Review Paper

Wildfire effects on water quality in forest catchments: A review with implications for water supply

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SUMMARY

Wildfires burn extensive forest areas around the world each year. In many locations, fire-prone forest catchments are utilised for the supply of potable water to small communities up to large cities. Following wildfire, increased erosion rates and changes to runoff generation and pollutant sources may greatly increase fluxes of sediment, nutrients and other water quality constituents, potentially contaminating water supplies. Most research to date has focused on suspended sediment exports and concentrations after wildfire. Reported first year post-fire suspended sediment exports varied from 0.017 to 50 t ha^{-1} year^{-1} across a large range of catchment sizes (0.021–1655 km^2). This represented an estimated increase of 1–1459 times unburned exports. Maximum reported concentrations of total suspended solids in streams for the first year after fire ranged from 11 to ~500,000 mg L^{-1}. Similarly, there was a large range in first year post-fire stream exports of total N (1.1–27 kg ha^{-1} year^{-1}) and total P (0.03–3.2 kg ha^{-1} year^{-1}), representing a multiple change of 0.3–431 times unburned, while NO$_3^-$ exports of 0.04–13.0 kg ha^{-1} year^{-1} (3–250 times unburned) have been reported. NO$_3^-$, NO$_2^-$, and NH$_3$/NH$_4^+$ concentrations in streams and lakes or reservoirs may increase after wildfire but appear to present a generally low risk of exceeding drinking water guidelines. Few studies have examined post-fire exports of trace elements. The limited observations of trace element concentrations in streams after wildfire found high levels (well over guidelines) of Fe, Mn, As, Cr, Al, Ba, and Pb, which were associated with highly elevated sediment concentrations. In contrast, Cu, Zn, and Hg were below or only slightly above guideline values. Elevated Na$^+$, Cl$^-$ and SO$_4^{2-}$ solute yields have been recorded soon after fire, while reports of concentrations of these constituents were mostly confined to coniferous forest areas in North America, where maximum sampled values were well below recommended limits. Likewise, reported wildfire effects on dissolved organic carbon were generally minor and elevated concentrations largely reflected background conditions. Available cyanide concentrations in small streams may approach levels of concern soon after fire, but increases are likely to be of short duration. Post-fire concentrations of polycyclic aromatic hydrocarbons in streams have been found to increase but remained within the recommended limit. Large increases in exports and concentrations of some constituents after wildfire indicate the potential for impacts on water supply from forest catchments. In response, various water treatment measures may be required and in the absence of adequate treatment facilities or alternative sources, water supplies may be vulnerable to disruption.

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1. Introduction

Wildfire poses an important threat to life, property, infrastructure, and natural resources in fire-prone forest areas. Each year wildfires burn large areas of forest land around the world, particularly in western North America, south-eastern Australia, and the Mediterranean (FAO, 2001). For example, in Canada the annual area of forest burned ranged between 0.29 and 7.56 million hectares from 1975 to 2005 (Canadian Council of Forest Ministers, 2006). Similarly, in recent years there have been extensive forest areas burned by wildfire in the United States. Between 1997 and 2008, 65 wildfires greater than 100,000 acres (40,469 ha) occurred, many of which included forest areas (National Interagency Fire Centre, 2009). Notably, in the western United States there has been a significant increase in the number, size and severity of wildfires since 1990 (Joint Fire Science Program, 2004).

In south-eastern Australia, wildfire events in 2003, 2006/2007 and 2009 burned a combined area of over 3 million hectares across largely forested upland regions (Victorian Department of Sustainability and Environment, 2009). In response to recent fires there has been an increased focus on all aspects of fire and forest management in Australia, from which the potential impact of wildfire on the quality of water supplies has emerged as one area of concern (Ellis et al., 2004; Parliament of Victoria, 2008; Victorian Bushfires Royal Commission, 2009). Similarly, wildfire events elsewhere have drawn attention to the risk to water supplies in burned forest regions (Scott et al., 1998; Moody and Martin, 2001, 2004; Ice et al., 2004; Rhoads et al., 2006).

Forest catchments are an important source of potable water to communities around the world and in many cases are managed specifically for this purpose (Dudley and Stolton, 2003). It has been reported that for the world's 105 largest cities (top 25 from the Americas, Africa, Asia, Europe and five from Australia) approximately one third (33 cities) obtained a significant proportion of their drinking water from protected forest areas, with a further five cities obtaining water from distant catchment areas which also contained protected forest (Dudley and Stolton, 2003). Furthermore, forest areas are estimated to contribute two-thirds of the freshwater supply in the United States (Committee on Hydrologic Impacts of Forest Management, 2008). Although a number of these forested water supply catchments (such as wet tropical rainforest areas) may be less fire-prone, other catchment areas are particularly susceptible to wildfire events, such as in south-eastern Australia and the western United States. For example, in south-eastern Australia wildfires have burned forested reservoir catchments which supply potable water to Sydney (2001 wildfire), Canberra (2003), Adelaide (2007), Melbourne (2009), as well as various regional towns.

An advantage of using largely undisturbed forest catchments for potable water supply is the generally high quality of water flowing from such regions (Neary et al., 2009). However, following wildfire, increased erosion rates and changes to runoff generation and sources of pollutants can greatly increase the amounts of sediment, nutrients and other constituents delivered to streams and reservoirs (e.g. Moody and Martin, 2001, 2009; Reneau et al., 2007; Sheridan et al., 2007b; Lane et al., 2008; Wilkinson et al., 2009), potentially resulting in water supplies that are unfit for consumption (White et al., 2006).

In this paper, we review changes to exports and concentrations of key physical and chemical constituents in water that may occur after wildfire in forest catchments. Wildfire effects on the generation and transfer of different constituents are considered in relation to post-fire rainfall and streamflow, erosion processes, and catchment controls reported in the literature. The review takes a particular focus on burned forest environments in south-eastern Australia, given the impact of recent wildfires on water supply catchments and the authors' access to reports and data for this region. The purpose of this review is not to outline the extent of published literature on the geomorphological and hydrological effects of wildfire, as this has been done previously (Shakesby and Doerr, 2006). Instead, we assemble information to provide an indication of the potential for water supply contamination by different constituents following wildfires. Compared to the expanding global research output on the hydrological and geomorphological effects of wildfire, published studies explicitly examining post-fire water quality in relation to drinking water supply, storage or treatment are limited (Moody and Martin, 2004; White et al., 2006).

The review is comprised of an initial section identifying key water quality constituents in forest catchments from a drinking water perspective. The next section examines post-fire exports and concentrations of individual constituents, with reference to hydrological and erosion processes contributing to observed changes after wildfire. The final section provides a summary of review findings and considers implications for water supply before outlining directions for future research.

2. Water quality constituents in forest catchments

This review focuses on a range of physical and chemical constituents that may be derived from forest environments and impact upon drinking water quality. These constituents are readily measurable and may be linked to catchment sources and delivery processes. Consideration of water-borne pathogens such as *Cryptosporidium*, *Giardia* and cyanobacteria is not included and routine monitoring for these specific pathogens is generally not recommended due to the difficulty and expense in their measurement (NHMRC, 2004). Sources of constituents in natural forest catchments include hillslopes, channels and floodplains, with additional inputs from disturbances such as access roads, forestry activities, prescribed burns, and wildfire. Major industrial or agricultural pollutant sources are excluded, with the exception of atmospheric inputs of mercury that have been recorded in forest areas. Forest fires may contribute additional pollutants (e.g. polycyclic aromatic hydrocarbons) that occur in otherwise negligible amounts or increase the store of pre-existing contaminants (such

as via ash deposits) which may be mobilised and delivered to streams. Ash is not identified as an individual water quality pollutant because it may contain various constituents but its contribution to post-fire water quality is considered in Section 3.2.

Key water quality constituents, the potential risks associated with each, and associated guideline values are outlined in Table 1. Guideline values were taken from the World Health Organisation (WHO) Guidelines for Drinking Water Quality (2008), and the Australian Drinking Water Guidelines (NHMRC, 2004) were used when a WHO guideline value was not given. Unless otherwise stated, guideline values are for the total amount of a constituent present, regardless of form (i.e. in solution or particle-associated). Total phosphorous (TP) and total nitrogen (TN) do not have drinking water guideline values and instead the Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines for freshwater aquatic ecosystems in south-eastern Australia were used for comparison with available post-fire data from this region (ANZECC, 2000). Although the ANZECC guidelines are not related to drinking water, they do provide a reference point for aquatic ecosystem health, with implications for eutrophication and algal blooms which may compromise drinking water quality.

The risks associated with different constituents range from aesthetic concerns (taste or appearance) to potential toxicity or carcinogenicity with prolonged exposure to elevated concentrations. For chemical constituents that present a health risk, calculation of guideline values is based on a Tolerable Daily Intake (TDI), which refers to the amount of substance in water and food per kilogram of body weight that may be ingested over a lifetime without appreciable health risk (WHO, 2008). In addition, body weight (default value of 60 kg; Australian guidelines use 70 kg), the proportion of TDI from drinking water (20% in the absence of exposure data), and a daily value for drinking water consumed (default value for an adult of 2 L) are also included (WHO, 2008). Uncertainty (safety) factors are used to account for interspecies (when transferring from animal studies to humans) and inter-individual variability in humans (both a factor of 10), with additional factors that may be applied covering inadequate databases as well as the nature and severity of effect (WHO, 2008).

Information on the health effects of short duration exposure to elevated chemical constituent concentrations is generally unavailable (NHMRC, 2004; WHO, 2008). Short-term consumption of water with concentrations exceeding guideline values may not necessarily present a health risk (depending on the chemical) assuming longer-term consumption does not exceed set levels (WHO, 2008). Generally, very high constituent concentrations following wildfires are short-lived, although elevated concentrations may persist for some time after post-fire rainfall events. Nonetheless, even exceeding guideline values in the short-term may be problematic for water authorities charged with responsibility for the provision of drinking water that conforms to set standards, particularly where treatment capacity may be insufficient to process polluted water, unaffected reserve supplies are inadequate, or the population is dependent upon a single water source that has been contaminated.

3. Wildfire effects on water quality

Research into wildfire impacts on water quality has tended to focus on suspended sediment and nutrients, with other constituents receiving less attention. Post-fire data for many of the constituents outlined in Table 1 is limited. For this reason, the focus on particular water quality constituents and their associated post-fire sources and delivery processes reflects the extent of available information on those constituents after wildfire. Furthermore, the likelihood of detecting periods when constituent concentrations exceed guideline values depends upon the sampling regime in relation to the rate and variability of constituent generation and the hydrological variability of the catchment system. In particular, the inclusion of storm event sampling may be critical to quantifying the magnitude and frequency of guideline exceedance for constituents in some systems, particularly in regions prone to high intensity rainfall events that generate rapid stormflow response. This can limit direct comparability between some studies and reported maximum concentration values should be considered in the context of both the sampling regime and the study environment.

3.1. Suspended sediment

Suspended sediment inputs to streams and reservoirs affect the colour and turbidity of water and may also transport various particle-associated contaminants (Horowitz and Elrick, 1987; Horowitz, 1991; Ongley et al., 1992). From a drinking water quality and treatment perspective, elevated suspended sediment concentrations may hinder detection of bacteria and viruses, promote bacterial growth from elevated levels of adsorbed nutrients, and limit effective disinfection (NHMRC, 2004). The strong tendency for many water quality constituents, particularly trace elements and P, to be bound to fine particles with low settling velocities necessitates a focus on fine sediment (Horowitz, 1991; Ongley et al., 1992). Furthermore, coarse material (>1 mm) flux has been shown to transport only a small fraction of TP exports after fire (Noske et al., 2010). As a result, consideration of post-fire bedload or coarse sediment flux is not included in this review.

Importantly, the transport of fine cohesive sediments through the fluvial system may be dominated by composite suspended sediment particles (flocs or aggregates) rather than primary particles (Droppo, 2001; Woodward and Walling, 2007). The structure of these composite particles incorporates microbial communities, organic and inorganic particles, and chemical constituents (Droppo, 2001), which may adversely affect water quality. Findings from burned forest environments indicate that soil aggregates exhibit significantly higher settling velocities than unburned particles of similar diameter, which has been attributed to an increase in burned aggregate density due to reduced organic content and pore space (Blake et al., 2007, 2009a). Furthermore, aggregation of clay particles into coarser composite particles has been attributed to the effect of soil heating (Dyrness and Youngberg, 1957; Giovannini et al., 1988; Giovannini and Lucchesi, 1997), thereby potentially increasing the concentration of contaminants bound to composite particles relative to primary particles of similar size (Blake et al., 2005). Potential consequences resulting from these fire effects on soil particles may include an increase in fine sediment storage and possibly the delayed release of contaminants after fire from degrading aggregates (Blake et al., 2009a).

Studies of wildfire effects on catchment suspended sediment exports and concentrations in streams and reservoirs have revealed a wide divergence of post-fire responses. This reflects the range of factors influencing post-fire erosion and sediment delivery (Shakesby and Doerr, 2006). However, much of the research to date into wildfire effects on sediment flux has focussed on plot to hillslope scales (e.g. Shakesby et al., 1993; Lavee et al., 1995; Prosser and Williams, 1998; Benavides-Solorio and MacDonald, 2001; Coelho et al., 2004; Dragovich and Morris, 2002; Cerdà and Lasanta, 2005; Campo et al., 2006; Sheridan et al., 2007a; Spigel and Robichaud, 2007; Smith and Dragovich, 2008; Blake et al., 2010), with fewer studies linking post-fire erosion response at this scale to downstream sediment delivery, sediment contributions from other sources, and catchment outputs (White and Wells, 1979; Moody and Martin, 2001; Blake et al., 2009b; Wilkinson et al., 2009). In this section, we present post-fire suspended sediment loads/yields

Table 1
Summary of water quality constituents from natural forest catchment sources that may occur in streams and reservoirs after wildfire[a].

Water quality constituent	Reason for risk	Sources	Concentration guideline value[b] (mg L^{-1} unless otherwise stated)	Treatment options	Examples of studies reporting concentrations/exports after wildfire
Aluminium (Al)	Neurotoxic	Leaching from soil and rock, aluminosilicates (insoluble)	No health-based value (0.2 for acid soluble Al for aesthetic reasons)	Flocculation and filtration. Note: Aluminium sulfate is used in water treatment as a coagulant	Gallaher et al. (2002)
Ammonia (NH$_3$)/ ammonium (NH$_4^+$)	Corrosion of copper pipes and fittings; food source for some microorganisms	Microbial metabolism, fertilisers and animal waste	0.5 (aesthetic limit)	Oxidation of ammonia to nitrate	Chessman (1986), Spencer and Hauer (1991), Bayley et al. (1992), Earl and Blinn (2003), and Bladon et al. (2008)
Arsenic (As)	Carcinogenic	Dissolution of minerals, industrial and mining effluent	0.01	Coagulation	Gallaher et al. (2002) and Leak et al. (2003)
Barium (Ba)	Vasoconstriction and peristalsis, convulsions and temporary paralysis	Soil and rock	0.7	Lime softening can remove more than 90%	Gallaher et al. (2002)
Chloride (Cl$^-$)	Aesthetic (taste and corrosion of pipes and fittings)	Dissolution of salt deposits	250 (aesthetic limit)	Removal not possible by conventional treatment processes, require distillation or reverse osmosis	Carignan et al. (2000), Ferreira et al. (2005), and Mast and Clow (2008)
Chromium (Cr)	Hexavalent chromium is carcinogenic, no evidence for carcinogenicity of trivalent chromium	Soils and rock; weathering and oxidation convert insoluble chromium oxide to soluble trivalent chromium	0.05 (for total chromium)	Coagulation/filtration	Gallaher et al. (2002) and Leak et al. (2003)
Copper (Cu)	Poisoning may occur at high concentrations, gastrointestinal symptoms at lower concentrations (3–5 mg L^{-1})	Soils and rock in the form of carbonate and sulphide minerals	2	Increase pH followed by coagulation and filtration	Gallaher et al. (2002)
Cyanide	Highly toxic, affects the thyroid and nervous system	Biomass burning is a major source of free cyanide (HCN and CN$^-$), as well as natural decomposition of some plants, and production by some microorganisms	0.07	No standard methods for the removal of cyanide from drinking water	Gallaher et al. (2002), Barber et al. (2003), and Crouch et al. (2006)
Iron (Fe)	Aesthetic issues (taste and colour) and staining of pipes and fittings	Soils and rock as oxide, sulphide, and carbonate minerals	0.3 (aesthetic limit)	Coagulation and filtration	Wasson et al. (2003), Townsend and Douglas (2004), and White et al. (2006)
Lead (Pb)	Toxic (affects the central nervous system)	Dissolution from natural sources	0.01	Coagulation or lime softening	Gallaher et al. (2002) and Leak et al. (2003)
Manganese (Mn)	Aesthetic (taste and staining of plumbing fixtures)	Soils and rock, exists as both soluble and insoluble compounds in divalent, tetravalent and heptavalent states	0.4	Convert soluble forms of Mn to insoluble precipitates and filter	Wasson et al. (2003), Townsend and Douglas (2004), and White et al. (2006)
Mercury (Hg)	Toxicity (kidneys). Methyl mercury may cause neurological disorders and mental disability	Atmospheric deposition from natural (e.g. volcanoes) and anthropogenic (e.g. coal-fired power plants) sources	WHO: 0.006 (inorganic Hg) Australian: 0.001 (total Hg)	Coagulation moderately effective for inorganic Hg; granular activated carbon effective for removing inorganic and organic Hg	Gallaher et al. (2002)
Nitrate (NO$_3^-$)/ nitrite (NO$_2^-$)	Toxicity associated with reduction of nitrate to nitrite, which affects the transport of oxygen to tissues	Nitrate: formation by oxidation of organic waste, nitrogen-fixing bacteria in soils, or lighting strikes. Nitrite (unstable): formation by reduction of nitrate in low oxygen water	NO$_3^-$: 50 (infants under 3 months) and 100 (adults and infants over 3 months) NO$_2^-$: 3	Conventional water treatment is not effective for nitrate removal	Chessman (1986), Spencer and Hauer (1991), Bayley et al. (1992), Hauer and Spencer (1998), Carignan et al. (2000), Ferreira et al. (2005), Bladon et al. (2008), Lane et al. (2008), and Mast and Clow (2008)
Total nitrogen (TN) [c]	Excessive algal growth and cyanobacterial blooms (eutrophication)	Soil and sediment associated N, organic material.	Not available for drinking water. Aquatic health trigger values for SE Australian freshwaters[c]: Upland river: 0.25 Lowland river: 0.5 Freshwater lake/reservoir: 0.35	Coagulation and filtration would remove most suspended solids and particle associated N	Spencer and Hauer (1991), Bayley et al. (1992), Townsend and Douglas (2004), Sheridan et al. (2007b), Bladon et al. (2008), Lane et al. (2008), and Mast and Clow (2008)

Table 1 *(continued)*

Water quality constituent	Reason for risk	Sources	Concentration guideline value[b] (mg L^{-1} unless otherwise stated)	Treatment options	Examples of studies reporting concentrations/exports after wildfire
Organic carbon (organic C)	Dissolved organic C (DOC) may contribute to the formation of disinfection by-products	Soils and sediment, organic material and ash	No WHO or Australian drinking water guideline available. 5 mg L^{-1} for DOC used in Ontario, Canada	Coagulation and filtration to remove particle-associated organic C	Carignan et al. (2000), McEachern et al. (2000), Minshall et al. (2001), Wasson et al. (2003), Petticrew et al. (2006), and Mast and Clow (2008)
Total phosphorous (TP)[c]	Excessive algal growth and cyanobacterial blooms (eutrophication)	Soil and sediment associated P, organic material, release of P from benthic sediments due to change in redox conditions	Not available for drinking water. TP aquatic health trigger values for SE Australian freshwaters[c]: Upland river: 0.02 Lowland river: 0.05 Freshwater lake/reservoir: 0.01	Coagulation and filtration would remove most suspended solids and particle associated P	Spencer and Hauer (1991), Bayley et al. (1992), Prepas et al. (2003), Townsend and Douglas (2004), Sheridan et al. (2007b), Bladon et al. (2008), Lane et al. (2008), Mast and Clow (2008), Blake et al. (2009b), and Gabos et al., (2001)
Polychlorinated dibenzodioxins and dibenzofurans (PCDD/Fs)[d]	Toxic, carcinogenic, undergo bioaccumulation, accumulate in soils and sediments	May be produced during forest fires. Estimated emissions of PCDD/Fs from wildfires of 0.5–28 µg per tonne of fuel burned[e]	Not available	Not available	
Polycyclic aromatic hydrocarbons (PAH)	Most toxicological studies examine benzo(a)pyrene (BaP), one type of PAH. PAHs are carcinogenic and mutagenic	Formed during incomplete combustion of organic matter, such as during forest fires or volcanic eruptions	0.0007 (for BaP); threshold values unavailable for other PAHs	Conventional treatment of coagulation, settling, and filtration are capable of reducing BaP concentrations and probably also for other PAHs	Olivella et al. (2006) and Vila-Escalé et al. (2007)
Sodium (Na$^+$)	Aesthetic (taste)	Dissolution of salt deposits	200 (aesthetic limit)	Removal not possible by conventional treatment processes, requires distillation or reverse osmosis	Carignan et al. (2000), Gallaher et al. (2002), and Ferreira et al. (2005)
Sulfate (SO$_4^{2-}$)	Aesthetic (taste), purgative effects may occur at high concentrations. It may contribute to increased corrosion of pipes.	Occurs naturally in various minerals. Aluminium sulfate is used as a flocculant in water treatment and copper sulfate is used as a control of blue-green algae in water storages	500 (250 for aesthetic reasons)	Most sulfate salts are highly soluble and cannot be removed from drinking water by conventional treatment processes, desalination methods are required	Carignan et al. (2000), Earl and Blinn (2003), Ferreira et al. (2005), and Mast and Clow (2008)
Total dissolved solids (TDS)	Aesthetic (taste). High TDS may also result in excessive corrosion of pipes and fittings.	TDS may comprise of inorganic salts, small amounts of organic material, clay particles, colloidal iron and manganese oxides and silica <0.45 µm (filter membrane)	500 (aesthetic limit)	TDS are difficult to remove, requires high energy/expensive reverse osmosis or distillation techniques	Chessman (1986), Lathrop (1994), and Gerla and Galloway (1998)
Turbidity/total suspended solids (TSS)	Health risk is associated with suspended material that may carry disease-causing microorganisms or particles that have adsorbed toxic organic or inorganic compounds	Turbidity in water results from the presence of suspended material, such as clay particles, plankton or other microorganisms	5 NTU (aesthetic limit)	Coagulation followed by filtration through granular media	Chessman (1986), Gerla and Galloway (1998), Gallaher et al. (2002), Lane et al. (2006), White et al. (2006), Wilkinson et al. (2006), Sheridan et al. (2007b) and Mast and Clow (2008)
Zinc (Zn)	Aesthetic (taste). Zn is an essential element for humans. Consumption of very high amounts may have gastrointestinal effects	Widely distributed in rocks	3 (aesthetic limit)	Coagulation using aluminium sulfate at pH 6.5–7 (30% removal) or lime softening at pH 9.5–10 (60% removal)	Gallaher et al. (2002)

[a] Table 1 was compiled from information provided by the World Health Organisation (WHO)'s Guidelines for Drinking-Water Quality (2008) and the Australian Drinking Water Guidelines (NHMRC, 2004).

[b] Concentration guideline values (as well as recommended values based on aesthetic concerns) are from WHO (2008), with NHMRC (2004) guidelines used in the absence of WHO guidelines (drinking water values for TP, TN and PCDD/Fs are not contained in these guidelines). Note that, unless otherwise stated, guideline values refer to the total amount of a substance present in a sample regardless of form (in solution or attached to suspended material).

[c] Information on TP and TN is from the Australian and New Zealand Environment and Conservation Council (ANZECC) guidelines (2000) for aquatic ecosystems in south-eastern Australia (chapter 3). The reported guidelines are for south-eastern Australian freshwater bodies and represent trigger values for adverse effects to aquatic ecosystems and not for human health or aesthetic effects.

[d] Information on PCDD/Fs is from the Australian Commonwealth Government's National Pollution Inventory (NPI) Fact sheet on PCDD/Fs: ⟨http://www.npi.gov.au/substances/polychlorinated-dioxins/index.html⟩.

[e] Sources of PCDD/Fs are from Environment Australia (2002).

Table 2
Summary of post-fire annual suspended sediment yields from catchment-scale studies in forest environments burned by wildfire.

References	Location	Catchment area (km²)	Method[a]	Annual suspended sediment yield in t ha⁻¹ year⁻¹ (and load in t) — First year after fire	Subsequent years after fire	Multiple increase relative to unburned yields (first year after fire)
Mayor et al. (2007)	Xortà Mountains, Eastern Spain	0.021	VS	0.65 (9.58); average sediment yield over 7 years		n/a
Blake et al. (2009b)	Blue Mountains, near Sydney, Australia	0.89	TB	0.58 (51.6)[b]	n/a	n/a
Scott (1993)	Southwestern Cape, South Africa	(a) 1.32 (b) 2.01	CM	(a) 5.2 (686) (b) 6.0 (1206)	n/a	n/a
Lane et al. (2006)	East Kiewa River, NE Victoria, Australia	(a) 1.36 (b) 2.44	CM	(a) 2.05 (280) (b) 0.88 (216)	2nd year: (a) 0.39 (56) (b) 0.35 (84)	8–9
Kunze and Stednick (2006)	Colorado Front Range, USA	(a) 2.2 (b) 3.9	RC	n/a	2nd year (summer): (a) 0.38 (84) b) 1.44 (562)	n/a
Reneau et al. (2007)	New Mexico, USA	16.6	VS[c]	9.0 (14,913)	2nd year: 5.2 (8637) 3rd year: 2.1 (3472) 4th year: 0.74 (1232) 5th year: 0.67 (1120)	106[b]
Moody and Martin (2001, 2004)	Colorado Front Range, USA	46.9[d]	VS[e]	50 (233,750)[e]	n/a	560
Lane et al. (2006)	West Kiewa River, NE Victoria, Australia	100.5	CM	0.29 (2890)	n/a	n/a
Petticrew et al. (2006)	Central British Columbia, Canada	135	CM	0.009 (115)[f]	n/a	1.7
Wilkinson et al. (2009)	Little River, Blue Mountains, Australia	183	RC	0.21 (3843)	2nd year: 1.02 (18,666)	104–250
Tomkins et al. (2007)	Nattai River, Blue Mountains, Australia	446	RC	1st year: 1968 wildfire: 0.74 (33,004) 2001/2002 wildfire: 0.017 (763)	2nd year: 1968 wildfire: 0.011 (508) 2001/2002 wildfire: 0.011 (486) 3rd year: 1968 wildfire: 0.106 (4741) 2001/2002 wildfire: 0.003 (120)	n/a
Sheridan et al. (2007b)	Victoria, Australia: (a) Ovens River (b) Tambo River-1 (c) Dargo River (d) Tambo River-2 (e) Mitta Mitta River (f) Kiewa River	(a) 495 (b) 523 (c) 676 (d) 895 (e) 1533 (f) 1655	OLE	1st year: (a) 0.83 (41,260) (b) 0.46 (24,147)[g] (c) 0.22 (14,854) (d) 0.37 (32,967) (e) 3.3 (511,559) (f) 0.11 (17,880)	2nd year: (a) 0.26 (12,678) (b) n/a (c) 0.30 (20,131) (d) 0.35 (31,386) (e) 0.54 (83,374) (f) n/a — 3rd year: (a) 0.05 (2588) (b) n/a (c) 0.40 (27,188) (d) 0.07 (6101) (e) 0.68 (104,170) (f) n/a	(a) 24 (b) 1459 (c) 21 (d) 171 (e) 168 (f) 1.3

[a] Method of sediment yield estimation: continuous monitoring of discharge and turbidity/TSS (CM); rating curve derived from measurements of discharge and TSS (RC); other load estimation techniques based on flow and TSS data (OLE); volumetric surveys of reservoirs converted to mass of sediment (VS); and sediment tracer budgeting (TB).

[b] Sediment yield estimate was based on ^{7}Be sediment budget (<63 μm fraction) for the first 3 months after wildfire (Blake et al., 2009b).

[c] The estimate of suspended sediment yield was based on the density of fine sediment (excluding ash) deposited in the reservoir (1.12 g cm⁻³), with the multiple increase value relative to the average annual unburned fine sediment input to the reservoir.

[d] The burn area of 46.9 km² (in two sub-catchments within a larger unburned river basin) was used to calculate the suspended sediment yield (determined by reservoir survey) based on the assessment that the burn dominated the post-fire sediment yield (Moody and Martin, 2001, 2004).

[e] Sediment volume was converted to mass using a density of 1.7 g cm⁻³ provided by the authors and the suspended sediment load was calculated from the difference between total load and bed load (Moody and Martin, 2001, 2004).

[f] Post-fire monitoring was for a 7 month period (covering the spring snowmelt) only (Petticrew et al., 2006).

[g] Suspended sediment yield data for the Tambo River (site 1) was for a 6 month period only (Sheridan et al., 2007b).

and suspended sediment concentration or turbidity data, with reference to reported erosion processes and hydrological drivers contributing to observed wildfire effects. Post-fire suspended sediment load, concentration and turbidity data inevitably incorporate an ash component (particularly in the first year after fire). Ash and its contribution to post-fire water quality are considered separately in the next section.

A summary of annual suspended sediment yields from catchment-scale studies in forest environments burned by wildfire is provided in Table 2. The yield data is presented in order of increasing catchment size (ranging from 0.021 to 1655 km^2) and, where available, for multiple years after the burn. The magnitude of suspended sediment yields in the first year after fire vary substantially, ranging from 0.017 to 50 t ha^{-1} year^{-1}, with an approximate peak in yields for catchments ranging in area from 1 to 50 km^2. Likewise, estimates of multiple increases (i.e. post-fire yield increases as multiples of reported pre-fire or unburned control catchment yields) for the first year after fire reveal large variation, ranging from 1 to 1459 times estimates of unburned annual suspended sediment yields.

The uncertainties associated with the estimation of suspended sediment yields (and multiple increase values) can be large and have been reported by some authors (Sheridan et al., 2007b; Wilkinson et al., 2009). Measurement of sediment yields using different techniques may also hinder comparability (Table 2). To address this, we limited the comparison to catchment-scale studies that report suspended sediment (using flow-based estimation techniques) or fine sediment (generally <63 µm for studies using volumetric surveys or tracer techniques) yields for the first year after fire (except where otherwise stated). This excluded studies that only reported total sediment loads after fire because the suspended or fine sediment fraction of the total load was unknown. For the purpose of comparison, suspended sediment yields are presented in t ha^{-1} year^{-1}. However, it should be noted that such a measure can be misleading, given that sediment supply may be dominated by localised sources (e.g. channels) rather than distributed uniformly across a catchment (Smith, 2008; Moody and Martin, 2009).

The range in suspended sediment yields after fire reflects various factors, including rainfall patterns, catchment burn extent and severity, erosion processes, sediment sources (location and connectivity to major tributary streams), and scale effects (such as increased opportunities for sediment storage with increasing catchment size). Suspended sediment yields following the first year after fire generally decline as vegetation cover is re-established and fire impacts on soil and hillslope hydrological properties (e.g. changes to soil water repellency and hillslope surface roughness) decline to pre-fire levels (Prosser and Williams, 1998; Lane et al., 2006; Sheridan et al., 2007a; Reneau et al., 2007). Increases in annual suspended sediment yields in subsequent years may reflect large rainfall events impacting on partially-recovered catchments and/or stormflows in tributary streams remobilising sediment deposits from previous post-fire flow events (Moody and Martin, 2001; Mayor et al., 2007; Wilkinson et al., 2009).

The magnitude, intensity and frequency of post-fire rainfall and associated flow events are key drivers of erosion and sediment delivery in many burned catchments (Robichaud et al., 2007; Malmon et al., 2007; Cannon et al., 2008; Moody and Martin, 2009). The importance of rainfall patterns for explaining the large variation in post-fire suspended sediment yields is apparent from a number of the studies shown in Table 2. For example, the large sediment yield (50 t ha^{-1}; 560 times unburned) reported by Moody and Martin (2004) for the first year after fire was mostly attributed (estimated 80%) to channel erosion and incision of unchannelised drainage lines in response to flash floods generated by short duration, high intensity summer storms (peak storm

intensity 90 mm h^{-1}) (Moody and Martin, 2001). Short, intense storm events were also important for generating the large post-fire suspended sediment yields (9.0 t ha^{-1}; 106 times unburned) reported by Reneau et al. (2007) and have been identified as triggers of post-fire runoff-generated debris flows which produce high sediment yields (Cannon et al., 2008). In south-eastern Australia, a single summer storm (>2 h; 43 mm; peak 15-min intensity of 80 mm h^{-1}) accounted for 45% (127 t) and 47% (101 t) of the total suspended sediment yield in the first year after fire from two small wet *Eucalyptus* forest mountain catchments (Lane et al., 2006). The impact of a longer duration rainfall event (3 days; 243 mm) was also captured by Lane et al. (2006) and accounted for a further 34% and 29% of total first year suspended sediment yields from the two burned study catchments.

In contrast, Tomkins et al. (2007) observed a first year suspended sediment yield of only 0.017 t ha^{-1} from the Nattai River catchment (446 km^2; >50% burned), part of the Lake Burragorang reservoir catchment (the main water supply of Sydney, Australia), during a period of below average rainfall following wildfire in 2001/2002. Reservoir sedimentation in the first 12 months after that fire was estimated to account for only 1–1.6% of total sedimentation since the reservoir was completed in 1960 (Wilkinson et al., 2007). Despite this, erosion estimates for burned hillslopes within the Nattai catchment ranged from 50 to 100 t ha^{-1} 5 months after the fire for areas of low to extreme fire severities (Shakesby and Doerr, 2006), revealing the extent to which colluvial and in-channel storage attenuated downstream suspended sediment delivery to the reservoir (Wilkinson et al., 2009). Transport limitation can be an important restriction on sediment delivery to streams and catchment outlets (Moody and Martin, 2001; Petticrew et al., 2006). In-channel storage of post-fire sediment deposits may result in lagged suspended sediment yield responses which are dependent upon the timing and magnitude of subsequent discharge events to remobilise these stores (Wilkinson et al., 2009). The increased stormflow response to rainfall events that may occur after fire also declines with post-fire recovery, thereby potentially reducing the capacity of subsequent flows to remove stored sediment and increasing residence times (Moody and Martin, 2001).

Reporting of post-fire suspended sediment concentrations (measured by the concentration of total suspended solids; TSS) or turbidity in streams and reservoirs is limited. Most studies included in Table 3 are from south-eastern Australia, which reflects both patterns in data reporting and the impact of multiple large wildfire events in this region. We focus on maximum values in order to quantify the effect of wildfire on TSS/turbidity (relative to pre-fire or unburned control maximums) and identify instances of guideline exceedance. This approach requires consideration of stream sampling regimes to assess the likelihood that sampling will have captured the brief periods of elevated TSS/turbidity, particularly during stormflows. Failure to reasonably characterise TSS/turbidity across a range of flows may result in underestimation of the risk to water quality, which is likely when routine weekly to monthly sampling is the only source of TSS/turbidity information for streams that respond rapidly to rainfall (Richards and Holloway, 1987; Walling et al., 1992; Robertson and Roerish, 1999). Such streams require additional storm-based sampling across event hydrographs to improve both load estimation and to characterise the magnitude of TSS/turbidity change (Robertson and Roerish, 1999).

Post-fire turbidity values for streams were less frequently reported than TSS but were more readily available for reservoirs. This presents a challenge, given that turbidity is the primary measure of the particulate content of water samples in drinking water guidelines (NHMRC, 2004; WHO, 2008), yet TSS is more commonly reported in the literature. This focus on TSS probably reflects the greater level of comparability and precision of TSS measurements

Table 3
Maximum total suspended solids (TSS) concentration and turbidity data from streams and reservoirs in forest environments burned by wildfire.

References	Location	Catchment area (km²) (B: burned and UB: unburned)	Post-fire sampling regime and duration of sampling	Pre-fire maximum TSS concentration (mg L^{-1})	Post-fire maximums			
					TSS concentration (mg L^{-1})			Turbidity (NTU)
Streams								
Gallaher et al., 2002	Jemez Mountains and Pajarito Plateau, New Mexico, USA	n/a (40 sites)	Storm events; 5 months	43,140	76,000 (12 out of 272 samples exceeded the pre-fire maximum)			n/a
Lane et al., 2006	East Kiewa River, NE Victoria, Australia	1.36, 2.44	Weekly to fortnightly and storm events; 3 years	n/a	47,152			n/a
Malmon et al., 2007[a]	Pueblo Canyon, Jemez Mountains and Pajarito Plateau, New Mexico, USA	22	Storm events during summer over 3 years	<15,000	~500,000			n/a
Gerla and Galloway, 1998	Yellowstone National Park, Wyoming, USA	B: 64 UB: 49	Weekly to monthly during spring to autumn; 5 years	n/a	n/a			B: 90 UB: 85
Mast and Clow, 2008	Glacier National Park, NW Montana	B: 96.4 UB: 47.3	2× weekly (snowmelt) to monthly; 4 years	n/a	B: 147 UB: 1497			n/a
Brown, 1972	Snowy Mountains, NSW, Australia	(a) 27 (b) 141	Low flow with some storm events; 5 years	(a) 7052 (b) 334	(a) 143,000 (b) 112,000			n/a
Wilkinson et al., 2006	Little River, Blue Mountains, Australia	183	Low flow and storm events; 12 months	82	2646			n/a
Chessman, 1986	Victoria, SE Australia	B: 40–750 (11 sites) UB: 110	Low flow and storm events; 3 months	n/a	B: 11–2300 UB: 163			130
Leak et al., 2003	Buckland River, Victoria, SE Australia	322	Single flow event	n/a	59,000			129,000
Sheridan et al., 2007b	Victoria, Australia		Low flow and storm events; 3 years	Event flow:	1st year	2nd year	3rd year	n/a
	(a) Ovens River	(a) 495		(a) 280	(a) 1700	(a) 330	(a) 83	
	(b) Tambo River-1	(b) 523		(b) 9	(b) 59,000	(b) n/a	(b) n/a	
	(c) Dargo River	(c) 676		(c) 34	(c) 1600	(c) 990	(c) 930	
	(d) Tambo River-2	(d) 895		(d) 90	(d) 15,000	(d) 9100	(d) 2000	
	(e) Mitta Mitta River	(e) 1533		(e) 260	(e) 43,000	(e) 2200	(e) 2000	
	(f) Kiewa River	(f) 1655		(f) 63	(F) 140	(f) n/a	(f) n/a	

(continued on next page)

Table 3 (continued)

References	Location	Catchment area (km²)/reservoir capacity (GL)	Post-fire sampling regime and duration of sampling	Pre-fire maximum turbidity (NTU)	Post-fire maximums	
					TSS concentration (mg L⁻¹)	Turbidity (NTU)
Reservoirs						
White et al., 2006	Bendora Reservoir, Cotter River, ACT, Australia	91.5/11.5	Not reported; 3 years	Previous maximums: –118 at bottom (>30 m depth) after 1983 wildfire and –18 at 15 m (above thermocline) after a 190 mm rain event in January 1995	n/a	3000 at bottom after January 2003 wildfire and 2 storm events (each approx. 60 mm) 1 month after fires
Goulburn-Murray Water/ Victorian Water Resources Data Warehouse[b,c]	(a) Lake William Hovel (b) Lake Buffalo (c) Lake Glenmaggie (d) Lake Eildon Victoria, Australia	(a) 332/13.7 (b) 1150/23.9 (c) 1891/178 (d) 3885/3334	Fortnightly to monthly; 2 years	(a) 4 (0%)[d] (b) 18 (16%)[d] (c) 130 (74%)[d] (d) 7 (2%)[d] (maximums prior to wildfire in 2006/2007)	n/a	(a) 5.7 (3.4%)[d] (b) 20 (22%)[d] (c) 1398 (100%)[d] (d) 16 (11%)[d]
Alexander et al., 2004	Dartmouth Reservoir, NE Victoria, Australia	3611/3900	5 sampling intervals; 7 months	1 (surface)	<4 (all depths to 100 m)[e]	5 (surface)

[a] Maximum pre and post-fire TSS concentrations were interpreted from figures presented by Malmon et al. (2007). The actual maximum values were not stated.

[b] Data was provided by Goulburn-Murray Water (L. Mattner, pers. comm., 18 May 2009) for Lakes William Hovel (LWH), Buffalo (LB) and Eildon (LE), while the Victorian Water Resources Data Warehouse (http://www.vicwaterdata.net/vicwaterdata/home.aspx) was accessed to obtain data for Lake Glenmaggie (LG).

[c] Turbidity data prior to the 2006/2007 wildfire was available for 2003–2006 for LWH, LB and LE, whereas a longer record (1978–2006) was available for LG which included the effect of a wildfire that burned 17% of the catchment in 1998. Note that the catchment of LB was burned by wildfire in 2003 (35% of the catchment area) and again in 2006/2007 (near complete burn). Reservoir sampling was at off-take points or outflows.

[d] Percentages indicate the proportion of turbidity values exceeding 5 NTU (guideline value) relative to the total number of pre and post-2006/2007 wildfire measurements.

[e] Data from depth-stratified sampling located near the dam wall (Alexander et al., 2004).

and the limited transferability of turbidity–TSS relationships between different catchments. Durations of elevated TSS/turbidity levels in streams after wildfire were also generally not reported. While periods of maximum TSS/turbidity may occur briefly during stormflow events, elevated levels may persist beyond initial event timescales, particularly where there are large post-fire inputs of sediment to streams leading to in-channel storage and remobilisation by subsequent flow events (e.g. Lyon and O'Connor, 2008). Large storage reservoirs are more likely to reduce the magnitude and rate of change of TSS/turbidity inputs from tributary streams, resulting in generally lower peak TSS/turbidities that may persist for longer, with potential implications for recovery times and water supply restrictions (White et al., 2006). In contrast to stream sampling, fixed-interval measurements in large reservoirs may better capture the more gradual changes in TSS/turbidity resulting from the attenuating effect of these large water bodies on variable turbid inflows.

Maximum reported TSS concentrations measured during stormflows in the first year after fire range from 11 to $\sim$500,000 mg L^{-1} for streams with catchments varying in size from 1.36 to 1655 km^2 (Table 3). The highest reported maximum TSS values after fire occurred during flash floods in a semi-arid ephemeral stream located in New Mexico (Malmon et al., 2007). In all studies, with the exception of that by Mast and Clow (2008), TSS maximums after fire exceeded unburned (or pre-fire) maximums. Mast and Clow (2008) found that wildfire did not affect TSS concentrations in a subalpine conifer forest catchment in Montana, USA, suggesting that sediment yields from subalpine streams may be less affected by fire than lower elevation streams because of a slower rate of water release during spring snowmelt (the main sediment transport period).

The highest TSS concentrations in streams after wildfire generally occurred in response to erosion events triggered by intense summer storms (Gallaher et al., 2002; Leak et al., 2003; Lane et al., 2006; Malmon et al., 2007; Sheridan et al., 2007b). For example, following wildfire in 2003, runoff-generated debris flows that scoured channels to bedrock (Fig. 1) were produced from a cluster of small catchments (<200 ha) by a very high intensity, short duration storm event (estimated at 150 mm in 1 h) in the upper Buckland River, north-eastern Victoria (Lyon and O'Connor, 2008; Tryhorn et al., 2008; Nyman et al., in press). A pulse of highly turbid water, with a peak TSS concentration of 59,000 mg L^{-1} (129,000 NTU) measured 30 km downstream (Leak et al., 2003), travelled down the Buckland and into the Ovens River. Subsequently, turbidity levels in the Ovens River remained well in excess of the guideline value (5 NTU), with the peak turbidity (2370 NTU)

at Wangaratta (located over 150 km downstream from the source) recorded 12 days after the event (Leak et al., 2003; Lyon and O'Connor, 2008). Similarly, following wildfire in summer 2006/2007, debris flows occurred in response to high intensity rainfall in small tributary sub-catchments of the upper Macalister River catchment (1891 km^2; 90% burned) in eastern Victoria (Nyman et al., in press). The debris flows delivered large quantities of ash and sediment to the Macalister River, which impacted on water quality in Lake Glenmaggie (located 60 km downstream). The maximum TSS and turbidity recorded at the Glenmaggie Dam outflow after the debris flows was 280 mg L^{-1} and 1398 NTU, respectively. Mean pre-fire monthly turbidity at the outflow for the period 1978–2006 was 10.7 NTU, whereas for the first year after the fire it was 29 times greater at 306 NTU. This included the impact of both the debris flows and large flood events in June (estimated at >100 years recurrence interval) and November 2007 (Parliament of Victoria, 2008).

Further examples of impacts on TSS and turbidity levels in reservoirs may be identified following the 2003 and 2006/2007 wildfires in south-eastern Australia. White et al. (2006) reported an increase in turbidity in the Bendora reservoir (supplying drinking water to Canberra) on the Cotter River (482 km^2; 98% burned) following the 2003 wildfire. This was 30 times the previously recorded maximum turbidity (after wildfire in 1983 that burned 12% of the upper Cotter catchment). White et al. (2006) also reported that 5 NTU (at 3 m depth) was exceeded 2% of the time during the pre-fire period (1967–2002) and 10% during the post-fire period (2003–2005). Water quality in the reservoir returned to near pre-fire conditions within 2 years of the fire. Examination of data for four reservoirs with catchments burned during the 2006/2007 wildfire in Victoria revealed all experienced a post-fire increase in the proportion of samples exceeding 5 NTU (Table 3). The largest increase occurred in Lake Glenmaggie, rising from 74% of samples before the fire (1978–2006) to 100% (2007–2009) after the fire.

Large water quality impacts in reservoirs do not automatically result from inflows with high TSS/turbidity. This was apparent in the case of the large Dartmouth reservoir (3900 GL capacity) in north-east Victoria, with a 3611 km^2 catchment that was 95% burned in 2003. Only a small change in turbidity was recorded near the dam wall and TSS remained <4 mg L^{-1} at all depths measured up to 12 months after the fire (Alexander et al., 2004). In contrast, the maximum TSS concentration recorded in the Mitta Mitta River (the main tributary of Dartmouth) was 43,000 mg L^{-1}, while an estimated suspended sediment load of 511,559 t (approximately 168 times pre-fire yields) passed the nearest gauging site 35 km upstream of the reservoir (Sheridan et al., 2007b). This contrast between inflows and the reservoir water quality near the dam wall probably reflects the considerable capacity for a reservoir of this size to mitigate impacts on water quality from contaminated inflows.

3.2. Ash

Ash deposits on hillslopes after wildfire may form a large store of particulate carbon (charcoal, black carbon) and contain elevated concentrations of various nutrients, trace elements, as well as other potential contaminants (Amiro et al., 1996; Johansen et al., 2003; Goforth et al., 2005). The ash layer from a severe burn in forest areas with high fuel loads may be 2–10 cm thick and equate to 1–5 kg m^{-2} (Doerr et al., 2008), with depths of up to 20 cm having been reported (Gabet and Sternberg, 2008). The low density ash may be readily entrained by overland flow and result in water quality impacts soon after fire, with the potential for rapid depletion of the hillslope ash store (Reneau et al., 2007). Hillslope ash beds have also been identified as a factor in post-fire changes to hillslope runoff generation and erodibility, which has implications

Fig. 1. Debris flow-scoured channel in a tributary of the upper Buckland catchment, eastern Victoria, following wildfire in 2003. Note the person standing in the middle of the channel for scale (photo provided by Gary Sheridan).

for the magnitude of water quality impacts. For example, reduced soil infiltration rates have been attributed to ash clogging of soil macropores and the formation of surface crusts (Mallik et al., 1984; Neary et al., 2005; Onda et al., 2008). Following sufficient rainfall and saturation of the ash layer, it has been found that the transport capacity of runoff is enhanced due to increased fluid density with incorporation of ash into the flow. This may lead to a positive feedback mechanism of increased ash entrainment and possibly the generation of progressively bulked debris flows with high erosion rates and the potential for large downstream water quality impacts (Cannon, 2001; Gabet and Sternberg, 2008).

The store of constituents in burned forest ash and the effect of ash on water quality have received only limited attention. A difficulty in assessing the chemical composition of ash is the high level of variability, which reflects the type of vegetation and part of the plant (bark, wood or leaves) burned, climate, soil type, and combustion conditions (Someshwar, 1996; Demeyer et al., 2001). For this reason, examples are presented only to give an indication of the type and range of constituents that may be present in ash. In a review of the properties of wood ash (white ash), Demeyer et al. (2001) reported that calcium carbonate ($CaCO_3$) is the major compound of ash along with other carbonates and oxides of metals and silica. Wood ash is highly alkaline, increasing soil pH by up to 3 pH units immediately after burning compared to unburned soils (Ulery et al., 1993). Someshwar (1996) reported a median concentration of P in bark ash of 1.57% (15,700 mg kg^{-1}), while Ferreira et al. (2005) found 8900 mg kg^{-1} of P in ash under burned *Pinus pinaster* stands in Portugal. Khanna et al. (1994) reported P and N concentrations in ash from *Eucalyptus* litter of 160–12,000 and 300–14,000 mg kg^{-1}, respectively. Concentrations of organic C and N in ash will depend on the extent of combustion, with dark ash formed by incomplete combustion (Demeyer et al., 2001; Goforth et al., 2005). For example, Goforth et al. (2005) reported average organic C and N concentrations in dark ash that were 7–17 and 5.2–10.5 times greater than in white ash, respectively, for burned conifer forest and pine–oak woodland in southern California. The concentrations of organic C and N in white or dark ash were 3900–66,300 mg kg^{-1} and 400–4200 mg kg^{-1}, respectively (Goforth et al., 2005).

Microelement concentrations in ash may also vary greatly. Fe is the most abundantly present and is probably part of the structural framework of ash (Someshwar, 1996; Demeyer et al., 2001). Ferreira et al. (2005) reported concentrations of Fe (1466 mg kg^{-1}), Mn (2570 mg kg^{-1}), Zn (201 mg kg^{-1}), and Cu (57 mg kg^{-1}) for pine ash. Higher maximum concentrations of Fe (600–10,000 mg kg^{-1}), Mn (60–100,000 mg kg^{-1}), Zn (20–370 mg kg^{-1}) and Cu (12–340 mg kg^{-1}), were recorded for ash formed by burning *Eucalyptus* litter (Khanna et al., 1994). These authors also reported Al and Pb concentrations in ash of 1000–18,000 and 4–152 mg kg^{-1}, respectively. Someshwar (1996) measured mean levels of Hg, Cd and As in wood ash and reported concentrations of <0.5, <10, and 23 mg kg^{-1}. The solubility of various constituents was examined by Plumlee et al. (2007), who undertook water leachate tests (1:20 ash to deionized water) on ash samples collected after wildfires in chaparral vegetation in southern California. These authors reported mean solution concentrations (mg L^{-1}) for Cl$^-$ (32), NO$_3^-$ (4.1), and SO$_4^{2-}$ (179), with low (<1 mg L^{-1}) concentrations of P, Al, Ba, Cu and Fe and very low (i.e. <0.005 mg L^{-1}) concentrations of As, Cr, Hg, Mn, Pb, and Zn.

Ash can form a significant component of suspended material flux within the first year after fire, with the rate of removal of ash from hillslopes dependent upon the erosivity of post-fire rainfall and wind events (Reneau et al., 2007; Woods and Balfour, 2008). For example, Cerdà and Doerr (2008) reported that 153 mm of rainfall over a 6 day period removed most of a 36 mm ash layer only 3 weeks after a high severity fire in eastern Spain.

At the catchment-scale, the ash yield in the first year after fire was 2.1 t ha^{-1} (3567 t or 19% of fine sediment exported) for a mixed conifer catchment (16.6 km^2) in New Mexico, which accounted for >90% of total ash exports (Reneau et al., 2007). Even where only a small increase in post-fire mineral sediment delivery occurred, significant increases in the export and in-channel storage of organic material (associated with formation of in-stream biofilms and possible delivery of black carbon from ash washed off hillslopes) were observed in British Columbia in the first 15 months after fire (Petticrew et al., 2006).

The specific effect of ash on stream water quality is difficult to quantify given that the contribution of ash is rarely distinguished from that of mineral sediment delivered to streams. An exception to this is a study by Earl and Blinn (2003) in burned forest within the Gila River catchment, New Mexico. This study involved both an experimental ash input (1140 L ash slurry delivered over a 1.25 h period) to a first-order stream as well as monitoring of streams in burned catchments. The experimental ash input resulted in an immediate change to water quality, with peak concentrations (in mg L^{-1}) exceeding pre-ash concentrations for various ions, including NH$_3$/NH$_4^+$ (1.35 and pre-ash: below detectable limit), NO$_3^-$ (0.08 and pre-ash: below detectable limit), and SO$_4^{2-}$ (50.5 and pre-ash: 15.8). Following the ash experiment, most parameters returned to levels similar to the reference stream within 24 h and concentrations in burned catchment streams returned to pre-fire levels within 4 months (Earl and Blinn, 2003). Atmospheric deposition of ash may also result in short-term impacts on stream water quality (Spencer and Hauer, 1991; Lathrop, 1994; Hauer and Spencer, 1998). Notably, Spencer and Hauer (1991) reported that TP and TN concentrations increased 5–60 times over background levels within two days of a wildfire in north-west Montana. Peak concentrations of 0.21 mg L^{-1} (TP), 0.35 mg L^{-1} (TN), 0.06 mg L^{-1} (NO$_3^-$), and 0.26 mg L^{-1} (NH$_3$/NH$_4^+$) were recorded (below drinking water guideline values for NO$_3^-$ and NH$_3$/NH$_4^+$), with concentrations declining to background levels within several days to weeks after the fire.

3.3. Nitrogen and phosphorous

Increased exports and concentrations of N and P in various forms following wildfire can be problematic for managers of water supply catchments. Elevated concentrations of NO$_3^-$ and NO$_2^-$ present a potential risk to human health primarily through reduction of NO$_3^-$ to NO$_2^-$, which may affect oxygen transport in red blood cells, while high concentrations of NH$_3$/NH$_4^+$ may corrode copper pipes and fittings. N and P are limiting nutrients for growth of aquatic plants, algae and cyanobacteria (blue-green algae) in water bodies. Eutrophication increases the risk of potentially toxic blooms, with implications for human health, aesthetics (taste, odour and colour), and aquatic ecosystem function (NHMRC, 2004; Drewry et al., 2006). Post-fire increases in stream exports and concentrations of N and P may result from atmospheric and runoff inputs of ash, as well as soil erosion and remobilisation of sediment stored in colluvial deposits, channels and floodplains. Studies do not distinguish ash, soil, and remobilised sediment contributions to fluxes of these nutrients, and for this reason we do not partition N and P loads as ash or sediment-derived.

Burning can result in changes to soil N and P, which may affect post-fire exports and concentrations of these constituents in runoff and streams. Wildfire may cause a very substantial reduction of N (92%) and P (76%) stores in forest floor surface organic material (O horizon) due to combustion and volatilization, with smaller reductions (N 31–51%) recorded under lower intensity burns (Murphy et al., 2006a,b). Burning and heating may reduce surface soil TN (Chambers and Attiwill, 1994; Murphy et al., 2006b; Baird et al., 1999), while increasing NH$_4$–N (from combustion), which tends

Table 4

Summary of post-fire exports of phosphorous [total P (TP), particulate P (PP)] and nitrogen [total N (TN), NO_3^-, NH_3/NH_4^+] in various forms from plot and catchment-scale studies in forest environments burned by wildfire.

References	Location	Vegetation	Sampling regime	Burnt plot (m²) or catchment area(s) (km²)	P or N form	First year export (kg ha⁻¹ year⁻¹) after wildfire (multiple increases over pre-fire or unburned control) P	First year export (kg ha⁻¹ year⁻¹) after wildfire (multiple increases over pre-fire or unburned control) N
Plot-scale							
Thomas et al., 2000	Águeda River, central Portugal	Eucalyptus globulus plantation	Storm events	16	NO_3^-	n/a	0.75–1.4 (3–5×)[a]
Ferreira et al., 2005	Águeda River, central Portugal	Pinus pinaster plantation	Storm events	16	NO_3^-	n/a	0.49 (29×)[b]
Spigel and Robichaud, 2007	Bitterroot National Forest, west-central Montana, USA	Ponderosa pine/Douglas-fire	Storm events	100	TN	n/a	Mean: 48 Range: 1–100[c]
Catchment-scale							
Bayley et al., 1992	North-west Ontario, Canada	Boreal pine forest	Weekly (ice-free period, May–October)	0.12, 0.56, 1.7	TP NO_3^- NH_3/NH_4^+ TN	TP: 0.03–0.13 (1.4×)[d]	NO_3^-: 0.04–0.42 (5.9×)[d] NH_3/NH_4^+: 0.05–0.11 (1.9×)[d] TN: 1.14–2.05 (1.6×)[d]
Ferreira et al., 2005	Águeda River, central Portugal	Pinus pinaster plantation	Weekly and storm events	1.1	NO_3^-	n/a	2.5 (250×)[b]
Lane et al., 2008	East Kiewa River, NE Victoria, Australia	Wet Eucalyptus forest	Weekly–fortnightly and storm events	1.36, 2.44	TP NO_3^- TN	1.67 (4–5×)	NO_3^-: 13.0[e] TN: 15.3 (6×)
Bladon et al., 2008	Rocky Mountain region, SW Alberta, Canada	Montane to subalpine coniferous forests	Snowmelt (10–14 days), winter (1–2 monthly) and storm events	3.6, 7.1, 8.2	NO_3^- NH_3/NH_4^+ TN	n/a	NO_3^-: 3.4–12.4 (12.7×)[f] NH_3/NH_4^+: 0.06–0.15 (1.9×)[f] TN: 8.2–27.1 (8.8×)[f]
Blake et al., 2009b	Blue Mountains, near Sydney, Australia	Dry Eucalyptus forest	n/a	0.89	PP	0.49 [g]	n/a
Prepas et al., 2003	Central Alberta, Canada	Mixed species boreal forest	Pre-fire: 2× daily Post-fire: 1–2 times weekly and storm events	248	PP TP	PP: 0.04 (1.5×)[h] TP: 0.06 (1.3×)[h]	n/a
Townsend and Douglas, 2004	Kakadu National Park, Northern Australia	Tropical savannah, open dry *Eucalyptus* forest	Every 3 days and storm events	6.6	TP TN	0.03 (1.8×)	1.2 (1.6×)
Mast and Clow, 2008	Glacier National Park, NW Montana	Subalpine coniferous forest	2× weekly (during snowmelt) to monthly	96.4	TP TN	0.03 (0.3×)	2.95 (2×)
Sheridan et al., 2007b	(a) Ovens River (b) Tambo River-1 (c) Dargo River (d) Tambo River-2 (e) Mitta Mitta River (f) Kiewa River Victoria, SE Australia	Dry and wet Eucalyptus forests, subalpine woodland	Pre-fire: monthly Post-fire: fixed interval and storm events	(a) 495 (b) 523 (c) 676 (d) 895 (e) 1533 (f) 1655	TP TN	(a) 1.1 (9×) (b) 0.6 (431×)[i] (c) 0.41 (10×) (d) 0.26 (30×) (e) 3.2 (37×) (f) 0.13 (1×)	(a) 8.2 (9×) (b) 2.7 (94×)[i] (c) 2.8 (7×) (d) 1.4 (12×) (e) 14.1 (20×) (f) 2.2 (1×)

[a] NO_3^- exports (converted from NO_3–N) were based on two plots for the first year after fire, with the multiple increase relative to the mean annual export from an unburnt control plot (Thomas et al., 2000).
[b] Monitoring undertaken for 14 months after the wildfire (Ferreira et al., 2005).
[c] TN exports from plots were based on the mean and range of data reported from four sites (six plots at each site) (Spigel and Robichaud, 2007).
[d] Reported post-fire values are 3 years averages from three catchments, with pre-fire data collected over 2-years. Multiple increases are based on mean values from the three catchments (Bayley et al., 1992).
[e] Lane et al. (2008) reported NO_3–N exports, which have been converted to NO_3^- for comparison.
[f] Multiple increases are based on mean export values from three burnt and two unburnt control catchments (Bladon et al., 2008).
[g] PP export based on a sediment budget (<63 µm fraction) derived using ^7Be for the first 3 months after wildfire (Blake et al., 2009a,b).
[h] Exports were calculated from the sum of baseflow and summer storm period export data (Prepas et al., 2003).
[i] TP and TN data for the Tambo River (site 1) are for a 6 month period only (Sheridan et al., 2007b).

to decline in the months after the fire as NO_3–N increases from nitrification of the additional NH_4–N (Khanna and Raison, 1986; Weston and Attiwill, 1990; Covington and Sackett, 1992). Soil P generally does not decrease after wildfire because the volatilization temperature of P (>550 °C) is much greater than N (200 °C) (Raison, 1979; Hernandez et al., 1997; Murphy et al., 2006b). However, fire may increase soil concentrations of available P (Chambers and Attiwill, 1994; Cade-Menun et al., 2000), while up to 20% of particulate P in burned sediment has been shown to be potentially bioavailable (Blake et al., 2010). This latter point has implications for water quality and potential eutrophication of water bodies with increased post-fire erosion and downstream sediment flux after fire.

The reported effect of wildfire on exports of N and P from different forest environments varies markedly. Fire effects on stream exports of TN and TP range from a small decline or minor increase (multiple change of 0.3–2 times unburned exports) to substantial increases (20–431 times unburned) from large burned catchments in south-eastern Australia (Bayley et al., 1992; Townsend and Douglas, 2004; Sheridan et al., 2007b; Mast and Clow, 2008). TN and TP stream exports in the first year after fire were 1.1–27 and 0.03–3.2 kg ha^{-1} year^{-1}, respectively (Table 4). Higher exports of TN have been reported at the plot-scale than at the catchment-scale, ranging from 1 to 100 kg ha^{-1} year^{-1} for the first year after fire in central-west Montana (Spigel and Robichaud, 2007). NO_3^- exports range from 0.49–1.4 kg ha^{-1} year^{-1} at the plot-scale in Eucalyptus and pine plantations to 0.04–13.0 kg ha^{-1} year^{-1} at the catchment-scale, representing a multiple increase of 3–250 times unburned exports (Table 4). Data on post-fire exports of NH_3/NH_4^+ is limited, with two studies from conifer forest catchments in Canada reporting low exports (0.05–0.15 kg ha^{-1} year^{-1}) and a multiple increase of 1.9 times unburned levels in both cases (Bayley et al., 1992; Bladon et al., 2008). The range in catchment exports of N and P may reflect differences in burn area and severity, erosion processes, the extent of delivery to streams, soil and forest vegetation types, as well as rates of pre-fire atmospheric deposition (Baird et al., 1999; Stephens et al., 2004; Townsend and Douglas, 2004; Mast and Clow, 2008).

The available data suggests that the particle-associated fraction may dominate exports of N (69%) and P (77–94%) during stormflow events in the first year after fire (Prepas et al., 2003; Lane et al., 2008). In contrast, Mast and Clow (2008) reported that 90% of

the TN exported from a burned subalpine coniferous forest catchment in north-west Montana was in dissolved form and similar to the unburned control catchment. However, that fire did not change suspended sediment concentrations, thereby minimising the effect of the fire on particulate N export. The contribution of particulate forms of N and P may be expected to decline with time since fire as vegetation recovery reduces erosion rates and runoff generation leading to a decline in sediment delivery (Lane et al., 2008).

Studies reporting concentrations of N and P in various forms in streams and reservoirs after wildfire are more extensive than those estimating exports (e.g. Wright, 1976; Chessman, 1986; Gerla and Galloway, 1998; Hauer and Spencer, 1998; Townsend and Douglas, 2000; Minshall et al., 2001; Gallaher et al., 2002; Malmer, 2004; Wilkinson et al., 2006; Lane et al., 2008). Generally, reported concentrations of NO_3^-, NO_2^-, and NH_3/NH_4^+ do not exceed the concentration guideline values for drinking water (Table 1). No drinking water guidelines exist for TN or TP (or other P forms), and instead the aquatic health guideline values for freshwaters in south-eastern Australia were used for comparison with studies from this region (ANZECC, 2000). Unfortunately, information is rarely provided on the duration over which observed concentrations are sustained, although elevated concentrations during flow events (stormflow or snowmelt events) may be expected to occur over periods in the order of hours to days. An important caveat on this assessment is the sampling regime adopted (Table 4) and the hydrological and constituent variability of the systems examined, which may greatly affect the ability of monitoring to capture the full range of post-fire constituent concentrations.

In subalpine coniferous forest streams, Mast and Clow (2008) observed first year post-fire NO_3^- concentrations 10 times higher than an unburned control catchment, with a maximum concentration of 0.8 mg L^{-1}, which was well below the guideline level of 50 mg L^{-1}. These authors noted that NO_3^- concentrations decreased fourfold during the 4 years study period but still remained elevated compared to the unburned catchment at the end of monitoring. In contrast, TP and NH_3/NH_4^+ concentrations appeared to be unaffected by the fire. Hauer and Spencer (1998), also working in burned conifer forest mountain catchments (1.8–238 km^2) in north-west Montana, reported maximum NO_3^- (1.9 mg L^{-1}) and NH_3/NH_4^+ (0.29 mg L^{-1}) concentrations for a 5 years period after wildfire. Maximum concentrations for both constituents were
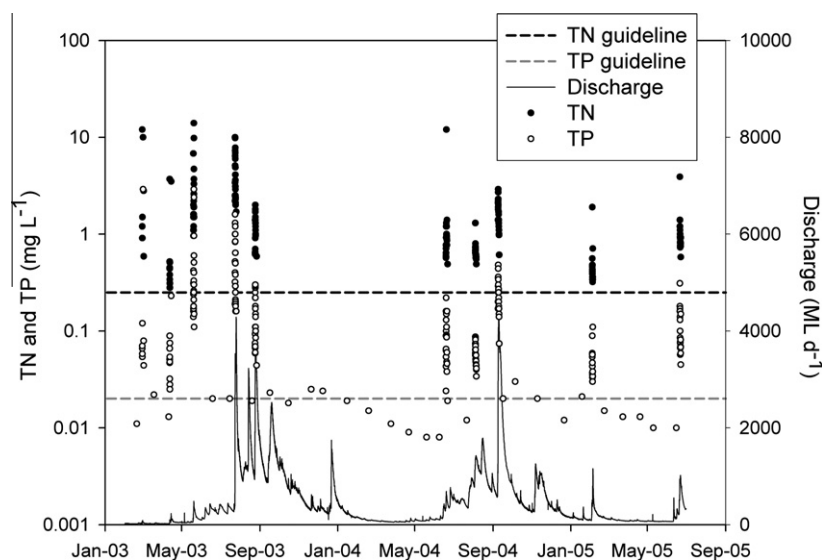

Fig. 2. Discharge, TN and TP concentrations for the Ovens River at Bright, north-east Victoria, for a period of two and a half years after the 2003 wildfires. Data provided by North East Water (T. Smith, pers. comm., 22 May 2009).

below drinking water guidelines. For similar regional settings in Canada, Bladon et al. (2008) observed a peak concentration for NO_3^- of 1.7 mg L^{-1} and Gluns and Toews (1989) reported 0.93 mg L^{-1}, both following high severity wildfire and both well below the drinking water guideline value.

Elevated concentrations of N and P in various forms have been reported after wildfire in south-eastern Australia. Based on storm-flow sampling in *Eucalyptus* forest catchments (40–750 km^2) in eastern Victoria, Chessman (1986) recorded maximum concentrations for NO_3^- (5.3 mg L^{-1}), NO_2^- (0.36 mg L^{-1}), NH_3/NH_4^+ (4.0 mg L^{-1}), and TP (0.82 mg L^{-1}) during a 3 month period after wildfire. The maximum TP and NH_3/NH_4^+ concentrations exceeded guideline thresholds by a factor of 41 times the aquatic health value for upland streams in south-eastern Australia and 8 times the drinking water value, respectively, whereas NO_3^- and NO_2^- concentrations were below the drinking water guideline values. Following the post-fire debris flows in the Buckland River catchment (referred to in Section 3.1), maximum TN and TP concentrations of 410 and 110 mg L^{-1} (1640 and 5500 times the regional aquatic health guideline values, respectively) were recorded at a water supply off-take point 30 km downstream (Leak et al., 2003). In contrast, the Ovens River upstream of the junction with the Buckland River (495 km^2; 55% burned), which was unaffected by the debris flows, recorded much lower maximum TN and TP concentrations. However, during the subsequent 2.5 years after the fire, TN and TP concentrations in the Ovens River exceeded the aquatic health guideline values during all automatically sampled flow events (Fig. 2). Despite declining annual exports of TN and TP in the Ovens (Sheridan et al., 2007b), only a slight reduction in monitored flow event concentrations was apparent. In contrast, Townsend and Douglas (2004), in a very different setting, reported no significant difference between pre and post-fire concentrations of TN and TP in open *Eucalyptus* forest streams in the wet/dry tropics of northern Australia, which they attributed to the low severity of the burn and the low relief in the 6.6 km^2 catchment.

Post-fire N and P concentration changes in lakes and reservoirs have not been widely reported, with published work largely confined to natural lakes in temperate and boreal forest regions of North America. McColl and Grigal (1977) observed that N concentrations in tributary streams and three oligotrophic lakes (area: 29–40 ha; depth: 4–5 m) in north-east Minnesota were unaffected by wildfire, with differences between tributary streams attributed largely to contrasting soil types. For the same lake systems, Wright (1976) recorded a post-fire increase in TP inputs from tributary streams but no increase in TP concentrations in the lakes, which were dominated (95%) by atmospheric inputs of P. In contrast, McEachern et al. (2000) observed a significant increase in mean TN (1.2 times unburned) and TP (2.6 times unburned) concentrations in lakes with burned catchments 2 years after fire compared to unburned boreal subartic lakes of northern Alberta. Likewise, Carignan et al. (2000), reported higher median concentrations of NO_3^- (maximum 60 times) and TP (2–3 times) in lakes with burned catchments compared to similar undisturbed lakes in the boreal forests of Québec, Canada, and noted that impacts were proportional to the catchment area burned divided by the lake volume or area.

In south-eastern Australia, data on post-fire changes to nutrient concentrations in lakes is largely confined to periodic measurements at reservoir off-take points or outflows. For example, nutrient data from water sampling in four reservoirs before and after the 2006/2007 wildfire in eastern Victoria is presented in Fig. 3, with the data for Lakes William Hovell, Buffalo and Eildon provided by Goulburn-Murray Water (L. Mattner, pers. comm., 18 May 2009), while data for Lake Glenmaggie was obtained through the Victorian Water Resources Data Warehouse. These measurements indicated that post-fire increases in mean NO_x, TN, and
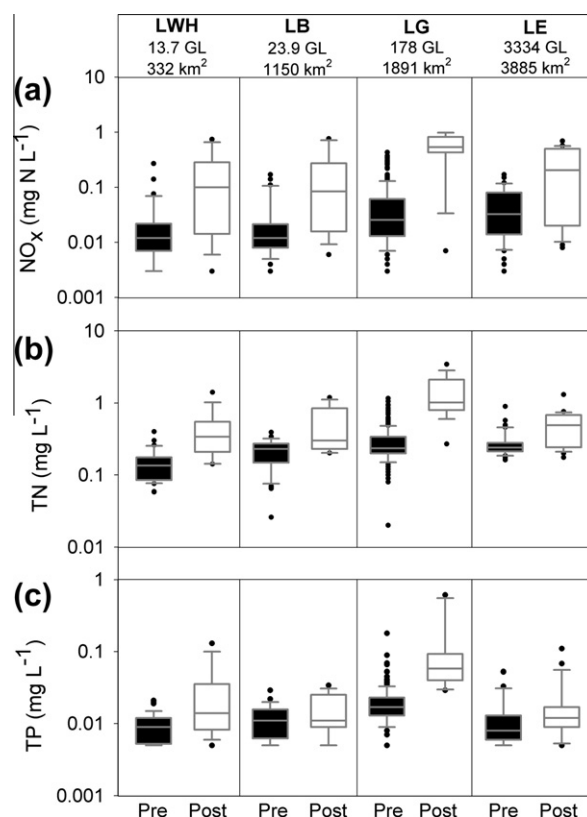

Fig. 3. Box plots of pre and post-fire (first year after fire) measurements at two–four weekly intervals of (a) NO_x; (b) TN; and (c) TP concentrations in four reservoirs (LWH: Lake William Hovell; LB: Lake Buffalo; LG: Lake Glenmaggie; LE: Lake Eildon) with catchments burned by wildfire in 2006/2007 in Victoria, south-eastern Australia. Box plots display the 10th, 25th, median, 75th, 90th percentiles and outliers. The reservoir capacity (GL) and catchment area (km^2) are listed below each reservoir label. Consecutive data prior to the 2006/2007 fire was available for 2003–2006 for LWH, LB and LE, whereas a longer record (1991–2006) was available for LG which included the effect of a wildfire that burned 17% of the catchment in 1998. Note that the catchment of LB was burned by wildfire in 2003 (35% of the catchment area) and again in 2006/2007 (near complete burn). Data for LWH, LB and LE was provided by Goulburn-Murray Water (L. Mattner, pers. comm., 18 May 2009), while data for Lake Glenmaggie was obtained through the Victorian Water Resources Data Warehouse (http://www.vicwaterdata.net/vicwaterdata/home.aspx).

TP concentrations, ranging from 1.3 to 11 times pre-2006/2007 fire levels, occurred in all reservoirs. These increases were statistically significant in all cases (Mann Whitney U test, 0.05 probability level), except for TP in two of the reservoirs (Lakes Buffalo and Eildon). Maximum concentrations in the first year after the 2006/2007 fire for NO_x, TN, and TP exceeded maximums recorded prior to this fire (Fig. 3), with both pre and post-fire monitoring occurring during a period of generally below average rainfall in south-eastern Australia (except for the longer pre-fire record at Lake Glenmaggie). The largest post-fire impact on reservoir water quality occurred in Lake Glenmaggie (catchment 90% burned) following debris flow (February 2007) and subsequent high magnitude floods (June and November 2007), with TN and TP concentrations in outflows only approaching pre-fire levels after 3 years (Fig. 4). In contrast, following the 2003 wildfires, Alexander et al. (2004) monitored nutrient levels in Dartmouth Reservoir (north-east Victoria) during the first year after fire. These authors reported elevated concentrations of TN (maximum 21 mg L^{-1}) and TP (8 mg L^{-1}) in tributary streams during flow events, but only a minor change was recorded in the reservoir near the dam wall.
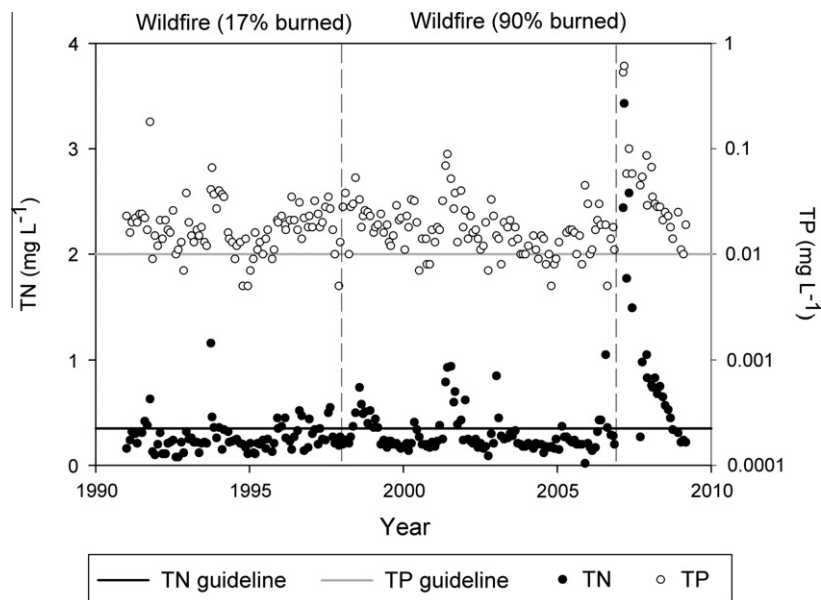
Fig. 4. Monthly TN and TP concentrations measured at the Lake Glenmaggie outflow (Macalister River) before and after wildfires in 1998 and 2006/2007, Victoria, south-eastern Australia. The TN and TP guidelines indicated are for freshwater lakes or reservoirs in south-eastern Australia (ANZECC, 2000). Data was obtained through the Victorian Water Resources Data Warehouse (http://www.vicwaterdata.net/vicwaterdata/home.aspx). Note the difference in scales between TN and TP.

3.4. Trace elements

Contamination of streams and water supply reservoirs by post-fire inputs of various trace elements may be problematic for both health and aesthetic reasons. Elevated concentrations of Fe, Mn and Zn cause aesthetic problems (taste, colour, staining of pipes and fittings), whereas poisoning may occur from continued consumption of water containing high concentrations of Ba and Cu, with less severe gastrointestinal symptoms possible with Cu concentrations of 3–5 mg L^{-1} (NHMRC, 2004; WHO, 2008). As and Cr (specifically hexavalent Cr) may be carcinogenic, while Al, Pb and Hg are toxic when consumed in sufficient quantities for prolonged periods. However, information on the health effects of short duration exposure to highly elevated concentrations of these elements is generally unavailable (NHMRC, 2004; WHO, 2008).

The store of total and water soluble Mn has been reported to increase in soils following fire through additions of Mn in ash from burned vegetation and the physiochemical breakdown of Mn complexed with organic matter (Chambers and Attiwill, 1994; Parra et al., 1996). A similar pattern may be expected for Fe, Cu, and Zn (Certini, 2005). Notably, water soluble Mn levels were observed to decrease to control levels within 1–2 months, probably as a result of a rapid increase in microbial population and oxidation of Mn to less soluble higher oxides (Chambers and Attiwill, 1994). Hg may accumulate in forest ecosystems (largely in soil organic matter) by atmospheric deposition from natural (e.g. volcanoes) and anthropogenic (e.g. coal-fired power plants) sources and can be subsequently revolatilized by wildfires (Rea et al., 2002; Schwesig and Matzner, 2000; Biswas et al., 2008). Gaseous mercury (Hg^0) is oxidised to Hg^{2+} which is rapidly deposited and may be reduced to the toxic methylmercury, which bioaccumulates (Garcia and Carignan, 1999; Lin and Pehkonen, 1999; Biswas et al., 2008). Wildfire effects on levels and mobility of other trace elements in soils (Al, As, Ba, Cr, Pb) have not been reported.

Information on post-fire exports of trace elements is largely unavailable. A notable exception is the study by Townsend and Douglas (2004), which reported first year post-fire Fe and Mn exports of 1.2 kg ha^{-1} year^{-1} and 0.022 kg ha^{-1} year^{-1}, respectively, from a tropical dry savannah forest catchment (6.6 km^2), which represented a negligible change from pre-fire exports. Following

the 2003 wildfire in south-eastern Australia, Wasson et al. (2003) estimated individual event loads of Fe and Mn entering Corin reservoir (197 km^2; 98% burned), which supplies Canberra, from three storm events within 2 months after the fire. The storms generated approximate Fe/Mn loads of 139/35 kg (59 mm/day; 25.8 ML inflow), 755/19 kg (72 mm/2 days; 280 ML), and 476/143 kg (n/a rainfall; 153 ML). The first post-fire event was a high intensity, localised convective storm and generated maximum Fe and Mn concentrations of 131 and 34 mg L^{-1} (sampled one day after the event), respectively, which were well in excess of guideline values. Notably, the largest Fe load was generated by the second event, a lower intensity, widespread rainfall event with much higher inflows, despite much lower concentrations (Wasson et al., 2003).

Reports of post-fire concentrations of trace elements in streams and reservoirs are also limited. A small number of samples were collected following intense summer storm events within 1 month of the 2003 wildfire in north-east Victorian streams and analysed for various elements. Leak et al. (2003) recorded maximum Fe, As, Cr, and Pb concentrations (mg L^{-1}) of 740 (2470×), 0.28 (28×), 0.92 (18×), and 0.98 (98× WHO guideline value), respectively, in a sample taken from the Buckland River 30 km downstream of debris flows. These concentrations should not be considered event maximums given that the sample was collected on the receding limb of the hydrograph at 1.5 m^3 s^{-1}, compared to the peak flow of 68 m^3 s^{-1} that occurred less than 12 h earlier (Leak et al., 2003). Following another storm event 2 days later in the nearby Ovens River catchment (495 km^2; 55% burned), samples were collected upstream of the debris flow water quality impacts from the Buckland River (a tributary to the Ovens) and analysed for the local water authority. Concentrations of Fe (30 mg L^{-1}) greatly exceeded the WHO guideline value (100×), and remained highly elevated 6 weeks later at 22 mg L^{-1}, exceeding the pre-storm event (but post-fire) concentration of 0.64 mg L^{-1} (2.2× guideline value) (T. Smith, pers. comm., 22 May 2009). In contrast, both post and pre-storm event concentrations (mg L^{-1}) of Cu (post: 0.032/pre: 0.001), Zn (0.1/<0.002), and Cr (0.04/0.001) were below the guideline values, while post-event As (0.012/0.003) and Pb (0.033/0.001) concentrations exceeded the guideline by 1.2 and 3.3 times, respectively (T. Smith, pers. comm., 22 May 2009).

Monitoring of Fe and Mn concentrations in the Bendora reservoir (supplying Canberra) on the Cotter River (482 km^2; 98% burned) was undertaken following the 2003 wildfire in south-eastern Australia (White et al., 2006). Major water quality problems occurred, with post-fire Fe and Mn concentration peaks exceeding all previous peaks by factors of 3 and 5, respectively (White et al., 2006). The high influx of these metals, combined with the post-fire inflow of water that was five degrees warmer than previously recorded near the reservoir bottom, contributed to increased release of Fe and Mn from bottom sediments under increasingly anoxic conditions (White et al., 2006).

Following the Cerro Grande wildfire in 2000, which burned mixed-conifer forest near Los Alamos in New Mexico, USA, manual and automatic sampling of summer storm runoff immediately after the fire was undertaken by the Los Alamos National Laboratory (LANL) at 40 ephemeral stream sites and analysed for a large range of constituents (Gallaher et al., 2002). Maximum reported concentrations for Fe, Mn, and Zn were 560 (1870×), 102 (255×), and 3.6 (1.2× WHO guideline value) mg L^{-1}, respectively, from a total of 85 unfiltered samples (Gallaher et al., 2002). Elevated concentrations of As, Al, Ba, Cr, and Pb in stormflows were also reported, with maximum concentrations of 0.14 (14×), 995 (guideline value not available for total Al concentration), 20.7 (30×), 0.51 (10×), and 1.2 (120× WHO guideline value) mg L^{-1}, respectively. The maximum concentration of Hg (0.0013 mg L^{-1}) slightly exceeded the total Hg guideline value (0.001 mg L^{-1}), while Cu (0.61 mg L^{-1}) was below the guideline value. All element concentrations except Hg exceeded pre-fire maximums and were attributed to natural sources rather than from the LANL facility located downstream (Gallaher et al., 2002). Maximum filtered sample concentrations (n = 57) were also reported and were well below unfiltered concentrations (Gallaher et al., 2002). WHO guideline levels were exceeded by maximum filtered concentrations (mg L^{-1}) of Fe, Mn and As, which were 6.9 (23×), 2.0 (5×), and 0.012 (1.2× guideline value), respectively, while Ba, Cu, Cr, Hg, Pb, and Zn were all below guideline values. The unfiltered concentration of Al was high at 11.5 mg L^{-1}. Unfiltered element concentrations generally corresponded with the suspended sediment concentration (Gallaher et al., 2002). While the storm runoff in the Los Alamos streams is not used for water supply, downstream contaminant transport is a concern (Gallaher et al., 2002).

Further information on post-fire concentrations of trace elements in streams and reservoirs is not readily available but warrants investigation, particularly in fire-prone forest regions susceptible to high magnitude erosion events (such as debris flows or large floods) that may generate large suspended sediment loads. Information on the duration of elevated concentrations in streams and reservoirs is also required to determine the longevity of post-fire water quality impacts (White et al., 2006).

3.5. Chloride, sulfate and sodium

Elevated concentrations of Na$^+$ and Cl$^-$ in water supplies present aesthetic concerns (taste and corrosion of pipes and fittings), whereas SO$_4^{2-}$ may be problematic for both aesthetic (taste at concentrations over 250 mg L^{-1}) and health reasons (purgative effects are possible with concentrations over 500 mg L^{-1}) (WHO, 2008). Additional solutes (namely Ca^{2+}, Mg^{2+}, K$^+$) that have been measured in streams after fire do not have individual guideline values but are captured by measurement of total dissolved solids (TDS), although TDS is rarely reported in studies of fire effects on water quality (Neary et al., 2005).

Burning may increase the concentration of SO$_4^{2-}$ in surface soils through oxidation of sulfur in soil organic matter, while Na$^+$, SO$_4^{2-}$ and Cl$^-$ may be leached from burned plant litter (Khanna and Raison, 1986; Murphy et al., 2006b). Ash deposits contribute to Na$^+$, Cl$^-$ and SO$_4^{2-}$ concentrations in burned soil, with 3, 9 and 3-fold increases, respectively, measured in soil solutions under ash beds compared to unburned sites (Khanna and Raison, 1986; Khanna et al., 1994).

Increased post-fire exports of Na$^+$, Cl$^-$ and SO$_4^{2-}$ relative to unburned areas have been observed. Ferreira et al. (2005) reported solute yields in overland flow after wildfire in Pinus pinaster forests of central Portugal, finding SO$_4^{2-}$ exports of 18.1 kg ha^{-1} year^{-1} (278 times unburned) at the plot-scale (16 m^2) and 13.2 kg ha^{-1} year^{-1} (4400 times unburned) at the catchment-scale (burned: 1.1 km^2; unburned: 0.61 km^2). These authors also measured burned catchment Na$^+$ and Cl$^-$ exports of 30.1 kg ha^{-1} year^{-1} (11 times unburned) and 39.9 kg ha^{-1} year^{-1} (19 times unburned), respectively. Most of the solute loss occurred during the first 4 months after fire and was associated with the removal of the easily mobilised surface ash layer (Ferreira et al., 2005). In contrast to this Mediterranean environment, increased solute losses in burned conifer catchments in North America have been associated with soil leaching and transfer to streams via sub-surface flow rather than storm-generated overland flow (Mast and Clow, 2008).

Reports of wildfire effects on solute concentrations in streams and lakes are mostly from coniferous forests in North America. Mast and Clow (2008) observed that wildfire increased levels of Cl$^-$ (4 times unburned) and SO$_4^{2-}$ (2.7 times unburned) in the first year after fire based on biweekly to monthly sampling in a subalpine conifer catchment (47.3 km^2) in north-west Montana. Despite the post-fire increase, maximum sampled Cl$^-$ concentrations remained <0.8 mg L^{-1}, with the peak occurring in the first spring snowmelt period after the fire. In a semi-arid environment, Gallaher et al. (2002) found post-fire maximum Cl$^-$ (53.2 mg L^{-1}) concentrations in stormflows only slightly exceeded the pre-fire maximum, while the maximum SO$_4^{2-}$ (16.7 mg L^{-1}) and Na$^+$ (14 mgL^{-1}) concentrations were below pre-fire. In the humid tropics (Malaysian Borneo), Malmer (2004) observed the effect of wildfire in small (<20 ha) forest catchment streams, which resulted in a small rise in Cl$^-$ baseflow concentrations for 2 months after fire.

Solute concentrations in lakes show a mixed response to burning. Wildfire increased exports of Cl$^-$ and SO$_4^{2-}$ to boreal shield lakes in Québec, Canada, resulting in up to a 6-fold increase in concentrations of both constituents over reference lakes, which declined over the 3 years monitoring period (Garcia and Carignan, 1999; Carignan et al., 2000; Lamontagne et al., 2000). However, maximum sampled concentrations of Cl$^-$ (<0.53 mg L^{-1}) and SO$_4^{2-}$ (<8 mg L^{-1}) after fire were very low by drinking water standards (Carignan et al., 2000). In contrast, Lathrop (1994) found negligible impact on all three solutes from a wildfire that burned 25% of the catchment areas of two lakes in Yellowstone National Park, and attributed this to the large size of the lakes which diluted increased post-fire inputs. Likewise, McEachern et al. (2000) found that fire did not significantly affect Cl$^-$ or Na$^+$ concentrations in boreal subarctic lakes of northern Alberta, Canada.

The effect of lower intensity prescribed fires on solute concentrations in streams has received some attention. Williams and Melack (1997) reported a 4 and 16-fold increase in volume-weighted mean concentrations of Cl$^-$ and SO$_4^{2-}$ (based on pre and post-fire 1–2 weekly sampling), respectively, in the first year after prescribed burning in a 13 ha mixed-conifer forest catchment in the Sierra Nevada, California. The large increase in mean concentration values was attributed to a succession of rain and snow events 4 months after the fire, although the maximum post-fire SO$_4^{2-}$ concentration (59 mg L^{-1}) was still well below the recommended limit. Stephens et al. (2004) also reported increased SO$_4^{2-}$ concentrations (13 times the mean of the unburned control) after a moderate-severity prescribed burn in the Lake Tahoe basin, California, but the maximum sampled concentrations remained

<10 mg L^{-1} and fire effects lasted for only 3 months. In contrast, both Davis (1989) and Richter et al. (1982), working in a 28 ha chaparral catchment (Arizona) and a 160 ha coastal pine forest catchment (South Carolina), respectively, reported no effect on Na$^+$, Cl$^-$ and SO$_4^{2-}$ concentrations from weekly stream sampling.

Based on this limited range of studies it is difficult to characterise Na$^+$, Cl$^-$ and SO$_4^{2-}$ responses to wildfire. It appears that exceedance of the recommended drinking water limits (based on aesthetic concerns) in streams and lakes after fire is unlikely to occur in North American coniferous forest catchments, while post-fire responses in most other forest environments are largely unknown.

3.6. Organic carbon

Wildfire may cause significant changes to the store of organic carbon on the forest floor and in surface soils, which, in conjunction with increases in post-fire erosion and leaching from the ash/soil, has implications for post-fire exports and concentrations of particulate and dissolved organic C (POC/DOC). No specific WHO drinking water guideline exists for POC or DOC, with turbidity used to infer the amount of suspended matter and colour (after 0.45 μm filtering) used as a general indicator of the level of dissolved organic matter in natural waters. In Ontario, Canada, a DOC guideline value of 5 mg L^{-1} has been set for aesthetic reasons, as well as to minimise disinfection by-products (Ministry of Environment, 2003). Chlorination of water with elevated dissolved organic matter can be problematic due to the formation of chlorinated by-products (e.g. trihalomethanes) that may present health concerns (NHMRC, 2004).

Forest floor (O horizon) loss of organic C after wildfire may be near complete, with reductions of >90% reported (Baird et al., 1999; Murphy et al., 2006a). However, deposited ash, primarily dark ash, may contain a sizable organic C component, with measured concentrations of up to 6.6% in ash from woodland burned at high severity in California (Goforth et al., 2005). Depending on the burn intensity, the effect on soil organic matter may range from minor (volatilization of some constituents), with charring (leading to formation of black carbon) and complete oxidation at higher temperatures (Giovannini et al., 1988; Tomkins et al., 1991; Giovannini and Lucchesi, 1997; Certini, 2005). Baird et al. (1999) reported soil organic C reduction of 10–17% in the surface horizons relative to unburned controls in conifer forests in Washington State, USA, 1 year after severe wildfire. An estimated 1% of the A-horizon organic C was lost through post-fire soil erosion relative to unburned control sites.

Increased post-fire erosion, combined with inputs of highly-erodible dark ash containing elevated organic C concentrations, may lead to increased POC exports after wildfire. Petticrew et al. (2006) reported a post-fire increase in the suspended organic sediment yield (burned: 3.2; unburned: 1.9 kg ha^{-1}) from a 135 km^2 catchment in British Columbia during a 7 month monitoring period, which included the spring snowmelt. These authors also observed a significantly higher proportion of organic matter in burned catchment composite suspended sediment (<500 μm) samples, which was attributed to either increased biofilm development (with increased light penetration following canopy removal by fire) and/or the transport of black carbon from ash deposits. In the 6 month period after the 2003 wildfire in south-eastern Australia, Wasson et al. (2003) estimated approximately 137 t (6.7 kg ha^{-1}) of POC entered the Corin reservoir based on sedimentation measurements, which exceeded the pre-fire mean annual POC input of 1.2 kg ha^{-1} year^{-1}.

More extensive research has been undertaken which examines DOC concentrations in streams and lakes after wildfire in the conifer forests of North America. Mast and Clow (2008) reported that wildfire had a minimal effect on DOC concentrations (burned

mean: 1.1 mg L^{-1}; unburned: 0.7 mg L^{-1}) over a 4 years period, with a concentration range of 0.5–2.3 mg L^{-1} in the burned catchment and <0.2–3.5 mg L^{-1} in the unburned. Comparison of paired burned and unburned streams in mixed conifer Rocky Mountain forest, central Idaho, revealed increased DOC levels in streams in burned catchments in the first year after fire during the peak spring flow period, although maximum sampled concentration remained <4 mg L^{-1} (Minshall et al., 2001). In boreal forests of northern Alberta, McEachern et al. (2000) observed a statistically significant 1.6-fold increase in DOC concentrations of burned catchment lakes (mean: 25 mg L^{-1}) compared to reference lakes (mean: 16 mg L^{-1}), as well as a 2.3-fold increase in mean colour of fire-affected lakes, with additional colour related to DOC in all lake waters. By contrast, Lamontagne et al. (2000) and Carignan et al. (2000) reported that wildfire did not cause a statistically significant increase in DOC stream exports or DOC concentrations (sampled maximums <11 mg L^{-1}) in burned catchment lakes compared to unburned in boreal forests, Québec, Canada.

Post-fire inputs of organic C to streams and reservoirs may be problematic, particularly from a water treatment perspective. POC flux appears to be contingent on the magnitude and timing of storm events after fire, and may be most problematic in those burned forest environments susceptible to large increases in runoff and erosion. Elevated post-fire DOC concentrations may reflect inputs from both overland and sub-surface flows. However, for the forest environments studied, wildfire effects on DOC were comparatively minor, with the concentrations measured in Canadian boreal lakes largely reflecting background conditions.

3.7. Cyanide

Cyanide release into stream water after burning as a result of ash leaching or aerial deposition has been reported (Barber et al., 2003). These authors found that available cyanide concentrations (comprised of free cyanides, HCN and CN$^-$, and several cyano-metal complexes) in ash leachate greatly exceeded that of unburned fuel in laboratory test burns. Furthermore, free cyanide concentrations measured in runoff from the first storm event following wildfire in North Carolina were an order of magnitude (mean 0.049 mg L^{-1}) higher than runoff from unburned areas (Barber et al., 2003). Both the test burns and wildfire generated available cyanide in similar or higher concentrations than the 96-h median lethal concentration for North American rainbow trout (0.045 mg L^{-1}) (Barber et al., 2003). Elevated post-fire concentrations of cyanide were also recorded following the Cerro Grande wildfire near Los Alamos (Gallaher et al., 2002). Maximum available cyanide (measured as cyanide amenable to chlorination) and total cyanide concentrations of 0.062 (median: 0.004) and 0.176 (median: 0.0116) mg L^{-1} were reported, respectively. The maximum available cyanide concentration, representing the potentially toxic fraction of cyanide, was only slightly below the WHO guideline value (0.07 mg L^{-1}). Importantly, increases in cyanide concentrations in stream water are probably of short duration during initial post-fire rainfall events and more likely to occur in small catchments where dilution is limited (Barber et al., 2003). In addition, Crouch et al. (2006) reported that the application of fire retardant to forest fires in North America had minimal effect on stream water quality, with cyanide concentrations in post-fire storm runoff unaffected by the presence of ferrocyanide in the retardant.

3.8. PAHs, PCDD/Fs and PCBs

Polycyclic aromatic hydrocarbons (PAHs), polychlorinated dibenzo-ρ-dioxins and dibenzofurans (PCDD/Fs) and polychlorinated biphenyls (PCBs) may be produced during combustion in forest fires and released into the atmosphere or deposited in ash and

soil/sediments (Freeman and Cattell, 1990; Gabos et al., 2001; Kim et al., 2003; Gosh, 2007). Concern about PAHs, PCDD/Fs, and PCBs relates to their toxicity, carcinogenicity, environmental persistence, and tendency to bioaccumulate (Gabos et al., 2001; Olivella et al., 2006; Vila-Escalé et al., 2007).

Once released, PAHs are subject to various transformations and the different physiochemical properties of PAHs result in different interactions with ash, sediment and biota in streams (Olivella et al., 2006). Data on the release of PAHs after fire is limited. Kim et al. (2003) reported increased levels of PAHs in forest soils (15.5 times unburned maximums) 1 month after wildfire in Korea, with the difference attributed to ash inputs. Gosh (2007) concluded that elevated PAHs in sediment samples can be largely explained by the black carbon content. Vila-Escalé et al. (2007) observed increased post-fire PAH inputs to streams associated with increased overland flow and soil erosion in a Mediterranean region (Spain). PAH concentrations approached background levels 15 months after the fire and never reached levels of toxicological concern (Vila-Escalé et al., 2007). These authors also suggested low rainfall after fire may contribute to increasing PAH concentrations. This is supported by Olivella et al. (2006) who reported an increase in PAH concentrations in stream waters associated with atmospheric deposition of ash (in the absence of rainfall) 1 month after forest fires (also in Spain). With subsequent heavy rainfall, PAH concentrations decreased significantly due to dilution (which was not offset by increased PAH input with eroded ash and sediment), as well as because of the degradation of PAHs (due to photo-oxidation) and their adsorption onto deposited particles. Total PAH concentrations measured at all sampling sites and on all sampling dates remained within the European drinking water limit (Olivella et al., 2006).

PCDD/Fs and PCBs concentrations in sediment were studied following wildfire in northern Alberta, Canada (Gabos et al., 2001). These authors observed very low concentrations of both PCDD/Fs and PCBs that were consistent with background levels at all sites investigated. However, Kim et al. (2003) reported increased PCDD/Fs levels (2.3 times unburned maximums) in forest soils following wildfire in Korea 1 month after fire, with concentrations comparable to unburned soils 5 months later. The short-term increase was attributed to inputs of PCDD/Fs from ash (ash concentrations were higher than for corresponding soils), which was subsequently removed by wind and runoff.

4. Wildfire impacts on water supply: summary and future research

This review reveals a wide range of wildfire effects on levels of various physical and chemical constituents in forest catchment streams and reservoirs. Any examination of the risk of contamination and disruption to water supply following wildfire is confronted by this large range of potential water quality impacts. This reflects the many variables that influence post-fire runoff, erosion and downstream constituent flux combined with the capacity of receiving waters to attenuate increased constituent inputs. The likelihood of water supply disruption after fire also depends upon water treatment capacity, the availability of alternative supplies, and the size of the population dependent upon the catchment source. Attempting to determine the contamination risk associated with wildfire is an important step in evaluating the security of water supplies in fire-prone forest environments. This may guide decision making concerning future infrastructure investment or the diversification of water sources. In concluding this review, we (1) summarise findings and consider the potential contamination risk for various constituents after fire; (2) examine impacts on water supply following wildfires in a highly fire-prone region; and (3) outline directions for future research.

4.1. Summary of review findings

The following points summarise information on water quality changes after wildfire in forest catchments, the associated hydrological drivers, and the likelihood of contamination by various constituents from a drinking water supply and treatment perspective.

1. Catchment-scale constituent exports vary greatly after wildfire, as do estimates of export increases relative to unburned or pre-fire conditions. The majority of studies examining wildfire effects on exports focus on suspended sediment, which displayed a large range in first year post-fire export increases (1–1459 times unburned exports). Information on exports of various forms of N and P is less extensive, with reported changes in TN and TP yields after fire ranging from 0.3 to 431 times unburned. Published studies of post-fire exports of trace elements are largely unavailable. Changes in trace element fluxes after fire may be expected to reflect the magnitude of change in suspended sediment exports, given the strong tendency for sediment-association. There are also few studies that provide information on Na^+, Cl^-, SO_4^{2-} or POC exports, although short-term increases after fire may occur.

2. Constituent concentrations in streams and reservoirs after wildfire also vary widely. While TSS concentrations in streams have been extensively measured and large increases observed after wildfire, drinking water guidelines focus on turbidity. Limited post-fire turbidity data from reservoirs indicates the potential for significant increases at off-takes or outflows. NO_3^-, NO_2^-, and NH_3/NH_4^+ concentrations in streams and lakes or reservoirs may increase after wildfire but appear to present a generally low risk of exceeding drinking water guideline values. Large post-fire increases in TN and TP concentrations in streams and reservoirs have been observed, particularly in response to large erosion events. High post-fire suspended sediment concentrations were associated with elevated trace element concentrations (well in excess of guideline values), particularly for Fe, Mn, As, Cr, Al, Ba, and Pb. In contrast, Cu, Zn, and Hg have been recorded below or only slightly above guidelines, indicating a potentially lower risk of contamination for these elements. Reports of Na^+, Cl^- and SO_4^{2-} concentrations are mostly confined to coniferous forest areas in North America, where the maximum sampled values were well below recommended limits. Similarly, wildfire effects on DOC were generally minor but observations were restricted to studies in North America where concentrations largely reflected background conditions. Available cyanide concentrations may briefly approach levels of concern soon after fire in small catchments where dilution is limited. Post-fire stream concentrations of PAHs have been found to increase but did not exceed guidelines, while PCDD/Fs and PCBs in soils and sediments (in the absence of stream concentration data) remain unchanged or may experience a small increase over low levels measured under unburned conditions. Caveats on these findings include the location-specific nature of data, differences in sampling regimes, and the limited data available for many constituents. Therefore, the absence of impacts or guideline exceedance should not be interpreted as indicating that there is no post-fire risk of contamination by a particular constituent.

3. Large increases in exports and concentrations of some constituents after wildfire indicate the potential for substantial impacts on water supply. Sizable export and concentration increases at both small and large catchment scales demonstrate that post-fire impacts may extend large distances downstream. Thus wildfire in forested headwater sub-catchments may affect water supplies in downstream lowland areas, and distance from source may not necessarily act as an effective buffer against

Table 5
Summary of management actions taken to secure drinking water supplies in response to water quality impacts following wildfires in 2003 and 2006/2007 in south-eastern Australia.

Wildfire	Catchment	Management actions[a]
2003	Ovens River, Victoria	Boil water notices; increased water restrictions (level 4 of 4)
	Cotter River, ACT (Canberra)	Switched supply (1 year); water restrictions; new water treatment plant ($38 million AUD)
2006/2007	Ovens River, Victoria	Boil water (6 months); temporary water treatment plant installed at Bright
	Goulburn River, Victoria	Boil water notice for Eildon township (supplied from Lake Eildon)
	Macalister River, Victoria	Water treatment plant taken offline (Lake Glenmaggie); increased water restriction level; Water carting (February–September 2007)
	Mitchell River, Victoria	Water unsuitable for harvest to off-river storage; Installed water clarifiers and five settling dams ($6 million AUD); utilised groundwater to ensure supply

[a] White et al. (2006), Southern Rural Water (2007), North East Water (2008), State Government of Victoria (2008, 2009).

post-fire impacts on water quality. Large increases in post-fire exports and highly elevated constituent concentrations generally reflect the impact of large erosion events after wildfire. In particular, this review has emphasised the importance of short duration, high intensity storms as drivers of high magnitude, localised erosion processes (such as debris flows or flash floods) that have resulted in major water quality impacts after wildfires. The extent of downstream delivery of sediment and associated constituents is also dependent on the timing and magnitude of post-fire flow events. In-channel storage may occur in response to insufficient transport capacity of flows, potentially reducing or delaying the downstream flux of particulates. This combination of post-fire storm events, erosion processes, and the transport capacity of flow events may largely dictate the magnitude, duration and downstream extent of post-fire water quality impacts.

4. The risk to water supplies associated with contamination by various constituents after wildfire depends on the likelihood that concentrations will exceed guidelines, the duration of exceedance, and the availability of adequate treatment facilities to process contaminated water. Suspended sediment and sediment-associated contaminants (particularly nutrients, organic carbon and various trace elements) represent the constituents that are most likely to occur at elevated levels in water supplies after fire. In response, water treatment by coagulation (through use of coagulants such as aluminium sulfate or polyaluminium chloride) and filtration to remove flocculated particles may be required. Notably, for most toxic metals, elevated concentrations may reflect high sediment concentrations, which once removed would greatly reduce levels of these contaminants. At very high TSS/turbidity levels, treatment problems may be encountered that reduce the rate of water processing (e.g. Leak et al., 2003), potentially causing difficulties in maintaining a continuous supply of potable water. In the absence of adequate treatment facilities, water supplies may be vulnerable to more prolonged disruption from large post-fire increases in suspended sediment flux. Elevated turbidity levels may also necessitate increased disinfection and oxidation of metals or organics using various disinfectants/oxidants such as chlorine, ozone or hydrogen peroxide. This may result in the increased formation of disinfection by-products. For other constituents, such as NO_3^-, NO_2^-, Na^+, Cl^-, SO_4^{2-}, cyanide, PAHs, PCDD/Fs and PCBs, it appears less likely that concentrations of concern may occur after wildfire, although conventional treatment options are not available for many of these constituents.

4.2. Case study of wildfire impacts on water supply

The large wildfires that occurred in south-eastern Australia over the last decade burned numerous forest catchments containing streams and reservoirs utilised for water supply. This provides a case study of the extent to which wildfires may disrupt water supplies in a highly fire-prone region. Post-fire impacts on water quality that led to difficulties in maintaining the supply of potable water and the associated management actions taken in response are summarised in Table 5. Most notable was the disruption to the supply of drinking water to Canberra from reservoirs within the Cotter River catchment following the 2003 wildfire (White et al., 2006; Wade et al., 2008). Fortunately, an alternative unburned water storage reservoir was available, which, in conjunction with mandatory water restrictions, enabled the continued supply of potable water to the city. However, the impact of the fire was sufficient to prompt rapid construction of a $38 million (AUD) flocculation and filtration plant within 18 months that was capable of treating 270 ML d^{-1} of water with turbidity up to 20 NTU (White et al., 2006). Water supply difficulties also arose after the 2003 and 2006/2007 wildfires in Victoria (Table 5). In contrast, the 2009 wildfires burned 30% of Melbourne's water supply catchments and 9 months later no water quality impacts had been observed in storage reservoirs, with brief storm event turbidity peaks rapidly returning to baseline conditions (Frame et al., 2009). Similarly, the impact on water quality in Lake Burragorang (supplying Sydney) was negligible following wildfire in 2001, despite increased flow event TSS, TN and TP concentrations in tributaries (Wilkinson et al., 2007). Following the fire, rainfall and river flows were well below average, with much of the material generated by increased post-fire hillslope erosion entering channel storage (Wilkinson et al., 2009). This range of wildfire impacts on water supply across south-eastern Australia highlights both the dependency of water quality changes on post-fire rainfall events and the regional diversity of landscape controls (e.g. burn patterns, topography, soils, vegetation and recovery rates) influencing post-fire runoff, erosion and downstream constituent flux.

4.3. Future research

Given the current limitations in the understanding of wildfire effects on water quality, there are several areas which may form a focus for future research. Firstly, there is a paucity of data on stream exports or concentrations for various constituents (particularly trace elements) that may be generated and delivered to streams in burned forest catchments. In the absence of such data, assessing the potential contamination risk to water supplies across the range of constituents of concern is difficult. Establishing continuous monitoring programs after unplanned fire events is challenging, particularly from research funding and logistical perspectives. Nonetheless, such data from different forest environments is critical for identifying the range of post-fire impacts and for model validation.

Secondly, an expanded focus on research linking post-fire soil and hillslope hydrological properties to changes in runoff generation and erosion processes may contribute to development or

adaptation of models to better represent post-fire runoff, erosion and constituent delivery from hillslopes to streams (Istanbulluoglu et al., 2004; Rulli and Rosso, 2005; Doerr et al., 2006; Robichaud et al., 2007; Moody et al., 2009).

Thirdly, channel contributions have been shown to represent a large part of post-fire catchment sediment budgets in some forest environments (Moody and Martin, 2009). However, few studies have reported channel erosion responses to flow events after fire (White and Wells, 1979; Moody and Martin, 2001), although post-fire debris flow processes that scour steep, low-order stream channels have received more attention (Cannon and Reneau, 2000; Cannon, 2001; Meyer et al., 2001; Gartner et al., 2008; Santi et al., 2008). For this reason, greater insight into both post-fire channel adjustments and changes in catchment hydrology that influence the scour and transport capacity of flows are needed if the sources and transport of constituents are to be adequately represented in catchment models.

Finally, following from the previous point, there is a need for better accounting of the dominant sources of sediment and other constituents entering and transported through stream networks after wildfire, particularly at larger (10–100s km^2) catchment scales (Blake et al., 2009b; Wilkinson et al., 2009). Methods such as sediment fingerprinting/tracing (Collins and Walling, 2004), in conjunction with catchment monitoring of discharge and constituent fluxes, offer approaches suited to quantifying sources and fluxes at these larger spatial scales. Such information is particularly relevant to efforts to determine the potential magnitude of wildfire impacts on water resources at the scale of catchments commonly used for water supply.

Addressing these various areas would contribute to both our understanding of fundamental processes and add to the catalogue of reported wildfire impacts. It may also drive model development in a direction that services both an understanding of interactions between components within catchment systems disturbed by wildfire, while also providing guidance to managers concerned about the risk of water supply contamination after fire.

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