



From burned slopes to streams: how wildfire affects nitrogen cycling and retention in forests and fire-prone watersheds

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Abstract Wildfire is a major driver of nitrogen (N) cycling and export from terrestrial to aquatic systems. While fire is a natural process in many watersheds, it can still degrade water quality by rapidly flushing N to streams. This can be particularly problematic in watersheds that experience high levels of N deposition or where climate change is promoting larger and more severe fires. The extent and duration of postfire N export, and the potential consequences for downstream water quality, depend on how N inputs, internal cycling, and outputs vary before, during, and after fire. Here we review the major factors controlling N cycling and retention in forests and adjacent shrublands, and how fire modifies these

controls. We connect burned slopes to streams to describe how fire exports N to aquatic environments. We also consider the implications for municipal watersheds and water resources management. We close by identifying critical knowledge gaps in projecting how fire will affect watershed N cycling and retention in the future.

Keywords Wildfire · Nitrogen cycling · Nitrogen retention · Water quality · Drinking water

Introduction

Forests and shrublands dominate many mountainous watersheds and act as natural treatment systems for drinking water. It is estimated that forests save 4.1 trillion U.S. dollars yearly in water treatment costs (Bladon et al. 2014). However, these landscapes are also fire-prone, which can periodically limit their filtration capacity and degrade water quality (Burton et al. 2016). Wildfires release many solutes, such as nitrogen (N), sodium (Na^+), chlorine (Cl^-), and sulfate (SO_4^{2-}) (Smith et al. 2011), which can create impure water and generate harmful byproducts during drinking water treatment processes (Hohner et al. 2019). In this review, we focus on N, which can fertilize and pollute streams, lakes, and reservoirs following wildfire, putting drinking water

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infrastructure, aquatic ecosystems, and recreational activities at risk. However, N is also a critical element for recovering upslope vegetation and therefore, global change processes that alter the way N cycles and moves through a watershed can have large consequences for ecosystems from slopes to streams.

Climate change is causing larger, more severe fires in many ecosystems (Westerling et al. 2006; Abatzoglou and Williams 2016; Hanan et al. 2021) and therefore increasing fire-induced N export to streams (Hanan et al. 2017). Many studies predict that this trend will continue through the twenty-first century (Rogers et al. 2011; Barbero et al. 2015; Parks et al. 2016; Halofsky et al. 2018). In western North America, the effects of climate change and wildfire on N export can vary substantially among watersheds that experience different climate conditions and differ in topography, vegetation, and associated fire regimes (Gresswell 1999). For example, areas that historically burned under a low frequency, high severity fire regime (i.e., wet forests west of the Cascade Mountains in Oregon and Washington or high elevation forests in the northern Rocky Mountains) may be particularly vulnerable because large amounts of N are contained in abundant aboveground biomass/fuels (Rozendaal et al. 2017). Similarly, areas that experience high levels of N deposition (i.e., many chaparral ecosystems in southern and central California) can also experience intense postfire N export (Fenn et al. 2003). Understanding and predicting how postfire N export varies among ecosystems and watersheds, and how N export influences water quality, is important for wildfire mitigation and postfire management.

Excess contaminant loading to drinking water sources, including inputs resulting from wildfires, increases treatment costs (Nunes et al. 2018) and is forcing municipalities to adapt their treatment methods to handle higher concentrations of N, sediment, and dissolved organic matter (Robinne et al. 2019). Nutrient-enriched streams and drinking water can have negative consequences for human health (i.e., by causing methemoglobinemia in infants), aquatic ecosystems (i.e., by causing algal blooms), and recreation. Usually, dissolved N (and other solutes, suspended sediments, etc.) are diluted as water moves downstream, meeting other uncontaminated rivers (Samuels et al. 2006). However, relying on dilution may not be adequate for addressing further increases in nutrient inputs that result from human-driven

changes to the N cycle (e.g., increasing rates of deposition and fertilization), larger and more severe wildfires, downstream land disturbances, and other future environmental changes (Li et al. 2016).

Wildfire-caused nutrient export is a key concern for many western U.S. water managers (Sham et al. 2013). If nutrient levels exceed maximum contaminant guidelines, the water is rendered temporarily unpotable. The U.S. Clean Water Act sets the Maximum Contaminant Levels (MCLs) at 10 mg L⁻¹ for NO₃⁻ and 1 mg L⁻¹ for NO₂⁻ in surface drinking water (Clean Water Drinking Act, 1972). However, postfire streamwater NO₃⁻ concentrations have been found to well-exceed this EPA threshold (e.g., 220 mg L⁻¹) (Teclé and Neary 2015). Even when postfire concentrations reach just below the threshold (e.g., > 9 mg L⁻¹), they can still be problematic for water treatment (U.S. Geological Survey 2012). While these effects are often transient, elevated nutrient levels in source watersheds have been recorded for more than ten years after severe fires (Emelko et al. 2011; Rhoades et al. 2019). Given that the number of large forest fires and length of wildfire season has increased, coupled with the uncertainties about climate change, managing water supplies from fire-prone watersheds is an evolving challenge (Hallema et al. 2018).

While periodic wildfire and subsequent N fluxes are a natural component of many forested and shrubland systems, environmental changes such as anthropogenic N deposition and climate change can exacerbate N export to streams. Here we review N cycling and fluxes in fire-prone watersheds during three key stages: before fire, during fire, and after fire (Fig. 1). Processes occurring at each of these stages can influence N cycling and retention in subsequent stages. First, we describe N cycling and fluxes in undisturbed watersheds. Then, we briefly review how fire alters ecosystem N storage and fluxes through changes to soil, vegetation, and in-stream processing. We also discuss mechanisms of postfire N export to streams, focusing primarily on forested watersheds in the western U.S., which range from semi-arid to mesic and many contain shrub/chaparral cover at lower elevations. Our goal is to review prefire N conditions and cycling, discuss how wildfires can affect N cycling and export, and explore the consequences of climate change-caused alterations to wildfire characteristics and the potential for increased N export. Increased

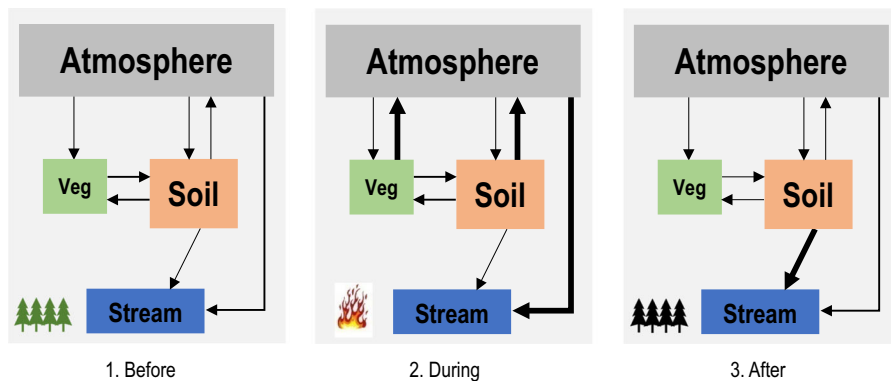


Fig. 1 Conceptual diagram of N stores and fluxes in a watershed (1) before, (2) during, and (3) after fire. (1) Before fire, N is stored in the atmosphere, vegetation, soils (including litter and rocks), and streams (which are small, often transient stores relative to the others). During this stage, there can be relatively small N fluxes to streamwater depending on the N status of the system (N-saturated or N-limited). (2) Fire

transforms and moves N through combustion, oxidation, and volatilization back to the atmosphere while simultaneously depositing N with ash on burned hillslopes or in surface waters. Much N also remains in the soil. (3) After fire, N is transported with soil erosion, surface runoff, and leaching to the streams or taken up by recovering vegetation

understanding of these processes has key implications for water quality planning and management.

Before fire

Nitrogen is distributed among multiple pools within an ecosystem or watershed, including the atmosphere, vegetation (live and dead), rocks, soil, and streams (which are small, often transient stores relative to the others). N availability is dictated by local environmental drivers (i.e., aquatic and terrestrial habitat compositions and precipitation regimes) and in a recently undisturbed watershed, N pool sizes and fluxes vary with season, input/output rates, soil properties, geology, and climate. Therefore, the distribution of N can vary substantially among ecosystems and over time (Gessel et al. 1973; Cole and Rapp 1981; Johnson and Lindberg 1992; Brockley et al. 1992; Klopatek et al. 2006; Johnson and Turner 2014). For example, some forests in coastal Oregon are hypothesized to be N-saturated because of high rates of N-fixation by red alder (*Alnus rubra*) (Compton et al. 2003). Similarly, in western Wyoming, human sources, such as artificial fertilizer or septic/sewage effluent, cause N enrichment (Eddy-Miller et al. 2013). In other watersheds (e.g., many in central Idaho), aquatic N is low due to factors such as low NO_3^- concentrations in the surrounding soils, smaller populations of anadromous fish, which release N when

they decompose, and wildfire suppression, which would otherwise release N to the soil and stream water (Delwiche 2010).

Ecosystems can be N-limited or N-saturated, which affects how N is internally cycled and subsequently lost (in the absence of a large disturbance) (Aber et al. 1989). An ecosystem becomes N-saturated when N inputs exceed the biotic demands (Fenn et al. 2008), which can promote leaching (Riggan et al. 1985; Aber et al. 1989; Stoddard 1994). N-saturated systems can also have elevated NO_3^- concentrations in the soil (Aber et al. 1989) and tend to lose NO_3^- with runoff (Jin-yan and Jing 2003). However, N leaching is not a perfect indicator of N saturation in many xeric and Mediterranean climates where seasonal N flushes are decoupled from the active growing season when plants would take up the exported N (Homyak et al. 2014). This decoupling can be particularly pronounced following fire when supply increases and demand decreases (Hanan et al. 2017). Therefore, to understand the relationship between prefire N status and postfire N export, we must consider N stores and pathways from the atmosphere, vegetation, rocks, and soil to streams.

Atmosphere

Nitrogen enters terrestrial systems via atmospheric deposition (wet and dry), abiotic and biological fixation of dinitrogen (N_2) gas, parent material

weathering, and industrial fixation (i.e., fertilizer). Different pathways may dominate in different regions depending in large part on their precipitation regimes (Cook et al. 2018). For example, wet deposition accounts for 21–64% of total N deposition in moist regions whereas in the American southwest, up to 75% of total deposition occurs as dry deposition (Li et al. 2016). Deposition can provide critical nutrients to N-limited ecosystems but too much N can have undesirable environmental effects, such as promoting non-native plant invasion, altering ecosystem function, and degrading surface waters (Aber 1992; Fenn and Poth 1998; Baron 2006; Eshleman et al. 2013; Eshleman and Sabo 2016).

Nitrogen can move from the canopy to the forest floor via throughfall deposition. Measuring throughfall deposition is useful for understanding ecosystem N status because it includes both wet and dry deposition and informs how N deposition directly influences soil biogeochemical and plant processes (Fenn et al. 2013); it is also strongly positively correlated with increased NO_3^- leaching to streams (Fenn et al. 2008). However, quantifying throughfall deposition only provides an estimate of the lower bound of tolerance for forest ecosystems because canopies retain some deposited N (Lovett and Lindberg 1993). Thus, throughfall measurements may underrepresent deposition in drier systems where precipitation is highly stochastic (Cook et al. 2018).

Vegetation

Nitrogen limitation is a major control on vegetation growth and can trade off with other factors that promote or constrain growth, such as light, water, and other nutrients. For example, in a disturbed old growth forest, regeneration is co-limited by light and N availability (Soto et al. 2017). Additionally, with elevated CO_2 concentrations in the atmosphere fertilizing plants, some researchers have found that N will be the main resource constraining growth in many ecosystem types in the future (Feng et al. 2015; Terrer et al. 2019). However, others have shown that with warmer temperatures, soil N availability might increase with increasing microbial activity, leading to greater productivity (Noyce et al. 2019). The effects of climate warming on plant-soil feedbacks is an active area of research and many uncertainties remain (Sistla et al. 2012; Pugnaire et al. 2019).

Ecosystem type and environmental conditions are critical factors regulating N cycling in fire prone watersheds. For example, aridity can modulate canopy cover and soil pH (McCulley and Jackson 2012; Delgado-Baquerizo et al. 2013) and as a result N is often limiting in very arid stands (Wang et al. 2014). To offset these types of environmental limitations, unique combinations of ecosystem characteristics can emerge. For example, some plant species can assimilate NO_3^- through their leaves (Black et al. 2002), which can substantially reduce N limitation as N deposition increases in the future (Bourgeois et al. 2019).

Soils

In addition to the ocean and atmosphere, soil is another major store of organic and inorganic N. Soil N concentrations vary with climate and biotic processes. N is distributed throughout the soil profile but most of it is found in the O and A horizons—the top two layers of the soil composed of mainly organic material and minerals. The top meter of global soil is estimated to contain 9.5×10^{13} kg N; in warm deserts, soils store about 0.2 kg m^{-3} N, while wet forest soil can contain 1.6 kg m^{-3} soil N (Post et al. 1985). Old growth forests and wetter forests tend to have greater amounts of soil N than young and/or dry forests (Gessel et al. 1973; Moghimian et al. 2020). Unmanaged old growth forests have more coarse woody debris and other downed biomass, which contribute more N to soils (Fisk et al. 2002). For example, in a variety of Washington and Oregon soils, total N in the O-horizons varies from 100 to 2000 kg ha^{-1} in response to forest age, elevation, and temperature and moisture levels (Gessel et al. 1973). Other factors influencing soil N loads include plant community composition and soil physical, chemical, and biological properties.

Soil properties also influence soil microbial community composition and N cycling processes. Most N exists in biologically unavailable forms (i.e., bound in complex organic molecules comprising vegetation, litter, and soil organic matter). Therefore, N must be transformed into usable forms—predominantly NH_4^+ and NO_3^- —through microbially-mediated redox reactions in the soil. When N is limiting, most inorganic N exists as NH_4^+ and can be immobilized by microbes, making it unavailable for plants. When N is less limiting, nitrification is carried out by highly

specialized chemoautotrophic microorganism that oxidize NH_4^+ into NO_2^- and NO_3^- . Nitrification releases H^+ ions into the soil and is therefore acidifying. Nitrifiers are very sensitive to soil pH, with optimal rates occurring when pH is greater than 6.0 (Sahrawat 2008; Nguyen et al. 2017). If nitrification rates are high, soil acidification can create a negative feedback where nitrifiers are inhibited by low pH and nitrification rates decline (Jin-yan and Jing 2003). However, when NH_4^+ substrate is extremely abundant, nitrification can happen even in very acidic soils (De Boer and Kowalchuk 2001; Hanan et al. 2016a; Li et al. 2018, b). For example, at Hubbard Brook Experimental Forest, New Hampshire, high rates of N deposition increased nitrification even when pH was as low as 4.3 (Likens et al. 1970).

We often overlook rocks as a source of N to ecosystems and watersheds but their contribution can be substantial (Morford et al. 2011). Rocks can contain 0.3 to 34% of total N in soils (Whitney and Zabowski 2004) and as they weather, rocks can increase ecosystem N budgets by 8–26% (Houlton et al. 2018). For example, in the Lake Tahoe Basin, California and Nevada, rocks hold 19% of the total soil N and rock content is directly proportional to the total C and N concentrations in the top 2 mm of soil (Johnson et al. 2012). N concentration in rocks can be $> 1000 \text{ mg N kg}^{-1}$ (Holloway and Dahlgren 2002) and in areas with N-rich rocks ($350\text{--}950 \text{ mg N kg}^{-1}$), they can elevate the N content of the soil by up to 50% (Morford et al. 2011). Worldwide, N geochemistry and regional climate controls the magnitude of rock weathering and consequent N contributions to ecosystems (Houlton et al. 2018).

Outputs

When N is limiting, plants and microorganisms compete for available N through uptake and immobilization, respectively. N can also be lost through denitrification, runoff, and leaching; these outputs typically increase with decreasing N limitation. Complete denitrification is a microbial process that removes N from an ecosystem by transforming NO_3^- to gaseous N_2 and in the process, releases N_2O and NO_x (Parton et al. 2001). Rates of denitrification depend on the ecosystem N status, its vegetation composition, seasonality, soil moisture,

temperature, and the availability of organic material. Because the riparian zone is a major site of denitrification, managers sometimes design vegetated riparian buffer zones to promote biological N removal, thereby reducing N fluxes to adjacent streams (Martin et al. 1999; Mayer et al. 2005).

Nitrogen can also leave an ecosystem through runoff and leaching (Cameron et al. 2013). Runoff is the process of water moving over land surfaces; at equilibrium, N leaching is a function of how much N is available (e.g., the extent to which an ecosystem is N-saturated) and how quickly plants and soil microbes take it up (Aber et al. 1991). Factors, such as wildfires, other disturbances, ecosystem management, climate change, and changes in N deposition rates can perturb equilibrium leaching rates (Dirnböck et al. 2016; Schleppi et al. 2017). In N-limited or balanced ecosystems, NO_3^- levels in runoff rarely exceeded $0.2 \text{ mg NO}_3^- \text{ L}^{-1}$ (Fenn et al. 2008; Rhoades et al. 2011). Ecosystems are considered “leaky” when runoff has more than $0.5 \text{ mg NO}_3^- \text{ L}^{-1}$ (Gundersen et al. 2006).

Nitrogen concentrations in runoff and leachate vary seasonally with temperature and moisture (Klopatek et al. 2006; Skorbilowicz and Ofman 2014). For example, rainfall can increase rates of mineralization and nitrification (Fisk et al. 2002; Chen et al. 2017). However, during the growing season, much of this N can be taken up or immobilized by plants and soil microbes, which reduces N leaching and runoff (Kelley et al. 2017; Lin et al. 2019). In regions where elevated temperatures correspond with elevated precipitation, N is taken up rapidly for growth, which reduces fluxes to streams (Skorbilowicz and Ofman 2014). However, in areas with wet winters, N export to streams can occur even when ecosystems are N-limited because plants are less active due to low temperatures. For example, in snow-dominated ecosystems, there is less N demand from plants and soil microbes when temperatures are cooler. As a result, N concentrations in surface runoff and lateral flows are higher during snowmelt (Rhoades et al. 2011), in part due to higher rates of N leaching. Leaching can also increase after autumn leaf fall (Fisk et al. 2002).

In ecosystems where spring and summer precipitation coincide with the peak growing season, elevated stream NO_3^- concentrations can indicate an N-saturated system (Fenn and Poth 1998; Fenn et al. 2008).

However, this metric breaks down in semiarid and/or Mediterranean climates, which are characteristic of many of the ecosystems in the western U.S. In these systems, the timing of precipitation is decoupled from the peak growing season, which can generate substantial N fluxes to streams (even in unpolluted systems) at the onset of winter rainy season when plant growth is slow (Homyak et al. 2014). This seasonal pulse can be exacerbated following fire due to increased levels of ash-deposited N on soil surfaces, coupled with a loss of vegetation that might otherwise be able to take up some of the available N, even if growth is slow (Hanan et al. 2017).

During fire

Fires actively change the form and distribution of biomass in ecosystems and watersheds. The extent of these changes is generally described as fire severity (Parson et al. 2010). Fire severity is a metric for quantifying fire effects on ecosystems. Because it is an indirect measurement (i.e., fire effects rather than a measure of the fire temperature itself), it is not ideal for deriving direct relationships between heating and biogeochemical processes (Smith et al. 2016). However, researchers still frequently use it to characterize fire because direct measurements of radiant energy and/or residence time are often not available.

Atmosphere

When combusted, N held in vegetation, litter, and soil can be volatilized or attached to smoke particles; remaining N can be deposited on soil surfaces with ash (Baird et al. 1999; Caon et al. 2014; Lindaas et al. 2021). Because N volatilizes at relatively low temperatures, fire always reduces total ecosystem N content (Neary et al. 1999; Kirkman 2011; Johnson and Turner 2014). However, the amount of N that is volatilized depends on fire characteristics, including temperature, fuel moisture and composition, and high-heat residence time (Binkley and Fisher 2000). Thus, severe fires, which typically burn at higher temperatures and have longer residence times, tend to volatilize more N (Knicker et al. 1996; Baird et al. 1999; Neary et al. 1999).

The N that is not volatilized is left as white and black or grey ash, which increases N availability (Bodí

et al. 2014). Complete combustion of organic material forms white ash. Black or grey ash is the organic ash mixture that comes from only partially combusted organic materials (i.e. the ash weighs 30–90% of the original material's weight) (Ranalli 2004). Because NH_4^+ has higher volatilization temperature than NO_3^- , it is typically more concentrated in ash following fire. Although ash can provide a N source for recovering ecosystems, it can also be easily mobilized by wind or precipitation events (Balfour et al. 2014; Neris et al. 2021).

Vegetation

Severe fires affect more components of an ecosystem than low severity surface fires. Fire severity and the path a fire travels is controlled by environmental factors, such as high fuel moisture levels, slope, wind, fuel surface to volume ratios, and fuel packing ratios (Butler et al. 2007; Banerjee et al. 2020). Low severity ground fires typically only consume vegetation and other organic matter found on the surface. High severity fires crown fires (i.e., fires that burn the canopy) can affect more parts of an ecosystem, such as both the under- and overstory vegetation and ladder fuels (i.e., fuels that bring a fire from the surface to the canopy). As a result, high severity crown fires typically have more intense and longer-lasting effects on watershed N dynamics than low severity fires.

Soil

Fire has immediate effects on soil physical and chemical properties, which can affect postfire N cycling. Depending on the temperature of the fire and soil type, fire can reduce soil sulfur, phosphorus, N, and amino acid concentrations. It can also kill bacteria and fungi, and char organic matter (Neary et al. 2005). Because N tends to volatilize, if a high severity fire burns hot and long enough, it can affect soil N compounds below the top two horizons, and likely disrupt microbial communities deep in the soil profile (Sharma et al. 2017). In addition to mortality effects, fire can create a water-repellent soil layer by combusting waxy or resinous hydrophobic compounds that distill downwards in the soil and inhibit water infiltration for several years postfire (Pierson et al. 2003; Bodí et al. 2014). Fire can also immediately increase soil pH by destroying organic acids and

depositing base-forming cations with ash (Giovannini et al. 1990). Many of these changes can persist for years after a fire occurs (Hanan et al. 2016a).

Postfire

After fire, various pathways exist for N movement through a watershed. These pathways depend on prefire ecosystem conditions, fire temperature and duration, and the rate of postfire recovery (N uptake and immobilization). The challenge is connecting changes in upland N storage to N delivery to streams. Many of the mechanisms that govern N cycling and export in an undisturbed ecosystem also persist once an ecosystem is disturbed (i.e., postfire). However, the magnitude and timing of N cycling and export usually change after a fire (i.e., rates of leaching and denitrification can respond to changes in soil biogeochemical cycling and vegetation demand) (Hanan et al. 2016a).

Vegetation

Ecosystem composition and fire characteristics influence postfire vegetation recovery rates in upland areas. Hardwoods and seed bank shrub species, such as *Ceanothus* spp., regrow rapidly and can dominate early successional communities, particularly if early colonizers are associated with N-fixing bacteria (Hanson and Stuart 2005; Shatford et al. 2007). Burn severity is also an important control on vegetation recovery. For example, four years after the Hayman Fire in Colorado, ponderosa pine (*Pinus ponderosa*) seedlings were observed in 52% of the plots that burned at low severity and 37% of the plots that burned at moderate severity (Rhoades et al. 2011). However, in areas burned at high severity, it takes longer for seedlings to reestablish and regrow, likely because seed banks have been destroyed (Radek 1997; Rhoades et al. 2011).

Regrowth of riparian zones is a critical factor in buffering against long-term N export to streams (Stephens et al. 2004). Because riparian zones are unique in their biophysical characteristics and moisture regimes, they are often very productive areas, burn at a lower severity than surrounding areas, and can even act as a natural fire break (Kobziar and McBride 2006; Hunsaker and Long 2014). Riparian zones act as physical buffers against nutrient and

debris inputs to the stream and take up much of the N that is delivered downslope from burned areas, which prevent N from entering streams (Stephan et al. 2015; Pinay et al. 2018; Hill 2019). Riparian zones are also a hotspot for denitrification because of their high soil moisture (Burgin et al. 2010). Therefore, even if riparian vegetation is consumed in fire and N plant uptake is reduced, denitrification can still persist and some of the N solutes arriving from upslope ecosystems may still be removed before they enter the stream (Pettit and Naiman 2007). Thus, the preservation and/or rapid regrowth of vegetation is key for preventing large fluxes of N to streams (Robichaud et al. 2021).

Time since fire is also a major control on streamwater chemistry. Systems with rapid vegetation recovery (i.e., during the first growing season) usually have less elevated streamwater N concentrations compared to systems with slower and more prolonged regrowth (Rodríguez-Cardona et al. 2020). N concentrations in streamwater diminish over time as vegetation and microbial communities reestablish and the soil loses its hydrophobic properties (Oliver et al. 2012; Santos et al. 2019). While high severity fire may leave more available N on soil surfaces, it also destroys plant biomass and reduces photosynthetic potential, which can in turn prolong streamwater N export (Jiang et al. 2015). The extent of vegetation removal and rate of recovery also change the light exposure and stream temperature (in-stream effects will be discussed in Sect. 6), which also has further implications for downstream water quality (Betts and Jones 2009; Cooper et al. 2015).

Soil

Fire releases N from vegetation, litter, and surface soils and deposits it on soil surfaces with ash (Turner et al. 2007). Fire also makes N more available in soil by releasing it from organo-clay minerals and promoting rapid mineralization from increased substrate inputs and reduced competition with plants (Giovannini et al. 1990; Turner et al. 2007; Dijkstra et al. 2017). Many studies have observed increases in soil NH_4^+ and NO_3^- concentrations immediately after fire (in both N-saturated and N-limited watersheds) (Christensen 1973; Hobbs and Schimel 1984; Kutieli and Naveh 1987; Turner et al. 2007; Delwiche 2010; Stephan et al. 2015; Fernelius et al. 2017; Goodridge et al. 2018). For example, after a wildfire in Rocky

Mountain mixed conifer forests, soil NH_4^+ concentrations were $44.2 \pm 29.1 \text{ mg kg}^{-1}$, relative to $6.8 \pm 5.6 \text{ mg kg}^{-1}$ in unburned sites; NH_4^+ and NO_3^- concentrations remained elevated for 3 years postfire (Stephan et al. 2015). However, pulses in N availability do not always occur immediately after fire in N-limited systems (Boerner 1982). For example, after a slash burning and prescribed fire in a ponderosa pine forest, soil NO_3^- concentrations did not increase; however, one year after the fire, nitrification rates increased and NO_3^- concentrations became 20 times greater than in unburned areas. Within five years, both NH_4^+ and NO_3^- returned to prefire conditions (Covington et al. 1991; Covington and Sackett 1992).

Elevated postfire nitrification can be problematic for water quality because NO_3^- is highly mobile in soils. Fires typically accelerate nitrification by increasing NH_4^+ availability and soil pH and decreasing vegetation uptake (due to plant mortality) (Rhoades et al. 2011; Hanan et al. 2016a). For example, in a ponderosa pine forest in Northern Arizona, annual net nitrification was found to be 1.98 to $3.51 \text{ g N m}^{-2} \text{ yr}^{-1}$ in burned areas compared to 0.15 to $1.87 \text{ g N m}^{-2} \text{ yr}^{-1}$ in unburned areas (Kurth et al. 2014). While elevated nitrification can last for multiple years following high severity fires (Hanan et al. 2016b), soils typically return to prefire conditions more rapidly after low severity fires (sometimes on the order of months) (Certini 2005). However, elevated postfire nitrification has been observed to last for up to a decade after a strand-replacing fire (Kurth et al. 2014).

Soil microbial biomass can also influence postfire N cycling and export to streams but fire effects on microbial biomass can be highly variable because microbial biomass responds to both direct heating and biogeochemical changes in the soil environment over the course of recovery (DeBano et al. 1998; Choromanska and DeLuca 2002; Smith et al. 2008). The extent to which heat energy penetrates the soil profile and destroys soil microbial biomass is a function of prefire soil water content (drier soils transmit more heat energy) (Massman 2015). For example, severe fire has been found to almost completely sterilize the surface soil layer (0–5 cm) and reduce microbial biomass by 50% in the lower horizon (5–10 cm). Four years after the fire, microbial biomass in the surface layer and lower horizons were 70% and 45% of prefire

biomass content, respectively (Prieto-Fernández et al. 1998).

The rate that microbial biomass recovers and immobilizes or mineralizes available N depends on both prefire N status and postfire litter and carbon inputs. Microbial biomass can rebound quickly when postfire litter and carbon inputs are sufficient (Stirling et al. 2019). Some studies have found that microbial biomass begins returning to prefire conditions one year after fire and that the magnitude of recovery depends on both burn severity and postfire resource availability (i.e., C and N) (Dumontet et al. 1996; Smith et al. 2008). Alternatively, ash inputs can lead to a short-term spike in microbial biomass because it supplies C and other nutrients; however, that can be followed by sharp decline after the first postfire growing season when those resources become depleted (Hanan et al. 2016a). Rapid recolonization of soil microbes in N-limited systems may in turn promote N immobilization and therefore decelerate N cycling and losses to streams (Hanan et al. 2016b; Stirling et al. 2019). However, in systems that are less N-limited, biomass recovery may instead enhance decomposition and N mineralization, which may promote nitrification and leaching (Vourlitis and Hentz 2016).

Water repellent soil conditions or hydrophobic layers generated during fire can continue to reduce hydraulic conductivity and infiltration capacity for years after fire. Hydrophobic layers increase runoff and erosion (Robichaud et al. 2010) while simultaneously reducing N leaching because less water is transported through the soil profile (Imeson et al. 1992; Robichaud 2000; Certini 2005; Balfour et al. 2014; Fernelius et al. 2017). However, water repellency can also delay the reestablishment of vegetation after the fire, which might otherwise take up mobilized N (Fernelius et al. 2017). Thus, if hydrophobic layers enable NO_3^- to accumulate in soil microsites that are hydrologically disconnected from plant roots, leaching may occur once hydrophobic layers breakdown. Hydrophobic layers have been observed for 5 months (McNabb et al. 1989), 15 months (Rodríguez-Alleres et al. 2012), 2 years (Huffman et al. 2001), and 6 years (Dyrness 1976) following fire. The persistence, depth, and size of a hydrophobic layer depends on fire severity (high severity fires have more persistent, larger, and deeper hydrophobic layers), topography (steeper slopes have large repellent layers), and soil

properties (Rodríguez-Alleres et al. 2012; Li et al. 2021).

N export to streams

Watershed and fire characteristics interact to control the extent and duration of water quality impacts (Elliot 2013). For example, strong hydrologic connectivity and gradients from steep slopes can tightly couple burned hillslopes to streams (Bladon et al. 2008; Serpa et al. 2020). The most persistent and pronounced effects on water quality occur when fires are severe, there are strong winds during fire, there is heavy precipitation following the fire, fire occurs on steep slopes, and in places where the soil has low cation-exchange capacity (Ranalli 2004; Oliver et al. 2012). These conditions result in the most persistent and pronounced effects because they make N more available and mobile. Finally, prefire N status can influence postfire N export. For example, in systems that are already N-saturated, postfire N export to streams may be substantially higher (Johnson et al. 2008).

Snowmelt, the timing of precipitation, and high flow periods postfire are major factors influencing spikes in streamwater N. Large runoff events from high intensity rainstorms can flush N from soils, exporting 14 times more N than undisturbed areas (Earl and Blinn 2003; Murphy et al. 2015; Neris et al. 2021). Therefore, a few dry years after a severe fire can reduce the negative effects on water quality (Bladon et al. 2008; Engle et al. 2008; Oliver et al. 2012). Similarly, snowmelt delivers significant amounts of inorganic N from burned areas to streams. Many studies have observed similar postfire N export patterns: an increase in NO_3^- concentrations in runoff during rainfall events and in the months following wildfire, reaching peak concentrations during snowmelt one to two years postfire, and then declining to prefire concentrations in subsequent years (Tiedemann et al. 1978; Feller and Kimmins 1984; MacKay and Robinson 1987; Gluns and Toews 1989; Brass et al. 1996; Williams and Melack 1997; Gerla and Galloway 1998; Ranalli 2004; Bladon et al. 2008; Bayley et al. 2011; Minshall et al. 2011).

Erosion and ash transport are additional postfire nutrient export mechanisms. Erosion is a function of fire severity, topographic characteristics, and climate

(Robichaud et al. 2010). Eroded soils carry adsorbed nutrients, which can be transported downslope and deposited in waterbodies (Certini 2005); NO_3^- is often the main form of N adsorbed to and carried with soils (Pacheco et al. 2015). Nutrients lost via postfire erosion have been estimated to be 1% of the A horizon's C and N content through the displacement of 15 to 18 Mg soil ha^{-1} (Baird et al. 1999). Others have found N mass loss can range from 3.3 to 110 kg ha^{-1} (Pierson et al., 2019). Ash also tends to mobilize postfire through wind and precipitation, and can be directly deposited on soil surfaces and waterbodies (Spencer and Hauer 1991; Rhoades et al. 2011; Neris et al. 2021). When ash is deposited on a waterbody during and after fire, it can cause an immediate spike in N concentrations (Earl and Blinn 2003; Burton et al. 2016).

Fire effects on in-stream processes

Generally, with low severity fires, N levels in adjacent streams are not as persistently elevated as they are following high severity fires. For example, prescribed burns typically only cause a brief pulse of NO_3^- to streams within the first year (Stephens et al. 2004; Delwiche 2010). Much of the N that is mobilized following low severity fires is rapidly taken up and immobilized by surviving vegetation and soil microbes (Rhoades et al. 2011). Following high severity fires, NO_3^- concentrations in streams can be an order of magnitude higher (Stephan et al. 2012), or 50 times greater (Hauer and Spencer 1998), than in unburned watersheds. For example, in temperate conifer forests, N concentrations in streamwater at burned sites were $337 \pm 337 \mu\text{g L}^{-1}$ while unburned sites were $41 \pm 60 \mu\text{g L}^{-1}$ (Stephan et al. 2012).

Fires can also increase stream temperatures by reducing canopy and riparian cover, which allows more light to reach the stream. Streams can be warmer for a decade or longer after a severe wildfire (Rhoades et al. 2019), which can in turn alter in-stream processes. For example, five years after the Hayman Fire, summertime streamwater temperatures in burned catchments were on average 4 °C warmer than unburned catchments (Rhoades et al. 2011). When coupled with increases in nutrient concentrations, warmer streams can increase productivity, alter aquatic food webs and increase algal biomass (Silins

et al. 2014; Cooper et al. 2015). When fire consumes riparian vegetation, algal biomass can be 5–10 times (Klose et al. 2015) or up to 71 times greater (Silins et al. 2014) than in unburned basins. However, when riparian vegetation is not consumed, algal biomass has been found to be much lower (i.e., 10–30% of that in an unburned catchment) (Klose et al. 2015). Similarly, after a high severity fire in Idaho conifer forests, in-stream moss N concentrations increased by about 40% (Stephan et al. 2012). Such increases in gross primary productivity can actually protect against water degradation by reducing downstream NO_3^- delivery (Stephan et al. 2015).

Streams vary in how efficiently they uptake and cycle NO_3^- and NH_4^+ , which is a control on downstream N delivery (Ribot et al. 2017). Recent fire history can alter physical and biological processes that regulate in-stream nutrient uptake and retention (Diemer et al. 2015). Increased levels of NO_3^- in streamwater from fire change the ratios of streamwater dissolved organic matter (DOM), dissolved organic N (DON), and dissolved organic C (DOC) to NO_3^- . Because NO_3^- uptake depends on streamwater biogeochemistry, changes in the relative concentrations of DOM, DOC, and other nutrients can reduce in-stream NO_3^- uptake efficiency and subsequently increase N delivery (Rodríguez-Cardona et al. 2020). Shorter fire return intervals (time in between fires) can delay full recovery of nutrient stoichiometry, which may alter N retention and export (Diemer et al. 2015).

Conclusions

Climate change is affecting wildfire regimes by changing regional climate and weather patterns, drying fuels, and lengthening the fire season (Westerling et al. 2006; Dennison et al. 2014; Abatzoglou and Williams 2016; Hanan et al. 2021). Severely burned areas are a major concern because of high potential for flash floods or surface erosion (Moody et al. 2008; Miller et al. 2012; Neris et al. 2021), either of which could deliver significant quantities of N to streams. As temperatures continue to rise, the greatest increase in area burned in the U.S. will likely occur in the highly productive, historically cold and temperate, wet forests of the Pacific Northwest (Littell et al. 2018). In these forests, wildfire is likely to increase in response to two major drivers: (1) warmer

temperatures, which decrease fuel moisture, and (2) or less precipitation falling as snow, which decreases moisture availability in warmer months (Westerling et al. 2006; Cansler and McKenzie 2014; Littell et al. 2016). The potential consequences of these changes are not entirely known, particularly in areas that experience stand-replacing fire regimes (Halofsky et al. 2018). However, because such forests have naturally high biomass, nutrients, and fuel loads, increases in the occurrence of large, severe fires is likely to promote N export that could devastate drinking water supplies (Lewis et al. 2014).

A major challenge in planning for these events is anticipating where the fire may occur—nearby or far away from municipal drinking water intake facilities—because that will change the magnitude and timing of N fluxes arriving at a facility (Neris et al. 2021). A severe fire occurring adjacent to a higher order stream or upslope from drinking water intake facilities would have a much more significant impact on water quality compared to a fire near a first order stream or further away from intake facilities because the nutrients would have less time to dilute. However, large rivers can still be affected by upstream disturbances when fires are severe (Emmerton et al. 2020). Decision-making support tools linking watershed disturbance to downstream pollutant delivery are being developed, which may be leveraged to assist with this planning (Nunes et al. 2018); however, they are currently limited.

To evaluate the risks of N export from wildland fires, and to help watershed and resource managers prioritize areas for fuel and fire management, we need to further develop and validate predictive tools. Tools exist for predicting nutrient and sediment export, upland erosion, and runoff associated with surface and subsurface flow and deep seepage (Dun et al. 2009; Srivastava et al. 2013; Elliot et al. 2015; Lew et al. 2019). However, there are currently no methods for water utility managers to quantify the amount of N that may be delivered following a wildfire to a water intake or reservoir some distance downstream from the fire. The Water Erosion Prediction Project (WEPP) model is under development to include the Soil and Water Assessment Tool (SWAT) water quality algorithms; WEPPcloud-WATAR (Wildfire Ash Transport And Risk) is also being developed to incorporate an ash transport model and ash loading maps (<https://wepp.cloud/weppcloud/>; <https://swat.tamu.edu>) (Neitsch

et al. 2011; Neris et al. 2020, 2021). Given the importance of ash in a postfire landscape, this is a critical development in modeling nutrient transport from burned slopes to streams. Understanding and predicting postfire N export is critical because excess N in waterbodies can have consequences for human and ecosystem health. Enhanced stream N export can promote eutrophic water bodies and harmful algal blooms (HABs) (Conley et al. 2009). HABs can have major socioeconomic and ecological costs because they are harmful for fish, humans, and other organisms that live in the water, use it recreationally, or consume it (Carmichael and Boyer 2016). For humans, consuming water with too much N can cause blue babies syndrome or have other toxic effects on the body (Knobeloch et al. 2000). Additionally, HABs can lead to the creation of disinfection byproducts during water treatment processes, which can make water unsafe to drink (Foreman et al. 2021). Consequently, algal blooms are a significant public health concern. Several areas of research should be further explored to improve our understanding of wildfire, N cycling, and water quality. These research questions include:

1. How quickly, in what form, and in what concentration will N arrive at a water intake or storage reservoir following a major upland runoff and/or erosion event postfire?
2. What is the best method to predict N movement from fire-disturbed landscapes to municipal water storage reservoirs and water intakes?
3. Under what circumstances will increased N deposition delay or accelerate watershed recovery to prefire N retention?
4. How will climate change-enhanced disturbances and precipitation regimes effect N export from watersheds in the future?

In summary, fire is a powerful force for transforming N cycling in many ecosystems and severe fires have longest lasting and most pronounced effect on N cycling and surface water quality. For municipal watersheds, fire has the capacity to interrupt operations and change drinking water treatment procedures, which may be worsened by climate change. Connecting burned slopes to streams and understanding the mechanisms behind postfire surface water degradation are necessary for managers to protect water resources.

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Conflict of interest The authors know of no conflict of interest.

References

- Abatzoglou JT, Williams AP (2016) Impact of anthropogenic climate change on wildfire across western US forests. *Proc Natl Acad Sci USA* 113(42):11770–11775. <https://doi.org/10.1073/pnas.1607171113>
- Aber JD (1992) Nitrogen cycling and nitrogen saturation in temperate forest ecosystems. *Trends Ecol Evol* 7(7):220–224. [https://doi.org/10.1016/0169-5347\(92\)90048-G](https://doi.org/10.1016/0169-5347(92)90048-G)
- Aber JD, Nadelhoffer KJ, Steudler P, Melillo JM (1989) Nitrogen saturation in northern forest ecosystems. *BioScience* 39(6):378–386. <https://doi.org/10.2307/1311067>
- Aber JD, Melillo JM, Nadelhoffer KJ, Pastor J, Boone RD (1991) Factors controlling nitrogen cycling and nitrogen saturation in northern temperate forest ecosystems. *Ecol Appl* 1(3):303–315. <https://doi.org/10.2307/1941759>
- Baird M, Zabowski D, Everett RL (1999) Wildfire effects on carbon and nitrogen in inland coniferous forests. *Plant Soil* 209(2):233–243. <https://doi.org/10.1023/A:1004602408717>
- Balfour VN, Doerr SH, Robichaud PR (2014) The temporal evolution of wildfire ash and implications for post-fire infiltration. *Int J Wildland Fire* 23(5):733–745. <https://doi.org/10.1071/WF13159>
- Banerjee T, Heilman W, Goodrick S, Hiers JK, Linn R (2020) Effects of canopy midstory management and fuel moisture on wildfire behavior. *Scientific Reports* 10(1):17312. <https://doi.org/10.1038/s41598-020-74338-9>
- Barbero R, Abatzoglou JT, Larkin NK, Kolden CA, Stocks B (2015) Climate change presents increased potential for very large fires in the contiguous United States. *Int J Wildland Fire* 24(7):892. <https://doi.org/10.1071/WF15083>
- Baron JS (2006) Hindcasting nitrogen deposition to determine an ecological critical load. *Ecol Appl* 16(2):433–439. [https://doi.org/10.1890/1051-0761\(2006\)016\[0433:HNDTDA\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2006)016[0433:HNDTDA]2.0.CO;2)
- Bayley SE, Schindler DW, Beaty KG, Parker BR, Stainton MP (2011) Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds: Nitrogen and phosphorus. *Can J Fish Aquat Sci*. <https://doi.org/10.1139/f92-068>

- Betts EF, Jones JB (2009) Impact of wildfire on stream nutrient chemistry and ecosystem metabolism in boreal forest catchments of interior Alaska. *Arct Antarct Alp Res* 41(4):407–417. <https://doi.org/10.1657/1938-4246-41.4.407>
- Binkley D, Fisher RF (2000) Ecology and management of forest soils, 4th edn. Wiley, New York
- Black BL, Fuchigami LH, Coleman GD (2002) Partitioning of nitrate assimilation among leaves, stems and roots of poplar. *Tree Physiol* 22(10):717–724. <https://doi.org/10.1093/treephys/22.10.717>
- Bladon KD, Silins U, Wagner MJ, Stone M, Emelko MB, Mendoza CA, Devito KJ, Boon S (2008) Wildfire impacts on nitrogen concentration and production from headwater streams in southern Alberta's Rocky Mountains. *Can J For Res* 38(9):2359–2371. <https://doi.org/10.1139/X08-071>
- Bladon KD, Emelko MB, Silins U, Stone M (2014) Wildfire and the future of water supply. *Environ Sci Technol* 48(16):8936–8943. <https://doi.org/10.1021/es500130g>
- Bodí MB, Martin DA, Balfour VN, Santín C, Doerr SH, Pereira P, Cerdà A, Mataix-Solera J (2014) Wildland fire ash: Production, composition and eco-hydro-geomorphic effects. *Earth Sci Rev* 130:103–127. <https://doi.org/10.1016/j.earscirev.2013.12.007>
- Boerner REJ (1982) Fire and nutrient cycling in temperate ecosystems. *Bioscience* 32(3):187–192. <https://doi.org/10.2307/1308941>
- Bourgeois I, Clément J-C, Caillon N, Savarino J (2019) Foliar uptake of atmospheric nitrate by two dominant subalpine plants: Insights from in situ triple-isotope analysis. *New Phytol* 223(4):1784–1794. <https://doi.org/10.1111/nph.15761>
- Brass JA, Ambrosia VG, Riggan PJ, Sebesta PD (1996) Consequences of fire on aquatic nitrate and phosphate dynamics in Yellowstone National Park. In: *Proceeding of the 2nd Biennial Conference on the Greater Yellowstone Ecosystem*, pp 53–57. https://www.fs.fed.us/psw/publications/riggan/psw_1996_riggan001.pdf
- Brockley RP, Trowbridge RL, Ballard TM, Macadam AM (1992) Nutrient management in interior forest types. In: Chappell WHN, Weetman GF, Miller RE (eds) *Proc. forest fertilization: sustaining and improving nutrition and growth of western forests*, Seattle. Institute of For. Resources Contrib. 73, College of For. Resources, Univ. Wash., Seattle, WA, pp 43–64
- Burgin AJ, Groffman PM, Lewis DN (2010) Factors regulating denitrification in a riparian wetland. *Soil Sci Soc Am J* 74(5):1826–1833. <https://doi.org/10.2136/sssaj2009.0463>
- Burton CA, Hoefen TM, Plumlee GS, Baumberger KL, Backlin AR, Gallegos E, Fisher RN (2016) Trace elements in stormflow, ash, and burned soil following the 2009 station fire in Southern California. *PLoS ONE* 11(5):e0153372. <https://doi.org/10.1371/journal.pone.0153372>
- Butler BW, Anderson WR, Catchpole EA (2007) Influence of slope on fire spread rate (Proceedings RMRS-P-46CD). USDA Forest Service, Rocky Mountain Research Station, Ft. Collins, CO
- Cameron KC, Di HJ, Moir JL (2013) Nitrogen losses from the soil/plant system: a review. *Ann Appl Biol* 162(2):145–173. <https://doi.org/10.1111/aab.12014>
- Cansler CA, McKenzie D (2014) Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern cascade range, USA. *Ecol Appl* 24(5):1037–1056. <https://doi.org/10.1890/13-1077.1>
- Caon L, Vallejo VR, Ritsema CJ, Geissen V (2014) Effects of wildfire on soil nutrients in mediterranean ecosystems. *Earth Sci Rev* 139:47–58. <https://doi.org/10.1016/j.earscirev.2014.09.001>
- Carmichael WW, Boyer GL (2016) Health impacts from cyanobacteria harmful algae blooms: implications for the North American Great Lakes. *Harmful Algae* 54:194–212. <https://doi.org/10.1016/j.hal.2016.02.002>
- Certini G (2005) Effects of fire on properties of forest soils: a review. *Oecologia* 143(1):1–10. <https://doi.org/10.1007/s00442-004-1788-8>
- Chen J, Xiao G, Kuzyakov Y, Jenerette G, Ma Y, Liu W, Wang Z, Shen W (2017) Soil nitrogen transformation responses to seasonal precipitation changes are regulated by changes in functional microbial abundance in a subtropical forest. *Biogeosciences* 14:2513–2525. <https://doi.org/10.5194/bg-14-2513-2017>
- Choromanska U, DeLuca TH (2002) Microbial activity and nitrogen mineralization in forest mineral soils following heating: evaluation of post-fire effects. *Soil Biol Biochem* 34(2):263–271. [https://doi.org/10.1016/S0038-0717\(01\)00180-8](https://doi.org/10.1016/S0038-0717(01)00180-8)
- Christensen NL (1973) Fire and the nitrogen cycle in California chaparral. *Science* 181(4094):66–68. <https://doi.org/10.1126/science.181.4094.66>
- Clean Water Drinking Act, 141.11 § Title 40. Chapter 1. Subchapter D. Part 141. Subpart B: Maximum Contaminant Levels (1972). <https://www.govinfo.gov/content/pkg/CFR-2019-title40-vol25/pdf/CFR-2019-title40-vol25-part141-subpartB.pdf>
- Cole DW, Rapp M (1981) Elemental cycling in forest ecosystems, Chapter 6. Dynamic properties of forest ecosystems. Cambridge University Press, Cambridge, p 70
- Compton JE, Church MR, Larned ST, Hogsett WE (2003) Nitrogen export from forested watersheds in the Oregon coast range: the role of N₂-fixing red alder. *Ecosystems* 6(8):773–785. <https://doi.org/10.1007/s10021-002-0207-4>
- Conley DJ, Paerl HW, Howarth RW, Boesch DF, Seitzinger SP, Havens KE, Lancelot C, Likens GE (2009) Controlling eutrophication: nitrogen and phosphorus. *Science* 323(5917):1014–1015. <https://doi.org/10.1126/science.1167755>
- Cook EM, Sponseller R, Grimm NB, Hall SJ (2018) Mixed method approach to assess atmospheric nitrogen deposition in arid and semi-arid ecosystems. *Environ Pollut* 239:617–630. <https://doi.org/10.1016/j.envpol.2018.04.013>
- Cooper SD, Page HM, Wiseman SW, Klose K, Bennett D, Even T, Sadro S, Nelson CE, Dudley TL (2015) Physicochemical and biological responses of streams to wildfire severity in riparian zones. *Freshw Biol* 60(12):2600–2619. <https://doi.org/10.1111/fwb.12523>
- Covington WW, Sackett SS (1992) Soil mineral nitrogen changes following prescribed burning in ponderosa pine. *For Ecol Manage* 54(1):175–191. [https://doi.org/10.1016/0378-1127\(92\)90011-W](https://doi.org/10.1016/0378-1127(92)90011-W)

- Covington WW, DeBano LF, Huntsberger TG (1991) Soil nitrogen changes associated with slash pile burning in pinyon-juniper woodlands. *For Sci* 37(1):347–355
- De Boer W, Kowalchuk GA (2001) Nitrification in acid soils: micro-organisms and mechanisms. *Soil Biol Biochem* 33(7):853–866. [https://doi.org/10.1016/S0038-0717\(00\)00247-9](https://doi.org/10.1016/S0038-0717(00)00247-9)
- DeBano LF, Neary DG, Ffolliott PF (1998) Fire effects on ecosystems. Wiley, New York
- Delgado-Baquerizo M, Maestre FT, Gallardo A, Quero JL, Ochoa V, García-Gómez M, Escolar C, García-Palacios P, Berdugo M, Valencia E, Gozalo B, Noumi Z, Derak M, Wallenstein MD (2013) Aridity modulates N availability in arid and semiarid Mediterranean grasslands. *PLoS ONE* 8(4):e59807. <https://doi.org/10.1371/journal.pone.0059807>
- Delwiche J (2010) After the fire, follow the nitrogen. *JFSP Briefs* 92:1–7. <https://digitalcommons.unl.edu/jfspbriefs/80/>
- Dennison PE, Brewer SC, Arnold JD, Moritz MA (2014) Large wildfire trends in the western United States, 1984–2011. *Geophys Res Lett* 41(8):2928–2933. <https://doi.org/10.1002/2014GL059576>
- Diemer LA, McDowell WH, Wymore AS, Prokushkin AS (2015) Nutrient uptake along a fire gradient in boreal streams of Central Siberia. *Freshw Sci* 34(4):1443–1456. <https://doi.org/10.1086/683481>
- Dijkstra FA, Jenkins M, de Courcelles VR, Keitel C, Barbour MM, Kayler ZE, Adams MA (2017) Enhanced decomposition and nitrogen mineralization sustain rapid growth of Eucalyptus regnans after wildfire. *J Ecol* 105(1):229–236. <https://doi.org/10.1111/1365-2745.12663>
- Dirnböck T, Kobler J, Kraus D, Grote R, Kiese R (2016) Impacts of management and climate change on nitrate leaching in a forested karst area. *J Environ Manage* 165:243–252. <https://doi.org/10.1016/j.jenvman.2015.09.039>
- Dumontet S, Diné H, Scopla A, Mazzatura A, Saracino A (1996) Post-fire soil microbial biomass and nutrient content of a pine forest soil from a dunal Mediterranean environment. *Soil Biol Biochem* 28(10):1467–1475. [https://doi.org/10.1016/S0038-0717\(96\)00160-5](https://doi.org/10.1016/S0038-0717(96)00160-5)
- Dun S, Elliot WJ, Robichaud PR, Flanagan DC, Frankenberger JR, Brown RE, Xu AD (2009) Adapting the water erosion prediction project (WEPP) model for forest applications. *J Hydrol* 336:45–54. <https://doi.org/10.1016/j.jhydrol.2008.12.019>
- Dyrness CT (1976) Effect of wildfire on soil wettability in the high Cascades of Oregon (Research Paper PNW-202). USDA Forest Service, Pacific Northwest Station, Seattle, WA
- Earl SR, Blinn DW (2003) Effects of wildfire ash on water chemistry and biota in South-Western U.S.A. streams. *Freshw Biol* 48(6):1015–1030. <https://doi.org/10.1046/j.1365-2427.2003.01066.x>
- Eddy-Miller CA, Peterson DA, Wheeler JD, Edmiston CS, Taylor ML, Leemon DJ (2013) Characterization of water quality and biological communities, Fish Creek, Teton County, Wyoming, 2007–2011 (Scientific Investigations Report No. 2013–5117; Issues 2013–5117). US Geological Survey, Renton, p 90
- Elliot WJ (2013) Erosion processes and prediction with WEPP technology in forests in the Northwestern U.S. *Trans ASABE* 56(2):563–579. <https://doi.org/10.13031/2013.42680>
- Elliot, W. J., Brooks, E. S., Traeumer, D., & Dobre, M. (2015). Extending WEPP technology to predict fine sediment and phosphorus delivery from forested hillslopes. SEDHYD 2015 Interagency Conference, Reno, NV
- Emelko MB, Silins U, Bladon KD, Stone M (2011) Implications of land disturbance on drinking water treatability in a changing climate: demonstrating the need for “source water supply and protection” strategies. *Water Res* 45(2):461–472. <https://doi.org/10.1016/j.watres.2010.08.051>
- Emmert CA, Cooke CA, Hustins S, Silins U, Emelko MB, Lewis T, Kruk MK, Taube N, Zhu D, Jackson B, Stone M, Kerr JG, Orwin JF (2020) Severe western Canadian wildfire affects water quality even at large basin scales. *Water Res* 183:116071. <https://doi.org/10.1016/j.watres.2020.116071>
- Engle D, Sickman J, Moore C, Esperanza A, Melack J, Keeley J (2008) Biogeochemical legacy of prescribed fire in a giant sequoia - Mixed conifer forest: A 16-year record of watershed balances. *J Geophys Res.* <https://doi.org/10.1029/2006JG000391>
- Eshleman KN, Sabo RD (2016) Declining nitrate-N yields in the Upper Potomac River Basin: what is really driving progress under the Chesapeake Bay restoration? *Atmos Environ* 146:280–289. <https://doi.org/10.1016/j.atmosenv.2016.07.004>
- Eshleman KN, Sabo RD, Kline KM (2013) Surface water quality is improving due to declining atmospheric N deposition. *Environ Sci Technol* 47(21):12193–12200. <https://doi.org/10.1021/es4028748>
- Feller MC, Kimmins JP (1984) Effects of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. *Water Resour Res* 20(1):29–40. <https://doi.org/10.1029/WR020i001p00029>
- Feng Z, Rütting T, Pleijel H, Wallin G, Reich PB, Kammann CI, Newton PCD, Kobayashi K, Luo Y, Uddling J (2015) Constraints to nitrogen acquisition of terrestrial plants under elevated CO₂. *Glob Change Biol* 21(8):3152–3168. <https://doi.org/10.1111/gcb.12938>
- Fenn ME, Poth MA (1998) Indicators of nitrogen status in California forests. In: Bytnerowicz A, Arbaugh MJ, Schilling SL (eds) *Tech. Coords Proceedings of the international symposium on air pollution and climate change effects on forest ecosystems*. Gen. Tech. Rep. PSW-GTR-166. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA, pp 123–130, 166. <https://www.fs.usda.gov/treearch/pubs/26958>
- Fenn ME, Baron JS, Allen EB, Rueth HM, Nydick KR, Geiser L, Bowman WD, Sickman JO, Meixner T, Johnson DW, Neitlich P (2003) Ecological effects of nitrogen deposition in the Western United States. *Bioscience* 53(4):404. [https://doi.org/10.1641/0006-3568\(2003\)053\[0404:EEONDI\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2003)053[0404:EEONDI]2.0.CO;2)
- Fenn ME, Jovan S, Yuan F, Geiser L, Meixner T, Gimeno BS (2008) Empirical and simulated critical loads for nitrogen

- deposition in California mixed conifer forests. *Environ Pollut* 155:492–511. <https://doi.org/10.1016/j.envpol.2008.03.019>
- Fenn ME, Ross CS, Schilling SL, Baccus WD, Larrabee MA, Lofgren RA (2013) Atmospheric deposition of nitrogen and sulfur and preferential canopy consumption of nitrate in forests of the Pacific Northwest, USA. *For Ecol Manage* 302:240–253. <https://doi.org/10.1016/j.foreco.2013.03.042>
- Fernelius KJ, Madsen MD, Hopkins BG, Bansal S, Anderson VJ, Eggett DL, Roundy BA (2017) Post-fire interactions between soil water repellency, soil fertility and plant growth in soil collected from a burned piñon-juniper woodland. *J Arid Environ* 144:98–109. <https://doi.org/10.1016/j.jaridenv.2017.04.005>
- Fisk MC, Zak DR, Crow TR (2002) Nitrogen storage and cycling in old- and second-growth northern hardwood forests. *Ecology* 83(1):73–87. [https://doi.org/10.1890/0012-9658\(2002\)083\[0073:NSACIO\]2.0.CO;2](https://doi.org/10.1890/0012-9658(2002)083[0073:NSACIO]2.0.CO;2)
- Foreman K, Renwick DV, McCabe M, Cadwallader A, Holsinger H, Kormondy C, Albert R (2021) Effects of harmful algal blooms on regulated disinfection byproducts: findings from five utility case studies. *AWWA Water Sci* 3(3):e1223. <https://doi.org/10.1002/aws2.1223>
- Gerla PJ, Galloway JM (1998) Water quality of two streams near Yellowstone Park, Wyoming, following the 1988 Clover-Mist wildfire. *Environ Geol* 36(1):127–136. <https://doi.org/10.1007/s002540050328>
- Gessel SP, Cole DW, Steinbrenner EC (1973) Nitrogen balances in forest ecosystems of the Pacific Northwest. *Soil Biol Biochem* 5(1):19–34. [https://doi.org/10.1016/0038-0717\(73\)90090-4](https://doi.org/10.1016/0038-0717(73)90090-4)
- Giovannini C, Lucchesi S, Giachetti M (1990) Effects of heating on some chemical parameters related to soil fertility and plant growth. *Soil Sci* 149(6):344–350
- Gluns DR, Toews DA (1989) Effect of a major wildfire on water quality in southeastern British Columbia. In: *Proceedings, symposium on headwaters hydrology*. American Water Resources Association, Bethesda, MD, pp 487–499
- Goodridge BM, Hanan EJ, Aguilera R, Wetherley EB, Chen Y-J, D'Antonio CM, Melack JM (2018) Retention of nitrogen following wildfire in a chaparral ecosystem. *Ecosystems* 21(8):1608–1622. <https://doi.org/10.1007/s10021-018-0243-3>
- Gresswell R (1999) Fire and aquatic ecosystems in forested biomes of North America. *Trans Am Fish Soc* 128:193–221. [https://doi.org/10.1577/1548-8659\(1999\)128%3c0193:FAAEIF%3e2.0.CO;2](https://doi.org/10.1577/1548-8659(1999)128%3c0193:FAAEIF%3e2.0.CO;2)
- Gundersen P, Schmidt IK, Raulund-Rasmussen K (2006) Leaching of nitrate from temperate forests effects of air pollution and forest management. *Environ Rev*. <https://doi.org/10.1139/a05-015>
- Hallema DW, Robinne F-N, Bladon KD (2018) Reframing the challenge of global wildfire threats to water supplies. *Earth's Future* 6(6):772–776. <https://doi.org/10.1029/2018EF000867>
- Halofsky JS, Conklin DR, Donato DC, Halofsky JE, Kim JB (2018) Climate change, wildfire, and vegetation shifts in a high-inertia forest landscape: Western Washington, U.S.A. *PLoS ONE* 13(12):e0209490. <https://doi.org/10.1371/journal.pone.0209490>
- Hanan EJ, D'Antonio CM, Roberts DA, Schimel JP (2016a) Factors regulating nitrogen retention during the early stages of recovery from fire in coastal chaparral ecosystems. *Ecosystems* 19(5):910–926. <https://doi.org/10.1007/s10021-016-9975-0>
- Hanan EJ, Schimel JP, Dowdy K, D'Antonio CM (2016b) Effects of substrate supply, pH, and char on net nitrogen mineralization and nitrification along a wildfire-structured age gradient in chaparral. *Soil Biol Biochem* 95:87–99. <https://doi.org/10.1016/j.soilbio.2015.12.017>
- Hanan EJ, Tague CN, Schimel JP (2017) Nitrogen cycling and export in California chaparral: the role of climate in shaping ecosystem responses to fire. *Ecol Monogr* 87(1):76–90. <https://doi.org/10.1002/ecm.1234>
- Hanan EJ, Ren J, Tague CL, Kolden CA, Abatzoglou JT, Bart RR, Kennedy MC, Liu M, Adam JC (2021) How climate change and fire exclusion drive wildfire regimes at actionable scales. *Environ Res Lett* 16(2):024051. <https://doi.org/10.1088/1748-9326/abd78e>
- Hanson JJ, Stuart JD (2005) Vegetation responses to natural and salvage logged fire edges in Douglas-fir/hardwood forests. *For Ecol Manage* 214(1/2/3):266–278
- Hauer FR, Spencer CN (1998) Phosphorus and nitrogen dynamics in streams associated with wildfire: a study of immediate and long-term effects. *Int J Wildland Fire* 8:183–198
- Hill AR (2019) Groundwater nitrate removal in riparian buffer zones: a review of research progress in the past 20 years. *Biogeochemistry* 143(3):347–369. <https://doi.org/10.1007/s10533-019-00566-5>
- Hobbs NT, Schimel DS (1984) Fire effects on nitrogen mineralization and fixation in mountain shrub and grassland communities. *J Range Manage* 37(5):402. <https://doi.org/10.2307/3899624>
- Hohner AK, Rhoades CC, Wilkerson P, Rosario-Ortiz FL (2019) Wildfires alter forest watersheds and threaten drinking water quality. *Acc Chem Res* 52(5):1234–1244. <https://doi.org/10.1021/acs.accounts.8b00670>
- Holloway JM, Dahlgren RA (2002) Nitrogen in rock: occurrences and biogeochemical implications. *Glob Biogeochem Cycles* 16(4):65–165–17. <https://doi.org/10.1029/2002GB001862>
- Homyak PM, Sickman JO, Miller AE, Melack JM, Meixner T, Schimel JP (2014) Assessing nitrogen-saturation in a seasonally dry chaparral watershed: limitations of traditional indicators of N-saturation. *Ecosystems* 17(7):1286–1305
- Houlton BZ, Morford SL, Dahlgren RA (2018) Convergent evidence for widespread rock nitrogen sources in Earth's surface environment. *Science* 360(6384):58–62. <https://doi.org/10.1126/science.aan4399>
- Huffman EL, MacDonald LH, Stednick JD (2001) Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado front range. *Hydrol Process* 15(15):2877–2892. <https://doi.org/10.1002/hyp.379>
- Hunsaker CT, Long JW (2014) Forested riparian areas. Gen. Tech. Rep. PSW-GTR-247. US Department of Agriculture Forest Service Pacific Southwest Research Station, Albany, CA, pp 323–340
- Imeson AC, Verstraten JM, van Mulligen EJ, Sevink J (1992) The effects of fire and water repellency on infiltration and

- runoff under mediterranean type forest. *CATENA* 19(3):345–361. [https://doi.org/10.1016/0341-8162\(92\)90008-Y](https://doi.org/10.1016/0341-8162(92)90008-Y)
- Jiang Y, Rastetter EB, Rocha AV, Pearce AR, Kwiatkowski BL, Shaver GR (2015) Modeling carbon–nutrient interactions during the early recovery of tundra after fire. *Ecol Appl* 25(6):1640–1652. <https://doi.org/10.1890/14-1921.1>
- Jin-yan Y, Jing F (2003) Review of study on mineralization, saturation and cycle of nitrogen in forest ecosystems. *J for Res* 14(3):239–243. <https://doi.org/10.1007/BF02856838>
- Johnson DW, Lindberg SE (eds) (1992) Atmospheric deposition and forest nutrient cycling: a synthesis of the integrated forest study. Springer, New York
- Johnson D, Turner J (2014) Nitrogen budgets of forest ecosystems: a review. *For Ecol Manage* 318:370–379. <https://doi.org/10.1016/j.foreco.2013.08.028>
- Johnson DW, Fenn ME, Miller WW, Hunsaker CF (2008) Chapter 18 fire effects on carbon and nitrogen cycling in forests of the Sierra Nevada. In: Bytnerowicz A, Arbaugh MJ, Riebau AR, Andersen C (eds) *Developments in environmental science*, vol 8. Elsevier, Amsterdam, pp 405–423
- Johnson DW, Walker RF, Glass DW, Miller WW, Murphy JD, Stein CM (2012) The effect of rock content on nutrients in a Sierra Nevada forest soil. *Geoderma* 173–174:84–93. <https://doi.org/10.1016/j.geoderma.2011.12.020>
- Kelley CJ, Keller CK, Brooks ES, Smith JL, Orr CH, Evans RD (2017) Water and nitrogen movement through a semiarid dryland agricultural catchment: Seasonal and decadal trends. *Hydrol Process* 31(10):1889–1899
- Kirkman K (2011) Burning issues: sustainability and management of Australia's southern forests. *Afr J Range Forage Sci* 28(3):155–156. <https://doi.org/10.2989/10220119.2011.642098>
- Klopatek JM, Barry MJ, Johnson DW (2006) Potential canopy interception of nitrogen in the Pacific Northwest, USA. *For Ecol Manage* 234(1–3):344–354. <https://doi.org/10.1016/j.foreco.2006.07.019>
- Klose K, Cooper SD, Bennett DM (2015) Effects of wildfire on stream algal abundance, community structure, and nutrient limitation. *Freshw Sci* 34(4):1494–1509. <https://doi.org/10.1086/683431>
- Knicker H, Almendros G, González-Vila FJ, Martin F, Lüdemann H-D (1996) ¹³C- and ¹⁵N-NMR spectroscopic examination of the transformation of organic nitrogen in plant biomass during thermal treatment. *Soil Biol Biochem* 28(8):1053–1060. [https://doi.org/10.1016/0038-0717\(96\)00078-8](https://doi.org/10.1016/0038-0717(96)00078-8)
- Knobloch L, Salna B, Hogan A, Postle J, Anderson H (2000) Blue babies and nitrate-contaminated well water. *Environ Health Perspect* 108(7):675–678. <https://doi.org/10.1289/ehp.00108675>
- Kobziar LN, McBride JR (2006) Wildfire burn patterns and riparian vegetation response along two northern Sierra Nevada streams. *For Ecol Manage* 222(1–3):254–265. <https://doi.org/10.1016/j.foreco.2005.10.024>
- Kurth VJ, Hart SC, Ross CS, Kaye JP, Fulé PZ (2014) Stand-replacing wildfires increase nitrification for decades in southwestern ponderosa pine forests. *Oecologia* 175(1):395–407. <https://doi.org/10.1007/s00442-014-2906-x>
- Kutiel P, Naveh Z (1987) The effect of fire on nutrients in a pine forest soil. *Plant Soil* 104(2):269–274. <https://doi.org/10.1007/BF02372541>
- Lew R, Dobre M, Brooks ES, Robichaud PR, Elliot WJ, Srivastava A (2019) WEPPcloud Watershed. Moscow ID: University of Idaho and US Department of Agriculture, Forest Service, Rocky Mountain Research Station. <https://wepp1.nkn.uidaho.edu/weppcloud/> or <https://forest.moscowfsl.wsu.edu/fswepp/>
- Lewis DB, Castellano MJ, Kaye JP (2014) Forest succession, soil carbon accumulation, and rapid nitrogen storage in poorly remineralized soil organic matter. *Ecology* 95(10):2687–2693. <https://doi.org/10.1890/13-2196.1>
- Li Y, Schichtel BA, Walker JT, Schwede DB, Chen X, Lehmann CMB, Puchalski MA, Gay DA, Collett JL (2016) Increasing importance of deposition of reduced nitrogen in the United States. *Proc Natl Acad Sci USA* 113(21):5874–5879. <https://doi.org/10.1073/pnas.1525736113>
- Li Y, Chapman SJ, Nicol GW, Yao H (2018) Nitrification and nitrifiers in acidic soils. *Soil Biol Biochem* 116:290–301. <https://doi.org/10.1016/j.soilbio.2017.10.023>
- Li Q, Ahn S, Kim T, Im S (2021) Post-fire impacts of vegetation burning on soil properties and water repellency in a pine forest, South Korea. *Forests* 12(6):708. <https://doi.org/10.3390/f12060708>
- Likens GE, Bormann FH, Johnson NM, Fisher DW, Pierce RS (1970) Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook Watershed-ecosystem. *Ecol Monogr* 40(1):23–47. <https://doi.org/10.2307/1942440>
- Lin J, Compton JE, Leibowitz SG, Mueller-Warrant G, Matthews W, Schoenholtz SH, Evans DM, Coulombe RA (2019) Seasonality of nitrogen balances in a Mediterranean climate watershed, Oregon, US. *Biogeochemistry* 142(2):247–264. <https://doi.org/10.1007/s10533-018-0532-0>
- Lindaas J, Pollack IB, Garofalo LA, Pothier MA, Farmer DK, Kreidenweis SM, Campos TL, Flocke F, Weinheimer AJ, Montzka DD, Tyndall GS, Palm BB, Peng Q, Thornton JA, Permar W, Wielgasz C, Hu L, Ottmar RD, Restaino JC, Fischer EV (2021) Emissions of reactive nitrogen from Western U.S. wildfires during summer 2018. *J Geophys Res* 126(2):e2020JD032657. <https://doi.org/10.1029/2020JD032657>
- Littell JS, Peterson DL, Riley KL, Liu Y, Luce CH (2016) A review of the relationships between drought and forest fire in the United States. *Glob Change Biol* 22(7):2353–2369. <https://doi.org/10.1111/gcb.13275>
- Littell JS, McKenzie D, Wan HY, Cushman SA (2018) Climate change and future wildfire in the Western United States: An ecological approach to nonstationarity. *Earth's Future* 6(8):1097–1111. <https://doi.org/10.1029/2018EF000878>
- Lovett GM, Lindberg SE (1993) Atmospheric deposition and canopy interactions of nitrogen in forests. *Can J for Res* 23(8):1603–1616. <https://doi.org/10.1139/x93-200>
- MacKay SM, Robinson G (1987) Effects of wildfire and logging on streamwater chemistry and cation exports of small forested catchments in Southeastern New South Wales, Australia. *Hydrol Process* 1(4):359–384. <https://doi.org/10.1002/hyp.3360010405>

- Martin TL, Trevors JT, Kaushik NK (1999) Soil microbial diversity, community structure and denitrification in a temperate riparian zone. *Biodivers Conserv* 8(8):1057–1078. <https://doi.org/10.1023/A:1008899722286>
- Massman WJ (2015) A non-equilibrium model for soil heating and moisture transport during extreme surface heating: the soil (heat–moisture–vapor) HMV-model version 1. *Geosci Model Dev* 8(11):3659–3680. <https://doi.org/10.5194/gmd-8-3659-2015>
- Mayer PM, Reynolds S, Canfield T, McCutchen M (2005) Riparian buffer width, vegetative cover, and nitrogen removal effectiveness: a review of current science and regulations. Environmental Protection Agency, Washington DC, p 40
- McCulley RL, Jackson RB (2012) Conversion of tallgrass prairie to woodland: consequences for carbon and nitrogen cycling. *Am Midl Nat* 167(2):307–321. <https://doi.org/10.1674/0003-0031-167.2.307>
- McNabb DH, Gaweda F, Froehlich HA (1989) Infiltration, water repellency, and soil moisture content after broadcast burning a forest site in southwest Oregon. *J Soil Water Conserv* 44(1):87–90
- Miller M, MacDonald L, Robichaud P, Elliot W (2012) Predicting post-fire hillslope erosion in forest lands of the western United States. *Int J Wildland Fire* 20:982–999. <https://doi.org/10.1071/WF09142>
- Minshall GW, Robinson CT, Lawrence DE (2011) Postfire responses of lotic ecosystems in Yellowstone National Park, U.S.A. *Can J Fish Aquat Sci* 54:2509–2525. <https://doi.org/10.1139/f97-160>
- Moghimian N, Jalali SG, Kooch Y, Rey A (2020) Downed logs improve soil properties in old-growth temperate forests of northern Iran. *Pedosphere* 30(3):378–389. [https://doi.org/10.1016/S1002-0160\(17\)60424-7](https://doi.org/10.1016/S1002-0160(17)60424-7)
- Moody JA, Martin DA, Haire SL, Kinner DA (2008) Linking runoff response to burn severity after a wildfire. *Hydrol Process* 22(13):2063–2074. <https://doi.org/10.1002/hyp.6806>
- Morford SL, Houlton BZ, Dahlgren RA (2011) Increased forest ecosystem carbon and nitrogen storage from nitrogen rich bedrock. *Nature* 477(7362):78–81. <https://doi.org/10.1038/nature10415>
- Murphy SF, Writer JH, McCleskey RB, Martin DA (2015) The role of precipitation type, intensity, and spatial distribution in source water quality after wildfire. *Environ Res Lett* 10(8):084007. <https://doi.org/10.1088/1748-9326/10/8/084007>
- Neary DG, Klopatek CC, DeBano LF, Ffolliott PF (1999) Fire effects on belowground sustainability: a review and synthesis. *For Ecol Manage* 122(1–2):51–71. [https://doi.org/10.1016/S0378-1127\(99\)00032-8](https://doi.org/10.1016/S0378-1127(99)00032-8)
- Neary DG, Ryan KC, DeBano LF (2005) Wildland fire in ecosystems: Effects of fire on soils and water (RMRS-GTR-42-V4). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ft. Collins, CO
- Neitsch SL, Arnold JG, Kiniry, JR, Williams JR (2011). Soil and Water Assessment Tool Theoretical Documentation Version 2009 (Technical Report No. 406, p. 647). Texas Water Resources Institute
- Neris J, Lew R, Doerr SH, Robichaud PR, Elliot WJ, Santin C, Sheridan G (2020). WEPPcloud-PEP WATAR Wildfire Ash Transport and Risk. University of Idaho and US Department of Agriculture, Forest Service, Rocky Mountain Research Station. <https://wepp1.nkn.uidaho.edu/weppcloud/> or <https://forest.moscowfsl.wsu.edu/fswepp/>
- Neris J, Santin C, Lew R, Robichaud PR, Elliot WJ, Lewis SA, Sheridan G, Rohlf A-M, Ollivier Q, Oliveira L, Doerr SH (2021) Designing tools to predict and mitigate impacts on water quality following the Australian 2019/2020 wildfires: insights from Sydney's largest water supply catchment. *Integr Environ Assess Manage*. <https://doi.org/10.1002/ieam.4406>
- Nguyen T, Xu C, Tahmasbian I, Che R, Xu Z, Zhou X, Wallace H, Hosseini Bai S (2017) Effects of biochar on soil available inorganic nitrogen: a review and meta-analysis. *Geoderma*. <https://doi.org/10.1016/j.geoderma.2016.11.004>
- Noyce GL, Kirwan ML, Rich RL, Megonigal JP (2019) Asynchronous nitrogen supply and demand produce nonlinear plant allocation responses to warming and elevated CO₂. *Proc Natl Acad Sci USA* 116(43):21623–21628. <https://doi.org/10.1073/pnas.1904990116>
- Nunes JP, Doerr SH, Sheridan G, Neris J, Santin C, Emelko MB, Silins U, Robichaud PR, Elliot WJ, Keizer J (2018) Assessing water contamination risk from vegetation fires: challenges, opportunities and a framework for progress. *Hydrol Process* 32(5):687–694. <https://doi.org/10.1002/hyp.11434>
- Oliver AA, Reuter JE, Heyvaert AC, Dahlgren RA (2012) Water quality response to the Angora Fire, Lake Tahoe, California. *Biogeochemistry* 111(1/3):361–376
- Pacheco FAL, Santos RMB, Sanches Fernandes LF, Pereira MG, Cortes RMV (2015) Controls and forecasts of nitrate yields in forested watersheds: a view over mainland Portugal. *Sci Total Environ* 537:421–440. <https://doi.org/10.1016/j.scitotenv.2015.07.127>
- Parks SA, Miller C, Abatzoglou JT, Holsinger LM, Parisien M-A, Dobrowski SZ (2016) How will climate change affect wildland fire severity in the western US? *Environ Res Lett* 11(3):035002. <https://doi.org/10.1088/1748-9326/11/3/035002>
- Parson A, Robichaud PR, Lewis SA, Napper C, Clark JT (2010) Field guide for mapping post-fire soil burn severity (RMRS-GTR-243). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ft. Collins, CO
- Parton WJ, Holland EA, Grosso SJD, Hartman MD, Martin RE, Mosier AR, Ojima DS, Schimel DS (2001) Generalized model for NO_x and N₂O emissions from soils. *J Geophys Res* 106(D15):17403–17419. <https://doi.org/10.1029/2001JD900101>
- Pettit NE, Naiman RJ (2007) Fire in the riparian zone: Characteristics and ecological consequences. *Ecosystems* 10(5):673–687. <https://doi.org/10.1007/s10021-007-9048-5>
- Pierson FB, Robichaud PR, Spaeth KE, Moffet CA (2003) Impacts of fire on hydrology and erosion in steep mountain big sagebrush communities. First interagency conference on research in the watersheds: October 27–30, 2003. U.S. Dept. of Agriculture Agricultural Research Service, Washington DC, pp 625–630

- Pierson DN, Robichaud PR, Rhoades CC, Brown RE (2019) Soil carbon and nitrogen eroded after severe wildfire and erosion mitigation treatments. *Int J Wildland Fire* 28(10):814–821. <https://doi.org/10.1071/WF18193>
- Pinay G, Bernal S, Abbott BW, Lupon A, Marti E, Sabater F, Krause S (2018) Riparian corridors: a new conceptual framework for assessing nitrogen buffering across biomes. *Front Environ Sci*. <https://doi.org/10.3389/fenvs.2018.00047>
- Post WM, Pastor J, Zinke PJ, Stangenberger AG (1985) Global patterns of soil nitrogen storage. *Nature* 317(6038):613–616. <https://doi.org/10.1038/317613a0>
- Prieto-Fernández A, Acea MJ, Carballas T (1998) Soil microbial and extractable C and N after wildfire. *Biol Fertil Soils* 27(2):132–142. <https://doi.org/10.1007/s003740050411>
- Pugnaire FI, Morillo JA, Peñuelas J, Reich PB, Bardgett RD, Gaxiola A, Wardle DA, van der Putten WH (2019) Climate change effects on plant-soil feedbacks and consequences for biodiversity and functioning of terrestrial ecosystems. *Sci Adv* 5(11):eaaz1834. <https://doi.org/10.1126/sciadv.aaz1834>
- Radek KJ (1997) Soil erosion following wildfires on the Okanogan national forest-initial monitoring report. Unpublished report. U.S. Department of Agriculture Forest Service, Okanogan National Forest, Okanogan, WA
- Ranalli AJ (2004) A summary of the scientific literature on the effects of fire on the concentration of nutrients in surface water (Open-File Report No 2004–1296; Issues 2004–1296). US Department of the Interior and US Geological Survey, Reston
- Rhoades CC, Entwistle D, Butler D (2011) The influence of wildfire extent and severity on streamwater chemistry, sediment and temperature following the Hayman fire, Colorado. *Int J Wildland Fire* 20(3):430. <https://doi.org/10.1071/WF09086>
- Rhoades CC, Chow AT, Covino TP, Fegle TS, Pierson DN, Rhea AE (2019) The legacy of a severe wildfire on stream nitrogen and carbon in headwater catchments. *Ecosystems* 22(3):643–657. <https://doi.org/10.1007/s10021-018-0293-6>
- Ribot M, von Schiller D, Martí E (2017) Understanding pathways of dissimilatory and assimilatory dissolved inorganic nitrogen uptake in streams. *Limnol Oceanogr* 62(3):1166–1183. <https://doi.org/10.1002/lno.10493>
- Riggan PJ, Lockwood RN, Lopez EN (1985) Deposition and processing of airborne nitrogen pollutants in Mediterranean-type ecosystems of southern California. *Environ Sci Technol* 19(9):781–789. <https://doi.org/10.1021/es00139a003>
- Robichaud PR (2000) Fire effects on infiltration rates after prescribed fire in northern Rocky Mountain forests, USA. *J Hydrol* 231–232(1–4):220–229
- Robichaud PR, MacDonald LH, Foltz RB (2010) Fuel management and erosion. Chapter 5 (Gen. Tech. Rep. RMRS-GTR-231) U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ft. Collins, CO, pp 79–100
- Robichaud PR, Bone ED, Lewis SA, Brooks ES, Brown RE (2021) Effectiveness of post-fire salvage logging stream buffer management for hillslope erosion in the U.S. Inland Northwest Mountains. *Hydrol Process*. <https://doi.org/10.1002/hyp.13943>
- Robinne F-N, Bladon K, Silins U, Emelko M, Flannigan M, Parisien M-A, Wang X, Kienzle S, Dupont D (2019) A regional-scale index for assessing the exposure of drinking-water sources to wildfires. *Forests*. <https://doi.org/10.3390/f10050384>
- Rodríguez-Alleres M, Varela ME, Benito E (2012) Natural severity of water repellency in pine forest soils from NW Spain and influence of wildfire severity on its persistence. *Geoderma* 191:125–131. <https://doi.org/10.1016/j.geoderma.2012.02.006>
- Rodríguez-Cardona BM, Coble AA, Wymore AS, Kolosov R, Podgorski DC, Zito P, Spencer RGM, Prokushkin AS, McDowell WH (2020) Wildfires lead to decreased carbon and increased nitrogen concentrations in upland arctic streams. *Sci Rep* 10(1):8722. <https://doi.org/10.1038/s41598-020-65520-0>
- Rogers BM, Neilson RP, Drapek R, Lenihan JM, Wells JR, Bachelet D, Law BE (2011) Impacts of climate change on fire regimes and carbon stocks of the U.S. Pacific Northwest. *J Geophys Res*. <https://doi.org/10.1029/2011JG001695>
- Rozendaal DMA, Chazdon RL, Arreola-Villa F, Balvanera P, Bentos TV, Dupuy JM, Hernández-Stefanoni JL, Jakovac CC, Lebrija-Trejos EE, Lohbeck M, Martínez-Ramos M, Massoca PES, Meave JA, Mesquita RCG, Mora F, Pérez-García EA, Romero-Pérez IE, Saenz-Pedroza I, van Breugel M, Bongers F (2017) Demographic drivers of aboveground biomass dynamics during secondary succession in neotropical dry and wet forests. *Ecosystems* 20(2):340–353. <https://doi.org/10.1007/s10021-016-0029-4>
- Sahrawat KL (2008) Factors affecting nitrification in soils. *Commun Soil Sci Plant Anal* 39(9–10):1436–1446. <https://doi.org/10.1080/00103620802004235>
- Samuels WB, Amstutz DE, Bahadur R, Pickus JM (2006) RiverSpill: a national application for drinking water protection. *J Hydraul Eng* 132(4):393–403. [https://doi.org/10.1061/\(ASCE\)0733-9429\(2006\)132:4\(393\)](https://doi.org/10.1061/(ASCE)0733-9429(2006)132:4(393))
- Santos F, Wymore AS, Jackson BK, Sullivan SMP, McDowell WH, Berhe AA (2019) Fire severity, time since fire, and site-level characteristics influence streamwater chemistry at baseflow conditions in catchments of the Sierra Nevada, California, USA. *Fire Ecol* 15(1):3. <https://doi.org/10.1186/s42408-018-0022-8>
- Schleppi P, Curtaz F, Krause K (2017) Nitrate leaching from a sub-alpine coniferous forest subjected to experimentally increased N deposition for 20 years, and effects of tree girdling and felling. *Biogeochemistry*. <https://doi.org/10.1007/s10533-017-0364-3>
- Serpa D, Ferreira RV, Machado AI, Cerqueira MA, Keizer JJ (2020) Mid-term post-fire losses of nitrogen and phosphorus by overland flow in two contrasting eucalypt stands in north-central Portugal. *Sci Total Environ* 705:135843. <https://doi.org/10.1016/j.scitotenv.2019.135843>
- Sham CH, Tuccillo ME, Rooke J (2013) Report on the effects of wildfire on drinking water utilities and effective practices for wildfire risk reduction and mitigation. Water Research Foundation and U.S. Environmental Protection Agency, pp 1–118. <https://www.bendoregon.gov/Home/ShowDocument?id=14309>

- Sharma U, Sharma J, Devi M (2017) Effect of Forest fire on soil nitrogen mineralization and microbial biomass: a review. *J Pharmacogn Phytochem* 6(3):682–685
- Shatford J, Hibbs DE, Puettmann K (2007) Conifer regeneration after forest fire in the Klamath-Siskiyou: how much, how soon? *J For* 105:139–146
- Silins U, Bladon KD, Kelly EN, Esch E, Spence JR, Stone M, Emelko MB, Boon S, Wagner MJ, Williams CHS, Tichowsky I (2014) Five-year legacy of wildfire and salvage logging impacts on nutrient runoff and aquatic plant, invertebrate, and fish productivity. *Ecohydrology* 7(6):1508–1523. <https://doi.org/10.1002/eco.1474>
- Sistla SA, Asao S, Schimel JP (2012) Detecting microbial N-limitation in tussock tundra soil: implications for Arctic soil organic carbon cycling. *Soil Biol Biochem* 55:78–84. <https://doi.org/10.1016/j.soilbio.2012.06.010>
- Skorbiłowicz M, Ofman P (2014) Seasonal changes of nitrogen and phosphorus concentration in Supraśl river. *J Ecol Eng* 15:26–31. <https://doi.org/10.12911/22998993.1084172>
- Smith NR, Kishchuk BE, Mohn WW (2008) Effects of wildfire and harvest disturbances on forest soil bacterial communities. *Appl Environ Microbiol* 74(1):216–224. <https://doi.org/10.1128/AEM.01355-07>
- Smith HG, Sheridan GJ, Lane PNJ, Nyman P, Haydon S (2011) Wildfire effects on water quality in forest catchments: a review with implications for water supply. *J Hydrol* 396(1):170–192. <https://doi.org/10.1016/j.jhydrol.2010.10.043>
- Smith AMS, Kolden CA, Paveglio TB, Cochrane MA, Bowman DM, Moritz MA, Kliskey AD, Alessa L, Hudak AT, Hoffman CM, Lutz JA, Queen LP, Goetz SJ, Higuera PE, Boschetti L, Flannigan M, Yedinak KM, Watts AC, Strand EK, van Wagtenonk JW, Anderson JW, Stocks BJ, Abatzoglou JT (2016) The science of firescapes: achieving fire-resilient communities. *Bioscience* 66(2):130–146. <https://doi.org/10.1093/biosci/biv182>
- Soto DP, Jacobs DF, Salas C, Donoso PJ, Fuentes C, Puettmann KJ (2017) Light and nitrogen interact to influence regeneration in old-growth Nothofagus-dominated forests in south-central Chile. *For Ecol Manage* 384:303–313. <https://doi.org/10.1016/j.foreco.2016.11.016>
- Spencer CN, Hauer FR (1991) Phosphorus and nitrogen dynamics in streams during a wildfire. *J N Am Benthol Soc* 10(1):24–30. <https://doi.org/10.2307/1467761>
- Srivastava A, Dobre M, Wu JQ, Elliot WJ, Bruner EA, Dun S, Brooks ES, Miller IS (2013) Modifying WEPP to improve streamflow simulation in a Pacific Northwest watershed. *Trans ASABE* 56(2):603–611. <https://doi.org/10.13031/2013.42691>
- Stephan K, Kavanagh KL, Koyama A (2012) Effects of spring prescribed burning and wildfires on watershed nitrogen dynamics of central Idaho headwater areas. *For Ecol Manage* 263:240–252. <https://doi.org/10.1016/j.foreco.2011.09.013>
- Stephan K, Kavanagh KL, Koyama A (2015) Comparing the influence of wildfire and prescribed burns on watershed nitrogen biogeochemistry using ^{15}N natural abundance in terrestrial and aquatic ecosystem components. *PLoS ONE* 10(4):e0119560. <https://doi.org/10.1371/journal.pone.0119560>
- Stephens SL, Meixner T, Poth M, McGurk B, Payne D (2004) Prescribed fire, soils, and stream water chemistry in a watershed in the Lake Tahoe Basin, California. *Int J Wildland Fire* 13(1):27–35. <https://doi.org/10.1071/WF03002>
- Stirling E, Macdonald LM, Smernik RJ, Cavigliaro TR (2019) Post fire litters are richer in water soluble carbon and lead to increased microbial activity. *Appl Soil Ecol* 136:101–105. <https://doi.org/10.1016/j.apsoil.2018.12.021>
- Stoddard JL (1994) Long-term changes in watershed retention of nitrogen. *Environmental chemistry of lakes and reservoirs*, vol 237. American Chemical Society, Washington DC, pp 223–284
- Teclé A, Neary D (2015) Water quality impacts of forest fires. *J Pollut Eff Control*. 3(3):1–7. <https://doi.org/10.4172/2375-4397.1000140>
- Terrer C, Jackson RB, Prentice IC, Keenan TF, Kaiser C, Vicca S, Fisher JB, Reich PB, Stocker BD, Hungate BA, Peñuelas J, McCallum I, Soudzilovskaia NA, Cernusak LA, Talhelm AF, Van Sundert K, Piao S, Newton PCD, Hovenden MJ, Franklin O (2019) Nitrogen and phosphorus constrain the CO_2 fertilization of global plant biomass. *Nat Clim Change* 9(9):684–689. <https://doi.org/10.1038/s41558-019-0545-2>
- Tiedemann AR, Helvey JD, Anderson TD (1978) Stream chemistry and watershed nutrient economy following wildfire and fertilization in Eastern Washington. *J Environ Qual* 7(4):580–588. <https://doi.org/10.2134/jeq1978.00472425000700040023x>
- Turner MG, Smithwick EAH, Metzger KL, Tinker DB, Romme WH (2007) Inorganic nitrogen availability after severe stand-replacing fire in the Greater Yellowstone ecosystem. *Proc Natl Acad Sci USA* 104(12):4782–4789. <https://doi.org/10.1073/pnas.0700180104>
- U.S. Geological Survey. (2012). Wildfire effects on source-water quality—lessons from fourmile canyon fire, Colorado, and implications for drinking-water treatment (Fact Sheet 2012–3095; Issue Fact Sheet 2012–3095, pp. 1–4)
- Vourlitis GL, Hentz CS (2016) Impacts of chronic N input on the carbon and nitrogen storage of a postfire Mediterranean-type shrubland. *J Geophys Res Biogeosci* 121(2):385–398. <https://doi.org/10.1002/2015JG003220>
- Wang C, Wang X, Liu D, Wu H, Lü X, Fang Y, Cheng W, Luo W, Jiang P, Shi J, Yin H, Zhou J, Han X, Bai E (2014) Aridity threshold in controlling ecosystem nitrogen cycling in arid and semi-arid grasslands. *Nat Commun* 5(1):1–8. <https://doi.org/10.1038/ncomms5799>
- Westerling AL, Hidalgo HG, Cayan DR, Swetnam TW (2006) Warming and earlier spring increase Western U.S. forest wildfire activity. *Science* 313:1–5
- Whitney N, Zabowski D (2004) Total soil nitrogen in the coarse fraction and at depth. *Soil Sci Soc Am J* 68(2):612–619. <https://doi.org/10.2136/sssaj2004.6120>
- Williams MR, Melack JM (1997) Effects of prescribed burning and drought on the solute chemistry of mixed-conifer forest streams of the Sierra Nevada, California. *Biogeochemistry* 39(3):225–253