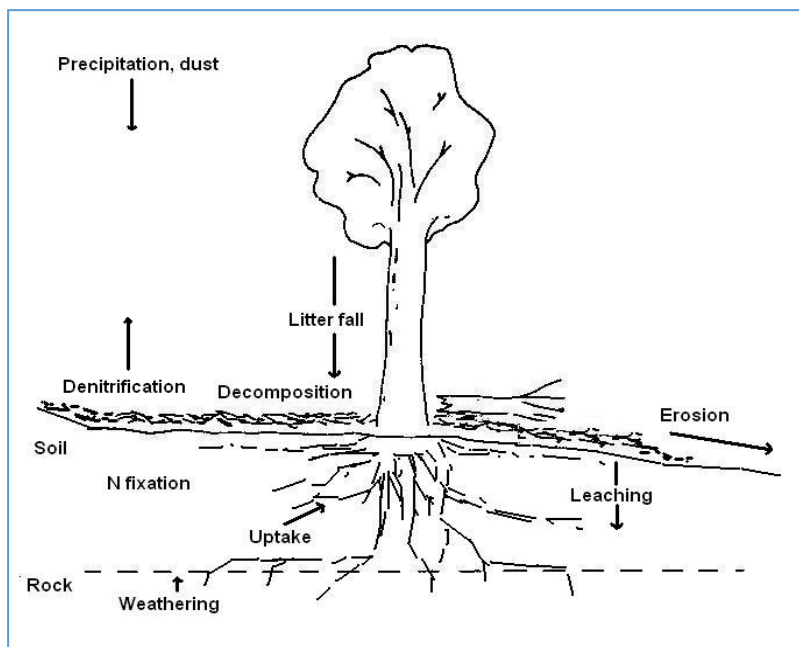
Figure 1. Nutrient cycling in a forest. Adapted from Attiwill (1980); Attiwill & Leeper (1987).

Studies of soil-biomass-nutrient systems have tended to focus on quantifying the size of nutrient pools within different components of the ecosystem, and the pathways of cycling between these pools. Nutrient cycling in unburnt environments is largely regulated by the relatively slow processes of biological decomposition and mineralisation, whereby nutrients in dead biomass and soil organic matter are transformed into forms available to plants. The rate of

decomposition varies widely depending on moisture, temperature, and the types of organic matter. Nutrient losses from unburned ecosystems by erosion, leaching, and denitrification are usually low (DeBano *et al.* 1998).

However, these cycling processes may be disrupted by disturbances, such as fire. Nutrients may be lost from soils during and following fire by:

- volatilisation (vaporisation)
- convection (lost to the atmosphere as particulates in smoke)
- leaching (downwards into groundwater and throughflow)
- erosion (of ash and nutrient-rich surface soil) (Figure 2).

Volatilisation and convection can remove nutrients from living biomass, as well as from the litter-soil system. Leaching and erosion affect largely mineralised materials.
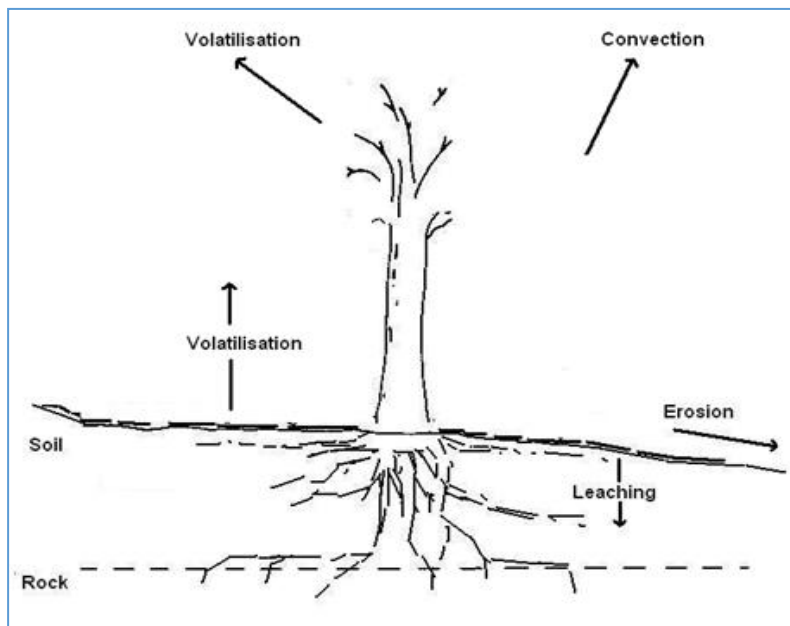
Figure 2. Nutrient losses from a burnt forest. Adapted from Attiwill (1980); Attiwill & Leeper (1987).

The fate of nutrients volatilised or mineralised during the burning of vegetation and litter has been the subject of an enormous body of research since at least the 1930s, particularly in the USA, where much of this work has been undertaken in the context of production forestry (e.g. Fowells & Stephenson 1934; Heywood & Barnette 1934; Heyward 1937; Isaac & Hopkins 1937; Finn 1943; Bruce 1947; Burns 1952; Austin & Baisinger 1955; Fuller *et al.* 1955; Knight 1966; Brender & Cooper 1968; De Bell & Ralston 1970; Wells 1971; Lewis 1974; Grier 1975; Wells & Jorgensen 1975; Christensen 1977; Kodama & VanLear 1980; McKee 1982; Covington & Sackett 1984; Boyer & Miller 1994; see especially the major reviews by Lutz & Chandler 1946; Wells *et al.* 1979; Christensen 1994 and Gessel & Harrison 1999).

The fate of nutrients is particularly important in Australia, where most forests grow on nutrient-poor soils that receive very low rates of nutrient input from weathering, rainfall or N fixation. The issue has been examined in relation to the management of pine plantations on poor, sandy soils, where the margin for error is slight (e.g. Stephens & Bond 1957; Flinn *et al.* 1979), and extensively researched in the context of native forestry (e.g. Floyd 1964; Humphreys 1966; Henry 1961; Pryor 1963; Harwood & Jackson 1975; Raison 1980; Adams & Attiwill 1984; see especially the major reviews by Humphreys & Craig 1981; and Attiwill & Leeper 1987).

Much of this work has investigated the biogeochemical cycle of N and phosphorous (P), with lesser attention directed to elements such as potassium (K), calcium (Ca), magnesium (Mg),

copper (Cu), iron (Fe), manganese (Mn) and zinc (Zn). Much of the research has examined the effects of fire on particular components of the nutrient cycle, such as: the nutrient content of litter (O'Connell *et al.* 1979); changes in the size of the soil nutrient store and its availability (Hatch 1960; Humphreys & Lambert 1965; Grove 1977; Hatch 1977); the role of recolonising plants in nutrient conservation (Turner & Lambert 1977); the effects of soil fauna and soil microbiology (Renbbuss *et al.* 1973; Springett 1976); soil N transformations (Raison 1976); and losses from the system in smoke (Vines *et al.* 1971; Harwood & Jackson 1975).

However, measurement of nutrient losses resulting from fires is extremely difficult and the results are often highly variable (Attiwill & Leeper 1987; Adams & Attiwill 2011). Furthermore, due to the complexity of nutrient transformations, it has proven to be extremely difficult to arrive at firm conclusions and consensus regarding the effects of any nutrient losses, and many research areas related to fire-nutrient cycling remain contested. Raison (1980) for example, focused on measuring changes in the rates of nutrient cycling processes, and using these as an index of the effects of fire on forest ecosystems, whereas Attiwill & Leeper (1987) have focused on the availability of nutrients and measurable productivity outcomes.

Impact of Fire on Soil Organic Matter and Nutrient Losses

Most of the nutrients in a soil are located in the O horizon and the A horizon. Organic matter is the main source of virtually all the available N and most of the available P and S in terrestrial soils (Flinn *et al.* 1979; DeBano *et al.* 1998) and is an important source of other nutrients, such as ammonium (NH_4^+), K, Ca, Mg, Cu, Fe, Mn and Zn. Moreover, despite its limited depth range, soil organic matter can provide over 50% of the cation exchange capacity of some forest soils (DeBano *et al.* 1998).

The size of these nutrient pools will vary depending on the type of soil-ecosystem, which in turn will be determined by factors including regolith and soil type, climate, and time – generally, the longer the period without fire, the greater the size of the nutrient pool contained in the litter and O/A layers (Adams & Attiwill 2011). Therefore, in order to understand the effects of fire on nutrient pools and cycling, it is necessary to understand the impacts of fire on soil organic matter in the topsoil layers.

Effects of Soil Heating and Volatilisation

The impact of fire on soil organic matter (and other aspects of soil chemistry), including how much of the various elements are lost in gaseous form, is dependent on fire severity (Flinn *et al.* 1979; Humphreys & Craig 1981; Certini 2005; Keeley 2009) and the temperature of vaporisation for each element.

Actual soil temperatures however will depend on the duration of the fire and soil-related factors, including:

- the thermal conductivity of the soil
- the bulk density, including porosity, of the soil
- the moisture content of the soil (Humphreys & Craig 1981).

Dry soils, sandy soils, and soils with low bulk density have a lower specific heat and greater thermal conductivity than moist soils, clay soils and soils with higher bulk density. Generally however, even high intensity fires typically heat only the uppermost 100 mm of soil (Henderson & Golding 1983; Walker *et al.* 1986; DeBano 2000), with steep temperature gradients down the profile so that temperatures at 50 mm in the mineral soil rarely exceed 150 ° (DeBano 2000; Certini 2005).

Loss of organic matter commences at relatively low temperatures (<100 ^{0}C) (Table 1), and certain elements including C, N (as nitrate NO_3^-) and S, begin to volatilise at temperatures of ~100-200 ^{0}C (Tiedemann 1987). Therefore, these elements are susceptible to loss by even low severity burns. At temperatures between ~200 and 300 ^{0}C, destructive distillation results in the

loss of ~85% of organic matter, and nutrient constituents that are not volatilised at these temperatures are mineralised.

At temperatures of ~450-500 ^{0}C, ~99% of organic matter is combusted (Fernandez et al. 1997; DeBano et al. 1998). Inorganic P and K are volatilised at moderate temperatures (~770 ^{o}C); and inorganic Ca and Mn at much higher temperatures (1484 ^{o}C and 1962 ^{o}C respectively) (Raison et al. 1985a; Neary et al. 1999).

Soil Property	Threshold Temperature ^{o}C	Source
Plant roots	48-54	Hare 1961
Fungi	60-80	Dunn et al. 1985
Bacteria	80-120	Dunn & DeBano 1977
Organic matter	100-220	Hosking 1938
Nitrogen	200	White et al. 1973
Soil hydrophobicity	250	DeBano & Krammes 1966
Soil structure	300	DeBano 1990
Sulfur	375	Tiedemann 1987
Clay alteration	460-980	DeBano 1990
Phosphorus & potassium	774	Raison et al. 1985a
Magnesium	1107	DeBano 1990
Calcium	1484	Raison et al. 1985a
Manganese	1962	Raison et al. 1985a

Table 1. Threshold temperatures for combustion/volatilisation/alteration of soil properties/elements (after DeBano et al. 1998).

N is essential for certain enzymatic reactions in plants, and symptoms of deficiency include poor growth, chlorotic leaves, lack of vigour and death. Due to the fact that virtually all the N in a soil is located in organic matter, the proportion of NO_3^- volatilised to N_2 (DeBell & Ralston 1970; Fisher & Binkley 2000) is related to the combustion of organic matter, and therefore, to fire severity (Certini 2005). Very little N is lost at temperatures <200 ^{0}C; complete loss occurs at >500 ^{0}C (White et al. 1973; Kutiel & Naveh 1987; Kutiel & Shaviv 1989; Kutiel & Shaviv 1992; Quintana et al. 2007). In high intensity fires, organic N is converted to NH_4^+, which later converts to the mobile NO_3^- as a result of nitrification (Covington & Sackett 1992), which can then be quickly leached downwards (Mroz et al. 1980; Prieto-Fernandez et al. 1993) and out of the available nutrient pool (see also De Bell & Ralston 1970; Grier 1975; Binkley et al. 1992; Kauffman et al. 1993). Such losses are likely to be greater where soil texture is coarse or where there has been considerable loss of organic matter (Christensen 1994). However, calculating the precise effect of fire on soil N pools has been difficult due to the complicated response of N to fire, and its potential relationship to site productivity (Wan et al. 2001).

Fire does not have the same impact on P, because fire converts organic P to insoluble orthophosphate (Cade-Menum et al. 2000), which binds to Al, Fe and Mn oxides, particularly in acidic soils. Furthermore, the threshold temperature of volatilisation for P is significantly higher than for N. Therefore, in many burns, losses of P due to volatilisation and leaching are generally

relatively minor (Certini 2005), although these may still be significant in the nutrient-poor Australian context.

The behaviour and availability of micronutrients such as Fe, Mn, Cu, Zn, B and molybdenum (Mo) with respect to fire is not fully understood however, because specific studies are lacking (Certini 2005).

Convection

Much of the nutrient losses from fire occurs in the convective smoke column. Allen (1964) found that approximately half of the C, N and S from a burnt heath was lost in smoke. Flinn *et al.* (1979) studied the effects of a moderately intense post-log fire in a pine plantation in south-west Victoria, and found that 72% of the N was lost due to ash convection and volatilisation. Losses of P, K, Ca, Mg, S, Fe and Mn were between 16 and 40%, which, it was considered, could affect productivity in the sandy podzols of the site (see also Lewis 1974; Harwood & Jackson 1975 and Clayton 1976). The most significant work carried out in Australia that has examined the nutrient content of smoke from prescribed burns has been undertaken by the CSIRO in the Brindabella Ranges (Raison *et al.* 1985a).

Leaching

A further potential mechanism for nutrient loss is the leaching of nutrients in the forms of ash and soluble elements down through the soil profile. Studies of soil leachate after burning have shown increased levels of total P (McColl & Grigal 1975; Knighton 1977; see also Fowells & Stephenson 1934; and Isaac & Hopkins 1937), although the relevance of this work to Australian conditions is unclear. Again, this process could be especially significant in sandy soils (Cole *et al.* 1975; Flinn *et al.* 1979), due to the low availability of inorganic cation exchange sites and low buffering capacity in such materials (Humphreys & Craig 1981).

Erosion

Fires, especially hotter fires, produce a highly nutrient-enriched fine grey ash. This material is predisposed to loss by water and wind erosion, and even small removals from a site either during or after a fire could result in significant losses from the system (Raison *et al.* 1983). For example, Leitch *et al.* (1983) found that a single intense rainstorm shortly after the Ash Wednesday fires removed ash and soil equivalent to 22 t/ha, which was equivalent to approximately one third of the N in the above-ground biomass.

Post-fire erosion losses may also occur as a consequence of individually less significant but repeated erosion losses following prescribed low severity burns, and these may cumulatively amount to a significant impact on nutrient supply (Thomas *et al.* 1999; see also Leitch *et al.* 1983). This matter is discussed in greater detail below.

Summary

Most of the nutrients in a soil are located in the O horizon and the A horizon. The pool size of topsoil fertility is therefore diminished by fire (Oswald *et al.* 1999; Badia & Marti 2003), particularly with respect to C, N and S. Nutrients may also be lost by convection, leaching and erosion of the nutrient-enriched ash. The relative impact of any given fire on the nutrient pool will be potentially greater where the fire is severe (Fernandez *et al.* 1997; Gonzalez-Peres *et al.* 2004; Terefe *et al.* 2008) and occurs in vegetation that has not been burnt for extended periods. However, the pool size of certain elements may be reduced as a result of even cool-moderate intensity fires, and concerns have been raised that frequent repeated burning may also cumulatively deplete sites of nutrients and interrupt nutrient cycling (Raison 1980).

Impact of Fire on Nutrient Bio-availability

The Ash Bed Effect and Bio-availability

Fires tend to reduce the size of soil-ecosystem nutrient pools. However, the concentration and bio-availability of many nutrients generally increases as a result of the combustion of soil organic matter and the mineralisation of nutrients (Humphreys & Lambert 1965b; Cromer 1967; Humphreys & Craig 1981; Khanna & Raison 1986; Tomkins et al. 1991; Chambers & Attiwill 1994; Christensen 1994; Khanna et al. 1994; Inglesias et al. 1997; Quintana et al. 2007; Pereira et al. 2012; Yusiharni & Gilkes 2012). This process of mineralisation and concentration is known as the 'ash-bed effect' (Fowells & Stephenson 1934; Pryor 1963).

The chemical processes involved and the resultant degree of concentration is highly contingent on a number of factors, including:

- the type of nutrient
- the type of fuel
- fuel densities
- fuel moisture content
- the amount of fuel consumed
- the severity of burning
- soil properties
- pathways of leaching and other forms of removal (Kutiel & Shaviv 1992; Gillon et al. 1995; Pereira et al. 2012).

It has been found following fire that Ca may be concentrated 10-50 times, Mg 10-35 times, and P 10 times (DeBano & Klopatek 1988; Etiegni et al. 1991; Ulery et al. 1993; Khanna et al. 1994; Giardina et al. 2000; Badia & Marti 2003; Ferran et al. 2005; Galang et al. 2010).

Due to volatilisation however, plant ash generally contains little N (Raison et al. 1985b; Khanna et al. 1994). However, fire may stimulate the rate of N fixation by encouraging the germination of nodulated plant species such as Acacias, Casuarinas and legumes (Adams & Attiwill 1984, 1986; Johnson & Curtis 2001; May & Attiwill 2003).

Fire may also cause mineralogical changes in soils subjected to fire (Ulery et al. 1996; Ketterings et al. 2000; Eggleton & Taylor 2008; Nornberg et al. 2009; Yusiharni & Gilkes 2012). It is thought that some of the compounds formed may have a high capacity to adsorb ions and may affect soil fertility (Ketterings et al. 2000), although these processes are poorly understood. Probably of greater significance is the formation of calcite and other salts in ash (Humphreys et al. 1987; Etiegni et al. 1991; Yusiharni et al. 2007), which tends to increase the electrical conductivity (EC) and pH of the soil (Ulery et al. 1993; Quintana et al. 2007), thereby increasing the availability of pH-dependent elements (Truog 1946). Fire may also be important in redistributing nutrients through the soil profile (Fisher and Binkley 2000).

The Significance of Nutrient Losses and Cycling

According to some, increased bio-availability of nutrients may offset losses from the system, especially in the short-term. Adams & Attiwill (2011) for example, considered that the most important principle in forest nutrition is the amount of bio-available nutrients in circulation, rather than the total size of the pool of nutrients in soil and woody material. This may be the case in relation to P, which is often bio-unavailable and limiting, but may be altered in favour of greater availability to plants after soil heating (Humphreys & Lambert 1965b) due to lowering of the N:P ratio caused by N losses (Adams & Attiwill 2011). However, there are differing views on the processes, magnitude and significance of P losses (see e.g. Raison 1980; Raison et al. 1985a,b; Bowman et al. 1986; Jackson 2000; cf. e.g. Attiwill 1992, 1994; Jurskis 2005a,b; Jurskis & Turner 2002). The former authors point to the depletion of supply; the latter argue that fire is essential to refresh P-cycling, species diversity and productivity.

Effects of Low Severity Fire on Nutrients

The relative magnitude and significance of the processes of combustion, volatilisation, convection, leaching and erosion therefore depend partly on fire severity and intensity (De Ronde 1990; Andreu *et al.* 1996), and partly on post-fire conditions.

In general, there are important differences in the effects of low severity and higher severity burns. Although there are fewer studies on the effects of low severity HR fire on nutrient levels (e.g. Richter *et al.* 1982; Covington & Sackett 1992; Murphy *et al.* 2006; Outeiro *et al.* 2008; Scharenbroch *et al.* 2012), it is clear that in relation to volatilisation, a smaller proportion of the nutrient supply is lost compared to wildfire, although as a proportion of that lost, more C, S and N are removed due to their lower temperatures of volatilisation. Similarly, losses in smoke are expected to be lower in most HR burns compared to a high severity burn. The potential for erosive losses is also greatly diminished in a properly conducted HR burn, due to partial retention of soil organic matter and ground cover.

However, some of these losses can still be significant in the Australian context. The nutrient content of smoke from a prescribed burn in a snow gum forest was studied by Raison *et al.* (1985a). They measured the transfer of N, P, K, Ca, Mg, Mn and boron (B) to the atmosphere and found that 54-75% of the N, 37-50% of the P, 43-66% of the K, 31-34% of the Ca, 25-49% of the Mg, 23-49% of the Mn, and 35-54% of the B were lost from the local ecosystem from the burn. They calculated that the replacement times for losses in the smoke was 11 years for N and 20 years for P. DeBano & Conrad (1978) measured 10% loss of N from a prescribed burn in southern Califormia; the loss of N was greater when soils were dry, compared to when the litter and soil were moist (DeBano *et al.* 1979). HR burns can also increase the mobility of inorganic ions in the litter and topsoil by either leaching or erosion via overland flow (Rowe & Hagel 1974; Adams & Attiwill 2011).

The actual ecosystemic or production impact of these losses has been harder to detect. Hatch (1959), working in jarrah forests of Western Australia, could find no significant differences in pH, soluble salts, organic C, and exchangeable Ca, Mg, K, and Na between sites that had been subjected to frequent burning compared to a site that had been protected from fire. Similarly, van Loon (1969), working near Taree in NSW, and van Loon & Love (1973), working near Whiporie in NSW, detected no changes from control plots in P, total N, Al, Ca, Mg, K and Na eight to 10 years after a burn. Even in low nutrient sites, Richards (1976) could not detect significant differences between sites on Fraser Island burnt in 1939 and 1974. In a study of low severity prescribed fire in pine stands in the US, Binkley *et al.* (1992), observed that site productivity was unaffected.

Humphreys & Craig (1981) however, realised that the effect could be significant on nutrient-poor sites as the quantity of mineral nutrients released would represent a greater proportion of that present in the soil (see also Raison 1980). It is also likely that any negative effects from HR burning may be more complex and take longer to become manifest. For example, Woods & Raison (1983) noted that major releases of bio-available organic nutrients did not occur until litter had undergone several years of decomposition in moist conditions. In dry sclerophyll forests however, the frequency of prescribed HR burns may not allow this process to complete (Raison *et al.* 1983), as repeated, frequent burning would tend to dry these deeper litter/organic layers and hence reduce decomposition. This complexity is underlined by a study into the role of fire and nutrient dynamics in Tasmanian forests, where McIntosh *et al.* (2005) concluded that nutrient loss by frequent fire encouraged fire-tolerant vegetation adapted to lower soil nutrient status, and this produced a feedback mechanism that caused further progressive soil nutrient depletion.

Views on the effects of prescribed low severity burning on nutrient supply and cycling are disparate and contingent on many factors. Certini (2005) concluded that '*if plants succeed in promptly recolonising the burnt area [i.e. if significant soil erosion losses are avoided], the pre-fire level of most properties can be recovered and even enhanced.*'

Similarly, Adams & Attiwill (2011) noted that *'the literature with respect to nutrients and fuel-reduction fires … has been interpreted by some as containing warnings about the risks of fuel-reduction fires due to nutrient losses. Yet these risks are so remote under the current, and any likely future, scenario of fuel reduction at the landscape scale as to be of minor consequence.'*

Nevertheless, nutrient-poor systems, and particularly those which depend for their long-term stability on efficient nutrient accumulation, retention and recycling processes, are likely to be more sensitive to disturbance caused by regular burning (Raison 1980).

The CSIRO summarised the dilemma that faces land managers in relation to fire and nutrient cycling: if burning occurs too frequently, there is a cumulative risk of loss of soil nutrition; if the interval between burns is too long, the risk is one of wildfire, with potentially greater nutrient loss and disruption to nutrient cycling, and a longer period before recovery is attained (CSIRO 1999).

Impact of Fire on the Carbon Cycle

Forms of Organic Carbon

The organic C in soils is stored in a number of forms, or pools, which vary in their rates of decomposition. The main forms are:

- living vegetative and microbial C
- labile (bio-available) C, with a turnover of less than five years
- residual and particulate C, including dead vegetative matter 0.053-2.0 mm diameter, with a turnover up to 40 years
- biologically stable organic C, typically in the form of humus, charcoal and 'black carbon', with a turnover in hundreds to thousands of years.

Soil organic matter comprises the labile, residual and biologically stable organic C forms, and is a significant source of C, often containing ~50 % C. C stored in stable forms is effectively removed from the C cycle. However, stable forms can have important functions. Humus, for example, whilst biologically stable, retains soil moisture, improves soil structure, and is important in cation exchange.

Factors Affecting Organic Carbon Pools

The size of the living vegetative pool is related to the rate at which plants fix and emit carbon dioxide (CO_2). This rate will vary between different vegetation types, which in turn are determined by factors including climate, soil, and the way the vegetation is managed, especially the frequency and severity of fires. The size of the labile pool is related to biological activity and nutrient cycling, particularly available N, which typically becomes available at a C:N ratio less than ~22:1 (Hoyle *et al.* 2006).

The capacity of soils to store organic matter is largely determined by land use, soil type, and site and landscape factors, and fire-related factors such as the seral stage of vegetation and soil recovery following previous fire. The main site factors related to the amount of organic matter are: the topographic position on the slope – lower slopes are relatively moister than upper slopes; and slope aspect – south- and east-facing slopes are relatively moister than north- and west-facing slopes. The percentage of clay in the soil is correlated with increased water-holding capacity of soils and therefore moisture status, which in turn is related to biological activity and the accumulation and cycling of soil organic matter. Thus, soil organic matter and C are generally greatest in sheltered slopes in higher rainfall regions on clay-rich soils.

Land use change will impact on soil organic C mainly due to its effects on C cycling. Forest soils in particular can store significant amounts of organic C in litter and upper soil layers. Conversion from native vegetation to agriculture typically reduces soil C by 20 to 70% (Luo *et al.* 2010; Sanderman *et al.* 2010). Total soil C tends to not change significantly unless there is a change in land use, but significant changes to the relative proportions of the various components of the

soil organic matter pool may be affected by changes in land management, including aspects of the fire regime.

Impact of Fire on Organic Carbon

Fire, even low severity prescribed burning, can potentially have significant impacts on soil organic matter and C due to the low temperature of volatilisation for C (Tiedemann 1987; Prentice *et al.* 2001). The amount of C consumed and volatilised is related to increased soil temperature, and therefore to fire intensity and severity (Figure 3) (Campbell *et al.* 1977; Giovannini & Lucchesi 1997; Hille & den Ouden, 2005; Homann *et al.* 2011). The different forms of soil organic matter can be impacted differentially by fire: Hoyle & Murphy (2006) for example, showed that the proportions of labile and microbial biomass C were reduced markedly after 17 years of repeated burning.
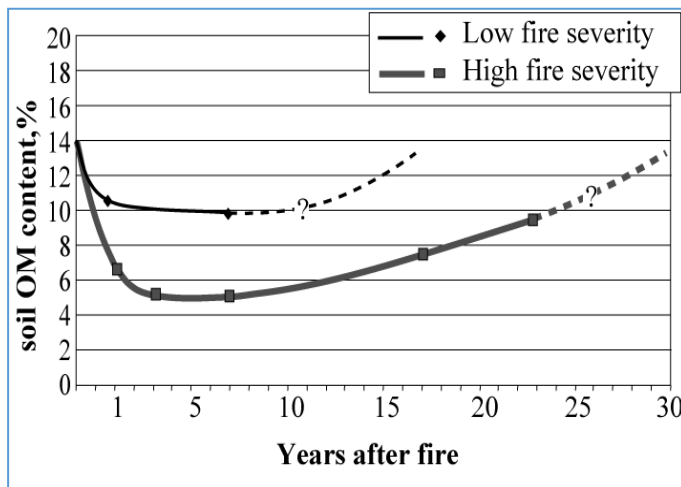
Figure 3. Temporal changes in soil organic matter content following low and high fire severity (after Tessler *et al.* 2008)

However, fire may also produce new, pyrogenic forms of C. Fire may transform soil organic C into stable C forms such as 'black carbon' (Gonzalez-Perez *et al.* 2004), produced by incomplete combustion of plant material, and which can be an important reservoir for nutrients, especially in tropical environments. Black C may comprise up to 30% of the total C stored in soils, and

may be an important long term C sink (Schmidt *et al.* 2001). Prescribed fires can convert variable amounts of biomass into a variety of black C forms. However, it appears that little is known about the forms of black C produced and the rate (and efficiency) of conversion.

Charcoal is formed by pyrolysis, in the absence of oxygen. Although charcoal represents a relatively minor portion of available biomass burned in wildfires and low severity burns, its recalcitrant properties confer residence times ranging from centuries to millennia, with significance for C sequestration in frequently burned forests (Lehman *et al.* 2008). It may also contribute to the water-holding capacity, ion exchange complex and active particle surface area of the soil environment (DeLuca & Aplet 2008). Fuel reduction burns are likely to result in an increase in the amount of 'recalcitrant' pyrogenic C (Preston & Schmidt 2006; Adams & Attiwill 2011). However, there have been no significant studies of the relationship between fuel reduction fires and pyrogenic C in Australia and international research is minimal (Adams & Attiwill 2011). Almost all of the data have come from research into bushfire and forest regeneration burns (Hopmans *et al.* 2005).

Perhaps the most significant work undertaken in south-eastern Australia is that of Volkova *et al.* (2014), working in dry *Eucalyptus* forests in eastern Victoria, who found significant differences in C emissions between sites subjected to HR burning compared to wildfire. Less than 3% of soil C was transferred to the atmosphere in HR burning; a subsequent wildfire transferred a further 6% to the atmosphere. In nearby site last burnt 25 years previously, the wildfire burning transferred 16% of forest C to the atmosphere, which suggested a significant potential for HR burns to mitigate greenhouse gas emissions. Fuel-reduction burning, whilst resulting in short-term C losses, can decrease the severity of a subsequent large fire and lead to smaller C

emissions over the long-term; fire-suppressed forests, on the other hand, can contain more C that is more vulnerable to catastrophic release (Hurteau *et al.* 2008; Read 2010).

Conclusions

High severity wildfires can impact on the C storage capacity of ecosystems and soils for decades to centuries. In comparison, individual prescribed HR burns typically result in lower losses of C by volatilisation, convection and other pathways. Prescribed low severity fire can potentially reduce catastrophic vaporisation of C stocks and C emissions from unplanned fires (Association for Fire Ecology 2009) and the use of prescribed burns for C emission abatement via mitigation of wildfire impacts has been discussed in North America (Hurteau *et al.* 2008) and in Europe (Narayan *et al.* 2007). Prescribed fire has been used in northern Australia for greenhouse gas emission abatement by reducing fuel through low intensity burns in the early dry season, thus decreasing the severity and extent of subsequent unplanned late dry season wildfires (Russell-Smith *et al.* 2009). The Savanna Burning methodology is an approved methodology for greenhouse gas emission abatement under the Australian Carbon Farming Initiative (CFI) (Department of Climate Change and Energy Efficiency 2012).

However, the potential for HR burning to reduce total C emissions and increase storage generally requires a more detailed understanding of the dynamics of combustible biomass pools in a range of different ecosystems (Bradstock & Williams 2009; Bradstock *et al.* 2012a,b). This potential of prescribed burning will vary between different ecosystems, owing to differences in the efficacy of HR burning in reducing unplanned fire activity and soil and landscape factors. In temperate eucalypt forests for example, the potential for HR burning to produce a net reduction in C emissions is considered to be low, according to Bradstock *et al.* (2012b), and in relation to tall forests, Underwood *et al.* (2008) concluded that insufficient research had been carried out into the differential impacts of wildfires and low severity burning on the C cycle. Even advocates for enhanced prescribed HR burning have concluded that further research is required to more fully understand C storage and emission trade-offs associated with managed versus wildfire regimes (Association for Fire Ecology 2009). Unfortunately, the various factors that comprise a fire regime, and the interactions of fire with different soils and vegetation types, make the research task difficult and complex.

Impact of Different Bushfire Regimes on Soils and Erosion

Background

Despite the differences in the literature concerning the relative significance of the effects of fire on the overall size of the litter-soil nutrient pool versus the increased bio-availability of certain nutrients, in many cases, post-fire erosion of soil, organic matter and ash can be the most significant impact of fires. This section therefore focuses on the erosion of surficial soil materials by overland flow.

There is a vast body of research into the effects of wildfires on soil erosion and related impacts. Much of this research has examined soil erosion processes operating at the plot or hillslope scale: (e.g. Prosser & Williams 1998; Benavides-Solorio & MacDonald 2001; Dragovich & Morris 2002; Cerda & Lasanta 2005; Doerr *et al.* 2006; Sheridan *et al.* 2007; Spigel & Robichaud 2007; Pierson *et al.* 2008; Smith & Dragovich 2008), or at the catchment scale (e.g. Scott *et al.* 1998; Moody & Martin 2001; Istanbulluoglu *et al.* 2004; Moody & Martin 2004; Kunze & Stednick 2006; Lane *et al.* 2006; Mayor *et al.* 2007; Rulli & Rosso 2007; Moody & Martin 2009).

However, while fire severity is recognised as being an important variable affecting post-fire runoff and erosion (Brown 1972; De Luis *et al.* 2003; Shakesby & Doerr 2006), there appears to be relatively few studies that have compared the effects of different fire severities (exceptions being Dragovich & Morris 2002; Benavides-Solorio & MacDonald 2005; Ferreira *et al.* 2005; Doerr *et al.* 2006), and even fewer studies specifically on the effects of prescribed low severity fire on soil properties and erosion rates (e.g. Robichaud 2000; Coelho *et al.* 2004; Cerda & Doerr 2005; Hubbert *et al.* 2006; Moffet *et al.* 2007).

Factors Affecting Soil Erosion

Precipitation may be intercepted by various strata of vegetation or the litter layer. That which reaches the soil may either infiltrate or run over the surface. The proportion of precipitation that results in overland flow, i.e. the runoff/infiltration ratio, and therefore produces soil erosion, is controlled by the intensity of the rainfall, and by a range of soil surface-related conditions. The key soil surface- and site-related factors that determine the runoff/infiltration ratio for any given rainfall intensity are:

- cover, including the litter layer
- soil-related factors, including
  - soil hydrophobicity
  - soil structure
  - soil depth to less permeable materials
  - antecedent soil moisture conditions
- the slope and slope length of the site.

Rainfall Erosivity

Raindrop size plays an important role in detaching individual soil particles, and therefore making them available for subsequent movement by water (Zuazo & Pleguezuelo 2009). The kinetic energy of a moving object (E) is equal to its mass x speed^2. As water droplets grow in size, their speed also increases, such that the energy of a five millimetre raindrop is potentially 500 times greater than that of a one millimetre drop. Rainfall erosivity (the R factor) is a combination of E and rainfall intensity I30 (the maximum intensity of rain in 30 minutes expressed in mm/hr). Rain splash erosion rates are significantly correlated with the rainfall erosivity index EI_{30} (Angulo-Martinez *et al.* 2012), a conclusion confirmed in the Mount Lofty Ranges in South Australia by Morris *et al.* (2010).

The largest surface runoff and erosion events occur during intense convective storms, where soil cover has been removed and where there is either no overstorey, or where the overstorey is greater than the height required for terminal raindrop velocity to be achieved. In these circumstances, the soil is exposed to the full potential energy of droplets. Soil losses due to significant erosive rainfall events in the first year after wildfire have been reported as being as high as 100,000 times greater than sediment yields from unburnt plots (Inbar et al. 1998). Brown (1972) found a thousand-fold increase in the rate of erosion after wildfire in the Snowy Mountains, and Atkinson (1984, 2012) measured soil losses up to 64.2 t/ha/yr when wildfire in the Royal National Park south of Sydney was followed by heavy rains.

In the case of low severity burns where the overstorey is maintained, E and the average velocity of droplets are lower, because terminal velocity is not reached. Raindrops that collide with overstorey foliage would also undergo fractionation upon collision with foliage, thereby reducing their size and further reducing E.

In extreme rainfall conditions, debris flows may result (Cannon 2001; Santi et al. 2008; Nyman et al. 2011; García-Ruiz et al. 2013; McInnes-Clarke et al. 2014). Debris flows are fast-moving, liquefied, channelised landslides of mixed and unconsolidated water and debris that can scour channels to bedrock. A great deal of attention has been directed towards debris flows, especially in the US and Canada (Cannon 2001; Cannon & Gartner 2005; Cannon et al. 2008; Cannon et al. 2010), due to their impact on catchment water quality and their potential to impact on public infrastructure and safety. Debris flows have also been noted in Australia: Nyman et al. (2011) studied a number of high magnitude erosive events in catchments subject to wildfire in eastern Victoria, and McInnes-Clarke et al. (2014) documented debris flows that resulted from extreme post-wildfire rainfall in February 2013 in the Warrumbungle National Park (Figure 4). Destablisation of hillslopes after fire may also produce a range of other mass movements, including rock falls, rock slides, slumping and mudflows.

Cawson et al. (2012) recorded debris flows that occurred in north-eastern Victoria following low intensity prescribed burns and a rainfall event with an I30 of 51 millimetres per hour (Figure 5). There was no evidence of erosion in an adjacent unburnt catchment.

In contrast, in the absence of significant rain, there is generally little effect on runoff and erosion. Blong et al. (1982b) for example, measured erosion from runoff plots following a bushfire in sandstone terrain similar to that of Atkinson (2012); however, post-fire rainfall was well below average, and sediment yields were therefore a comparatively low 2.5-8.0 t/ha/yr.

Figure 4. Debris flow in Warrumbungle National Park after wildfire and intense rain. Photo: M. Tulau
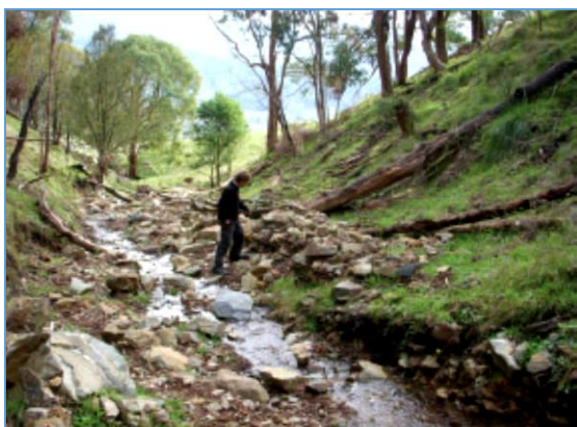
Figure 5. (Above and right) Debris flows in Victoria following a prescribed burn. Source: Cawson *et al*. (2011).

Cover Factor

In a post-fire situation, the amount of green vegetation and surface litter protecting the mineral soil surface from raindrop impact is reduced. This C factor is a crucial consideration in relation to the potential for post-fire soil erosion, as overall soil erosion rates in RUSLE calculations are far more sensitive to this factor than other factors such as soil erodibity, K. Benavides-Solorio & MacDonald (2005) for example found that the percentage of bare soil accounted for nearly two-thirds of the variability in sediment production rates, and Gilmour (1968) and Craig (1968) found that the cover factor was relatively more important in soils with lower infiltration rates (see also Brock & DeBano 1982). Zuazo & Pleguezuelo (2009) reviewed the literature relating to vegetation cover and runoff, and found wide variation according to the experimental conditions (Figure 6).

A threshold of 20-45% coverage is generally considered necessary to reduce soil from burnt plots (Dieckmann *et al*. 1992; Inbar *et al*. 1998; Prosser & Williams 1998; Mayor *et al*. 2007). In Australia, a groundcover of <30% is generally considered to be significant for soil erosion (Yang 2014), although Loch (2000) found that erosion rates were remained high up to 47% grass cover. Site-specific threshold figures will depend on differences in factors such as slope, aspect, soil type, and vegetation type and architecture (Cerda *et al*. 1995; Mayor *et al*. 2007; Wittenberg & Inbar 2009; Wittenberg *et al*. 2014).The C factor is strongly correlated with burn severity. A severe wildfire for example, can result in total loss of ground cover (e.g. Brown 1972; Atkinson 1984), a condition which may prevail for years where tree damage has been severe, and leaf litter input is reduced (Raison 2005).
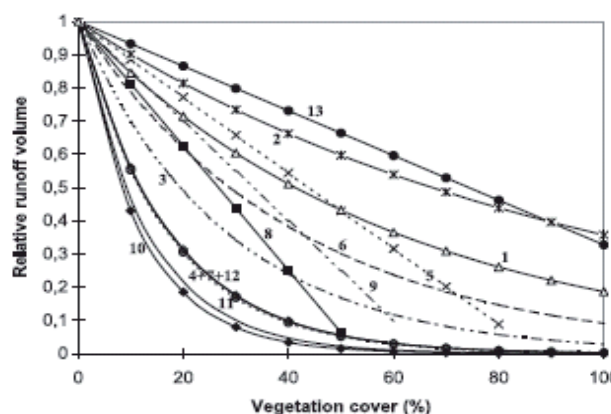
Figure 6. Relationship between plant cover and relative runoff. Source: Zuazo & Pleguezuelo (2009).

In contrast, a sufficient layer of litter and other organic matter may remain after a low severity burn to protect the site from accelerated surface erosion (DeBano *et al.* 1998; De Luis *et al.* 2003; Puigedefabregas 2005). A greater proportion of ground cover can be retained by burning when the surface soil and lower litter layers are moist. This can retain ground cover equivalent to a mass of 4-8 t/ha, which is valuable in protecting against soil loss in erodible landscapes (Gilmour 1968; Raison 2005).

Post-fire cover is related to fire intensity (Overton 1996). The Byram-Keetch Drought Index (BKDI) is a measure of the amount of rain that is required to return the soil to saturation, and is used for fire danger measurements across much of south-eastern Australia. With a BKDI of 75-125, fire consumes most surface litter along with a significant proportion of the organic soil material, leaving exposed mineral soil. A BKDI of 37.5-75 allows scattered patches of surface litter to remain in damp areas following a fire, and preserves the soil organic layer.

Pre-fire soil moisture conditions are often reflected in burn patterns. For any given burn type and conditions, fires tend to be hotter on the northern, drier aspects (Good 1973), and this is generally reflected in erosion and sediment production rates. Brown's (1972) experience in the Snowy Mountains is typical – the north facing slopes were severely eroded and largely denuded of topsoil after a fire, while the south-facing slope showed minor erosion (see also Wittenberg & Inbar 2007).

The potential for sediment movement is greater in the immediate post-fire period, before cover has recovered. After severe bushfires, sediment yields may remain elevated for at least ~5-10 years, by which time vegetation has regrown and leaf litter accumulated on the ground (Inbar *et al.* 1998; Mayor *et al.* 2007). In relation to HR burns, the recovery period for re-establishment of soil cover is generally significantly shorter. Raison *et al.* (1986a) quantified fine surface litter following prescribed burns in a sub-alpine eucalypt forest and found that the litter re-accumulated rapidly, reaching a mass of 10-12 t/ha within four to five years. The period of elevated erosion risk post-HR burn is correspondingly shorter compared to that of wildfires. Recovery periods are also likely related to vegetation type: Good (1973) for example, noted that grasslands in the Snowy Mountains had the potential to recover relatively quickly. Conversely, burnt *Callitris glauca* communities remained susceptible to erosion for longer periods.

Soil Hydrophobicity

The phenomenon of fire-induced soil hydrophobicity, or water repellency, occurs when heating of the surface and near-surface soil causes the vaporisation of organic compounds, which move downwards through the soil profile in response to temperature gradients, and subsequently condense and form a hydrophobic layer or coating around soil particles (Figure 7) (DeBano 1967, 1970; Savage 1974; Giovannini & Lucchesi 1983; Imeson *et al.* 1992; Giovannini 1994; Prosser & Williams, 1998; Doerr *et al.* 1998; DeBano, 2000; Huffman *et al.* 2001; Letey 2001; Pierson *et al.*, 2001; Mataix-Solera & Doerr 2004; Keizer *et al.* 2005; Doerr *et al.* 2009; Stoof *et al.* 2011; Bodi *et al.* 2012).
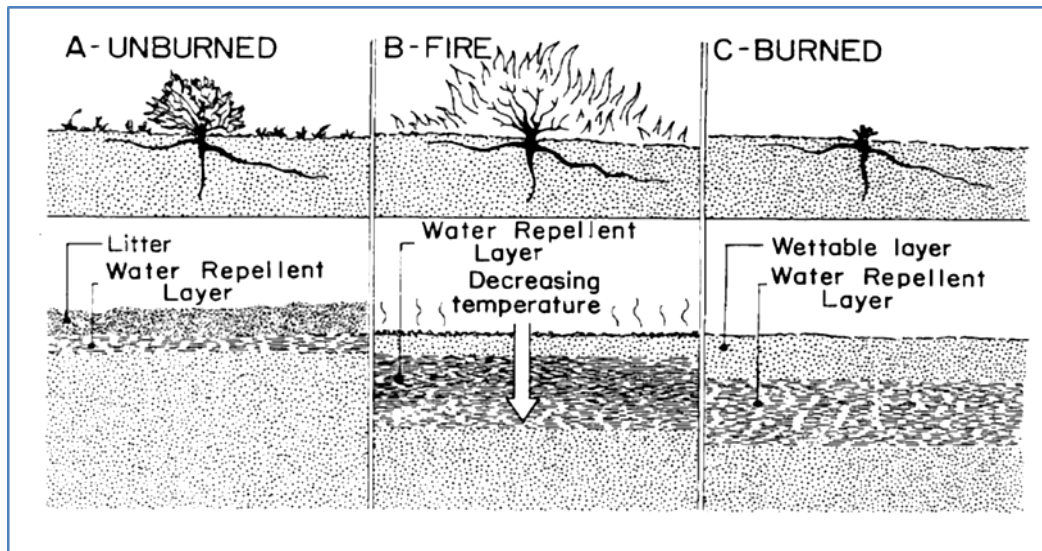
Figure 7. The development of water repellency. After DeBano (1969).

There has been a great deal of research in relation to this phenomenon over the decades, as the reviews by DeBano (2000) and DeBano *et al.* (2008) reveal. However, key aspects of soil hydrophobicity, including its formation and impacts, and relationship to fire remain contested in the literature. The property is common in a range of soil types, especially dry, sandy materials (Gilmour 1968; Roberts & Carbon 1972; DeBano 2000; Shakesby 2011; Rodriquez-Alleres *et al.* 2012), including in unburnt soils, as hydrophobic substances are leached from organic horizons (Doerr *et al.* 2000; Horne & McIntosh 2000).

Intense hydrophobicity can be generated by heating the soil to temperatures of approximately 200°C (Savage 1974; DeBano *et al.* 1976; DeBano 1981; Henderson & Golding 1983; Huffman *et al.* 2001; Letey 2001; Certini 2005). Fires which produce moderate increases in soil temperatures typical of HR burns could cause soil aggregates to become hydrophobic (Craig 1968; Giovannini & Lucchesi 1983; Giovannini 1994; Mataix-Solera & Doerr 2004).

Conversely, hydrophobicity is destroyed at temperatures >288 °C (Savage 1974; DeBano *et al.* 1976; DeBano 1981; DeBano 2000; Certini 2005) and combustion of organic matter is complete at 460°C (Giovannini *et al.* 1988). Shakesby *et al.* (2003), working in Nattai National Park in the Blue Mountains, found that surface repellency was destroyed at *c.*350 - 400°C. Heath *et al.* (2015) found that with an increase in burn severity, both C and water repellency in topsoils were reduced after wildfire in the Blue Mountains. Similarly, Tessler *et al.* (2012) found that water repellency in sites repeatedly burnt by wildfire was significantly lower than in the unburnt control sites. However, in hotter wildfires, a water repellent front can still occur in the subsoil, due to thermal gradients in soil profiles (Henderson & Golding 1983; Huffman *et al.* 2001). The spatial distribution of hydrophobicity will also vary across a site, due to localised variation in fire severity (Imeson *et al.* 1992; Martin & Moody 2001; Woods *et al.* 2007).

Hydrophobic conditions may persist for several months (McNabb *et al.* 1989) or even years following a mild fire (Henderson & Golding 1983; Huffman *et al.* 2001) and for up to several years following an intense fire (Dyrness 1976; DeBano, 2000; Pierson *et al.*, 2001). Leitch *et al.* (1983) found that soils affected by the Ash Wednesday fires in Victoria were hydrophobic for more than three months after that fire.

These hydrophobic coatings can significantly reduce raindrop penetration (Everett *et al.* 1995), permeability (Imeson *et al.* 1992) and hydraulic conductivity (Robichaud 2000), thereby resulting in low infiltration rates and enhanced overland flow (Leighton-Boyce *et al.* 2007), and post-bushfire increases in runoff generation and sediment yields have been partly

attributed to an increase in soil hydrophobicity (Osborn *et al.* 1964; Rustomji & Hairsine 2006). Soil hydrophobicity may increase runoff and rill formation at the plot scale in the post-fire period (DeBano 2000; DeBano *et al.* 2008), but there is limited research linking changes in soil hydrophobic properties to post-fire runoff and erosion at larger spatial scales (Prosser & Williams 1998; DeBano 2000; Shakesby *et al.* 2000; Doerr *et al.* 2003; Doerr & Moody 2004; Ferreira *et al.* 2005; Ferreira *et al.* 2008). Rustomji & Hairsine (2006) however did attribute increases in sediment yields following severe bushfire in the lower Cotter River catchment in the ACT to an increase in soil hydrophobicity, reduced vegetative cover and the low cohesion of ash and desiccated soil. Similarly, Neary *et al.* (2012) considered that hydrophobicity contributed significantly to massive soil erosion in northern Arizona in 2010.

Shakesby (2011) concluded in his review that the potential of water repellency to affect post-wildfire erosion was likely to vary between vegetation and soil types, sandy soil types being particularly susceptible. Despite the research effort directed to this issue, Certini's (2005) review concluded that in terms of erosion, the combustion of the vegetation and litter layer is likely to be more significant than hydrophobicity (see also Sevink *et al.* 1989; Scott & van Wyk 1990; Marcos *et al.* 2000). Other authors have also commented on the difficulty of distinguishing the importance of water repellency from other factors such as soil sealing and loss of vegetative cover (Doerr *et al.* 2003; Doerr & Moody 2004; Larsen *et al.* 2009).

Soil Structure

Soil structure refers to the spatial arrangement of soil aggregates, cracks and pores in a soil. These soil properties have a huge influence on water movement into and within the soil, the water-holding capacity of the soil (Boyer & Miller 1994; Boix Fayos 1997) and therefore, the hydrologic condition and behaviour of the catchment as a whole.

Fire can affect soil structure in a number of ways. These relate to the loss of organic matter, soil microbiology, water repellency, and effects on the physico-chemical properties and mineralogy of clays, including effects on soil structure imparted by sesquioxides (Figure 8) (Baver *et al.* 1972; Greene *et al.* 1990; Cerda & Doerr 2005; Neary *et al.* 2005; Mataix-Solera *et al.* 2011; Wittenberg 2012).
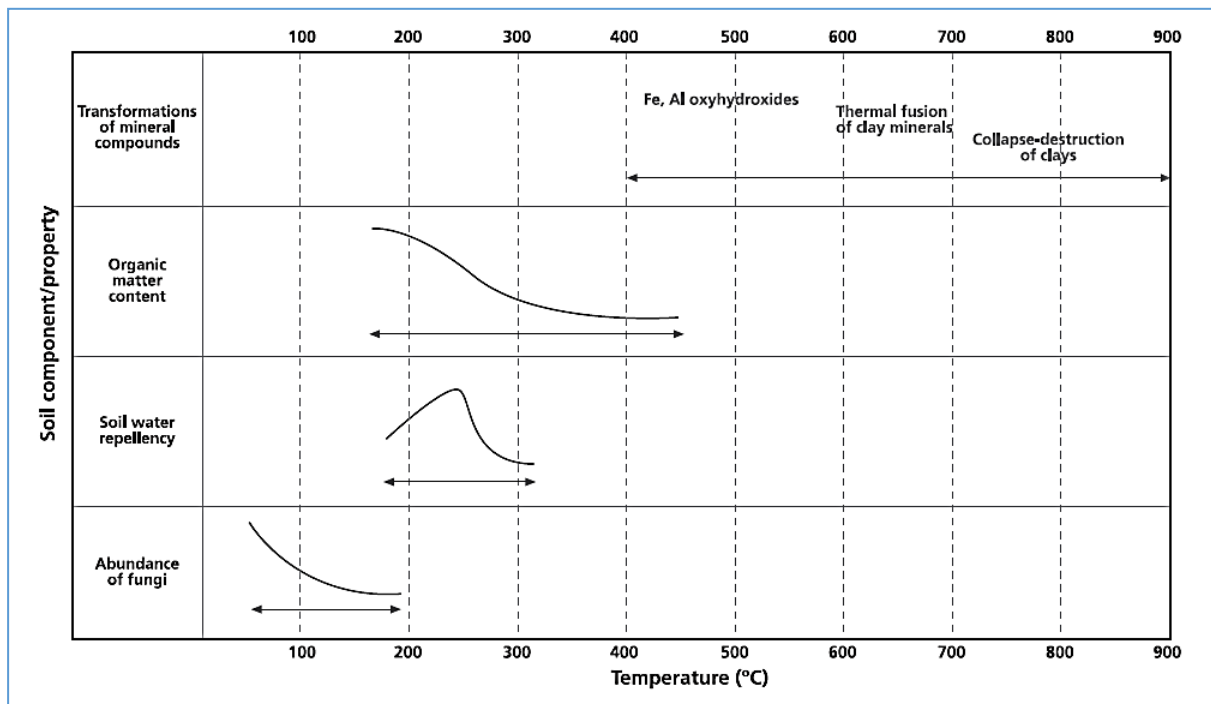
Figure 8. The main soil components or properties relevant to aggregation and their changes at different temperatures. Source: Mataix-Solera *et al.* (2011).

The impact of fire on soil organic matter is correlated with fire severity (Certini 2005). Consumption of organic matter commences in the range 200–250 ºC (Doerr *et al*. 1998) and is complete around 460 ºC (Humphreys & Craig 1981; Giovannini *et al*. 1988), causing loss of aggregate stability (Badia & Marti 2003; Blake *et al*. 2009), and producing a more easily erodible soil (DeBano *et al*. 1998; Scott *et al*. 1998; Neary *et al*. 1999). The bulk density of the soil (mass of soil/volume) increases as a result of the destruction of these organo-mineral aggregates (Giovannini *et al*. 1988) and mobilisation of disaggregated clays and/or ash materials can cause sealing of soil pores (Durgin & Vogelsand 1984). This results in a reduction in the hydraulic conductivity and water-holding capacity of the soil (Boyer & Miller 1994; Boix Fayos 1997), which in turn causes an increase in runoff and erosion (Martin & Moody 2001; see also Pillsbury 1953; Ramacharlu & Rao 1954; Tarrant 1956; Beaton 1959; Craig 1968).

The result is that high severity fires can induce important changes in soil structure and aggregate stability, but with different effects depending on the type of soil affected. The patterns observed can vary from disaggregation as a consequence of the organic matter destruction, to a strong aggregation if a recrystallisation of some minerals such as Fe and Al oxyhydroxides (Mataix-Solera *et al.* 2011).

In contrast, the impacts of low intensity fires are less marked, as fires that do not exceed a soil temperature of 200 ºC have little effect on soil organic matter, and therefore do not affect soil stability (Humphreys & Craig 1981; Mataix-Solera *et al.* 2011). Rather, some authors have suggested that low intensity fires can enhance structural stability in some cases due to the formation of organo-mineral aggregates (Mataix-Solera & Doerr 2004; Cawson *et al*. 2012). It also appears that the sesquioxide-related structure is generally little affected by cooler burns (Humphreys & Craig 1981). The effects of moderate intensity burns on organic matter-related soil structure appear to be similar to those of high intensity fires, albeit of a lesser magnitude. In Australia, Greene *et al*. (1990) found that fire caused a major decline in aggregate stability in the surface horizon. Boyer & Miller (1994) working in pine stands in the US, found that moderate intensity prescribed burning reduced macroporosity, increased the bulk density of surface soils, and decreased available moisture holding capacity of both surface and subsurface soils. This resulted in a reduction in tree growth rates due, at least in part, to increased moisture stress associated with changes in soil physical properties.

Antecedent Moisture Conditions

The antecedent soil moisture status of the soil profile relates to the proportion of the effective soil moisture storage capacity that is already occupied, and therefore the amount of further rain that can be accommodated. The significance of this in affecting runoff/infiltration ratios has been demonstrated by Yu (2015) in relation to the 2013 Warrumbungle National Park fire. Heavy rainfall immediately following a large wildfire produced relatively little erosion, but primed the landscape for massive runoff when the intensive storm occurred on 1 February 2013, causing widespread erosion and siltation of drainage lines (McInnes-Clarke *et al.* 2014).

Slope and Slope Length

Numerous studies (e.g. Musgrave 1947; Wischmeier 1972) have shown that sediment yield by surface wash increases dramatically with increasing slope (S) and increasing slope length ((L). Slope is a variable that can only be managed in terms of burn site selection. However, the slope length factor is alterable indirectly through manipulation of fire intensity. In severe fires, the effective slope length is greater due to complete combustion of slope length-limiting obstacles such as ground cover, litter and woody debris.

However, a patchy burn typical of a HR burn will retain many of these elements (Rab 1996), thus limiting slope length. Cawson *et al*. (2011), working in the upper Yarra River catchment in Victoria, found that unburnt patches trapped 92–99% of sediment for rainfall events with an ARI < 1 year, although for more intense rainfall events, unburnt patches were ineffective.

It is likely that the substantial soil erosion and runoff results detected in plot and hillslope-scale studies by some authors (e.g. Benavides-Solorio and MacDonald 2005; Morales *et al.* 2000; Robichaud 2000) are not always replicated at the sub-catchment scale (cf. e.g. Elliot & Vose 2005; Ronan 1986) as limited effective slope lengths and local sediment sinks often dilute the instream effects of burning (e.g. Townsend & Douglas 2000).

One of the most significant slope length-limiting features for a HR burn is the retention of riparian filter strips as control lines. The significance of riparian filter strips as retention lines for surface flow and sediment movement has been extensively studied (Hairsine 1996), and their effectiveness has long been recognised. They are routinely used in the context of forestry, having been incorporated into the Standard Erosion Mitigation Guidelines for Logging in NSW 1993 (Department of Conservation and Land Management 1995), and more recently, the Private Native Forestry Code of Practice for Northern NSW (DECC 2008a,b,c,d).

The Effects of Different Fire Regimes on These Factors

Severe wildfires are recognised as having a highly significant impact on soil erosion and sedimentation, due to:

- effectively greater rainfall erosivity due to canopy loss
- greater impact on the litter layer and soil cover
- loss of soil aggregate stability due to physico-chemical changes
- increased effective slope length due to combustion of obstacles.

As a result, catchment sediment yields are known to increase by up to several orders of magnitude following intense bushfires, both in Australia (Brown 1972; Leitch *et al.* 1983; Atkinson 1984, 2012) and overseas (Ahlgren & Ahlgren 1960; Diaz-Fierros *et al.* 1987; Inbar *et al.* 1998; Rulli *et al.* 2006; Mayor *et al.* 2007; Neary *et al.* 2012). Robichaud *et al.* (2009) found that high severity wildfires increase runoff and erosion rates by two or more orders of magnitude, while low and moderate severity burns have much smaller effects on runoff and sediment yields. Benito *et al.'s* (1991) review of post-wildfire soil erosion losses in the Mediterranean literature suggested soil losses up to 170 t/ha/yr, and Shakesby's (2011) review noted first year soil losses of 45-56 t/ha, figures that compare with those in Australia. Furthermore, the recovery period is generally extended after wildfire, where tree and shrub canopies have been killed (e.g. Brown 1972).

Erosion rates generally take up to a decade or more to return to near pre-disturbance conditions after wildfire (Gimeno-Garcia *et al.* 2007; Wittenberg & Inbar 2009; Shakesby 2011). Such is the impact of wildfire that it has been suggested (e.g. Shakesby 2011) that wildfire and its effects is often the single most important agent of geomorphological change.

In contrast, in most low intensity HR burns, soil cover remains higher, compared to that following wildfire. There is also a greater retention of organic matter (Figure 3) and therefore aggregate stability. In a well-conducted HR burn, effective slope lengths are generally shorter. On the other hand, low intensity burning may reduce macroporosity and increase the bulk density of surface soils, and therefore decrease the available water-holding capacity of both surface and subsurface soil layers (Boyer & Miller 1994).

Soil hydrophobicity may also occur as a result of cooler fires, but its significance is debated.

The result is that overall, lower runoff and erosion rates have been reported as a consequence of low severity burns (Copley *et al.* 1944; Biswell & Schultz 1957; Robichaud 2000; Dragovich & Morris 2002; Benavides-Solorio & MacDonald 2005). Robichaud *et al.* (2009) found that sediment yields from rainfall simulations on plots burned at low severity were an order of magnitude smaller than the values from the plots burned at high severity. Similar differences in sediment yields were reported by Benavides-Solorio and MacDonald (2005).

Low fire severity (Arkle & Pilliod 2010; Richter *et al.* 1982) and burn patchiness (Richter *et al.* 1982; Smith *et al.*2010) are often cited as potential explanations for the observed small impacts following low severity burns. Morris *et al.* (2014) examined prescribed burns in the Mount Lofty Ranges in SA, and found that fire severity was a highly significant environmental determinant for the presence of sediment movement after prescribed burning. It is also likely that low severity burns produce less soil erosion compared to more severe wildfires because the recovery period is shorter (Raison *et al.* 1986a); therefore, the chance of large erosion impacts following prescribed burns are less. Cawson *et al.* (2012) carried out an extensive review into the impacts of prescribed low severity burns, and were unable to locate reports of large impacts at the catchment scale following prescribed burning. They concluded that existing studies show that the effects of prescribed burning are usually minimal. However, this does always appear to be the case: at a site in the upper Yarra catchment in Victoria, Cawson *et al.* (2011) found that a high fire severity site delivered only 13% more than the low fire severity site, and that the low fire severity site had 1000 times more erosion than an unburnt site.

Post-HR burn impacts may occur, but would be expected to occur infrequently, and would require an intense storm in the burn area. There appears to be but one example in the literature of severe erosion following prescribed low severity burning; this occurred in eastern Victoria in 2010, due to intense, short duration rainfall events (Cawson *et al.* 2012). The frequency of these events and their importance from a management perspective is only likely to be quantified by surveys of many prescribed burn areas (Cawson *et al.* 2012). However, thus far, these have not been undertaken. Wittenberg & Inbar (2007) also suggested that cumulative impacts may result from repeated burning, leading to a prolonged recovery phase and a gradual increase in 'baseflow' sediment yields. However, this has not been tested in Australian conditions.

Impact of Different Bushfire Regimes on Quantity and Quality of Water

Introduction

Fire can affect the quantity and quality of water produced at the catchment scale by the destruction or modification of vegetation, litter and the organic horizons of soils, and by altering certain physical characteristics of the soil that are related to runoff-infiltration characteristics, namely: soil structure and the stability of soil materials; porosity and infiltration; and water-holding capacity. The removal of organic matter is particularly important, as this causes collapse of the soil structure, and a consequent reduction in soil porosity and therefore infiltration.

Organic matter can also be important in contributing to the water-holding capacity of the soil, and in protecting the soil surface from raindrop impact, which can cause detachment of fine material and consequent loss of porosity. The phenomenon of hydrophobicity has also been dealt with above. Depending on the magnitude of these changes to the soil interface, and their spatial extent and pattern, significant water quantity and water quality effects at the hillslope and catchment scales can result, which vary over the short to long term.

Water Quantity

Post-fire Water Use and Stream Flow Responses

Immediately following a fire, the reduction in leaf area causes a reduction in evapotranspiration losses from vegetation, and therefore lower water use, compared with that in the pre-fire forest. Lower water use leads to increased soil moisture storage and the recharge of groundwater systems. The result is that runoff becomes more responsive and runoff/infiltration ratios and stream flows generally increase, compared with runoff from the pre-fire period. This period of lower water use and increased runoff typically lasts ~2-10

years (Brown 1972; Watson *et al.* 1999) before restoration of leaf area causes water use to increase to at least pre-fire levels (see e.g. Ahlgren & Ahlgren 1960 and studies cited therein; see also Campbell *et al.* 1977; David 1977; Helvey 1980; Hibbert 1984; Lavabre *et al.* 1993; DeBano *et al.* 1998).

The magnitude of the increase in immediate post-fire stream flows generally correlates with burn severity i.e. greater reduction in evapotranspiration rates (Campbell *et al.* 1977). In cases of severe wildfire the increase in stream flows may be significant. Sinclair & Hamilton (1955) and Glendinning *et al.* (1961) for example noted post-wildfire increases of 200-45 000% on peak flows from chaparral vegetation in south-western United States. This effect has also been noted in Australia: McArthur (1964) examined the effects of a wildfire in subalpine vegetation on the Bogong High Plains in Victoria, and reported an increase in stream flows immediately following the fire. McArthur & Cheney (1965) noted increases in stream flow by 43-235% following fires in the Cotter River catchment in the Australian Capital Territory. Brown (1972) studied the effects of a severe bushfire that affected the Snowy Mountain in 1965 and similarly found pronounced changes in the shape of the flood hydrograph in the years immediately following the fire, including increased peak discharges, and much greater runoff overall.

In the medium- to longer-terms however, catchments affected by severe wildfires generally experience a reduction in water yields below pre-fire discharge levels as the vegetation enters a phase of rapid growth. Leaf area and evapotranspiration increases, so that water use is increased compared to that of a mature forest, and catchment flows are generally reduced. The actual magnitude of post-fire changes to the water cycle are dependent on the severity of the fire and the nature of consequent ecosystem succession, including the species of trees and other vegetation present, their susceptibility to death during the fire, and the environmental setting, including the rainfall zone.

These relationships have been particularly examined in the context of *Eucalyptus regnans* forests, which occur in many Victorian water catchment areas. Langford (1976) noted that regenerating forests burnt in the severe 1939 bushfires produced a 24% reduction in yield in the 21 years following the fires, and the dramatic effects of high severity fires on water yields have since been verified by a large body of research (e.g. Kuczera, 1985, 1987; Vertessy *et al.* 1993, 1996, 2001; Watson *et al.*, 1999a, 1999b, 2001; Cornish & Vertessy 2001; Lane & Mckay 2001). The age/stream flow relationship for *E. regnan*s forests was generalised by Kuczera (1985, 1987) using rainfall and runoff data collected from forested catchments burnt in 1939, and is represented by the well-known 'Kuczera curve' (Figure 9), which allows the reduction in mean annual stream flow to be estimated by forest age (Kuczera 1987; Watson *et al.* 1999a).

These reductions can be significant. Sinclair Knight Mertz (2004) used satellite imagery of the 2003 wildfire in upper Murray catchments in Victoria to model vegetation combustion and tree water usage. They predicted an initial flow increase of 14-106%, expected to persist for approximately 7 years, followed by a reduction in total inflows to the Murray of up to 129 GL/yr. Vertessy (2003), in an investigation of the same fire, concluded that inflows to the river would be reduced by approximately 430 GL/yr in 20 years' time.

This phase of reduced flows can persist for periods ranging from 20 to 250 years depending on a number of factors, including the vegetation communities affected, landscape factors, and the spatial extent and severity of the burn (Watson *et al.* 1999a; Cornish & Vertessy 2001; Roberts *et al.* 2001).
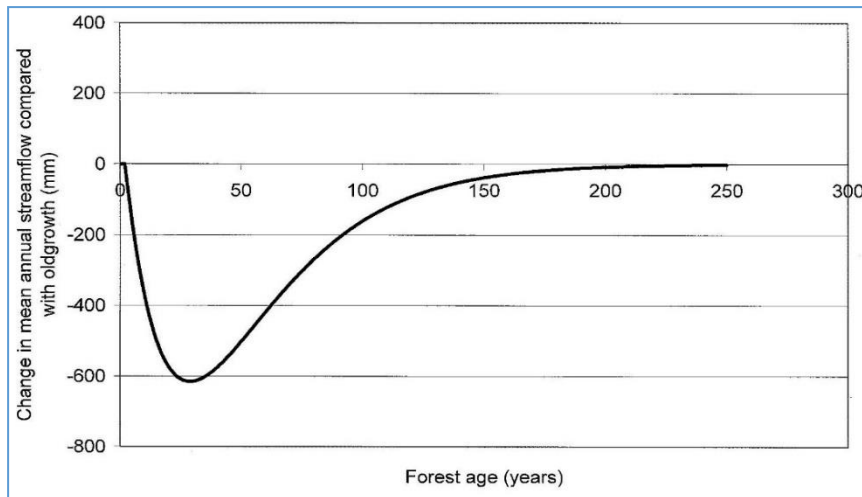
Figure 9. Kuczera curve.

However these results from Victorian catchments may not necessarily apply to different soils and landscapes: Heath *et al.* (2012), working in sandstone landscapes of the Nattai part of the southern Blue Mountains, found no such pattern resembling the Kuczera curve. Vegetation and water yield had recovered within two to five years after wildfire, possibly underlining the significance of vegetation type and its relationship to fire, specifically obligate resprouters rather than reseeders.

Effects of Low Severity Hazard Reduction Burning on Water Use and Stream Flow Responses

Although more significant hydrologic responses occur as a consequence of wildfire, it is likely that fire regimes that result in lower vegetation mortality rates, or altered vegetation composition, could produce similar, although more subtle, hydrologic effects (Scott 1993; DeBano *et al.* 1998; Zhang *et al.* 2001; Huxman *et al.* 2005). Feikema *et al.* (2008) therefore modelled the hydrologic responses of a number of Victorian catchments, assuming various vegetation mortality rates, and found significant cumulative reductions in water yield, even at low mortality rates (Figure 10). These results may be indicative of hydrologic reductions in some vegetation types after even low severity burns, especially where water use by the understorey is a significant proportion of total water use by the vegetation community (Vertessy *et al.* 2001; Wood *et al.* 2008; Pfautsch *et al.* 2010). Marcar *et al.* (2006), in a review of the hydrological impacts of bushfire in the upper catchments of the Murray River, concluded that hydrological responses depend largely on ecological responses to fire. They considered that further investigation was required into the ecological and physiological responses of vegetation to fire, in order to better validate models of forest evapotranspiration.

Hence, any changes in the structure of vegetation, even temporary changes, could potentially have implications for water use and the yield from catchments. However, despite the importance in many cases of the understorey to total stand water use, it appears that there has been little research on the actual hydrological effects of fuel-reduction burns (Adams & Attiwill 2011), although the general expectation is as shown in Figure 11. Notable exceptions are Gottfried & DeBano (1990), who failed to detect any significant change following a prescribed low severity fire in a ponderosa pine forest in Arizona, in which fine fuels were reduced by 70%, and Scott (1993), who in contrast, recorded a 15% increase in hydrologic response following a prescribed burn in fynbos in South Africa.
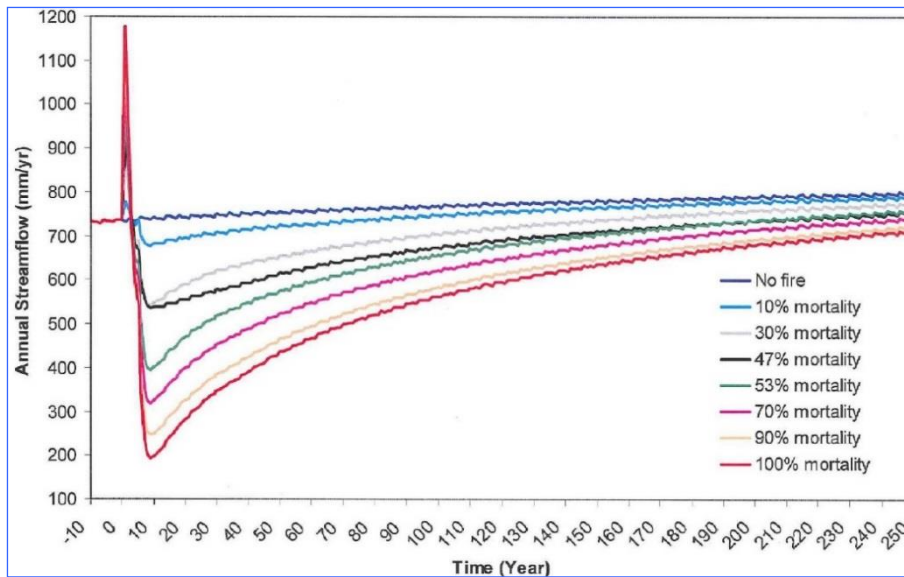
Figure 10. Hydrologic response curve assuming various mortality rates, from Feikema *et al*. (2008).
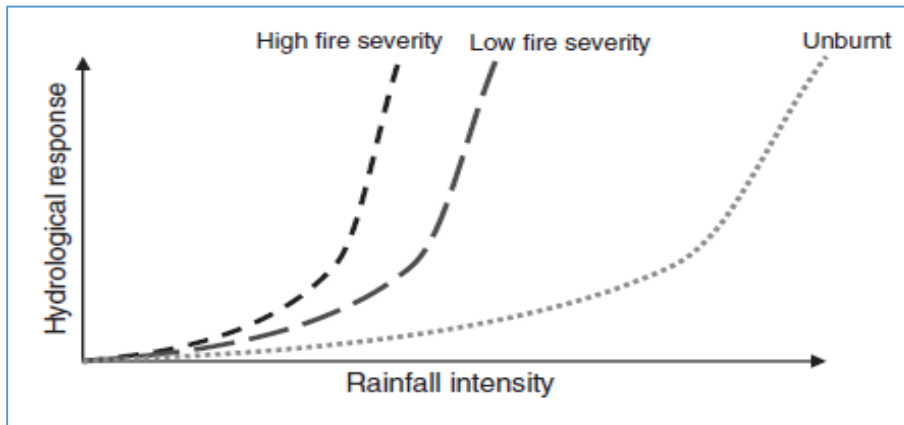
Figure 11. Conceptual model of the relationships between fire severity and rainfall intensity on post-fire runoff and erosion. From Cawson *et al.* (2012).

Conclusions

Cawson *et al*. (2012) undertook a review of the literature dealing with the effect of different fire regimes, including prescribed low severity burning, on surface runoff and confirmed that stream flow responses to prescribed fire do occur, but are generally smaller in magnitude compared to wildfire responses. However, relatively minor medium- to long-term reductions in flows caused by prescribed low severity fire may be compounded by climate change-related reductions in precipitation, particularly in south-east Australian catchments (Hennessy *et al*. 2005; Howe *et al*. 2005; Lane *et al*. 2011).

Water Quality

Introduction and Background

Water quality refers to the physical, chemical and biological characteristics of water. Post-fire, particularly post-wildfire, peak flows and total runoff tends to be greater compared with pre-fire runoff characteristics, and as a result, there is an increased potential for sediment movement, and with it, nutrients and other pollutants. Sediment may be sourced from incision and removal of headwater peats (Good 1973), increased erosion of hillslopes, or

from the increased competence of scouring flows on gullies, drainage lines and stream banks.

Coarse sediment is generally arrested on the slope or within the catchment. Conversely, finer silt and clay-sized particles may become entrained in flowing water. Many potential pollutants such as N, P, C, Ca, Mg and K may be adsorbed to these fine-grained organic and inorganic sediments (Palis *et al.* 1990; DeBano *et al.* 1998; Pettersson 1998; Webster *et al.* 2001), or may be transported in dissolved form. The fine sediment load is generally expressed as turbidity, which is strongly correlated to the concentration of fine-grained sediments <63 microns in size (Rustomji & Hairsine 2006; White *et al.* 2006).

A great deal of research has focused on the potential risks from fires to human water supplies, both in the US (Ice *et al.* 2004; Rhoads *et al.* 2006;) and in Australia (Wilkinson *et al.* 2004; White *et al.* 2006) (see also Anderson *et al.* 1976; Tiedemann *et al.* 1979; Richter *et al.* 1982; Chessman 1986; Baker 1990; Spencer & Hauer 1991; Bayley *et al.* 1992; Lathrop 1994; Gerla & Galloway 1998; McEachern *et al.* 2000; Minshall *et al.* 2001; Gallaher *et al.* 2002; Earl & Blinn 2003; Leak *et al.* 2003; Prepas *et al.* 2003; Wasson *et al.* 2003; Andreassian 2004; Townsend & Douglas 2004; Ferreira *et al.* 2005b; Petticrew *et al.* 2006; Sheridan *et al.* 2007b; Bladon *et al.* 2008; Lane *et al.* 2008; Mast & Clow 2008; Noske *et al.* 2010).

Nutrient Mobilisation by Wildfire

As a general principle, the management of water supply catchments has focussed on the minimisation of delivery of two important nutrients, N and P (Harris 2001) as well as dissolved organic C. Post-fire stream exports of total N may be significantly elevated (1.1 to 27 kg/ha/yr: Smith *et al.* 2011a), much of it in the form of NO_3^-, which is highly soluble and therefore available for mobilisation from a catchment into streams, where it may contribute to eutrophication of waters (Rustomji & Hairsine 2006). Smith *et al.* (2011a) found that post-fire NO_3^- exports ranged from 0.04 to 13.0 kg/ha/yr, or 3 to 250 times that from unburnt catchments.

Smith *et al.* (2011a) also examined the literature with respect to total P. Post-wildfire exports ranged from 0.03 to 3.2 kg/ha/yr, which represented up to 431 times that from unburnt catchments. See also Morris & Calliss (n.d.), who noted a slight rise in P following a fire at the Mount Bold reservoir in the Adelaide Hills.

Dissolved organic C deliveries to reservoirs have also been noted in the literature, particularly in a number of studies from the Mt Lofty Ranges in South Australia (Stevens *et al.* 1999; Flemming & Cox 2001; Cox & Pitman 2001). Elevated Na^+, Cl^- and SO_4^{2-} have also been recorded soon after fire (Smith *et al.* 2011a). Few studies have examined post-fire exports of trace elements, although high levels of Fe, Mn, Cr, Al, Ba and Pb have been associated with highly elevated sediment concentrations. Within reservoirs, anoxic conditions also trigger a series of chemical reactions that liberate a range of pollutants that are otherwise bound to sediment upon the reservoir floor, such as Mn and Fe (White *et al.*, 2006).

Again, most studies have looked at the effects of wildfire on nutrient mobilisation and pollution, due to the marked impact of wildfires in accelerating surface erosion, and therefore the amounts of sediment, nutrients and other contaminants delivered to streams and reservoirs (DeBano *et al.* 1998; Moody & Martin 2001, 2009; Reneau *et al.* 2007; Sheridan *et al.* 2007b; Lane *et al.* 2008; Wilkinson *et al.* 2007; White *et al.* 2006). An extensive review of this topic has recently been undertaken by Smith *et al.* (2011a), who found that reported first year post-fire suspended sediment exports ranged from 0.017 to 50 t/ha/yr, representing an increase of up to 1 459 times the exports from unburnt catchments.

Effects of Prescribed Low Severity Hazard Reduction Burning on Water Quality

In contrast, there are few studies specifically on the effects of prescribed low severity fire on stream water chemistry and water quality (e.g. Richter *et al.* 1982; Davis 1989; Williams & Melack 1997; Stephens *et al.* 2004; Smith *et al.* 2010). Broughton (1970) and Brown (1974) reviewed the relevant literature to that point and concluded that low intensity fires offered a much lower risk of adverse effects on water supplies than higher intensity wildfires. The most recent review was by Cawson *et al.* (2012), who found that despite the large areas burnt in Victoria every year, post-burn water quality has been measured in only two small sub-catchments, and during below-average rainfall conditions (Smith *et al.* 2010). This study involved a paired (control and treatment) catchment analysis of the effect of prescribed low severity fire on suspended sediment and nutrients (P and N as NO_3^-) export in a eucalyptus forest in south-eastern Australia. They found that the effect of the prescribed fires on stream exports of suspended sediment and nutrient from the study catchments was very minor. According to Cawson *et al.* (2011, 2012), the characteristics most likely to be important in affecting water quantity and quality are fire severity, burn patchiness, fire frequency and burn season.

The effects of an individual prescribed low severity HR burn may be small and only last for a short period (Beche *et al.* 2005; Stephens *et al.* 2004) but the cumulative effect of multiple smaller prescribed burns may be an issue (Cawson *et al.* 2012). Furthermore, while existing studies show that the effects of prescribed low severity burning on the quantity and quality of water are usually minimal, this may be because large erosion impacts following prescribed burns are rarely located and studied.

Conclusions

The vast majority of the literature deals with the effects of severe wildfire. This is because these fires can result in catastrophic impacts on nutrient pools, post-fire soil erosion, sedimentation and impacts on water quality, and long-term impacts on water quantity from affected catchments.

In contrast, relatively little research has been carried out into the effects of prescribed low severity HR burns, where the effects are generally more subtle and variable. Furthermore, much of the overseas literature is not particularly relevant to Australia, due to its generally thin, nutrient-poor soils.

Conclusions are presented below in response to the three Priority Research Questions addressed in the Background section of this report.

In Relation to Nutrient Cycling and the Carbon Cycle

What is the impact of different bushfire regimes on nutrient cycling and the carbon cycle?

Fire typically results in a reduction in the size of fuel and soil organic nutrient pools by depleting nutrients in native vegetation systems through:

- losses to the atmosphere by volatilisation and convection
- losses by runoff
- losses by leaching.

Fire may also:

- increase soil nutrient cycling rates
- redistribute nutrients through the soil profile.

The pool of topsoil fertility is diminished by fire. High severity wildfires can impact on the C ecosystem and soil storage capacity for decades to centuries. In comparison, prescribed low severity HR burns result in lower losses by volatilisation, convection and other pathways.

Due to their lower temperatures of volatilisation, C, S and N are particularly susceptible to fire, including low severity burns. Virtually all the N in a soil system is located in the soil organic matter, so the volatilisation of N is proportional to the combustion of soil organic matter. In contrast, P is not as susceptible to volatilisation and leaching. Especially in acidic soils, losses of P due to volatilisation and leaching are generally relatively minor, although these may still be significant in the Australian context, especially in nutrient-poor sandy soils.

The relative impact of any given fire on the nutrient pool will be potentially greater where the fire is severe and occurs in vegetation that has not been burnt for extended periods. However, concerns have also been raised that frequent repeated burning may cumulatively deplete sites of nutrients and interrupt rates of nutrient cycling.

Much of the nutrient losses occur in the convective smoke column. Lower severity burns tends to produce less smoke, and therefore lower convective nutrient losses, but again, these may still be significant in the Australian context, especially in nutrient-poor sandy soils.

Leaching of nutrients could be especially significant in sandy soils, due to the low inorganic cation exchange sites and low buffering capacity of such materials. Leaching through sandy soils is greatest in high rainfall zones.

The potential for erosive losses is also greatly diminished in a properly conducted HR burn, due to partial retention of soil organic matter and ground cover. However, these losses can still be significant.

Moist soils, and soil types with high specific heat (clayey soils), will be affected by fire through soil heating to a lesser extent than soils with greater thermal conductivity, including drier soils, soils with a lower bulk density, and soils with a lower specific heat (sandy soils).

Insufficient research has been carried out into the differential impacts of bushfires and prescribed low severity HR burning on the C cycle. In temperate eucalypt forests, the potential for prescribed burning to produce a net reduction in C emissions is considered to be low.

In Relation to Soil Erosion and Sedimentation

What is the impact of various bushfire regimes on different soil formations and on erosion and sedimentation rates (i.e. what soil types are vulnerable to what fire conditions)?

Wildfires are generally known to have a greater impact on soils than low severity fires; intense bushfires may increase catchment sediment yields by up to several orders of magnitude. However, there is surprisingly little recent Australian research, and few studies have been made of the rates of soil erosion following bushfires in sandstone catchments.

Most studies that have considered the effects of low severity burning on surface runoff and erosion show that the impacts are generally minimal, due to the retention of ground cover and soil organic matter, compared to more intense fire. However, severe erosive events following low severity burns have been recorded.

The key soil surface-related factors that determine the runoff/infiltration ratio for any given rainfall intensity are:

- the thickness and water-holding capacity of litter and organic layers
- soil structure, porosity and permeability
- soil depth to less permeable materials
- the antecedent litter and soil moisture conditions
- other factors which may enhance runoff, such as soil hydrophobicity.

For any given burn severity therefore, the soils that are the most susceptible to erosion are erodible sandy soils or soils with otherwise high K factors, that are thin, stony, and massive, with high bulk density.

In Relation to Water Quantity and Quality

What is the impact of different bushfire regimes on water flows (both quantity and quality)?

Fire can affect the quantity and quality of water produced at the catchment scale by the destruction or modification of vegetation, litter and organic horizons of soils, and by altering certain physical characteristics of the soil that are related to runoff-infiltration characteristics, namely: soil structure and the stability of soil materials; porosity and infiltration; and water-holding capacity. The removal of organic matter is particularly important. Depending on the magnitude of these changes to the soil interface, and their spatial extent and pattern, significant water quantity and water quality effects at the hillslope and catchment scales can result, which vary over the short to long term.

Immediately following a fire, runoff becomes more responsive and runoff/infiltration ratios and stream flows generally increase, compared with runoff from the pre-fire period. This period of lower water use and increased runoff typically lasts ~2-10 years. The magnitude of the increase in immediate post-fire stream flows generally correlates with burn severity. In cases of severe wildfire, the increase in stream flows may be significant.

In the medium to longer term however, catchments affected by severe wildfires generally experience a reduction in water yields below pre-fire levels as the vegetation enters a phase of rapid growth and catchment flows are generally reduced. The actual magnitude of post-fire changes to the water cycle is dependent on the severity of the fire and a range of other factors.

Although more significant changes to hydrologic responses occur as a consequence of wildfire, it is likely that fire regimes that result in lower vegetation mortality rates, or altered vegetation composition, could produce similar, although more subtle, hydrologic effects. Stream flow responses to low severity prescribed fire do occur, but are generally smaller in magnitude compared to wildfire responses. However, relatively minor medium to long term reductions in flows caused by prescribed low severity fire may be compounded by climate change-related reductions in precipitation, particularly in south-east Australian catchments.

Most studies have looked at the effects of wildfire on nutrient mobilisation and pollution, due to the marked impact of wildfires in accelerating surface erosion, and therefore the amounts of sediment, nutrients and other contaminants delivered to streams and reservoirs. Post-fire, particularly post-wildfire, peak flows and total runoff tends to be greater compared with pre-fire runoff characteristics, and as a result, there is an increased potential for sediment movement, and with it, nutrients and other pollutants. Many potential pollutants are adsorbed to fine-grained organic and inorganic sediments, or may be transported in dissolved form. First year post-fire suspended sediment exports may be several orders of magnitude greater than exports from unburnt catchments. The export of two important nutrients, N and P, may be hundreds of times greater than that from unburnt catchments.

There are few studies specifically on the effects of prescribed low severity fire on stream water chemistry and water quality. However, evidence suggests that low intensity fires offer a much lower risk of adverse effects on water supplies than higher intensity wildfires. The factors most likely to be important in affecting water quantity and quality are fire severity, burn patchiness, fire frequency and burn season.

Knowledge Gaps

Despite the enormous body of research that has examined soil-fire interactions, the vast majority of this research has overwhelmingly concentrated on the effects of wildfire, rather than on the effects of low severity burns such as HR burns. Though it has been demonstrated that the majority of erosion occur within a few storm events , little research has been done in estimating event-based soil loss and sediment delivery, and the exact causes of variation in post-fire erosion has not yet been quantified. Given the recent increase of HR measures in NSW and other jurisdictions, further research is urgently required into the effects of prescribed HR fire on soil-related issues such as nutrient cycling, soil erosion and sedimentation, and effects on water quantity and quality. Research should examine these issues across a range of different soil, vegetation and landscape types, and fire regimes. This work should include process-based studies, to better understand the factors involved; these studies should calibrate and validate broader-scale modelling used to extrapolate site-specific process studies to larger areas.

Research is required to define the period of disturbance as a result of HR burning, particularly in relation to vegetation and groundcover recovery for different fire intensities, soils and landscape types. Post-fire ground cover measurements should be used to determine appropriate factors for erosion modelling and risk mapping.

There are few studies specifically on the effects of prescribed low severity HR fire on stream water chemistry and water quality. More research is also required in relation to the magnitude and duration of post-fire water quantity effects from different soil, vegetation and landscape types subjected to different fire regimes.

The effects of HR burning on soil erosion and sedimentation, and its effects on water quantity and quality need to be examined in the context of high magnitude rainfall events. The potential under such circumstances for HR burning to contribute to elevated rates of erosion and sedimentation, including debris flows, and the potential for impacts on public safety and infrastructure, should be assessed. Spatially explicit maps of soil erosion in near-real time are needed in HR burn assessments.

There is need for research in relation to soil ecology, chemistry and physics to better understand the effects of repeated HR burning and the relationships between pre- and post-fire soil ecological function. This should be undertaken across a range of different soil, vegetation and landscape types, and fire regimes.

The impacts of HR burning on nutrient-poor systems need to be better understood, and particularly those which depend for their long-term productivity on efficient nutrient accumulation, retention and recycling processes, and which are likely to be more sensitive to disturbance caused by regular burning.

The potential for HR burning to reduce total C emissions and increase storage requires a more detailed understanding of the dynamics of combustible biomass pools in a range of different soil, vegetation and landscape types. Further research is required to more fully understand C storage and emission trade-offs associated with HR fire regimes.

The above work should contribute to the development of decision support tools, including the development of erosion risk mapping for environmental assessments of proposed HR burns. These decision support tools should enable a more systematic analysis of the risks associated with prescribed HR burning, with a view to minimising the risks to soil resources, water quality, public safety and infrastructure. This would allow decision makers to assess the extent and magnitude of post-burn soil erosion and to prioritise remedial activities after storm events.

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