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Modeling effects of wildfire and fire retardant on nutrients downstream in a watershed scale

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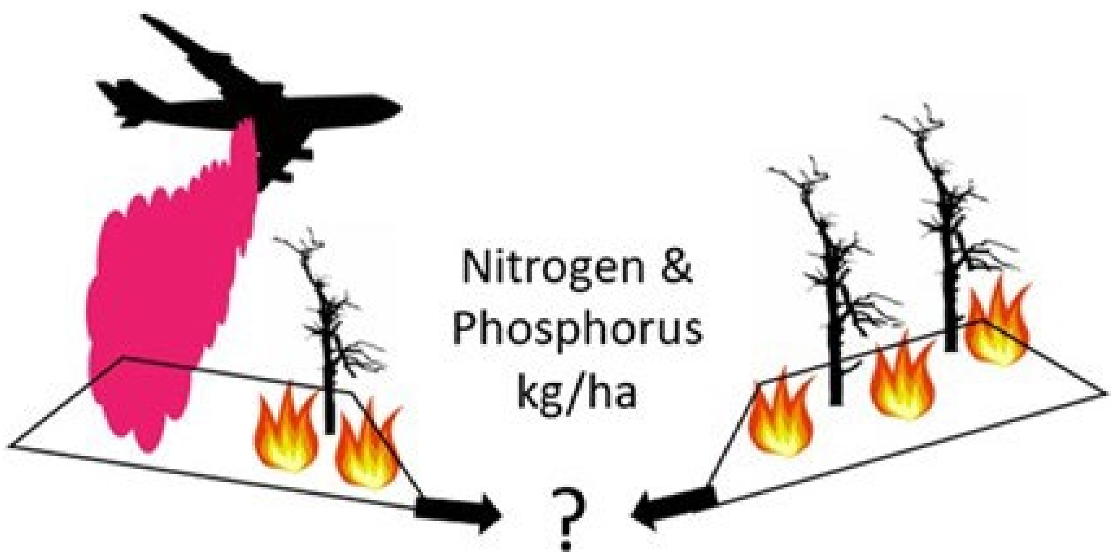

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14. ABSTRACT
The effects of wildfire and ammonium-phosphate (AP) fire retardants on loads of nutrients (nitrogen, N, and phosphorus, P) in downstream waters were evaluated using a critical literature review and watershed modeling. The literature review suggests retardant application in an average first-order watershed (320 hectares, ha) would add about 0.5 kg-N ha^{-1} and 1.2 kg-P ha^{-1} per aerial drop, much less than the typical ranges of N and P yield following wildfire. No studies have demonstrated N and P transport from AP fire retardants into surface water via overland flow. The updated PFHydro water-quality module was applied to part of the Cache Creek watershed, California, which burned during summer 2015. Daily time-series loads of total N (TN) and total P (TP) fit calibrated daily TN and TP loads for WY 2015 (pre-fire) and WY 2017 (post-fire) at the watershed outlet well. Annual TN and TP loads during WY 2017 were 4.5–6 times those during WY 2015. Modeling results indicate that application of AP fire retardants would theoretically increase TN and TP loads less than 1%, even with 100 aerial drops of fire retardant applied, a rate much higher than what was documented during the 2015 fires.

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Wildfire impacts, nutrients, total phosphorus, total nitrogen, fire-retardants, sediment load, modeling.

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Modeling effects of wildfire and fire retardant on nutrients downstream in a watershed scale

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Acronyms and Abbreviations

AP	ammonium phosphate
BAER	Burned Area Emergency Response
Bp	load bias percent
C	carbon
Ca	calcium
Cl	chlorine
CO_2	carbon dioxide
DAP	di-ammonium phosphate
DON	dissolved organic nitrogen
DWR	Department of Water Resources (California)
f	filtered
GIS	geographic information system
H	hydrogen
ln	natural logarithm
MAP	mono-ammonium phosphate
N	nitrogen
N_2	nitrogen (gas)
NADP	National Atmospheric Deposition Network
NAIP	non-apatite inorganic phosphorus
NH_3	ammonia
NH_4^+	ammonium ion
NO_2^-	nitrite ion
NO_3^-	nitrate ion
NO_x	nitrogen oxides
NSE	Nash-Sutcliffe Efficiency index
O	oxygen
OP	orthophosphate
org	organic
p-value (p)	probability of obtaining observed results, assuming null hypothesis
P	phosphorus
PO_4^{3-}	phosphate ion
Q	streamflow (discharge)
R^2	adjusted coefficient of determination Reclamation Bureau of Reclamation
SE	standard error
SRP	soluble reactive phosphorus
SS	suspended sediment

TDN	total dissolved nitrogen
TN	total nitrogen
TP	total phosphorus
u	unfiltered
U.S.	United States
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
WY	water year (October 1 through September 30)

Measurements

cm	centimeter
ft	foot
g	gram
gal	gallon
gpc	gallon per hundred square feet
ha	hectare
in	inch
kg	kilogram
kmol	kilomole
L	liter
mg	milligram
mo	month
mm	millimeter
ppm	part per million (equal to milligram per kilogram)
s	second
yr	year
%	percentage

Unit Conversion

Unit	
1 ton (2000 pounds)	0.907185 metric ton (1000 kilograms)
1 cubic foot per second (cfs)	0.028316847 cubic meter per second (m^3 s^{-1})
1 acre	0.40468564 hectare (ha)
1 inch (in)	2.54 centimeter (cm)

Contents

Executive Summary

Fire retardants have been used in wildland firefighting with increasing frequency in recent decades due to increased wildfire occurrence and size, especially in the western U.S. Increasing use will likely continue with the projected continuation of current wildfire trends. The retardants used are primarily made up of ammonium and phosphate, two nutrients that can lead to surface-water eutrophication when present in excess. There is concern that widespread use of ammonium- phosphate retardants could have negative effects on downstream waterbodies.

The objective of this report is to address the question “Do fire retardants have an effect on downstream waterbodies?” Evaluating this question is complex. Nutrient dynamics of a specific system must be understood with enough detail to assess the retardant addition effects. In most watersheds, there is not available information on soil nutrients, spatial nutrient loads, erosion estimates, or annual export information that would be needed to characterize nutrient transport in the terrestrial and aquatic environments. Adding the major disruption of wildfire greatly changes the watershed characteristics and compounds the complexity of the issue.

To address this complex topic, a literature review and modeling approach were combined to study fire retardants impacts on total nitrogen (TN) load and total phosphorus (TP) load.

According to the literature review, there is no empirical evidence that fire retardant applied to the landscape is transported to streams in runoff. After wildfire there may be large increases in nutrient loads (N and P) from transport of ash and erosion of bare soil. For both N and P, there is a possibility that under specific circumstances and in specific watersheds, the added fire retardant could increase the nutrient export, particularly in small watersheds with low post-fire nutrient exports.

Modeling the dynamic effects of wildfire disruption on hydrologic and biogeochemical processes in a watershed scale is difficult. For this project, the Cache Creek watershed in northern California was chosen because some limited nutrient data from years preceding and following the 2015 Rocky and Jerusalem Fires were available. Also, the team had previously modeled post-fire runoff and sediment transport in the area, and had an existing, calibrated watershed model for water years 2015 through 2017. However, the available nutrient data and the previously modeled time period did not overlap. The lack of available pre- and post-fire data for water quality and soil chemistry is a major gap in wildfire science and one of the reasons that post-fire water-quality modeling is scarce.

To fill the data gaps, annual sediment and nutrient loads were independently obtained using the model LOADEST. The LOADEST model used the limited available data to calculate daily and annual loads for several nitrogen and phosphorus species at several locations in the watershed, both pre-fire and post-fire. The output from LOADEST was then used to calibrate the PFHydro model to simulate the specific effects of wildfire in the watershed.

The outcome of the modeling process was a result from PFHydro that accounts for watershed

characteristics, burning characteristics, and post-fire hydrologic response, and simulates the sediment and nutrient loads with a high degree of agreement with the available data and LOADEST model results. The application of fire retardant was then considered in the context of the pre-fire and post- fire nutrient loads to evaluate possible changes in nutrient loading for a range of retardant application rates.

Detailed modeling of the Cache Creek watershed indicates that annual TN and TP loads would have increased by less than 1% even if 100 aerial drops of fire retardant had been made on the Rocky and Jerusalem fires in 2015. (The known number of drops was 7 over a 2-day period, and the likely total was probably less than 20.)

Both literature review and modeling results clearly demonstrate that the additive effect of aerial ammonium-phosphate (AP) fire retardant on nutrient loading to downstream waterbodies are small compared to nutrient load increases from burning alone.

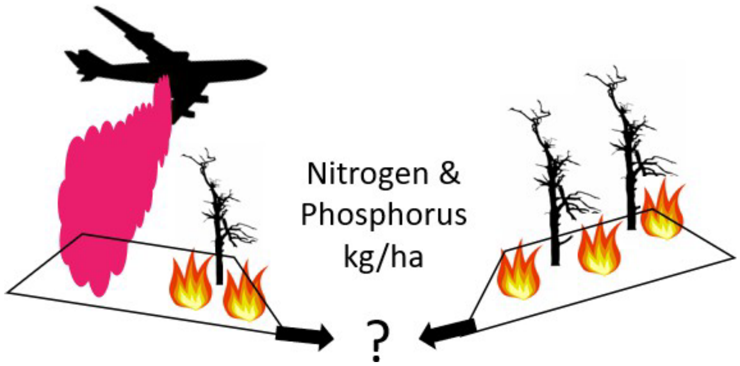
The report addresses the complexity of post-fire nutrient geochemistry and provides a detailed review of previous studies on fire retardant application and mobility in Chapter 1. The development of a conceptual model for nutrient inputs to the terrestrial system from wildfire ash are included in Chapter 2. In Chapter 3, the application of the LOADEST model and the updated PFHydro water quality module and results of the modeled retardant application are presented. The key findings of the project, chapter summaries, challenges and limitations, and direction for future work are presented in Chapter 4.

1. Literature Review

Abstract

Increasing fire occurrence and extreme fire behavior necessitates consideration of wildfire and wildfire suppression in water management. Though it is widely known that wildfire alters biogeochemical cycling of nutrients and other constituents in surface water, the potential fertilization effects from widespread use of ammonium-phosphate-based aerial fire retardant on downstream receiving waters remains understudied and is a concern among resource managers and the public. One common reason for persistent questions regarding fire-retardant safety is that contextualizing the effects of fire-retardant application is difficult due to the complexity of nutrient cycling and the compounding effects of major disturbance in watersheds following fire. Therefore, a synthesis of pre- and post-fire geochemical nutrient data and a review of existing literature on fire-retardant application are presented alongside theoretical calculations demonstrating the potential for fire- retardant fertilization. Review of nutrient pools and major transport pathways reveal that landscape N export from unburned watersheds is < 1 kg-N ha^{-1} yr^{-1} whereas burned watersheds may export > 68 kg-N ha^{-1} yr^{-1} immediately following burning. Similarly, unburned landscapes export 0.02–0.2 kg- P ha^{-1} yr^{-1} but this can increase to 1.7 kg-P ha^{-1} yr^{-1} immediately following burning. Retardant application by aircraft in a first-order watershed (around320 ha) would add about 0.5 kg-N ha^{-1} and 1.2 kg-P ha^{-1} per drop. In small watershed, particularly those with low P export, retardant could substantially increase export. However, the combination of nutrient retention on the landscape and dilution effects in most streams suggests that added nutrients from aerial retardants are not likely to cause measurable increases in surface-water nutrient concentrations draining burned landscapes.

Graphical Abstract



1.1. Introduction

The changing climate is causing increased wildfire occurrence and extreme fire behavior across the western U.S. and in similar climates globally (Westerling et al., 2006; Holden et al., 2018; Goss et al., 2020). The effects of the increased fire behavior are felt throughout society, particularly in the western U.S., in the form of increased news reporting, increased smoke events, evacuations, property destruction, and fatalities (Kramer et al., 2019; Burke et al., 2021). The increase in wildfire visibility has also increased public concern and scientific interest in the effects of fire, fire management policies, and the ability of firefighters to counter the spread of fire when lives and property are threatened.

In response to life and property threats from fires, wildland firefighters employ suppression strategies, including use of aerial firefighting tactics. One major component of aerial firefighting is the use of airplanes to deliver large amounts of ammonium-phosphate (AP) fire retardants to a fire's flank. (Details of AP composition are given below in section 3.1). Application of fire retardant often results in dramatic imagery during application and leave highly visible red streaks across the landscape due to the dyes added for determining the drop location (**Figure 1-1**). The increased use and high visibility to the public have raised questions about the environmental effects of retardant (e.g., Weiser, 2017; Davis, 2020; Heller, 2020). Specifically, one question is whether the application of retardant into wildland environments may cause fertilization effects and potential altering of nutrient cycles in downstream surface waters. However, the issue is complex as it requires the context of terrestrial and aquatic nutrient cycles, the effects of burning on these cycles, and the consideration of retardant application to both burned and unburned landscapes. Further, retardant transport from application sites to downstream waterbodies is likely linked to hydrologic conditions and post-precipitation response, which are also highly altered following burning.

Figure 1-1: Land surface after aerial deployment of ammonium-phosphate fire retardant. Napa County, California, Nov. 12, 2020 after Glass Fire and LNU Lightning Complex Fire. Photo by C.N. Alpers, USGS.

The alteration of surface soils and production of ash following heating contribute to increased nutrient concentrations downstream. Wildfire changes soil through the processes of heating, partial combustion, and full combustion; each of these processes result in various end-products described by "severity". Wildfire severity, from a soil perspective, is a function of two components: intensity (temperature) and duration of heating (Parsons, 2003). From a forestry perspective, severity may be defined similarly as a combination of intensity (energy released; kW per area) and the duration of burning, or more simply as the general degree of combustion completeness of above-ground fuels (Keeley, 2009). In general, the degree and abundance of combustion end products are a function of wildfire severity. Fire severity is typically assessed based on consumption of fuels load, degree of canopy remaining, and other visual indications of the degree of burning (Parsons, 2003). Above-ground and below-ground temperatures resulting from wildfire vary due to antecedent conditions such as available fuel, ambient temperature, relative humidity, and other local environmental factors. Charring of organic materials begins around 180°C, combustion of woody materials occurs at 460°C, and complete combustion occurs around 550°C (Certini, 2005). Partial combustion and full combustion result in the formation of ash, the remnants of biomass. The formation of ash end-products exists along a continuum of combustion completeness (Bodí et al., 2014). The loss of organic material during combustion releases elements otherwise associated with parent materials that are not volatilized at temperatures reached during burning of organic material. In this way, fire can liberate nutrients from fuels and change their geochemical cycling in soils and water.

Wildfire can significantly alter surface-water characteristics; previous reviews on post-wildfire water quality have summarized hydrologic responses (such as discharge per precipitation unit) as well as sediment, nutrient, metal, and carbon transport and cycling (Ice et al., 2004; Neary & Overby, 2006; Smith et al., 2011; Shakesby, 2011). In general, previous works show that fire causes increase in the transport of many water-quality constituents above pre-fire levels. Increases in sediment transport can be particularly dramatic during initial flushes, with post-fire sediment fluxes possibly many orders of magnitude greater than base-flow concentrations or the typical seasonal response. First flushes are usually the largest response for flows and mass exports, but for some constituents the total export remains elevated above pre-fire levels through the first wet season and beyond. Flows and mass export may also quickly decrease and approach background or normal seasonal levels as watersheds recover and vegetation is reestablished. Waterbodies receiving runoff from burned landscapes can undergo substantial water-quality changes and impairment, including algal blooms and severe oxygen depletion associated with eutrophication in receiving waters (Eamus et al., 2006; Oropeza and Heath, 2013).

The onset of eutrophication could be compounded by nutrient-based fire retardants. The downstream effects of AP fire retardants have been studied before, but there has not been a comprehensive review of the issue and there is still uncertainly as to whether the application of AP fire retardants on the landscape can influence downstream nutrient levels. To provide guidance on the topic of aquatic fertilization from aerial AP retardants, a summary of established nutrient cycling research is coupled with a condensed review of post-fire effects on nutrient cycling to provide context for a critical review of geochemical studies of AP fire-retardant application.

1.2. Nutrient Cycling Before and After Wildfire

1.2.1. Nitrogen

Nitrogen input and subsequent cycling in terrestrial and aquatic environments includes the processes of N deposition, fixation, nitrification, assimilation, mineralization, and denitrification (**Figure 1-2**). These processes are universal; however, quantities and rates may be site-specific. Landscape characteristics (plant species, soil carbon, moisture, other available nutrients, soil and water chemistry) have a great influence on N cycling (Hill and Shackleton, 1989). Adding to the complexity of nitrogen's biogeochemistry, it is rapidly converted between different oxidation states as it is cycled between atmospheric, terrestrial, and aquatic environments. Different forms of N (**Table 1-1**) have very different transport behavior and bioavailability; therefore, prediction of fertilization effects would need to account for conversion between forms to predict behavior.

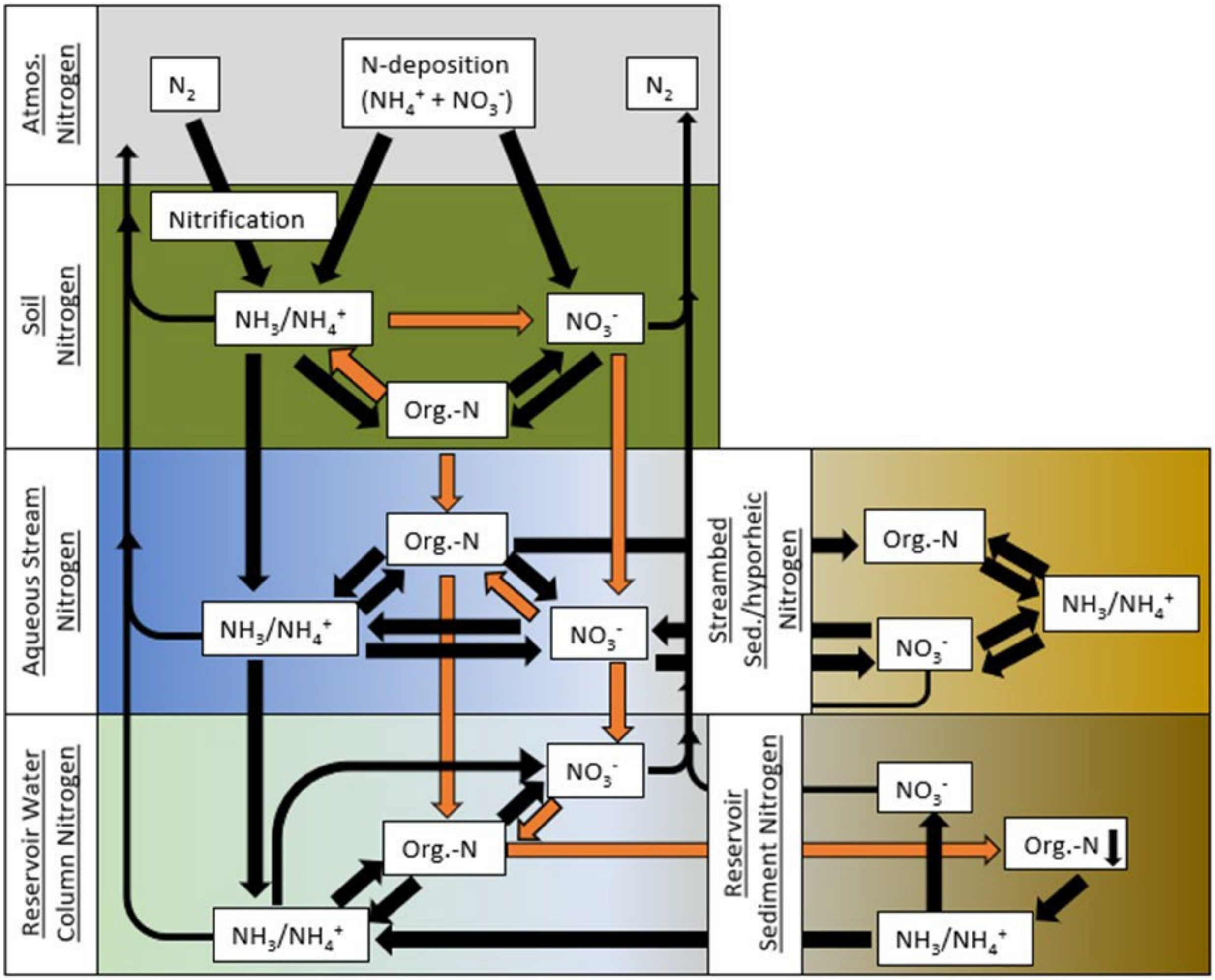


Figure 1-2. Nitrogen cycling across environmental compartments.

Nitrogen can enter the terrestrial environment through atmospheric deposition or through N fixation. Atmospheric N deposition can vary widely depending on proximity to air pollution sources such as agricultural lands and industrial sources. An undisturbed watershed likely receives no more than 2.5 kg-N ha^{-1} yr^{-1} and this is a typical deposition rate for most of the western U.S. (Lewis et al., 1999). The National Atmospheric Deposition Network (NADP) reported the 2019 total N deposition rates (wet + dry, all species) to be less than 4 kg-N ha^{-1} yr^{-1} across most forested land in the western U.S., with some forested areas receiving higher deposition from regional sources (NADP, 2020). In addition to direct deposition, N fixation can contribute to terrestrial and aquatic N loading. Fixation rates in CA forests are estimated to be 6.6 kg-N ha^{-1} yr^{-1} for forest, 2.4 kg-N ha^{-1} yr^{-1} for woodlands and grasslands, whereas chaparral and arid shrublands only received 1.5 kg-N ha^{-1} yr^{-1} (Cleveland et al., 1999; Sobota et al., 2009).

Once in the terrestrial environment, inorganic forms of N (primarily nitrate, NO_3^- and ammonium, NH_4^+) are assimilated to organic N and this plays a large role in watershed N retention. Organic N forms, such as amino acids, sugars, and N-containing organic molecules formed during organic matter breakdown, account for approximately 75-85% of soil N (Essington, 2015). The concept of N saturation suggests that plants and soil microbes will retain nearly all the soil N if the soil is undersaturated in relation to the uptake potential (Stoddard, 1994). Most terrestrial environments are undersaturated with respect to N, and application of N to terrestrial environments usually leads to rapid uptake and utilization of N through increased plant growth, leading to its retention on the landscape. Soil N retention will vary seasonally as well: watersheds have a much lower ability to retain N during winter and spring due to reduced biomass growth. Soils that are saturated (inputs are greater than uptake potential) can release large amounts of N (Stoddard, 1994). On average, temperate watersheds generate a watershed export yield of 0.01 to 0.04 kg ha^{-1} yr^{-1} of NH_4^+, 0.01 to 0.56 kg ha^{-1} yr^{-1} of NO_3^-, and 0.01 to 0.6 kg ha^{-1} yr^{-1} of total inorganic N (TIN; Lewis et al., 1999).

Stream-water concentrations of N vary widely depending on watershed characteristics such as vegetation type, land use, watershed disturbance, bedrock, and pollution sources. In-stream N is actively cycled between forms and can move from the water column to the sediment, be retained in the hyporheic zone, lost as a gaseous phase (N_2, NH_3 or NO_x), or assimilated, losing 5–50% of the total N-load daily, with smaller streams losing a larger percentage than large streams (Alexander et al., 2000). Across the U.S., stream-water N consists of about 30% NO_3^-,10% NH_4^+, and 60% dissolved organic N (DON; Binkley et al., 2004). Forested streams in the western U.S. were reported to have median NO_3^- concentrations of 0.03 mg-N L^{-1} whereas the concentrations of NH_4^+ in stream water were much lower with a median < 0.01 mg-N L^{-1} (Binkley et al., 2004).

Unlike terrestrial environments, growth stimulation from N addition to surface-water bodies is unusual unless the waterbody is an N-limited system (Stoddard, 1994). Lake inputs are typically dominated by NO_3^-, and primary production converts NO_3^- to algal N, which becomes the main source to sediments (Golterman, 2004). Because of the algal origin, most sediment N is in the organic form (>90%; Golterman, 2004). Sediment organic N can be mineralized to ammonia (NH_3) or ammonium (NH_4^+), which can be released back into the water column, although some stays as a cation associated with the sediment, a process that depends on the soil's cation exchange capacity (CEC; Golterman, 2004). In sediment, N can exist as NO_3^- or NO_2^- (nitrite), but it is rapidly converted to N_2 gas by anaerobic denitrification when oxygen is limited (Golterman, 2004).

1.2.2. Effects of Fire on Nitrogen Cycling

1.2.2.1. Combustion Effects on Nitrogen in Fuel

Fire can cause inorganic N to increase greatly in soil, while organic N decreases greatly. Decreases in organic N are a combination of volatilization and mineralization (Raison, 1979; Neary et al., 1999; Prieto-Fernández, 2004). Organic N begins to volatilize at 200°C and is 50% lost at 500°C (Neary et al., 1999). In severe fires, organic N is converted to NH_4^+, which can be converted to NO_3^- following the fire (Certini, 2005). Typically, the remaining NH_4^+ fraction will then adsorb to soil organic matter and the NO_3^- will leach into the subsurface (Certini, 2005). Organic N that remains in organic matter is likely incorporated into hetero-aromatic and cyclic structures (Knicker et al., 2011). Post-fire NH_4^+ persistence may be due to decreases in the abundance of nitrifying bacteria in soil (Prieto-Fernández, 2004; Gómez-Rey & González-Prieto, 2014), though others have found elevated nitrification as well (DeLuca et al., 2006).

1.2.2.2. Post-fire Nitrogen Cycling

Two synthesis papers (Wan et al., 2001; Smithwick, 2005) have compiled data on N cycling following fire and have identified several major trends. Wan et al. (2001) examined 87 studies published from 1955 to 1999 and extracted data on fuel N, soil N, soil NH_4^+ and soil NO_3^-. The analysis of historical data revealed that fire reduces fuel N by 58% on average, with higher average N losses (around 70%) in broadleaf and shrub, slightly lower average N losses (48%) in conifer vegetation types and with grasslands right on the 58% average. The average change in spatial loading of soil NH_4^+ was near 1100% and near 700% for soil NO_3^-, with increases primarily observed at 0–2.5 cm soil depths (Wan et al., 2001). Soil NH_4^+ is highest after fire and then asymptotically decreases with time. Soil NO_3^- is initially low post-fire, increases until about one year after fire and then gradually returns to pre-fire levels within five years (Wan et al., 2001).

Smithwick (2005) conducted a meta-analysis of studies on N cycling change following fire and revealed that NH_4^+ concentrations increased by 200 to 400% in stand-replacing wildfire in conifer forests but was even higher in non-stand-replacing fire, with increases as high as 2000%. Studies of woodland chaparral fire revealed 300 to 3000% increases of NH_4^+ whereas broadleaf forests had 100 to 1000% increases (Smithwick, 2005). Trends of NO_3^- were different than trends of NH_4^+ in conifer forest. Stand-replacing fire in conifer forests increase NO_3^- somewhat but was often not significant. In non-stand-replacing fires in conifer forests NO_3^- increases were 90 to 400%. In chaparral fires NO_3^- increases ranged from "not significant" to 4000% and in broad leafed forest increases ranged from "not significant" to 30% (Smithwick, 2005).

1.2.2.3. Concentration of Nitrogen in Soil Following Fire

Total N (TN) in burned soil and ash can vary widely, with most concentrations below 100 mg kg^{-1} and some much higher (**Table 1-5**). Concentrations of TN as high as 4000 mg kg^{-1} in ash from a eucalyptus forest in Australia have been reported (Santín et al., 2015). The organic N that is converted to NH_4^+ during heating and combustion can remain in resulting ash and soil. For example, NH_4^+ in ash and soil is typically below detection limit in low- and moderate-severity burning but may increase to 10 mg kg^{-1} in high severity fires (Santín et al., 2015). The increase in NH_4^+ occurs immediately following burning and elevated NH_4^+ may persist for years (Blank and Zamudio, 1998). Concentrations of NH_4^+ are typically in the range of 1 to 60 mg kg^{-1} following fire and tend to decrease with time (Kutiel & Naveh, 1987; Rodríguez et al., 2009; Stephan, 2012; Marañón-Jiménez, 2013; Gómez-Rey & González-Prieto, 2014).

Concentrations of NO_3^- as high as 60 mg kg^{-1} have been reported (Kutiel & Naveh, 1987), with most values in the range of 0.5 to 20 mg kg^{-1} (Rodríguez et al., 2009; Stephan, 2012; Marañón-Jiménez, 2013; Gómez-Rey & González- Prieto, 2014).

1.2.2.4. Spatial Loading of Nitrogen Following Fire

The spatial loading of total N is greatly changed by wildfire. Because N volatilizes during combustion, losses are expected in fuel and surface soil. Spatial total N loads in California chaparral

(kg ha^{-1}) are 130 kg ha^{-1}, but this pool decreased to 30 kg ha^{-1} following burning (Debano & Conrad, 1978). Changes in total N loading are limited to shallow soils (< 2 cm; Debano & Conrad, 1978).

1.2.2.5. Influence of Fire on Surface-Water Nitrogen

An increase in inorganic N species also occurs in downstream surface water following wildfire (**Table 1-4**; Hauer & Spencer, 1998; Brass et al., 1996; Lane et al., 2008; Mast & Clow, 2008; Meixner et al., 2006; Bladon et al., 2008; Coombs & Melack, 2013; Son et al., 2015; Mast et al., 2016). The concentration of NH_4^+ in surface water draining burned lands can increase 2- to 3-fold immediately following fire, but increases are short-lived and typically dissipate after a year as most export occurs in the first few storms (Hauer & Spencer, 1998; Coombs & Melack, 2013). The decrease in NH_4^+ is due to NH_4^+ oxidizing to NO_3^- in most environments. Accordingly, NO_3^- concentrations increase over time following burning and may even be higher in the early spring flushes during the second year following burning than the first flushes following burning (Mast et al., 2016). The largest contributing species to total dissolved N post-fire tends to be NO_3^-, typically accounting for 70 to 90% (Lane et al., 2008; Mast & Clow, 2008; Bladon et al., 2008). The amount of NO_3^- released is related to fire severity, with higher severity causing greater amounts of NO_3^- in downstream waters (Brass et al., 1996). NO_3^- concentrations have been shown to increase from 2- to about 40-fold, with post-fire concentrations typically in the range of 0.5 to 2 mg L^{-1} NO_3-N (Hauer & Spencer, 1998; Bladon et al., 2008).

Corresponding watershed yields of total N follow a pattern of increases in the first- and second-year total N export (Williams and Melack, 1997; Ferreira et al., 2005; Mast & Clow, 2008; Mast et al., 2016). Total N loads have been reported from 1 to 70 kg ha^{-1} in the first year following fire and then decrease in year 3 to baseline conditions (**Table 1-5**). The main forms of N in surface water following fire are NO_3^- and DON, consistent with the dominant forms reported for unburned watersheds (Binkley et al., 2004; Mast and Clow, 2008; Bladon et al., 2008). Although short-term effects are the most dramatic, changes to N cycling may persist for many years in aquatic systems downstream of burns (Meixner et al., 2006).

1.2.3. Phosphorus

1.2.3.1. Phosphorus Cycling

Phosphorus cycling plays an important role in governing aquatic geochemistry and water quality. As a limiting nutrient in many systems, input of readily bioavailable forms of P can lead to algal growth and initiate complex changes in in-stream ecosystems influencing water chemistry and stream biota through eutrophication. However, not all P is readily bioavailable and therefore the forms of P (**Table 1-2**) are important. Deposition and resuspension, biotic uptake, and longitudinal cycling are important for predicting downstream effects of P on water quality (**Figure 1-3**).

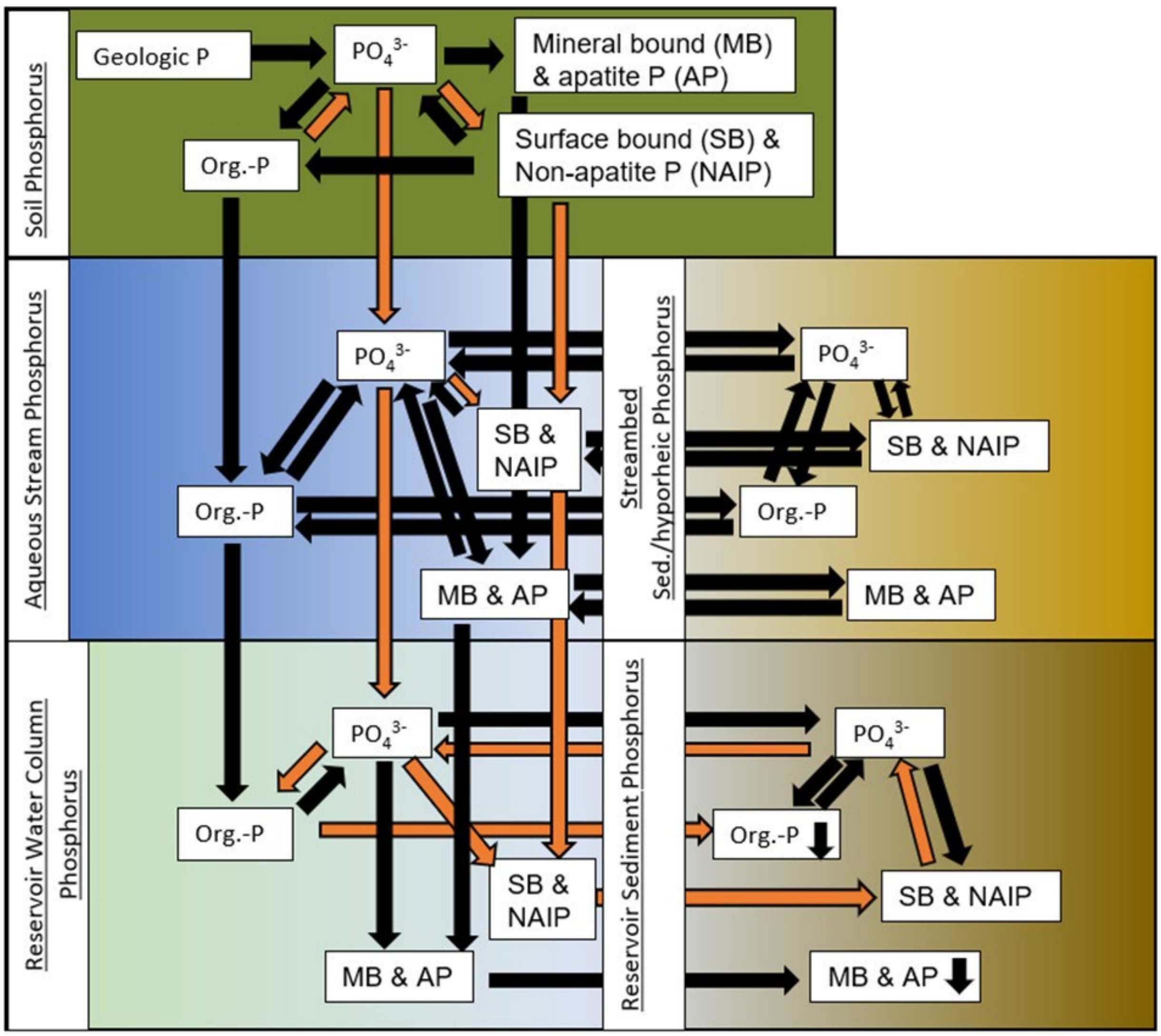


Figure 1-3. Phosphorus cycling across environmental compartments.

1.2.3.2. Phosphorus Inputs to Terrestrial Environments

The cycling of P in terrestrial systems is controlled by the soil P content, and more importantly, the mineral associations of P that limit availability. The two main sources of P to soils—parent rock weathering and atmospheric inputs—provide P to vegetation. Annual inputs of P to soil from weathering parent rock range from 0.01 to 1 kg ha^{-1} yr^{-1} (Newman, 1995). Natural atmospheric P inputs include deposition of fine spray from water bodies, crustal rock and soil particles (dust), and biomass aerosols (including plant particles of forest fires). P spatial loads range from 0.1 – 2 kg ha^{-1} with a median value of 0.6 kg ha^{-1} across the globe while soil P concentrations can range from 0.2 to 0.8 g kg^{-1} (Newman, 1995; Tiessen, 2008). Soil P concentrations are mainly driven by organic matter decomposition, which may be as high as 80% of the P input in forests (Tiessen, 2008). In a typical eucalyptus forest, above-ground P pools were determined to be about 50 kg ha^{-1}, with 10 kg ha^{-1} in the litter layer, whereas rock weathering inputs were estimated at 0.1 kg ha^{-1} yr^{-1}, demonstrating the accumulation of P in forests is in biomass (Attawill and Adams, 1993). Because P is limited, and biomass recycles landscape P via decay, forest ecosystems are reservoirs for landscape P and do not release large percentages of the spatial P load.

1.2.3.3. Phosphorus Speciation in Surface Water

P does not undergo changes in oxidation state in freshwater cycling; all forms of P present are pentavalent. P can exist as orthophosphate (ortho-P, PO_4^{3-}) with different protonation depending on pH. Ortho-P can associate with mineral surfaces where it is strongly bound to Al and Fe and can precipitate with Ca as the mineral apatite ($Ca_5(PO_4)_3(Cl/F/OH)$; House & Denison, 2002; Kopáček et al., 2005; Withers and Jarvie, 2008). Available P is defined as dissolved P and adsorbed P on calcium carbonate and Fe-oxide surfaces. Ortho-P is the only form of P that is available for biotic uptake and assimilation; once it is taken up in plant material, P exists as an organophosphate. Although only ortho-P is available for biotic assimilation, other forms of P may eventually transform into ortho-P (Correll, 1998). Thus, many forms of P could be considered as potentially bioavailable.

Particulate-associated phosphate represents the majority of P entering aquatic systems (Kronvang et al., 1997; House, 2003). Although P is transferred into surface water as particulate associated initially, free ortho-P can be released from the particulates. Non-apatite inorganic phosphorus (NAIP) can be characterized based on binding to specific environmental surfaces. For instance, loosely-sorbed P is bound by carbonate and organic complexes, reducible P is associated with Fe- and Mn-oxides subject to reduction, and metal-oxide-bound P is associated with Fe and Al. Phosphate can buffer in aquatic systems due to the surface interactions with sediments. As P concentrations increase, P is increasingly adsorbed to sediments. As P concentration decreases, P is released from sediments back into the water column (House, 2003). All NAIP is potentially available for assimilation as free phosphate based on environmental conditions (Correll, 1998). Other forms are not likely to increase ortho-P. For instance, apatite P is generally thought of as a sink for P and is relatively stable in the environment (Correll, 1998).

1.2.4. Fire Effects on Phosphorus Cycling

1.2.4.1. Effects of Combustion on Phosphorus Speciation

Wildfire can chemically transform phosphorus into soil (**Table 1-6**), but the total P soil abundance is not greatly changed. Loss of inorganic soil P to volatilization are reported to be minor, as P does not volatilize until a temperature of around 770°C. However, soil organic P can be converted to ortho-P during organic matter combustion (Caon et al., 2014), which is retained on soil particles due to strong complexation with environmental surfaces (Maurice, 2009).

1.2.4.2. Effects of Fire on Phosphorus Cycling in Soil

There are conflicting reports regarding P cycling in soils horizons following fire. O-horizon P has been reported to increase in some reports (Kutiel & Naveh 1987, Wienhold & Klemmedson, 1992) and to decrease or remain unchanged in others (Debano & Conrad, 1998; Murphy et al., 2006, Johnson et al., 2007; Otero et al., 2015). It is most likely that P concentrations in surface soils are dependent on ash deposition and persistence, as well as local site characteristics. Losses of total P (TP) from A-horizon soils appear to be uncommon (e.g., Thomas et al., 2000) though this has been reported (Santín et al., 2015).

Because P is not lost to the atmosphere it tends to become enriched in post-fire soil. P concentration in ash can be higher than soil. Typical soil P increases of 2- to 3-fold occur following fire due to ash enriched in P depositing in the soil (Kutiel & Naveh, 1987; Gómez-Rey & González- Prieto, 2014). It is likely that much of the TP in ash is in the form of ortho-P (Rodríguez et al.
2009).

Burn severity is an important factor in P mineralization. Heating of organic matter converts organic P into ortho-P (Schaller et al., 2015; Caon et al., 2014; Santín et al., 2015). The conversion of organic-P to ortho-P in ash causes enrichment compared to local soil (Santín et al., 2015). Fire severity also changes the forms of P, decreasing organic P and Ca-P while increasing oxide P and inorganic P (Otero et al., 2015). Interestingly, P bound with Al oxides seems to be the predominant form of P in ash (Santín et al., 2015), despite Ca being frequently more abundant than Al in ash (Raison, 1979). The amount of available P appears to increase following burning (Ferran et al., 2005). It is likely the mineral-associated P can be desorbed at increasing pH values, increasing P bioavailability (Ferran et al., 2005; Escudey et al., 2010). Rainfall on ash may also increase the availability of P, at least in the short term (Francos et al., 2016).

1.2.4.3. Effects of Ash Input on Soil Phosphorus

The length of persistence of altered P cycling post-fire is variable. Extractable P can be higher in soil following burning, but these effects appear to be seasonal, probably decreasing and increasing with wetting and drying or just steadily decreasing over a period of months (Gómez-Rey & González- Prieto, 2014; Francos et al., 2016). A sharp increase in soil P followed by a relatively quick return to pre-fire concentrations is not universal, as long-term increases in soil P post-fire have been reported at two years following fire (Marañón-Jiménez, 2013). The observed variation in persistence of post- fire P cycling is likely due to ash mobility by wind and rain, and deposition into deeper soils (Kutiel & Naveh, 1987; Santín et al., 2015).

1.2.4.4. Effects of Fire on the Spatial Loading of Phosphorus

The spatial loading of P does not appear to change during burning; P is transferred from biomass to surface soil as deposited ash. Evaluation of nutrients in chaparral (before and after wildfire found biomass P loads of 10 kg ha^{-1} that decreased to 0.8 kg ha^{-1} with a simultaneous increase of surface soil P from 22 to 31 kg ha^{-1}, suggesting direct transfer (Debano & Conrad, 1978). Soil P increases are not universal; Santín et al. (2015) studied eucalyptus forests that contained a litter layer with 1 kg-P ha^{-1} and a O horizon soil of 3 kg-P ha^{-1} and found that burned litter and O-horizon soil produced a TP pool of 2 kg-P ha^{-1}, suggesting a net loss of P from the location.

1.2.4.5. Effects of Fire on Downstream Surface Water Phosphorus

Concentrations of TP are typically elevated in surface water draining burned areas (**Table 1-7**), although these effects tend to be short lived. There can be large increases in TP concentrations in overland flow in the first year following fire, but TP then tends to decrease in subsequent years, in some cases TP concentrations decrease to below pre-fire conditions (McColl and Grigal, 1975). A two-year window for elevated P is common in the literature (Lane et al., 2008; Hauer & Spencer, 1998; Silins et al., 2014) and some have found altered P geochemistry even seven years following fire (Emelko et al., 2016). The instream concentrations caused by fire tend to increase TP over controls or pre-fire conditions from about 2- to 10-fold (McColl & Grigal, 1975; Hauer & Spencer, 1998; Brass et al., 1996). Although P export can increase following fire, if dilution is sufficient there may be no measurable change in P concentration in a downstream lake, or the loading may still be within the waterbody's annual loading variation (McColl & Grigal, 1975).

Not all studies have found elevated P in downstream surface water following burning (e.g. Mast & Clow, 2008). Erosion and stream energy both factor into observed TP; particulate P may increase in storm flows but not in base flow (Prepas et al., 2003). The in-stream concentration of TP may be driven more by erosion and overall sediment transport than changes in sediment P concentration.Concentrations of P on sediments from both burned and unburned watersheds have been shown to be in the range of 0.3 to 0.6 mg g^{-1} in coarse material regardless of burn history, whereas total sediment yields can vary by 40- to 70-fold (Noske et al., 2010).

Due to strong correlation with erosion and sediment transport, concentrations of P in surface water vary greatly across studies. While not all that helpful on their own due to differences in geographical setting, some of the reported concentration ranges are of interest. Brass et al. (1996) found TP levels that rose from 0.2 mg L^{-1} to 2 mg L^{-1} peak concentrations following fire, Lane et al. (2008) found peak levels as high as 40 mg L^{-1}, and Son et al. (2015) found concentrations as high as 100 mg L^{-1} TP. Concentrations of TP above about 20 μg L^{-1} are likely to lead to eutrophication in most waterbodies (Correll, 1998).

Changes in P chemical composition in aquatic systems due to fire may increase its reactivity. Multiple studies have measured 2- to 6-fold increases in soluble reactive phosphorus (SRP; ortho-P and loosely-sorbed ortho-P) that may last for years following fire (Hauer & Spencer, 1998; Silins et al., 2014). Following fire, NAIP concentrations tend to increase and may compose around 50% of particulate P (PP; Allin et al., 2012). The increased NAIP may remain elevated in sediments downstream of a burned watershed for prolonged periods (7 yr; Emelko et al., 2016). One reason that P is elevated in post-fire suspended and bed sediment may be that wildfire produces materials with greater sorption capacity for phosphate (Son et al., 2015). The increase in NAIP suggests a greater bioavailability of P in fire-affected sediments.

1.2.4.6. Effects of Fire on Watershed Export Yields of Phosphorus

The export of TP from burned watersheds tends to increase following wildfire (**Table 1-8**). Generally, the increases are short lived, with export returning to pre-fire levels or even below pre- fire levels within 2 to 4 years. The TP export loads have been reported from of 0.3 kg ha^{-1} to 2 (±0.07) kg ha^{-1} TP in the first year following fire (Wright, 1976; Williams and Melack, 1997; Thomas et al., 2000); Lane et al., 2008). The TP loads decrease by about 70% per year in each of the following years (Lane et al., 2008). The relationship between P in coarse material, suspended sediment (SS), and dissolved P is altered following fire, with SS loading accounting for 90% of the total export and dissolved P accounting for 5% initially. As the watershed recovers from fire impacts, the fraction of P associated with suspended sediment decreases to around 75%.. The remaining fraction (59%) of total P transported was associated with other course matter (carbon- based and soil aggregate) (Noveske et al., 2010).

1.3. Fire Retardant Application and Potential for Altered Nutrient Cycling

Wildland fire suppression has used fire retardants since the 1930s and began using aerial deployment of retardants in the 1950's. As early as the 1960s, the U.S. Department of Agriculture - Forest Service (USDA-FS) applied tens of millions of liters of AP fire retardant annually (USDA, n.d.). The western states (USDA-FS Regions 1-6) are the primary region of use. The average annual application from 2012 to 2019 on USDA-FS lands was 48×10^6 L (approximately 7,100 aerial drops per year) (with about 60% in USDA-FS region 5 (seventeen national forests in California) (USDA, n.d.). In California, CalFire (California Department of Forestry and Fire Protection) also applied 57×10^6 L of AP fire retardants and in addition to the USDA application during the 2016 fire year (Weiser, 2017).

Retardant is applied to the landscape by airplanes, helicopter, or ground crews; the most common application method is by airplane. There are several airtanker types; these are distinguished by the delivery capability which ranges from 3000 L to 30,000 L (800 to 8,000 gal) (USDA, n.d. B). The application rate of retardant depends on the fire intensity, which has a relation with fuel type. A typical PHOS-CHEK® delivery rate is 1 L m^{-2} for fires with an intensity of 2000 kw m^{-2} (Bell et al., 2005). The coverage is frequently measured in units of "coverage level" expressed as "gpc" (gallons per 100 square feet). One gpc (0.01 gal ft^{-2}) is equal to 0.4 L m^{-2}. According to the USDA-FS (2011), the typical application rate ranges from 2 to 6 gpc (0.8 L m^{-2} to 2.4 L m^{-2}). Application varies by region and fuel type, most western-state fuel types are targeted with 2 gpc, though application rates greater than 6 gpc may be used in chaparral fuel types. The average drop dimension is 15–23 m x 244 m (50–75 ft x 800 ft) (USFS, 2011).

1.3.1. Fire Retardant Composition and Mechanism

Initially, retardants were made using borate salts, though these were shown to have plant toxicity and were eventually phased out and replaced with AP salts; AP based fire retardants have been used since the 1960s (Norris & Webb, 1989). The chemical composition of the currently used AP salts can vary and some of the mixture additives remain proprietary. Generally, AP-based retardants consist of monoammonium phosphate (MAP), diammonium phosphate (DAP), ammonium polyphosphate (APP) or combinations of these compounds. Diammonium phosphate ($[NH_4]_2HPO_4$) and ammonium polyphosphate ($NH_4[PO_3]_n$) are the main constituents in fire retardants (Giménez et al., 2004; PHOS-CHEK® product website, 2021). Historically, the retardants have been marketed under a variety of names, all with slightly different compositions (Giménez et al., 2004). Although the primary ingredients are similar, other chemicals may be added to reduce corrosion, enhance wetting, adhere retardant to surfaces, and provide color for visual determination of the drop area. Though there are slight variations in composition, most fire retardants are about 85% water and 10% AP salts with 5% other additives such as thickeners, coloring agents, corrosion inhibitors, stabilizers, and bactericidal agents (USDA-FS, 2011).

The retardant is applied to biomass prior to combustion where it interacts with the fuel surface and reduces combustion. The application of AP onto woody materials decreases the pyrolysis temperature, causes the release of combustible gases below their ignition temperature, and decreases flaming combustion of fuel (George & Susott, 1971; Giménez et al., 2004; Liodakis et al., 2009; Scarff et al., 2012). As the fire burns fuels with retardant, some of the AP is consumed (Crouch et al., 2006; Liodakis et al., 2009). The NH_4^+ is lost as NH_3 gas and phosphoric acid is generated according to a temperature-dependent reaction series (Liodakis et al., 2009):

$$(NH_4)_2HPO_4 \Rightarrow NH_4H_2PO_4 + NH_3(g)\ (150°C)$$

$$2\ NH_4H_2PO_4 \Rightarrow (NH_4)_2H_2P_2O_7 + H_2O(g)\ (170°C)$$

$$(NH_4)_2H_2P_2O_7 \Rightarrow 2\ NH_4PO_3 + H_2O(g)\ (280°C)$$

$$2\ NH_4PO_3 \Rightarrow P_2O_5 + 2NH_3(g) + H_2O(g)\ (660°C)$$

$$P_2O_5 + 3H_2O \Rightarrow 2H_3PO_4$$

1.3.2. Review of Studies on the Mobility and Environmental Effects of Fire Retardant

The primary risk to aquatic organisms is the potential for NH_3 toxicity; thresholds for NH_3 toxicity are on the order of 1 mg L^{-1} (Labat, 2017). There are two primary pathways for retardants to enter waterbodies: (1) direct application to surface water and (2) runoff from surfaces following application.

The direct application of retardant to waterbodies has been the most studied of the two scenarios for aquatic contamination. Most fish kills related to retardant use are due to accidental application to waterbodies via airdrops or inadvertent chemical spills of retardant into waterbodies (USDA-FS, 2011; Labat, 2017). The dilution of the constituents, combined with the rapid oxidation of NH_3 to NO_3^- in surface water, quickly decreases the concentrations of NH_3 and these events are relatively short-lived(<24h) (USDA-FS, 2011; Labat, 2017). In addition to ammonia, early retardant mixtures included a corrosion inhibitor, sodium ferrocyanide ($Na_4Fe(CN)_6$), which is toxic to aquatic life. Because of the toxicity potential, the USDA-FS discontinued the use of retardants containing sodium ferrocyanide after the 2007 fire season (USDA-FS, 2011; Labat, 2017). To minimize the potential for water contamination the USDA-FS restricts aerial applications within 300 ft (91 m) of surface water (USDA-FS, 2011, 2022).

The second scenario for entry of AP fire retardants into waterways is more complex and depends on the persistence and mobility of AP salts in the environment. The application of retardant on soil is, effectively, a localized extreme-fertilization event. The application of fertilizer may have short- and long-term influence on nutrient cycling in the application area. In general, the vegetation response will depend on vegetation type and post-application weather patterns (Adams & Simmons, 1999).

Larson and Duncan (1982) used application rates of 381 kg ha^{-1} N and 408 kg ha^{-1} P to evaluate the effects of retardant on grassland forage yields and reported increases in growth in areas receiving applications. It was also noted that plots receiving retardant applications showed increased growth over similar treatments without application (burned or unburned), while increased forage was also strong in burned plots without retardant.

Bell et al. (2005) applied a mixture of 0.144 kg retardant per liter (12% w:v) with varying application rates of 0.5, 1, and 1.5 L m^{-2} and measured plant abundance on plots. They suggest anecdotal evidence for some fertilization effects but were unable to measure response by vegetation to the retardant application. The authors did acknowledge their methods may not have been able to detect subtle changes, so the possibility of effects was not ruled out (Bell et al., 2005). The observations of increased yield following applications suggest a greater amount of available nutrients in the soil from the retardant application, but also reveals the complexity of fire-caused alteration of nutrient cycling and the difficulty in assigning observed effects to a single cause.

Couto-Vázquez et al. (2006) examined the persistence of common fire retardants at 0–2 cm depth after 1, 3, and 12 months after application combined with prescribed fire. The experiments used a series of adjacent subplots which received one of the following treatments: (1) no burning and no retardant (control), (2) water only, (3) water and foaming agent (AUXQUIMIA® RFC-88), (4) water and FIRESORB® (proprietary terpolymer), and (5) water and APP, each applied at 2 L m^{-2}. When using APP, NH_4^+ was 1800 mg kg^{-1} after application compared to < 10 mg/kg in the control. Concentrations of NH_4^+ decreased by over 50% in the first 30 days (750 mg kg^{-1}) and was below 500 mg kg^{-1} at 90 days. After one year, NH_4^+ was indistinguishable from the control. NO_3^- was statistically higher than the control at 30 days, peaked at 140 mg kg^{-1} at 90 days, and was then indistinguishable from the control after one year. Available P was initially just over 800 mg kg^{-1} after application, then 200 mg kg^{-1} at 30 days, 150 mg kg^{-1} at 90 days, and 50 mg kg^{-1} after one year (still significantly greater than control (<10 mg kg^{-1}) (Couto-Vázquez et al., 2006).

Chemical data from soil profiles receiving retardant application show that AP-based retardants can leave lasting effects. Fernández-Fernández et al. (2015) reexamined the experiment plots of Couto- Vázquez et al. (2006) 10 years after the prescribed fire and found that soils with NH_4^+ polyphosphate addition had N concentrations that were not significantly different than the control. On the same plots, P remained considerably higher at 14 mg kg^{-1} compared to less than 5 mg kg^{-1} in the soils receiving the other applications.

Pappa et al. (2006) studied the leaching of P from soils with applied retardant in a laboratory setting that used intact soils cores, some with living trees, in pots. The pots were covered in natural materials, such as pine needles, and allowed to stabilize in the controlled setting with continued watering for a period of time. Once stabilized, selective application of FIRE-TROL® (AP retardant) and burning of the pot surfaces was carried out resulting in the following treatments: (1) unburned soil, no retardant, (2) unburned soil, retardant, (3) unburned with tree, and (4) and burned with tree. The experiments then simulated rainfall on the pots by irrigating every 12 hr for one month; leachate was collected from the bottom drain layer of each pot every three days for one month resulting in 10 leachates per pot. Three treatments (soil, burned tree, and soil with living tree) were found to leach 2.4%, 1.3% and 0.4 % of the P in retardant, respectively. Extrapolating the application rate to a coverage level of 890 kg P ha^{-1} corresponds to a maximum leaching rate of up to 25 kg P ha^{-1}, representing a small percentage of the initially applied P. The highest P leaching rate was from the pot with retardant and no trees, therefore the presence of vegetation seems to affect net release of P from soil.

Pappa et al. (2008) reported on N leaching from previously described bucket experiments (Pappa et al., 2006) and found that soils treated with retardant leached much greater amounts of N than those without retardant treatments. Of the initially applied inorganic N, average cumulative total leaching from the retardant-treated samples was found to be 30% from soils with retardant, 21% from soils with a tree and retardant, and 12% from soils with a tree, retardant, and fire. The forms of N affected the leaching behavior; as NO_3^- continued to evolve in the soil (from oxidation of NH_4^+, the leachate concentrations increased over 10-fold from the initial leachate concentration (5 mg L^{-1} to 70 mg L^{-1}). The average cumulative leaching of NH_4^+-N from all the treated samples was lower than 3% of the initially applied N. Extrapolating application and leaching values, Pappa et al. (2008) reported a retardant application rate of 450 kg N ha^{-1};

leaching rates of total inorganic N from treated pots ranged from 56 to 134 kg N ha^{-1} compared with leaching rates from the control pots of 10–12 kg N ha^{-1} over the 30-day leaching period. As with the study on P leaching (Pappa et al., 2006), N leaching rates from pots with retardant and no trees were higher than those from pots with trees (Pappa et al., 2008), that the presence of vegetation affects N release from soil by nutrient uptake.

Aerial fire retardant is not always applied directly to ground surfaces as much of it deposits on vegetation. Xanthopoulos et al. (2006) examined rainfall-induced removal of retardants from pine needles under controlled settings and found that most retardant could be removed from needles with moderate wetting over a period of less than an hour. In their experiment, FIRE-TROL® 936 was applied after dilution to 20% v:v with water. This mixture is a typical retardant used in ground application and is not gum thickened, therefore it may not be a good representation for thickened retardants used in aerial applications. The process used bundles of needles to which retardant was applied; the bundles were oven dried at 80°C and then exposed to natural precipitation for a timed period. Rainfall was measured and removal was regressed on rainfall amount. Xanthopoulos et al. (2006) found the removal was a function of time and rainfall amount, i.e., if one cm of rain is applied over a short duration (5 min) the removal is lower than the same cm of rain applied over a longer period (40 min).

The occurrence of retardant entering waterways via direct surface runoff from the terrestrial environment remains unsupported in the literature. Boulton et al. (2003) conducted a study on Kangaroo Island, South Australia, and studied three sites with varying retardant applications. The study included two locations where retardants had been applied over a stream channel and riparian area and a third location that was burned but received no retardant (control). Samples of water were collected within two weeks of application, but there were no marked differences in water quality among the sites. The authors noted that in one of the streams where fire retardants were applied, flow had ceased before the fire and dissolved, and total P concentrations were elevated at the site. It was unclear if this result had to do with the applications or drying of stream water. Otherwise, there were no observable differences in water chemistry attributable to retardant application (Boulton et al., 2003).

In a study of retardant movement following applications, Crouch et al. (2006) examined multiple watersheds across western North America that received AP fire-retardant application. The fires studied were the Cerro Grande Fire in New Mexico, the Silver Creek Fire in British Columbia, the Viveash Fire in New Mexico, and the Rodeo-Chedeski Fire in Arizona. The Cerro Grande fire burned 17,000 ha and 4.2 $x10^6$ L of retardant was applied; the Silver Creek Fire was 6400 ha and 2.8 x 10^6 L retardant was applied; the Viveash Fire in NM burned 11,700 ha and 1.56 x 10^6 L of retardant was applied; and the Rodeo Fire Arizona burned 195,000 ha, and 2.8 $x10^6$ L of retardant was applied. In addition to different landscapes, the hydrological effects varied as well among these settings, with some areas having large overland flow and erosion events and others not. Overall, the study evaluated a wide range of conditions over multiple forest types and hydraulic responses. Each sampling location was characterized as "burned no treatment", "burned treatment", or "unburned". Upon examination of NH_3 and total P in runoff, there were no statistical differences found attributable to the use of fire retardants. The group did report strong correlations between the measured constituents and calcium in surface water. The correlation between NH_3 and Ca was strong ($R^2 = 0.755$, $p <.001$) and total P was also strongly correlated with Ca ($R^2 = 0.741$, $p <.001$). The authors considered calcium a tracer for burning, thus an indication they were observing fire effects, but no retardant effects. The results of Crouch et al. (2006) provide the strongest evidence that properly applied AP fire retardants have little to no discernable effects on downstream nutrient levels under most settings.

One study has found evidence that N was elevated in surface water following application of aerial AP fire retardants. Tobin et al. (2015) investigated a 1500 ha fire in the Kaweah River watershed in the southern Sierra Nevada, CA, that received 20 applications of PHOS-CHEK® D75. The fire occurred in September 2008 and winter rains began in December 2008. Waters from multiple tributaries and drainages within the fire perimeter were sampled and NO_3^- concentrations were found to correlate positively with application of retardant. The authors noted that each basin had similar vegetation and burning, so NO_3^- should have been similar. Similar trends were not observed for other anions. The authors suggest that the results of their study had to do with the karst geology in the area and that when NO_3^- enters karst aquifers it can move through conduits behaving as a conservative tracer.

It is likely that the downstream effects of AP fire retardants are highly dependent on watershed characteristics. Small basins with small waterways (low-order streams) are probably more likely to have increased nutrient loads due to application of retardant than larger watersheds with greater stream order and flow rate.

The company Labat Environmental, Inc. has previously used the Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) model to estimate the effects of retardant application on surface water (Labat, 2017). The modeling effort was designed to evaluate the concentration of ammonia, the most toxic compound in retardant, in streams receiving either accidental application (direct application to streams) or runoff from application adjacent to streams. The GLEAMS model (Leonard et al., 1987, 1988) considers hydrology, erosion, nutrients, and pesticides using soil characteristics, and water movement through the soil column. Version 3.0 of the GLEAMS model (Knisel and Davis, 2000) improved handling of forested areas. The erosion of soil is considered based on particle size and organic matter content to account for constituent enrichment by size fraction over the bulk soil. The model was designed specifically for predicting fertilizer age and is a good tool for modeling retardant fate as AP fire retardants are very similar to agricultural fertilizers regarding their N and P constituents (Labat, 2017).

Labat (2017) ran the GLEAMS model to evaluate the concentration of retardant constituents in stream flow originating from non-accidental use (intended applications). Based on the modeling results, they reported: "no risks of acute toxic effects were predicted for aquatic wildlife from runoff to streams in drainage basins where mixed (diluted) retardant was applied as intended to fuels (vegetation) during firefighting activities." The modeled concentrations were not reported. For accidental application (retardant is dropped directly over waterbodies), maximum reported $NH_3 + NH_4^+$ concentrations were 3.3 $\mu g\ L^{-1}$ (PHOS-CHEK® 259 FX). Other forms of PHOS-CHEK® resulted in lower concentration (0.2 – 1.3 $\mu g\ L^{-1}$).

1.4. Summary of Literature Review

A review of forest N cycling reveals that total N deposition rates (wet + dry, all species) are less than 4 kg-N ha^{-1} yr^{-1} across most forested land in the western U.S (NADP, 2020). Most the of the N deposited in forested lands is retained on the landscape in biomass, therefore export yields are a fraction of the inputs at only 0.01 – 0.6 kg/ha yr total inorganic nitrogen (TIN; **Table 1-3**). Following fire, TIN concentration in soil increases, mainly as NH_4^+ by up to 2000 % (**Table 1-3**). The subsequent export of N is also greatly increased and has been reported to be as high as 68.8 kg N ha^{-1} (**Table 1-3**).

Each aerial drop of fire retardant covers about 0.5 ha (USDA-FS, 2011) and the N loading rate is 173 kg (assuming MAP) per drop. At the local scale (i.e., in the drop zone) the application rate of N is far greater than the typical natural annual loading rate and likely contributes more than the pre- existing pool of total N, so this may be considered to be an extreme fertilization event. However, deposition rates at the local scale do not necessarily translate into downstream effects. A typical 1st order stream has a watershed area of 320 ha (Hughes et al., 2011). To consider the added N and P based on an annual load, A single application (aerial drop) in a 1st order watershed would increase the total N level in the watershed by about 0.5 kg ha^{-1} average across the watershed, thus this would be the maximum addition to the annual export yield. If it is assumed that all added retardant N is mobilized from the watershed, the total export from the retardant alone would be around the typical unburned export level. It is possible that small watersheds with lower N export could have measurable increases in TN; however, considering the likely increases in N export following fire of between 0.05 – 70 kg N ha^{-} (**Table 1-3**), the increase in N from AP fire retardant would not be a major source compared to other watershed sources. Even with many applications it is unlikely that a measurable increase would be detected, particularly for increasingly larger watersheds (**Figure 1-4**). The atmospheric loading of P in terrestrial systems is not nearly as great as that for N, with a median value of 0.55 kg-P ha^{-1} across the globe (Newman, 1995). The above-ground forest P pool is expected to be around 50 kg ha^{-1} (Attawill and Adams, 1993) and of this total P pool only a small fraction is released into surface water annually (0.02 – 0.2 kg/ha; **Table 1-4**). The spatial loading of P does not appear to change much during burning; P is transferred from biomass to surface soil as deposited ash and as enriched soil (Schaller et al., 2015; Caon et al., 2014; Santín et al., 2015). As with nitrogen, increases in soil P following burning, coupled with loss of vegetation and soil stabilization, can mobile P causing increases in P export as high as 1.7 kg ha^{-1} (**Table 1-4**).

Each aerial retardant drop of MAP (from a Type-1 aircraft) deposits around 380 kg of TP. Based on the median 1st order watershed size (320 ha; Hughes, 2011) the watershed loading per drop would be around 1.2 kg ha^{-1} if considered across the entire watershed, this mass would be the additional annual export yield. Assuming all added P was mobilized from the small watershed, this would be near the high end of reported post-fire export yield (**Figure 1-5; Table 1-4**). Theoretically, in small watersheds, this loading could constitute a measurable amount of TP and observable increased P export would be more likely with increased aerial applications.

The assertion that measurable changes to nutrient export could occur downstream of retardant application is moderated by the acknowledgement that there are real-world constraints on the assumption that all added nutrients (N or P) would be mobilized to produce the results shown on **Figure 1-4** and **Figure 1-5**. The transport of all applied nutrients is not likely. Indeed, a review of studies on retardant applications found strong evidence that retardant is retained in soil for years following application and not directly mobilized in overland runoff (Fernández-Fernández et al., 2015). Further, previous studies of waterbodies downstream of retardant applications have not reported any measurable increases in N or P (Crouch et al., 2006). Currently, there is no empirical evidence that retardant produces measurable changes to nutrient exports.

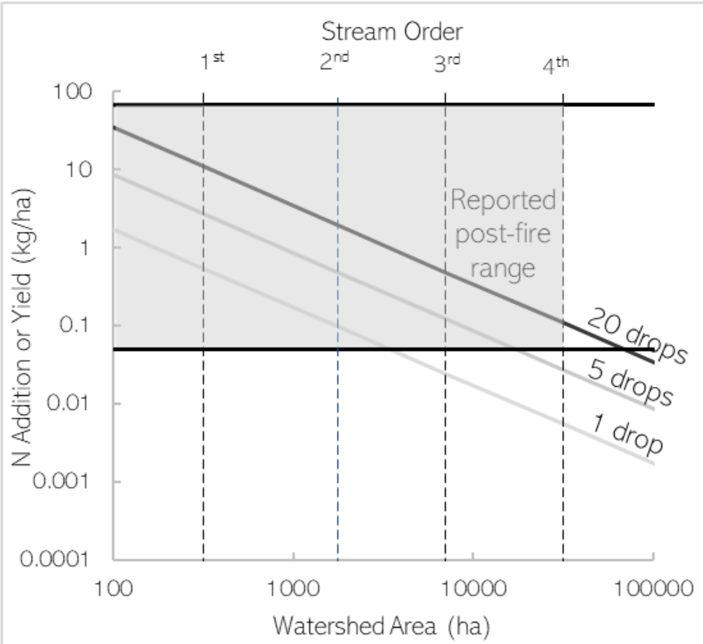


Figure 1-4. Maximum total nitrogen (TN) export from retardant application as a function of number of 11,000-liter (3,000-gallon) aircraft drops and watershed size.

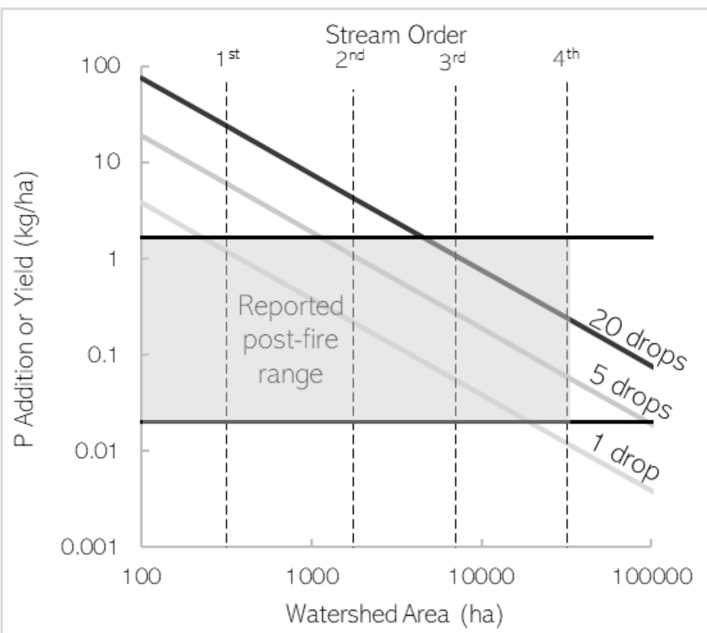


Figure 1-5. Maximum total phosphorus (TP) export from retardant application as a function of number of 11,000-liter (3,000-gallon) aircraft drops and watershed size.

1.5. References

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Tables

Table 1-1. Types of nitrogen referenced in various studies and their descriptions.

Form	Description
Organic N	$R\text{-}NH_2$ e.g., amino acids, biomass N
Dissolved organic N (DON)	Soluble molecules with organic N
Soil organic labile N	Biomass N + DON
Mineral N (Inorganic N)	$NH_3/NH_4^+ + NO_3^- + NO_2^-$
Total dissolved N	$NH_3/NH_4^+ + NO_3^- + NO_2^-$ + DON
Kjeldahl N	Organic N + NH_3/NH_4^+

Table 1-2. Types of phosphorus referenced in various studies and their descriptions

Form	Description
Organic P	$R\text{-}OPO_3$ e.g., biomass P
Orthophosphate (Ortho-P)	H_3PO_4 , $H_2PO_4^-$, HPO_4^{2-}, PO_4^{3-}
Particulate P	e.g. suspended sediment with P
Oxide-bound P	Aluminum and iron oxide bound P
Apatite P	Calcium bound P
Non-apatite inorganic P (NAIP)	All non-calcium bound ortho-P
Soluble reactive P (SRP)	Ortho-P + loosely sorbed ortho-P
Reductant soluble P	Iron and manganese oxide bound P
Loosely sorbed P	Calcium carbonate and organo-complexed P
Available P	Ortho-P, oxide-P, loosely sorbed P
Total P	All forms of P

Table 1-3. Summary of reported soil nitrogen concentrations and trends following wildfire

Year	Author	Fire	fuel type	Region	Matrix	Time	N conc.	N form	Trend
1987	Kutiel and Naveh	Unburned	Pine	Carmel NP, Israel	Soil	< 1yr	4000 mg/kg	TN	-
1987	Kutiel and Naveh	Unburned	Pine	Carmel NP, Israel	Soil	< 1yr	10-37 mg/kg	NH_4^+- N	-
1987	Kutiel and Naveh	Unburned	Pine	Carmel NP, Israel	Soil	< 1yr	5-12 mg/kg	NO_3^- - N	-
1987	Kutiel and Naveh	Burned	Pine	Carmel NP, Israel	soil	< 1yr	3000 – 4000 mg/kg	TN	-
1987	Kutiel and Naveh	Burned	Pine	Carmel NP, Israel	soil	< 1yr	20 - 60 mg/kg	NH_4^+- N	Increased, then decrease with time
1987	Kutiel and Naveh	Burned	Pine	Carmel NP, Israel	soil	< 1yr	17 - 55 mg/kg	NO_3^- - N	Increased through study
2005	Hatten et al.	Low	Pine/fir	Cascades, Washington, USA	soil (O -Horizon)	1 to 27 yr	15000 mg/kg	TN	Not significant from unburned
2005	Smithwick et al.	Stand replacing	Conifer	Multiple	Soil	Multiple	-	NH_4^+	196
2005	Smithwick et al.	Non-stand replacing	Conifer	Multiple	Soil	Multiple	-	NH_4^+	180
2005	Smithwick et al.	Burned	Broad leaf	Multiple	Soil	Multiple	-	NH_4^+	126
2005	Smithwick et al.	Burned	Woodland Chaparral	Multiple	Soil	Multiple	-	NH_4^+	257.52
2005	Smithwick et al.	Stand replacing	Conifer	Multiple	Soil	Multiple	-	NO_3^-	No Increase
2005	Smithwick et al.	Non-stand replacing	Conifer	Multiple	Soil	Multiple	-	NO_3^-	81
2005	Smithwick et al.	Burned	Broad leaf	Multiple	Soil	Multiple	-	NO_3^-	NS - +250%
2005	Smithwick et al.	Burned	Woodland Chaparral	Multiple	Soil	Multiple	-	NO_3^-	NS - +4400%
2009	Rodriguez et al.	Unburned	p. canariensis	Canary Islands, Spain	Soil - Ash	1 mo	4 mg/kg	NH_4^+	Increased
2009	Rodriguez et al.	Burned - Crown	p. canariensis	Canary Islands, Spain	Soil - Ash	1 mo	25 mg/kg	NH_4^+	-
2009	Rodriguez et al.	Unburned	p. canariensis	Canary Islands, Spain	Soil - Ash	1 mo	2 mg/kg	NO_3^-	Increased
2009	Rodriguez et al.	Burned - Crown	p. canariensis	Canary Islands, Spain	Soil - Ash	1 mo	18 mg/kg	NO_3^-	-
2009	Rodriguez et al.	Unburned	a. vicous	Canary Islands, Spain	Soil - Ash	1 mo	5 mg/kg	NH_4^+	Increased

2009	Rodriguez et al.	Burned - Crown	a. vicous	Canary Islands, Spain	Soil - Ash	1 mo	14 mg/kg	NH_4^+	
2009	Rodriguez et al.	Unburned	a. vicous	Canary Islands, Spain	Soil - Ash	1 mo	2 mg/kg	NO_3^-	Increased
2009	Rodriguez et al.	Burned - Crown	a. vicous	Canary Islands, Spain	Soil - Ash	1 mo	4 mg/kg	NO_3^-	
2012	Koyama et al.	Prescribed + wild	Mixed conifer	Western USA	Soil	2 yr	NS	NH_4^+	Study was 26 months after wildfire and 18 months after prescribed fire, so N was probably diminished.
2012	Koyama et al.	Prescribed + wild	Mixed conifer	Western USA	Soil	2 yr	NS	NO_3^-	
2012	Koyama et al.	Prescribed + wild	Mixed conifer	Western USA	Soil	2 yr	30 mg-N/kg	Biomass N	
2012	Stephan et al.	Prescribed	Pine, Fir	Idaho, USA	Soil	0 to 3 yr	21.5 mg/kg	NH_4^+	Persisted 1 to 2 months
2012	Stephan et al.	Prescribed	Pine, Fir	Idaho, USA	Soil	0 to 3 yr	0.8 mg/kg	NO_3^-	controls ND
2012	Stephan et al.	wildfire	Pine, Fir	Idaho, USA	Soil	0 to 3 yr	44.2 mg/kg	NH_4^+	Increase from 6.8 Persisted over 3 years
2012	Stephan et al.	wildfire	Pine, Fir	Idaho, USA	Soil	0 to 3 yr	9.4 mg/kg	NO_3^-	controls were ND
2013	Maranon-Jimenez et al.	High	Pine	SE Spain	Standing Dead	6 mo	1630 – 1890 mg/kg	TN	
2013	Maranon-Jimenez et al.	High	Pine	SE Spain	Soil	2 yr	600 - 2000 mg/kg	TN	no controls, just post fire in four sites
2013	Maranon-Jimenez et al.	High	Pine	SE Spain	Soil	2 yr	2.5 - 3.7 mg/kg	NH_4^+	
2013	Maranon-Jimenez et al.	High	Pine	SE Spain	Soil	2 yr	0.81 - 2.14 mg/kg	NO_3^-	
2014	Gomez-Rey & Prieto	Medium-High Severity	Pine	NW Spain	Soil (0-2 cm)	0-1 yr		NH_4^+	+ 4800% increases
2014	Gomez-Rey & Prieto	Medium-High Severity	Pine	NW Spain	Soil (0-2 cm)	0-1 yr	55 mg/kg / 67 mg/kg	NH_4^+	Initial/peak
2014	Gomez-Rey & Prieto	Medium-High Severity	Pine	NW Spain	Soil (0-2 cm)	0-1 yr		NO_3^-	+700% increase
2014	Gomez-Rey & Prieto	Medium-High Severity	Pine	NW Spain	Soil (0-2 cm)	0-1 yr	3.5 mg/kg / 8.5 mg/kg	NO_3^-	Initial/peak
2015	Santin et al.	Low	Eucalyptus	SE Australia	Ash	-	3900 mg/kg	TN	-
2015	Santin et al.	Moderate	Eucalyptus	SE Australia	Ash	-	3000 mg/kg	TN	-
2015	Santin et al.	High	Eucalyptus	SE Australia	Ash	-	3000 mg/kg	TN	-

2015	Santin et al.	Unburned	Eucalyptus	SE Australia	Soil (3 - 8 cm)	-	700 mg/kg	TN	-
2015	Santin et al.	Low	Eucalyptus	SE Australia	Soil (3 - 8 cm)	-	600 mg/kg	TN	-
2015	Santin et al.	Mod	Eucalyptus	SE Australia	Soil (3 - 8 cm)	-	600 mg/kg	TN	-
2015	Santin et al.	High	Eucalyptus	SE Australia	Soil (3 - 8 cm)	-	600 mg/kg	TN	-
2016	Francos et al.	Burned	Pine, Eucalyptus	NE Spain	Soil	3 d	3200 mg/kg	TN	-
2016	Francos et al.	Burned	Pine, Eucalyptus	NE Spain	Soil	After 1st rain	3400 mg/kg	TN	-
2016	Francos et al.	Burned	Pine, Eucalyptus	NE Spain	Soil	1 yr	3000 mg/kg	TN	-
2018	Santin et al.	Low	Eucalyptus	SE Australia	Ash	1 d	6400 mg/kg	TN	-
2018	Santin et al.	Moderate	Eucalyptus	SE Australia	Ash	2 d	6400 mg/kg	TN	-
2018	Santin et al.	High	Eucalyptus	SE Australia	Ash	3 d	6700 mg/kg	TN	-
2018	Santin et al.	Control	Eucalyptus	SE Australia	Soil (O)	Pre-fire	3900 mg/kg	TN	-
2018	Lui et al.	Control 0-10 cm	Grass	North-Central China	Soil	1 yr	3300 mg/kg	TN	-
2018	Lui et al.	Control 0-10 cm	Grass	North-Central China	Soil	1 yr	1200 mg/kg	NH_4^+	-
2018	Lui et al.	Control 0-10 cm	Grass	North-Central China	Soil	1 yr	25.4 mg/kg	NO_3^-	-
2018	Lui et al.	Burned 0-10 cm	Grass	North-Central China	soil	1 yr	3500 mg/kg	TN	-
2018	Lui et al.	Burned 0-10 cm	Grass	North-Central China	soil	1 yr	15,500 mg/kg	NH_4^+	-
2018	Lui et al.	Burned 0-10 cm	Grass	North-Central China	soil	1 yr	18.9 mg/kg	NO_3^-	-

Table 1-4. Summary of reported surface water nitrogen concentrations in recently burned watersheds

Year	Author	Fire severity	fuel type	region	Matrix	Time	N Conc.	N form	Trend	Notes
1998	Hauer and Spencer	Unburned	Mixed conifer	NW Montana, USA	Water	1 - 4 yr	< 0.001 – 0.125 mg/L	NO_3^-	streams flowing through burned areas often had an order of magnitude higher nutrient concentrations that persisted years after the fire.	Elevated P and N
1998	Hauer and Spencer	Burned	Mixed conifer	NW Montana, USA	Water	0 yr	< 0.001 – 0.065 mg/L	NO_3^-	burned had 2- 3-fold increased nitrate in years following fire, then decreased over time	Large variations for different stream
1998	Hauer and Spencer	Burned	Mixed conifer	NW Montana, USA	Water	1 - 4 yr	0.002 – 0.025 mg/L	NH_4^+	-	-
1998	Hauer and Spencer	During Fire	Mixed conifer	NW Montana, USA	Water	0 yr	0.250 mg/L	NH_4^+	-	TPN (total persulfate N - NO3 + Org N)
2003	Earl and Blinn	N/A	Pine, oak	New Mexico, USA	Water	1 mo	0.1 mg/L	NH_4^+	one measurement was above control	
						1 mo	0.08 mg/L	NO_3^-		
2006	Miexner et al.	Unburned	chaparral	S California, USA	Water	4 yr	< 0.06 to 4.3 mg/L	NO_3^-	7-10 years of elevated nitrate	Monitored 1988 to 2003
2006	Miexner et al.	Burned	chaparral	S California, USA	Water	4 yr	0.12 - 50 mg/L	NO_3^-	Largest difference in first year of monitoring (4 yr post fire)	These are the range of 1988 values (highest burn conc)
2008	Lane et al.	High	Eucalyptus	SE Australia, USA	Stream Storm flow	3 mo	5 mg/L	TKN	Peak was 12 mo after fire at 420 mg/L, but this was anomalous	Particulate N&P are major components to nutrient export
2008	Lane et al.		Eucalyptus	SE Australia, USA	Stream Base Flow	3 mo	1.5 mg/L	TDN	-	Nitrate 70% of TN
2008	Lane et al.				Stream Base Flow	3 mo	0.9 mg/L	NO_3^-	-	-

Year	Author	Fire severity	fuel type	region	Matrix	Time	N Conc.	N form	Trend	Notes
2008	Mast and Clow	Mixed	Spruce fir	Colorado Rockies, USA	Water	< 1 - 4 yr	0.19 - 0.45 mg/L	VWM TN	Increased conc in burned	No change in P, increase in N, Nitrate 90% of TN
2008	Mast and Clow	Unburned	Spruce fir	Colorado Rockies, USA	Water	< 1 - 4 yr	0.09- 0.15 mg/L	VWM TN		
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies, Canada	Stream Water	1,2,3 yr	1.1, 0.9, 0.4 mg/L	TN	N elevated for 3 years	Burn occurred in aug. - sept. 2003
2008	Bladon et al.	-	Mixed conifer	Alberta Rockies, Canada	Stream Water	1,2,3 yr	0.5, 0.5, 0.2 mg/L	NO_3^-	-	Sampling started at snow melt the following year
2008	Bladon et al.	-	Mixed conifer	Alberta Rockies, Canada	Stream Water	1,2,3 yr	0.009, 0.009, 0.004 mg/L	NH_4^+	-	-
2008	Bladon et al.	Unburned	Mixed conifer	Alberta Rockies, Canada	Stream Water	1,2,3 yr	0.2, 0.35, 0.23 mg/L	TN	-	-
2008	Bladon et al.	-	Mixed conifer	Alberta Rockies, Canada	Stream Water	1,2,3 yr	0.76, 0.13, 0.12 mg/L	NO_3^-	-	-
2008	Bladon et al.	-	Mixed conifer	Alberta Rockies, Canada	Stream Water	1,2,3 yr	0.006, 0.003, 0.003 mg/L	NH_4^+	-	-
2011	Fernandez et al.	Unburned	Eucalyptus	NW Spain	Stream Water	Pre fire	0.05 mg/L	TN	-	-
2011	Fernandez et al.	Moderate	Eucalyptus	NW Spain	Stream Water	1st yr	0.23 mg/L	TN	3 - 5-fold increase ave	-
2011	Fernandez et al.	Moderate	Eucalyptus	NW Spain	Stream Water	2nd yr ave	0.12 mg/L	TN	Decline over time	-
2011	Fernandez et al.	Moderate	Eucalyptus	NW Spain	Stream					-

	Water	3rd yr ave				0.04 mg/L TN -			
2013	Coombs and Melack	Burned	Chaparral	S California, USA	Stream Water	< 1 yr6.3 - 9.4 mg/L	NH_4^+	(32 - 38% TDN)	-

Year	Author	Fire severity	fuel type	region	Matrix	Time	N Conc.	N form	Trend	Notes
2013	Coombs and Melack	Burned	Chaparral	S California, USA	Stream Water	< 1 yr	1.6 – 3.9 mg/L	NO_3^-	5.5 increase over unburned	VWM of first three storms, peaks 740 uM (85% of TDN after first couple storms)
2013	Coombs and Melack	Burned	Chaparral	S California, USA	Stream Water	< 1 yr	1.5 – 6.6 mg/L	DON	2.8 increase over unburned	VWM of first three storms
2015	Son et al.	Mixed Severity	Mixed conifer	Colorado, USA	Storm Flow	1 mo	1 to 3 mg/L	TN	-	-
2015	Son et al.	Mixed Severity	Mixed conifer	Colorado, USA	Base Flow	1 mo	< 1 mg/L	TN	-	-
2016	Mast et al.	Pre fire	Spruce & fir	Colorado Rockies, USA	Water	-2 - 0 yr	0.08 - 0.18 mg/L	NO_3^-	average seasonal values from two years preceding and two years following	Spring runoff has highest N
2016	Mast et al.	Burned	Spruce & fir	Colorado Rockies, USA	Water	0 - 2 yr	0.12 - 0.33 mg/L	NO_3^-	nitrate was highest in second year following fire	-

Table 1-5. Export loads of nitrogen species following wildfire

Year	Author	Fire Status	Fuel Type	Region	Matrix	Time since fire	N conc	Units	Form	Notes
1997	Williams and Melack	Prescribed Burn	Mixed Conifer	CA, Sequoia NP	Water	1yr	0.5	kg/ha/yr	PN	-
		Control	Mixed Conifer	CA, Sequoia NP	Water	1 yr	0.1	kg/ha/yr	PN	-
1997	Williams and Melack	Prescribed Burn	Mixed Conifer	CA, Sequoia NP	Water	3 yr avg.	1.6	kg/ha/yr	NO_3^-	-
1997	Williams and Melack	Pre-burn	Mixed Conifer	CA, Sequoia NP	Water	7 year avg.	0.03	kg/ha/yr	NO_3^-	-
2000	Thomas et al.	Burned	Pine	north central Portugal	Water	1 yr	17.6-38.1	kg/ha/yr	TN	Average Annual load
2000	Thomas et al.	Burned	Pine	north central Portugal	Water	2 yr	6.5 -9.4	kg/ha/yr	TN	Average Annual load
2000	Thomas et al.	Burned	Eucalyptus	north central Portugal	Water	1 yr	14.44 -57.6	kg/ha/yr	TN	Average Annual load
2000	Thomas et al.	Burned	Eucalyptus	north central Portugal	Water	2 yr	31.7 - 68.8	kg/ha/yr	TN	Average Annual load
2005	Ferreira et al.	Unburned	Pine	north central Portugal	Water	1 yr	0.017	kg/ha/yr	NO_3^-	-
2005	Ferreira et al.	Burned	Pine	north central Portugal	Water	1 yr	0.49	kg/ha/yr	NO_3^-	-
2005	Ferreira et al.	Unburned	vegetation catchment	north central Portugal	Water	1 yr	0.01	kg/ha/yr	NO_3^-	-
2005	Ferreira et al.	Burned	vegetation catchment	north central Portugal	Water	1 yr	2.5	kg/ha/yr	NO_3^-	-
2008	Lane et al.	Eucalyptus	Eucalyptus	Victoira Australia	Water	1 yr	15.25	kg/ha/yr	TN	Decreases significantly each year
2008	Mast and Clow	High Severity	mixed conifer	Glacier NP, Montana	Water	3 yr avg.	2.7	kg/ha/yr	TN	-
2008	Mast and Clow	Unburned	mixed conifer	Glacier NP, Montana	Water	3 yr avg.	1.2	kg/ha/yr	TN	-
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 1	0.06 – 0.14	kg/ha/yr	NH_4^+	three burned watersheds compared to two control watersheds 200 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 2	0.09 – 0.45	kg/ha/yr	NH_4^+	400%

Year	Author	Fire Status	Fuel Type	Region	Matrix	Time since fire	N conc	Units	Form	Notes
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 3	0.02 - 0.08	kg/ha/yr	NH_4^+	200%
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 1	3.4 - 12.4	kg/ha/yr	NO_3^-	1300%
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 2	5.5 - 14.9	kg/ha/yr	NO_3^-	572 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 3	1.5 - 3.5	kg/ha/yr	NO_3^-	280 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 1	3.9 – 14.3	kg/ha/yr	DON	660 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 2	2.4 – 7.1	kg/ha/yr	DON	116 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 3	1.0 – 2.6	kg/ha/yr	DON	115 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 1	0.5 -1	kg/ha/yr	TPN	290 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 2	1.0 – 3.7	kg/ha/yr	TPN	220 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 3	0.4 – 0.6	kg/ha/yr	TPN	143 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 1	8.2 – 27.1	kg/ha/yr	TN	876%
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 2	16.0 – 25.2	kg/ha/yr	TN	276 %
2008	Bladon et al.	Burned	Mixed conifer	Alberta Rockies	Water	yr 3	2.8 – 5.7	kg/ha/yr	TN	208 %
2016	Mast et al.	Pre fire	Spruce & fir	Colorado Rockies	Water	6 yr pre-fire	0.29 - 0.59	mg-N/ha/yr	NO_3^--N	Report a 33% increase in yield yr 1 and 52 % yr2
2016	Mast et al.	Burned	Spruce & fir	Colorado Rockies	Water	1 yr ad 2 yr	0.64 -1.03	mg-N/ha/yr	NO_3^--N	Year 1 and 2 after fire
2020	Serpa et al.	High Severity	Eucalyptus	north central Portugal	Water	0 - 2 yr	0.4	kg/ha	NO_3^-	Total export during the 2 yr study
2020	Serpa et al.	High Severity	Eucalyptus	north central Portugal	Water	0 - 2 yr	0.1	kg/ha	NO_3^-	-
2020	Serpa et al.	High Severity	Eucalyptus	north central Portugal	Water	0 - 2 yr	10.4	kg/ha	TN	-
2020	Serpa et al.	High Severity	Eucalyptus	north central Portugal	Water	0 - 2 yr	19.8	kg/ha	TN	-

Table 1-6. Summary of reported soil phosphorus concentrations and trends following wildfire

Year	Author	Fire	Fuel Type	Region	Matrix	Time	Report P	P form	Trend
1987	Kutiel and Naveh	Unburned	Pine	Carmel NP, Israel	Soil	< 1yr	600 mg/kg	TP	
		Unburned	Pine	Carmel NP, Israel	Soil	< 1yr	1200 mg/kg	Soluble	
		Burned	Pine	Carmel NP, Israel	Soil	< 1yr	1800 mg/kg	TP	300% increase
		Burned	Pine	Carmel NP, Israel	Soil	< 1yr	400 mg/kg	Soluble	min
		Burned	Pine	Carmel NP, Israel	Soil	< 1yr	1.8 mg/kg	Soluble	max
2009	Rodriguez et al.	Unburned	p. canariensis	Canary Islands, Spain	Soil - Ash	1 mo	5 mg/kg	Ortho-P (as P)	Increase
		Burned - Crown	p. canariensis	Canary Islands, Spain	Soil - Ash	1 mo	15 mg/kg	Ortho-P (as P)	
		Unburned	a. vicous	Canary Islands, Spain	Soil - Ash	1 mo	4 mg/kg	Ortho-P (as P)	Increase
		Burned - Crown	a. vicous	Canary Islands, Spain	Soil - Ash	1 mo	20 mg/kg	Ortho-P (as P)	
2013	Maranon-Jimenez et al.	High	Pine	SE Spain	Standing Dead	6 mo	58 - 105 mg/kg	TP	No controls, four samples after burn
		High	Pine	SE Spain	Soil	2 yr	1.87 - 5.4 mg/kg	TP	
2014	Gomez-Rey & Prieto	Unburned	Pine	NW Spain	Soil (0-2 cm)	0 yr	8 mg/kg	TP	
		Medium-High Severity	Pine	NW Spain	Soil (0-2 cm)	0 yr	14.5 mg/kg	TP	
2015	Santin et al.	Low	Eucalyptus	SE Australia	Ash	3 mo	140 mg/kg	TP	
		Moderate	Eucalyptus	SE Australia	Ash	3 mo	136 mg/ Kg	TP	
		High	Eucalyptus	SE Australia	Ash	3 mo	153 mg/kg	TP	
		Unburned	Eucalyptus	SE Australia	Soil (3 - 8 cm)	3 mo	46.2 mg/kg	TP	
2015	Santin et al.	Low	Eucalyptus	SE Australia	Soil (3 - 8 cm)	3 mo	47 mg/kg	TP	
		Mod	Eucalyptus	SE Australia	Soil (3 - 8 cm)	3 mo	48 mg/kg	TP	
		High	Eucalyptus	SE Australia	Soil (3 - 8 cm)	3 mo	66 mg/kg	TP	

Year	Author	Fire	Fuel Type	Region	Matrix	Time	Report P	P form	Trend
2016	Francos et al.	Burned	Pine, Eucalyptus	NE Spain	Soil	3 d	45.1 mg/kg	TP	
		Burned	Pine, Eucalyptus	NE Spain	Soil	After 1st rain	65.5 mg/kg	TP	
		Burned	Pine, Eucalyptus	NE Spain	Soil	1 yr	19.8 mg/kg	TP	
2018	Santin et al.	Unburned (O-Horizon)	Eucalyptus	SE Australia	Soil (0-2 cm below litter)	1 d	130 mg/kg	TP	
		Unburned (A Horizon)	Eucalyptus	SE Australia	Soil (0-2 cm below litter)	1 d	102 mg/kg	TP	
		Burned (A Horizon)	Eucalyptus	SE Australia	Soil (0-2 cm below litter)	1 d	80 mg/kg	TP	
		Low	Eucalyptus	SE Australia	Ash	1 d	294 mg/kg	TP	
		Moderate	Eucalyptus	SE Australia	Ash	1 d	244 mg/ Kg	TP	
		High	Eucalyptus	SE Australia	Ash	1 d	426 mg/kg	TP	
2020	Lui et al.	Control 0-10 cm	Grass	North-Central China	Soil	1 yr	702 mg/kg	TP	
		Burned 0-10 cm	Grass	North-Central China	soil	1 yr	720 mg/kg	TP	

Table 1-7. Summary of studies reporting surface water phosphorus concentrations in recently burned watersheds

Year	Author	Fire severity	fuel type	Region	Matrix	Time	P Conc.	P form	Trend	Notes
1975	McColl and Grigal	Unburned	spruce, fir	Minnesota, USA	Overland Flow	1 yr ave	0.654 mg/L	TP		Lake change within annual variation. Note that TP was not statistically different for most waters
		Burned	spruce, fir	Minnesota, USA	Overland Flow	1 yr ave	1.881 mg/L	TP	300+ % increase	
		Unburned	spruce, fir	Minnesota, USA	soil Water	1 yr ave	0.106 mg/L	TP		
		Burned	spruce, fir	Minnesota, USA	soil Water	1 yr ave	0.543 mg/L	TP		
		Unburned	spruce, fir	Minnesota, USA	Stream Water	1 yr ave	0.009 mg/L	TP		
		Burned	spruce, fir	Minnesota, USA	Stream Water	1 yr ave	0.022 mg/L	TP		
		Unburned	spruce, fir	Minnesota, USA	Lake Water	1 yr ave	0.009 mg/L	TP		
		Burned	spruce, fir	Minnesota, USA	Lake Water	1 yr ave	0.0012 mg/L	TP		
		Burned	Mixed conifer	NW Montana, USA	Water	1 - 4 yr	0.005 to 0.045 mg/L	SRP	highest during spring runoff, increased in magnitude with time	
		During Fire	Mixed conifer	NW Montana, USA	Water	0 yr	0.135 mg/L	SRP	Up from < 3 ug/L background control	
2003	Earl and Blinn	N/A	Pine, oak	New Mexico, USA	Water	1 mo	0.6 mg/L	SRP	1-2 month after fire, only statistical result	
2008	Lane et al.	High	Eucalyptus	SE Australia	Stream Storm flow	3 mo	0.06 mg/L	TDP (Mean)	3 mo was highest for most, decreased asymptotically over time	Particulate P is major driver of export
			Eucalyptus	SE Australia	Stream Base Flow	3 mo	0.4 mg/L	PP (mean)		
2008	Mast and Clow	Mixed	Spruce fir	Colorado Rockies, USA	Water	< 1 - 4 yr	0.002 - 0.007 mg/L	VWM P	Decrease export from control	
		Unburned	Spruce fir	Colorado Rockies, USA	Water	< 1 - 4 yr	0.002 to 0.019 mg/L	VWM P		

Year	Author	Fire severity	fuel type	Region	Matrix	Time	P Conc.	P form	Trend	Notes
2011	Fernandez et al.	Unburned	Eucalyptus	NW Spain	Stream Water	Pre fire	0.01 mg/L	TP	Average did not change	
		Moderate	Eucalyptus	NW Spain	Stream Water	1st yr ave	0.01 mg/L	TP		
		Moderate	Eucalyptus	NW Spain	Stream Water	2nd yr ave	0.01 mg/L	TP		
		Moderate	Eucalyptus	NW Spain	Stream Water	3rd yr ave	0.01 mg/L	TP		
2013	Coombs and Melack	Burned	Chaparral	S California, USA	Stream Water	< 1 yr	0.3 mg/L	TP	VWM, 2.2 times increase over unburned	
2014	Silins et al.	Severe	Mixed conifer	Alberta, Canada	Stream Water	1 yr ave	0.003 – 0.025 mg/L	SRP	5.8-fold over control	All forms of P elevated for four years of study
		Severe	Mixed conifer	Alberta, Canada	Stream Water	1 yr ave	0.003 – 0.030 mg/L	TDP (Mean)	7.8-fold over control	1-year delay in PP and TP
		Severe	Mixed conifer	Alberta, Canada	Stream Water	1 yr ave	0.003 – 0.1 mg/L	PP (mean)	3.4-fold over control	PP elevated in storm flow
		Severe	Mixed conifer	Alberta, Canada	Stream Water	1 yr ave	0.010 – 0.120 mg/L	TP	3.7-fold over control	
2015	Son et al.	Mixed Severity	Mixed conifer	Colorado, USA	Storm Flow	1 mo	1 to 100 mg/L	TP storm		Fire increased bed sediment P sorption
		Mixed Severity	Mixed conifer	Colorado, USA	Base Flow	1 mo	< 1 mg/L	TP Base		
		Mixed Severity	Mixed conifer	Colorado, USA	Stream water	1 mo	0.2 - 1.6 mg/L	SRP		
		Mixed Severity	Mixed conifer	Colorado, USA	Storm Flow	1 mo	0.2 - 0.7 mg/L	DOP		
		Mixed Severity	Mixed conifer	Colorado, USA	Storm Flow	1 mo	2 - 100 mg/L	PP		
2016	Emelko et al.	Unburned	Mixed conifer	S Alberta, Canada	sediment	7 yr	619.2 mg/kg P	TPP		7 years after fire, water still showed increase

Year	Author	Fire severity	fuel type	Region	Matrix	Time	P Conc.	P form	Trend	Notes
		Burned - Severe	Mixed conifer	S Alberta, Canada	sediment	7 yr	651.9 mg/kg P	TPP		
		Unburned	Mixed conifer	S Alberta, Canada	sediment	7 yr	519.7 mg/kg P	TPP		
		Burned - Severe	Mixed conifer	S Alberta, Canada	sediment	7 yr	667.0 mg/kg P	TPP		

Table 1-8. Export load of phosphorus species following wildfire

Year	Author	Fire Characterization	fuel type	region	Matrix	Time	P Export	Units	Type
1997	Williams and Melack	Burned	Mixed Conifer	CA, Sequoia	Water	1yr	0.4	kg/ha/yr	PP
1997	Williams and Melack	Pre-burn	Mixed Conifer	CA, Sequoia	Water	1 yr	0.1	kg/ha/yr	PP
2000	Thomas et al.	Burned	Pine	N Portugal	-	yr 1	0.22 -0.44	kg/ha	TP
2000	Thomas et al.	Burned	Pine	N Portugal	-	yr 2	ND	kg/ha	TP
2000	Thomas et al.	Burned	Eucalyptus	N Portugal	-	yr 1	0.17-0.83	kg/ha	TP
2000	Thomas et al.	Burned	Eucalyptus	N Portugal	-	yr 2	ND	kg/ha	TP
2008	Lane et al.	Burned	Eucalyptus	Victoria Australia	Water	1 yr	1.67	kg/ha	TP
2008	Mast and Clow	Burned	mixed conifer	Glacier NP, Montana	Water	0 - 4 yr	0.018 - 0.043	kg/ha/yr	TP
2008	Mast and Clow	Unburned	mixed conifer	Glacier NP, Montana	Water	0 - 4 yr	0.019 - 0.237	kg/ha/yr	TP
2010	Noske et al.	Severe	Eucalyptus	Victoria Australia	Susp. Sediments	1 yr	2.14	kg/ha/yr	PP
2010	Noske et al.	Severe	Eucalyptus	Victoria Australia	Water	1 yr	0.12	kg/ha/yr	TDP
2010	Noske et al.	Severe	Eucalyptus	Victoria Australia	Susp. Sediments	1 yr	1.03	kg/ha/yr	PP
2010	Noske et al.	Severe	Eucalyptus	Victoria Australia	Water	1 yr	0.08	kg/ha/yr	TDP
2020	Serpa et al.	Burned P1	Eucalyptus	N Portugal	Water	0 - 2 yr	0.2	kg/ha	PO4
2020	Serpa et al.	Burned P2	Eucalyptus	N Portugal	Water	0 - 2 yr	2.5	kg/ha	PO4
2020	Serpa et al.	Burned P1	Eucalyptus	N Portugal	Water	0 - 2 yr	1.8	kg/ha	TP
2020	Serpa et al.	Burned P2	Eucalyptus	N Portugal	Water	0 - 2 yr	8.7	kg/ha	TP

2. Methods for Evaluating Post-fire Soil Nutrients and Retardant Additions to the 2015 Rocky Fire, Cache Creek, California

To evaluate the effects of aerial retardant on downstream water bodies, post-fire nutrient transport needs to be independently evaluated. This chapter details the development of a spatial modeling strategy to evaluate the loading of total nitrogen (TN) and total phosphorus (TP) in both pre-fire and post-fire conditions. The modeling approach is greatly simplified from the complex geochemical processes occurring in the soil and surface water. The simplified modeling approach assumes that TN and TP transport is directly related to the erosion of soil particles from the landscape. While it is acknowledged that the various forms of N and P can have both dissolved and particulate forms, there is strong empirical evidence that both total N and total P are particulate-dominated, especially during storm events, in the Cache Creek watershed (as discussed in Section 2.6). The PFHydro water quality module implements this simplified approach for the Cache Creek Watershed modeling area because storm events account for most mass transfer in the watershed pre- and post-fire.

The approach estimates the total load of N and total load of P based on preexisting conditions (soil nutrient concentrations) and ash inputs of N and P due to combustion of fuels and the subsequent transfer of biomass ash onto the soil surface. The approach uses the output from a combustion model, FOFEM 6 (First Order Fire Effects Model; Lutes and Keane, 2016), to account for nitrogen losses, though it is not considering particulate associated N loss to the atmosphere. From the antecedent soil concentrations and post-fire inputs, a soil nutrient concentration is generated that can be applied in the PFHydro water quality module. The approach used to generate nutrient concentration on particles are outlined below and discussed in detail in the chapter.

1. Determine watershed and model-space characteristics including, vegetation, carbon loads, pre-fire soil N & P content
2. Estimate post-fire nutrient mass based on: fuel characteristics (vegetation type, fuel loading), fire severity, average stoichiometric ratio in woody fuels (C:N, C:P)
3. Calculate ash N and P spatial loads from estimated ash nutrient content and an estimated post-fire spatial ash load
4. Calculate the soil concentration of total N and total P by adding the ash N and P loads to the pre-fire N and P spatial load and accounting for the depth of mixing and the estimated soil density
5. Apply fuel- and fire-characteristics to the study watershed spatially to estimate the amount of total N and total P transferred to the ground surface during the 2015 Rocky and Jerusalem fires, Cache Creek watershed, CA
6. Use the average pre- and post-fire soil concentrations in the model space and the

calibrated suspended sediment load to estimate pre- and post-fire total N and total P concentrations in Cache Creek.

7. Validate the approach by comparison with stream water data

2.1. Determining Watershed and Model-space Characteristics Including, Vegetation, Carbon Loads, Pre-fire Soil N & P Content

The Cache Creek drainage is an area of about 3,000 km^2 in the Coastal Range of California. The relative elevations in the model project area are 125 m at Rumsey, a frequent sampling location for USGS mercury studies (https://waterdata.usgs.gov/nwis/inventory/?site_no=11451800), and the top of Pine Mountain at 1,250 m at the highest point of the watershed. The modeling domain in the Cache Creek watershed has an average cover distribution of 69% Shrub/Scrub, 6% evergreen forest, 6 % mixed forest, 16 % herbaceous, and 1% developed, with less than 1% deciduous forest, woody wetlands, emergent wetlands, cultivated crops, barren, or open water (**Figure 2-1**). Using these general categories, the local species types for the watershed were identified and applied. The vegetation composition within the watershed boundary is a combination of shrub, conifer and hardwood forest, grassland, and oak- grassland. Conifer species including Douglas fir (*Pseudotsuga menziesii*) and Ponderosa pine (*Pinus ponderosa*) tend to be found at the higher elevations of the watershed, whereas lower elevations support grey pine (*Pinus sabiniana*) and knobcone pine (*Pinus attenuate*). Blue oak (*Quercus douglasii*) is the dominant hardwood in the oak-grassland land-cover type, preferring north-facing slopes; black oak (*Quercus kelloggii***)**, and canyon live oak (*Q*uercus chrysolepis), can be found throughout watershed. Examination of the model domain suggests that most of the landscape is primarily shrub-covered with interspersed hardwood and conifer woodland. All species analysis were performed using aerial imagery, so it was not possible to distinguish exact conifer types.

Vegetation classes for the region were assigned a class in the FOFEM model (First order Fire effects Model; Lutes and Keane, 2016). Using the landcover GIS layer and the five dominant classes identified (shrub/scrub, evergreen, mixed forest, herbaceous, and deciduous), vegetation types present in the area matching these covers were identified for use in the soil heating model. The “shrub/scrub” was modeled as scrub oak and mixed shrub (SRM207), “mixed forest” was modeled as mixed Blue Oak and Grey Pine (SRM 250), “deciduous” was modeled as Blue Oak (SRM 201), “herbaceous” was modeled as annual grassland (SRM215) and “evergreen” was modeled as Knobcone Pine (SAF 248).

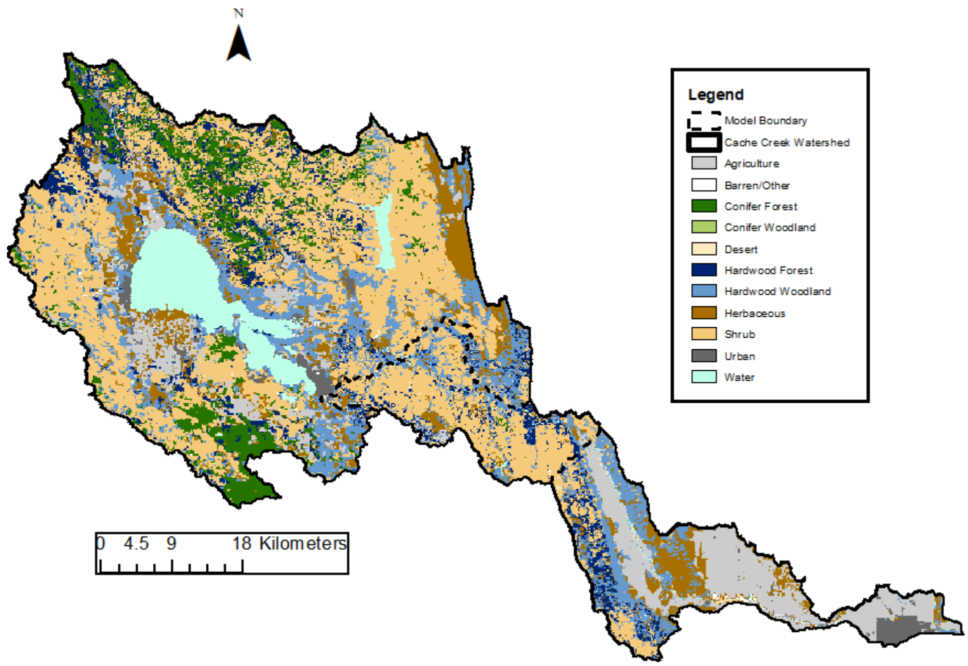

Figure 2-1. Vegetation types used in modeling

Nutrient concentrations for pre-fire conditions were determined using available data from USGS reports on the soil geochemistry in the area (Morrison et al., 2008; Wise et al. 2019). The locations of soil samples collected by Morrison et al. (2008) are not directly within the model space but were collected adjacent to the boundary. They are still considered to be the best representative of the lower watershed, as no other data were available.

Total P in regional surface soil (< 2 cm depth) was reported to be between 100 and 1300 ppm, with an average of 550 ± 200 ppm and a median concentration of 600 ppm (Morrison et al., 2008). The reported concentrations by Wise et al. (2019), indicate that there is a strong total P gradient in the watershed, with lower elevations having concentrations as low as 300 ppm and upper elevations having concentrations greater than 750 ppm. Because of the strong gradient, it cannot be assumed that particulate concentrations of total P in the sediment load are the same for the model boundary conditions and the model space. It is likely the upper boundary sediment total P is higher than the particulate concentrations that would be generated from erosion in the model domain. Based on the ranges reported, and the likely gradient, reasonable values to use in the model fall within the 500 to 750 ppm range. For the purposes of modeling, the median concentration of 600 ppm is used as a background level for modeled erosion, and a slightly higher concentration of 650 ppm is used for the upstream boundary conditions based on the TP gradient in the watershed.

Regional TP concentrations were obtained from Morrison et al. (2008) and were reported from 100 to 8800 ppm in the surface soil (<2 cm) adjacent to the model domain (lower Cache Creek watershed), though two sites had very large (34,000 and 53,000 ppm) concentrations that were excluded from the analysis as it appears that these high values are not representative of general soil N concentrations in the watershed and may have been agricultural areas. Average soil concentrations were 1800 ppm with a standard deviation of 1000. The median of the data was 1400 ppm, which seems to be an appropriate estimate based on the spread of values. The average soil N concentrations are used as an estimate for both the model domain and greater Cache Creek watershed. Like soil phosphorus, the soil nitrogen is estimated from sites adjacent to the model domain, thus it is possible that the data are not an exact representation of the model space and can only provide an order of magnitude estimate. To our knowledge, Morrison et al. (2008) represent the best available data.

2.2. Estimating Post-fire Soil Particle Nutrient Concentrations based on: Fuel Characteristics (Vegetation Type, Loading), Fire Severity, and Average Stoichiometric Ratios (C:N, C:P)

A general scheme for developing the post fire soil nutrient concentration is outlined in **Figure 2.2**.

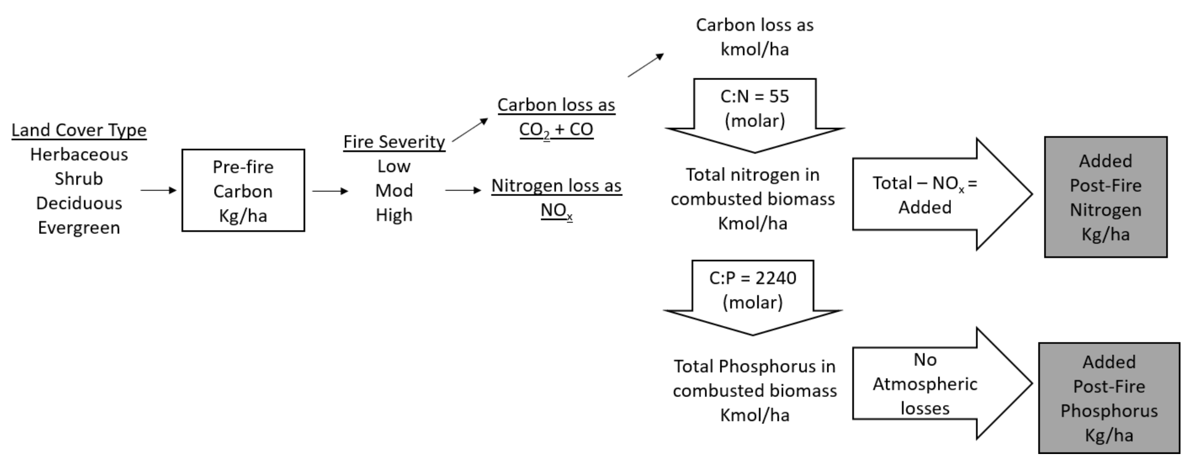

Figure 2-2. Modelling scheme for determining total N and total P in post-fire soils.

The land cover types described in section 2.1 were used to develop a carbon load on the landscape by vegetation type.

2.2.1. Determine Burn Characteristics

Burn severity was estimated using the Monitoring Trends in Burn Severity (MTBS) burn severity mapping GIS Layer and combining this with the landcover type. To increase the accuracy of the estimate, the watershed was divided into hydrologic units. For each unit within the watershed, a percentage of land cover was determined. The overall average cover in the Cache Creek watershed is 69% Shrub, 6% Evergreen, 6% Mixed Forest, and 16% Herbaceous. Many of the hydraulic units can vary from this average, and not all watershed units were burned, so the characteristics of individual burned watershed units were used. For each unit, the land cover type percentage was obtained, and the percent of low, medium, high, and severe burning was determined (**Table 2-1**). MTBS burn severity is determined using the dNBR or delta normalized burn ratio, which is the degree of green light reflectance before and after a fire. The degree of difference in green reflection is an indication of the loss of vegetation. For the purposes of estimating biomass combustion, this is a reasonable application of the dNBR data.

In this model, four severity categories (unburned, low, moderate, and high) are used to estimate the degree of combustion completeness; the "high" and "extreme" severity classes identified by MTBS layers are combined into one category. The model does not use a specific "severity" metric, but the vegetation moisture level can be used as a proxy to adjust the degree of combustion, which is essentially the definition of severity.

Carbon losses due to fuel combustion were estimated using the FOFEM (Lutes et al., 2012), which uses vegetation type and moisture inputs to estimate the degree of combustion during wildfire. The output of the model can be used to estimate carbon loss by cover class and fire severity. Each specific forest type has an assigned average fuel load that is used to calculate the carbon load (ton/ha) within the model. FOFEM estimates the pre-fire and post-fire carbon load on the landscape, the difference between the pre and post fire load is accounted for as the carbon oxidized to CO and CO_2.

2.2.2. Obtain the Amount of Nitrogen and Phosphorus Associated with the Combusted Fuels

To estimate the total nitrogen within the combusted fuel, the net carbon loss estimated above is converted to kmol ha^{-1} and is multiplied by a molar stoichiometric ratio of 55 (C:N) for woody materials. This molar ratio is an average of the reported range of 43.6 to 66.2 C:N (McGroddy et al., 2004).

Some nitrogen is lost to the atmosphere as NO_x and this is estimated within the FOFEM model. The NO_x emissions are given in terms of kg ha^{-1} and are converted to kmol per hectare. Based on the FOFEM output, the amount of nitrogen loss to the atmosphere is small compared to the total nitrogen in the combusted biomass (1-7%), and therefore most N is assumed to be transferred to the soil. Previous studies have shown nitrogen is converted to ammonium during heating and thus would not be in the gaseous form, supporting the assumption N is deposited (Santín, et al., 2015).

Table 2-1. Summary of land cover, fire severity, severity-based nutrient inputs (low, moderate, and high severity), and average total nutrient input by watershed unit.

Unit	Shrub/ Scrub	Conifer	Mixed forest	Herb	Low	Med	High	Low N kg ha^{-1}	Mod N kg ha^{-1}	High N kg ha^{-1}	Low P kg ha^{-1}	Mod P kg ha^{-1}	High P kg ha^{-1}	Ave Unit ΔN kg ha^{-1}	Ave Unit ΔP kg ha^{-1}
1	85%	2%	4%	8%	0%	0%	0%	320	319	303	18	18	18		
2	37%	0%	1%	60%	0%	0%	0%	150	150	132	8	8	8		
3	54%	0%	2%	39%	0%	0%	0%	208	208	192	12	12	12		
4	53%	4%	4%	31%	0%	0%	0%	220	218	191	12	12	12		
5	52%	1%	3%	40%	0%	0%	0%	202	201	184	11	11	11		
6	80%	2%	5%	8%	0%	0%	0%	300	299	283	17	17	17		
7	64%	0%	2%	29%	9%	5%	6%	241	240	227	14	14	14	49	3
8	72%	1%	5%	21%	24%	30%	12%	272	271	255	15	15	15	179	10
9	72%	0%	3%	24%	27%	31%	21%	269	268	255	15	15	15	208	12
10	54%	29%	7%	6%	0%	0%	0%	314	303	199	17	17	17		
11	54%	14%	10%	18%	0%	0%	0%	269	263	195	15	15	15		
12	77%	14%	4%	4%	0%	0%	0%	334	328	276	19	18	19		
13	81%	4%	5%	8%	1%	0%	0%	313	311	289	18	17	18	2	0
14	82%	1%	4%	10%	31%	9%	14%	307	306	291	17	17	17	164	9
15	67%	1%	3%	5%	33%	10%	23%	247	246	237	14	14	14	164	9
16	84%	5%	9%	2%	14%	12%	64%	336	333	299	19	19	19	278	17
17	86%	1%	5%	5%	20%	20%	47%	321	320	306	18	18	18	270	16
18	68%	4%	9%	18%	23%	22%	55%	277	274	242	16	15	16	256	15
19	86%	3%	7%	4%	14%	14%	72%	332	330	305	19	19	19	313	19
20	67%	17%	13%	2%	23%	12%	65%	327	319	242	18	18	18	270	18
21	59%	17%	15%	8%	9%	22%	69%	307	298	215	17	17	17	242	17
22	72%	9%	8%	9%	6%	11%	53%	308	303	259	17	17	17	188	12
23	67%	13%	7%	12%	19%	21%	46%	302	297	242	17	17	17	231	14
24	76%	7%	10%	5%	1%	0%	0%	321	317	273	18	18	18	4	0
25	67%	5%	7%	16%	1%	1%	0%	273	271	239	15	15	15	7	0

NOx losses are subtracted from the total N mass to obtain a total nitrogen delivered to the landscape. This total N calculation assumes all nitrogen in the fuel that is not accounted for in the NOx emissions is deposited locally. This assumption is probably not entirely accurate and particulate losses in smoke are likely, but there is no way to estimate this fraction. The total amount of nitrogen associated with the fire ash (kmol ha^{-1}) is then converted back to kg ha^{-1} and added to the pre-fire total N spatial load (**Table 2-1, Table 2-2**).

Total P inputs to soils during fire are also estimated by assuming a stoichiometric ratio C:P and the total carbon loss from burning. The stoichiometric ratio is 2239 C:P is used, based on the average of a reported range of 1334 to 3144 C:P in woody biomass (McRoddy, 2004). In the case of total P, no atmospheric losses are considered: it is assumed that all P associated with burned biomass is transferred to the soil surface in ash (**Table 2-1, Table 2-2**).

Table 2-2. Predicted input of total N and total P as a function of vegetation and fire severity. Species represent the cover types identified within the model boundary

	Severity	SRM 915 - Mixed Herb - Herbaceous	SRM 207 - Scrub Oak Mixed Chaparral	SAF 250 - Blue Oak - Grey Pine	SAF 248 - Knobcone Pine
Total N ash input (kg ha^{-1})	Low	28	354	169	273
Total N ash input (kg ha^{-1})	Medium	28	354	201	330
Total N ash input (kg ha^{-1})	High	28	354	220	362
Total P ash input (kg ha^{-1})	Low	2	20	9	15
Total P ash input (kg ha^{-1})	Medium	2	20	11	18
Total P ash input (kg ha^{-1})	High	2	20	12	20

2.3. Applying Fuel and Fire Characteristics to the 2015 Rocky Fire

For each hydrologic unit in the Cache Creek watershed, a landcover percent was determined and the predicted total N and total P transferred to the soil under each of the three severity scenarios was calculated (e.g. **Table 2-3**). The total predicted nutrient input to the soil for each hydrologic unit was estimated by multiplying the predicted input for a severity and cover class by the percent of area in the hydrologic unit, thereby weighting the input by cover type and area contribution. This calculation is done for each cover class and severity class in each hydrologic unit. The maximum input for a cover class and severity class is then multiplied by the percent of hydrologic unit area burned at that severity. The sum of the inputs is the average hydrologic unit element (N or P) input.

Table 2-3. Example calculation for the maximum input with high severity burning in a mixed-cover watershed. Similar calculations are repeated for low and moderate severity

	Land cover percent	Total N - high severity (kg ha^{-1})	ΔN input (kg ha^{-1})
Herbaceous	24%	354	85
Shrub/Scrub	72%	362	260
Mixed forest	3%	220	6
Evergreen forest	0%	28	0
Average unit input at 100% high burn	-	-	**sum=351**
Average unit input at 21% high burn	-	-	**0.21 × sum =74**

2.4. Calculating the Concentration of Total N and Total P in Post-fire Soil by Adding the Ash N and P Loads to the Pre-fire N and P Spatial Loads and Accounting for the Depth of Mixing and Soil Density

To assign an average increase in soil concentration, the input N or input P is converted from kg ha^{-1} to g kg^{-1} by assuming a specific weight of the soil and a depth of mixing of ash into the surface soil (<2 cm). The mechanism for mixing is not specifically defined here, though it seems that ash entrainment in soil is likely, and that rainfall is a likely mechanism for ash to integrate into the surface soil. Soil specific weight is variable and could be less dense than soil grain density due to ash and charcoal inputs (ash and charcoal can have densities < 1 kg L^{-1}, Santín et al., 2015) An average soil density is very likely to be in the 1.5-2 kg L^{-1}. The higher value of 2 kg L^{-1} is assumed to convert the aerial mass load into a soil concentration.

The depth of soil that the ash is mixing in is also important. For example, if the nutrients are only mixing into the first 0.5 cm, then the concentrations are higher than if a mixing depth of 2 cm is assumed. The assumption of mixing depth can greatly alter the concentration as a function of dilution with background soil nutrient concentrations. It is assumed that mixing does not occur beyond 2 cm. Using the conservative background soil concentration of 1.0 g kg^{-1} total N, and background soil P concentration of 0.6 g kg^{-1}, the summation of the background and post-fire nutrient input becomes the new soil concentration. A summary of the Cache Creek nutrient loading following the 2015 fire is presented in **Table 2-4**.

2.5. Evaluation of Approach

The concentration of total in-stream nutrient concentrations were evaluated in relation to the sediment concentrations. Secondly, the concentrations of particle-associated nutrients (g N L^{-1} water; g P L^{-1} water) were normalized by the suspended sediment concentration (g sediment L^{-1} water) to obtain the nutrient concentration on the sediment particles (g N g^{-1} sediment; g P g^{-1} sediment). Estimated nutrient concentrations associated with suspended sediments in Cache Creek were then compared to soil estimates.

Data from samples collected over the 2019 water year at Rumsey show strong positive correlations between sediment and total N and total P (**Figure 2-3**). The relationship is strongest between unfiltered phosphate and SS (R^2= 0.96) supporting the use of SS as a proxy for TP. The strong correlation between phosphate and sediment can be attributed to strong surface interactions that results in most P associated with particulates. Similarly, the total N and SS relationship is strong (R^2= 0.81) and can be slightly improved by considering only the particulate fraction of N (R^2 = 0.84; not plotted in Figure 2-3).

Table 2-4. Summary of post-fire soil concentrations in each unit based of assumed mixing depths of 0.5 and 2 cm

HUC	Ave HUC ΔN (kg ha^{-1})	Ave HUC ΔP (kg ha^{-1})	Ave HUC ΔN (g m^{-2})	Ave HUC ΔP (g m^{-2})	**Soil Density (Specific weight)**	ΔN 0.5 cm depth (g kg^{-1})	ΔN 2 cm depth (g kg^{-1})	ΔP 0.5 cm depth (g kg^{-1})	ΔP 2 cm depth (g kg^{-1})	Assumed BG Total N (g kg^{-1})	Assumed BG Total P (g kg^{-1})	Post Fire Total N 0-0.5 cm	Post Fire Total N 0-2 cm	Post Fire Total P 0-0.5 cm	Post Fire Total P 0-2 cm
1					**2**					1.4	0.6	1.4	1.4	1.4	1.4
2					**2**					1.4	0.6	1.4	1.4	1.4	1.4
3					**2**					1.4	0.6	1.4	1.4	1.4	1.4
4					**2**					1.4	0.6	1.4	1.4	1.4	1.4
5					**2**					1.4	0.6	1.4	1.4	1.4	1.4
6					**2**					1.4	0.6	1.4	1.4	1.4	1.4
7	49	3	5	0	**2**	0.5	0.1	0.0	**0.0**	1.4	0.6	1.9	1.6	1.6	1.5
8	179	10	18	1	**2**	1.8	0.4	0.1	**0.0**	1.4	0.6	3.2	2.3	2.0	1.8
9	208	12	21	1	**2**	2.1	0.5	0.1	**0.0**	1.4	0.6	3.5	2.4	2.1	1.9
10					**2**					1.4	0.6	1.4	1.4	1.4	1.4
11					**2**					1.4	0.6	1.4	1.4	1.4	1.4
12					**2**					1.4	0.6	1.4	1.4	1.4	1.4
13	2	0	0	0	**2**	0.0	0.0	0.0	**0.0**	1.4	0.6	1.4	1.4	1.4	1.4
14	164	9	16	1	**2**	1.6	0.4	0.1	**0.0**	1.4	0.6	3.0	2.2	1.9	1.8
15	164	9	16	1	**2**	1.6	0.4	0.1	**0.0**	1.4	0.6	3.0	2.2	1.9	1.8
16	278	17	28	2	**2**	2.8	0.7	0.2	**0.0**	1.4	0.6	4.2	2.8	2.3	2.1
17	270	16	27	2	**2**	2.7	0.7	0.2	**0.0**	1.4	0.6	4.1	2.8	2.3	2.1
18	256	15	26	2	**2**	2.6	0.6	0.2	**0.0**	1.4	0.6	4.0	2.7	2.3	2.0
19	313	19	31	2	**2**	3.1	0.8	0.2	**0.0**	1.4	0.6	4.5	3.0	2.4	2.2
20	270	18	27	2	**2**	2.7	0.7	0.2	**0.0**	1.4	0.6	4.1	2.8	2.3	2.1
21	242	17	24	2	**2**	2.4	0.6	0.2	**0.0**	1.4	0.6	3.8	2.6	2.2	2.0
22	188	12	19	1	**2**	1.9	0.5	0.1	**0.0**	1.4	0.6	3.3	2.3	2.0	1.9
23	231	14	23	1	**2**	2.3	0.6	0.1	**0.0**	1.4	0.6	3.7	2.6	2.2	2.0
24	4	0	0	0	**2**	0.0	0.0	0.0	**0.0**	1.4	0.6	1.4	1.4	1.4	1.4
25	7	0	1	0	**2**	0.1	0.0	0.0	**0.0**	1.4	0.6	1.5	1.4	1.4	1.4
											Model ave.	2.5	1.7	0.7	0.6

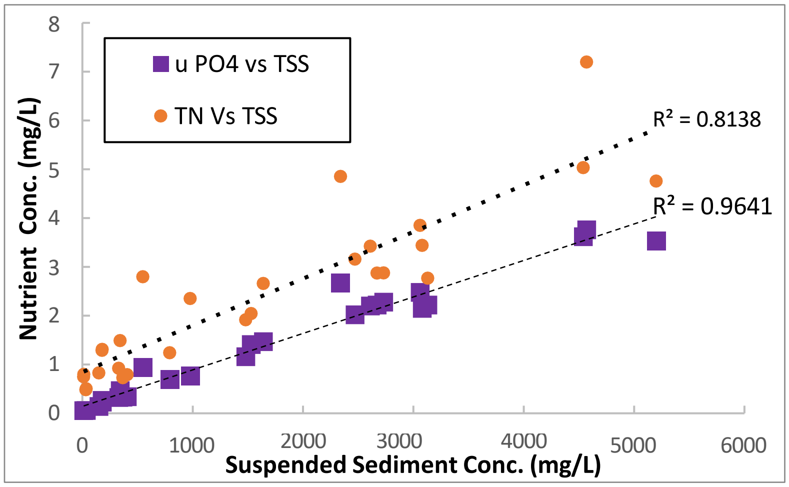

Figure 2-3. Correlations between total nitrogen, Total phosphate and suspended sediment measured at the Rumsey gage on Cache Creek during the 2019 water year (https://waterdata.usgs.gov/nwis/inventory/?site_no=11451800)

Sediment particulate nutrient concentrations can be derived from in-stream nutrient- and suspended sediment-concentrations for comparisons with bulk soil concentrations. In the case of phosphate, particulate P concentrations (calculated from the difference in P concentrations in unfiltered and filtered samples), in mg-P L^{-1}, is divided by the SS concentration (mg L^{-1}) and multiplied by 1×10^6 mg kg^{-1} to obtain an estimate of the sediment particulate concentration (mg-P kg^{-1}-sediment, or ppm). All P present in-stream is expected to originate within watershed soils, and so the previous calculation was repeated using total (i.e. unfiltered) P concentrations. The average sediment particulate P concentration was estimated to be 1100 ppm (+/- 700 ppm) and the average total P concentration (normalized by the SS concentration) was 1400 ppm. The median particle P concentration was 830 ppm and the median total P (normalized by SS) was 1100 ppm.

The median particulate P concentration was 830 ppm, higher than the median soil P concentration adopted for the study area (600 – 650 ppm). Soil P concentrations increase with elevation, and the upper watershed above the study area is expected to have soil P concentrations >750 ppm. A reasonable assumption is that some portion of the suspended sediments in Cache Creek originate in the upper reaches of the watershed, although without actual soil concentration data in the region no proportions can be estimated. Regardless, estimated in-stream particulate P concentrations based on empirical evidence can be reasonably compared to and explained by combining lower watershed and upper watershed median soil P concentrations of 580 ppm and >750 ppm, respectively.

Another possibility is a grain size effect that concentrates the P on particle. The sediment grains in the water column may be smaller diameter and therefore have a larger surface area to volume ratio, increasing the fraction of adsorbed phosphate. In the case of a grain size effect, grains would have a higher concentration compared to the bulk soil of the watershed.

From the measured stream water data collected at Rumsey in 2019, the average total N normalized by the suspended sediment concentration, calculated in an analogous fashion to sediment-normalized total P concentration is 7400 mg-N kg-sediment^{-1}, with a median value is 1800 mg-N kg- sediment^{-1}.

Estimated soil nitrogen concentrations for the region have an average of 1800 mg-N $kg\text{-}soil^{-1}$, median of 1400 mg-N $kg\text{-}soil^{-1}$. The concentrations of total N in soils are much more variable than the in-stream measurements for the watershed. Specifically, high soil-N concentrations may bias the overall mean soil N concentration. Regardless, the median soil-N concentration is close to the median particle N concentration, suggesting that using soil-N median concentrations for the watershed might be the most suitable for estimating the total N load.

2.6. Model Limitations

The determination of a single N or P concentration to assign to sediment fluxes to capture total nutrient export greatly simplifies the modeling approach. The simplification comes at the cost of providing modeled values that do not fully represent the complex cycling and various forms of each nutrient within a watershed. Notably, to account for a total nutrient export of N and P, both dissolved and particulate forms must be accounted for. To include both forms require crude assumptions:

> **Assumption 1** - Total N and total P in soil accounts for all forms of each nutrient. In both pre-and post-fire conditions, the exact form of nutrient does not matter. All nutrient forms are equally likely to mobilize.
> **Assumption 2** - Each of the forms of nutrient (particulate, dissolved, ions, organic, etc.) are released from the landscape and advected through the watershed simultaneously
> **Assumption 3** - Total nutrient in the surface water includes both dissolved and particulate forms and both are directly correlated to the sediment load.

The assumptions listed are not realistic. For example, N and P concentrations are likely to vary across the watersheds, specific areas have greater hydraulic connectivity and have a great contributions, and many forms of N and P are unlikely to mobilize readily or to mobilize simultaneously. However, when averaged over an annual load or large storm event, TN and TP are highly correlated and sediment is a reasonable proxy.

2.7. Use of Fire Retardant in the Cache Creek Watershed

Retardant used in 2015 is only referenced in one article, which states that seven flights dropped a total of 20,476 gallons of fire retardant on the Rocky Fire (Gabbert, 2015). It is possible that more was applied later in the fire, though this is unknown. Each aerial retardant drop of MAP (from a Type 1 aircraft) deposits around 380 kg of TP and 175 kg of TN (DAP would double this N amount)(USDA, n.d.). Aerial firefighting during the Rocky Fire is therefore expected to have increased the TP loading in the watershed by around 2,700 kg and the TN loading by 1,200 kg.

To evaluate a higher application, it might be assumed that a similar number of drops occurred for the next two days of the fire (this is the point forward progress was stopped). If this was the case, the expected number of drops would be around 20, corresponding to an additional watershed load of around 7700 kg TP and around 3500 kg TN. Using the 20-drop estimate, over the total watershed area (300,000 ha), the spatial input is 0.026 kg-P ha^{-1} and 0.012 kg-N ha^{-1}. Using the 20-drop estimate, over the modeling area (28,200 ha), the spatial input is 0.27 kg-P ha^{-1} and 0.12 kg-N ha^{-1}. In comparison, the average vegetation inputs in the burned area of the Rocky Fire were 191 kg-N ha^{-1} and 10.7 kg-P ha^{-1}. The increase in the spatial load of nutrients caused by the 20-drop estimate, retardant was calculated to be around 2.5% for TP and 0.06 % for TN (percent of increase based on vegetation input) within the burn area.

To evaluate the effects at the practical limits of application, the estimated drop number was increased to create a "maximum loading" scenario. In this case, 30 drops of retardant are assumed. The application of 30 drops would increase the nutrient mass by 5200 kg N and 11400 kg P. Over the modeling area, the increase in spatial load would be 0.18 kg-N ha^{-1} and 0.4 kg-P ha^{-1}. Beyond the practical application, an unrealistic scenario of 100 drops was considered to see what the effect would be. Using a 100-drop scenario the increase model space nutrient mass is17272 kg of N and 38200 kg P, which corresponds to an increased spatial load of 1.4 kg-N and 0.61 kg-P ha^{-1}.

A summary table of the average effects of burning and combined with retardant loading scenarios is presented in **Table 2-5**.

Table 2-5. Summary of applied retardant within 28,200 ha the modeling boundary. The effects of assumed mixing depth on average soil nutrient concentrations are included. Note the decimal places do not signify the precision of estimate (spurious digits) but were included to give a sense of the calculated effects of fire retardant and show the calculated effects are small. The digits outside of our two significant digit results are shown in subtext. The significant digits would be rounded when used.

	Post Fire TN (g kg^{-1})	Post Fire TN (g kg^{-1})	Post Fire TN (g kg^{-1})	Post Fire TN (g kg^{-1})	Post Fire TP (g kg^{-1})	Post Fire TP (g kg^{-1})	Post Fire TP (g kg^{-1})	Post Fire TP (g kg^{-1})
Pre-fire	1.4	1.4	1.4	1.4	0.6	0.6	0.6	0.6
Mixing depth (cm)	**0-0.5 cm**	**0-1 cm**	**0-1.5 cm**	**0-2 cm**	**0-0.5 cm**	**0-1 cm**	**0-1.5 cm**	**0-2 cm**
No Retardant	3.1_{67}	2.2_{83}	1.9_{89}	1.8_{42}	0.70_{88}	0.65_{39}	0.63_{63}	0.62_{70}
20 drops	3.1_{687}	2.2_{83}	1.9_{89}	1.8_{42}	0.71_{06}	0.65_{53}	0.63_{69}	0.62_{80}
30 Drops	3.1_{68}	2.2_{84}	1.9_{89}	1.8_{42}	0.71_{19}	0.65_{59}	0.63_{73}	0.62_{79}
100 drops	3.1_{73}	2.2_{86}	1.9_{91}	1.8_{43}	0.72_{14}	0.66_{07}	0.64_{05}	0.63_{03}

2.8. References

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3. Modeling Impacts of Wildfire and Fire Retardant on Total Nitrogen and Total Phosphorus Loads

3.1. Introduction and Previous Research

One of the major challenges in developing watershed hydrologic and water-quality models applicable to wildfire impacts is the difficulty in obtaining appropriate time-series data given the remoteness of the terrain being analyzed and the high cost of obtaining a comprehensive data set. This research faced the challenges of lack of total nitrogen (TN) load and total phosphorus (TP) load data in the modeling watershed outlet to verify the accuracy of the watershed modeling.

To evaluate the potential impact of ammonium-phosphate (AP)-based fire retardants on nutrient loads in a watershed affected by wildfire, we needed to know TN and TP loads both pre-fire and post-fire with and without fire retardants. The modeling area chosen for this analysis is the Upper Cache Creek watershed, CA, where modeling was done for a previous Reclamation S&T research project (Project ID:7112, Wang et al., 2019). The previous research developed the PFHydro (post- fire hydrological) model to simulate post-fire runoff and sediment load in the Cache Creek watershed. PFHydro was calibrated well for flow and sediment for one year pre-fire (WY 2015) and two years post-fire (WYs 2016 and 2017) based on monitoring data collected by the U.S. Geological Survey (USGS) for Cache Creek at Rumsey (USGS station 11451800) which is the outlet of the watershed where PFHydro was applied.

Although no water-quality data for nutrients are available for WYs 2015–17 at the Rumsey station, data collected by USGS at Rumsey and other monitoring stations in the Cache Creek watershed during WY 2011 and WY 2019 are used in this study to develop calibrated time-series estimates of daily pre-fire and post-fire loads of TN and TP for Cache Creek at Rumsey during WYs 2015–17. These calibrated daily nutrient loads at the Cache Creek watershed outlet at Rumsey are used to verify and calibrate the daily TN and TP loads simulated by the PFHydro model which, by incorporating estimates of soil N and P concentrations in the upper Cache Creek watershed (Chapter 2), provides a quantitative context for evaluation of possible impacts of AP fire retardants.

3.1.1. Modeling Application Area

The Cache Creek watershed covers portions of Lake County, Yolo County, and Colusa County in northern California (Figure 3-1). It has an area of 3,017 square kilometers (km^2), with elevations ranging from approximately 10 to 1800 meters above sea level and a total population of about 58,000 (Sacramento River Watershed Program, 2022). This region does not experience significant amounts of snowfall.

During July–August 2015, the Jerusalem and Rocky fires burned a combined 384 km^2 and about 214 km^2 within the Upper Cache Creek watershed downstream of Clear Lake (Figure 3-2a). The fires varied in intensity across the watershed, resulting in spatially variable changes of soil properties and reductions in vegetation cover. A sub-area of the upper Cache Creek watershed upstream of Rumsey was chosen for the model (PFHydro) application area, comprising 282 km^2 in which 163 km^2 were burned in 2015.

During summer 2018, some additional large fires affected the upper Cache Creek watershed. The Mendocino Complex Fire, which at the time was the largest fire in California history (now ranked third), burned 459,123 acres in Mendocino, Lake, Colusa and Glenn Counties between July 27 and September 18, 2018. The Mendocino Complex Fire had two sections, the River Fire to the west of Clear Lake and the Ranch Fire to the north and east of Clear Lake (Figure 3-2b). Prior to the Mendocino Complex Fire, the Pawnee Fire (Figure 3-2b) burned 15,185 acres in Lake County to the east of Clear Lake and adjacent to Indian Valley Reservoir during June 23 to July 8, 2018. The 2018 Mendocino Complex and Pawnee fires occurred upstream of the PFHydro modeling watershed which Rumsey is the watershed outlet (Figures 3-1 and 3-2b). Although much of the area burned in summer 2018 was upstream of Clear Lake and Indian Valley Reservoir, some of the burned area drained unregulated into lower North Fork Cache Creek below Indian Valley Reservoir (USGS gaging station 11451500, Figure 3-1), a tributary to Cache Creek. Therefore, conditions during WY 2019 for Cache Creek at Rumsey are considered to be post-fire with respect to the summer 2018 fires, similar in some ways to how WY 2017 is considered post-fire to the summer 2015 fires. The SS, TN, and TP loads during WY 2019 at Rumsey were used to estimate the TN, TP loads during WY 2017 at Rumsey, which is discussed in more detail in the following sections

3.1.1. Previous Modeling Results in Application Area

The pre-fire (WY 2015), and post-fire (WY 2017) daily time-series total phosphorus (TP) and total nitrogen (TN) load simulations for this study are based on the WY 2015 and WY 2017 runoff and suspended sediment (SS) load simulations of previous S&T research (Wang et al., 2019, Wang et al., 2020). Figures 3-3 through 3-6 are the results of the previous runoff and sediment load simulations. The pre-and post-fire runoff and sediment load simulation results matched the observations well, with values of the Nash-Sutcliffe Efficiency Index (NSE; McCuen et al., 2006) ranging from 0.79 to 0.93.

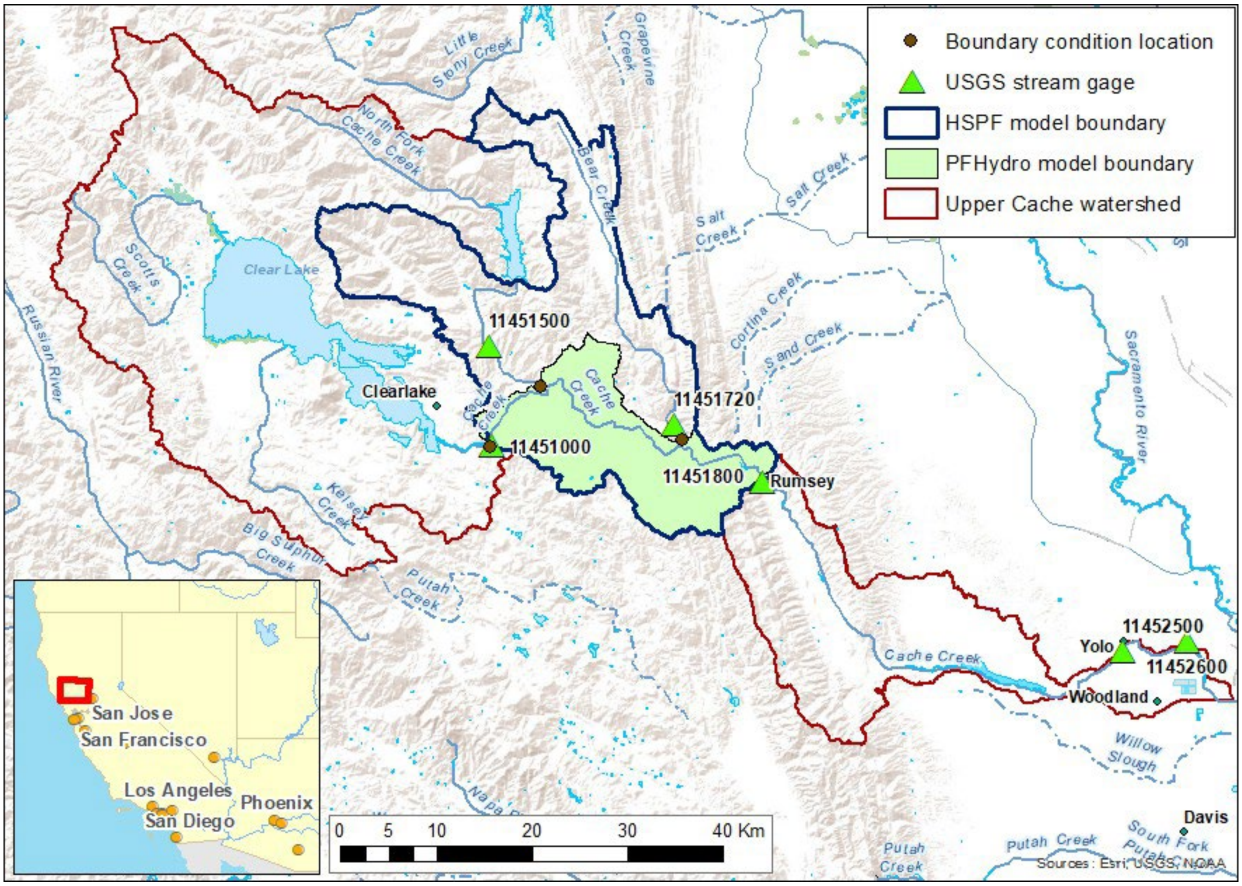

Figure 3-1. Location of upper Cache Creek watershed in northern California showing boundaries for the HSPF model and PFHydro model (modified from Wang et al., 2019). The map also shows flow monitoring stations located along Cache Creek with outflow from the study watershed at the Rumsey gage (green triangles), along with locations for hourly time-series flow from the HSPF model used as boundary conditions for the PFHydro model runoff and sediment load simulation. The Yolo gage (USGS ta for total nitrogen (TN) and total phosphorus (TP).

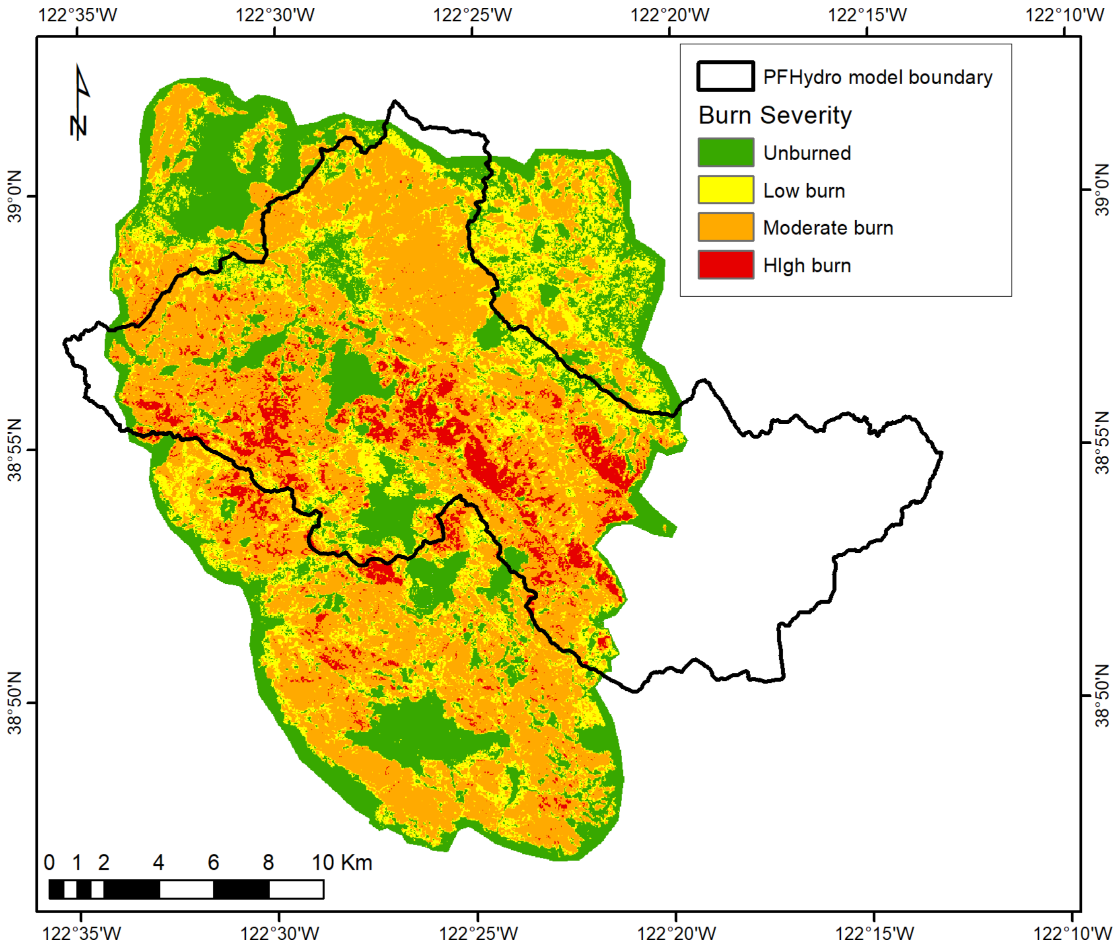

Figure 3-2a. Burn severity classifications for combined Rocky and Jerusalem Fires. The model application area boundary is shown for reference on the map in the area outlined in black. From Wang et al. (2019).

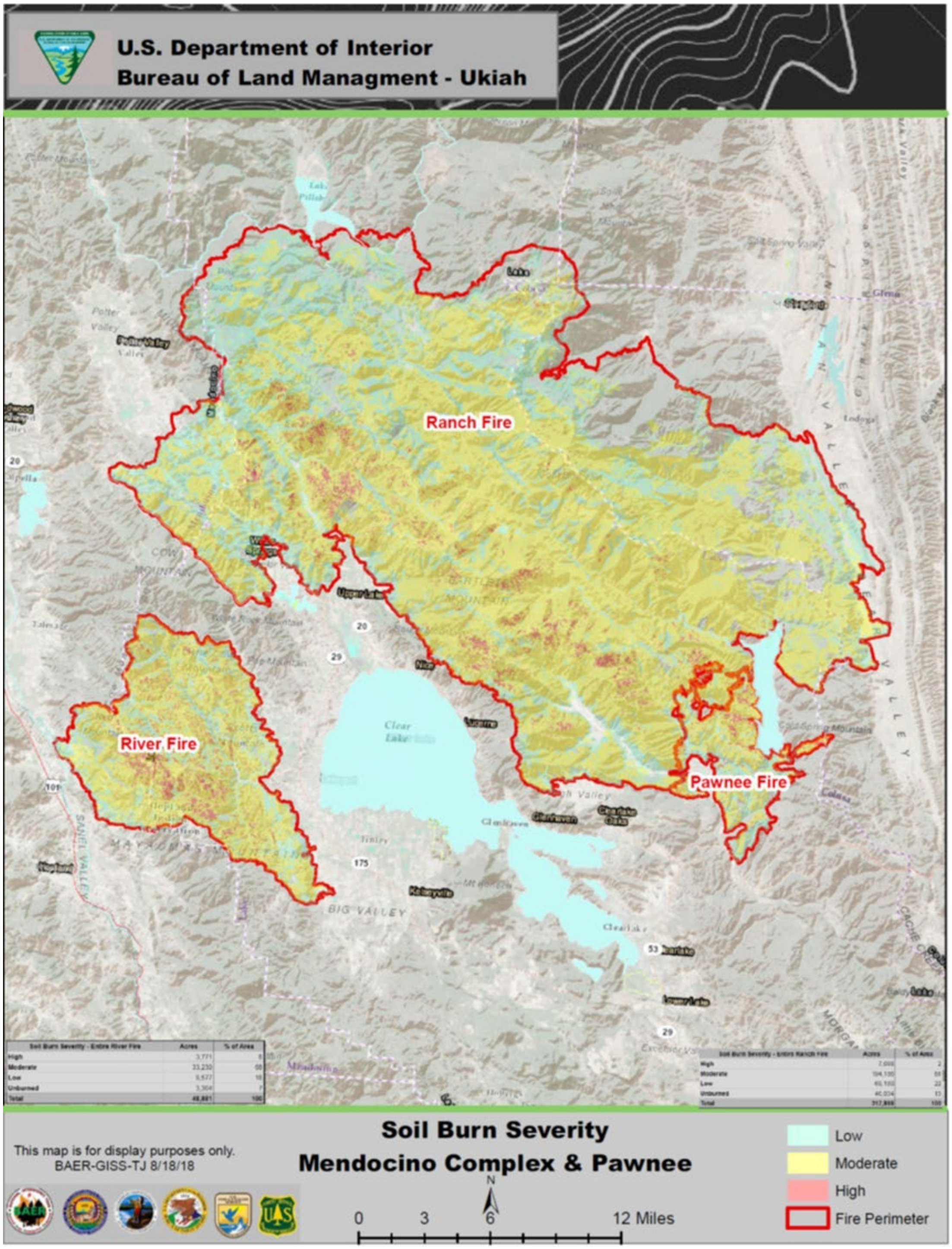

Figure 3-2b. Burn severity classifications for fires affecting the upper Cache Creek watershed during summer 2018. The River Fire and Ranch Fire are considered parts of the Mendocino Complex Fire; the Pawnee Fire is also shown. From BAER (2018).

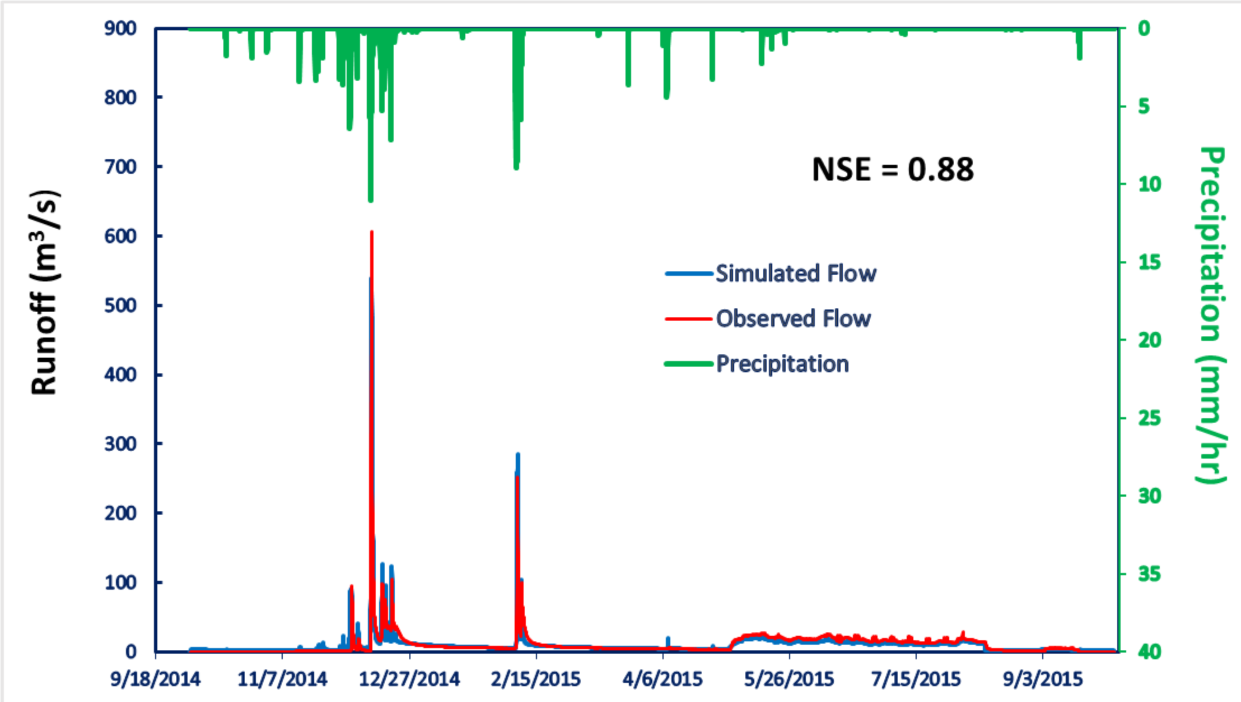

Figure 3-3. Hydrograph showing comparison of observed and simulated pre-fire runoff in the upper Cache Creek watershed (Rumsey outlet) for WY 2015. From Wang et al. (2019).

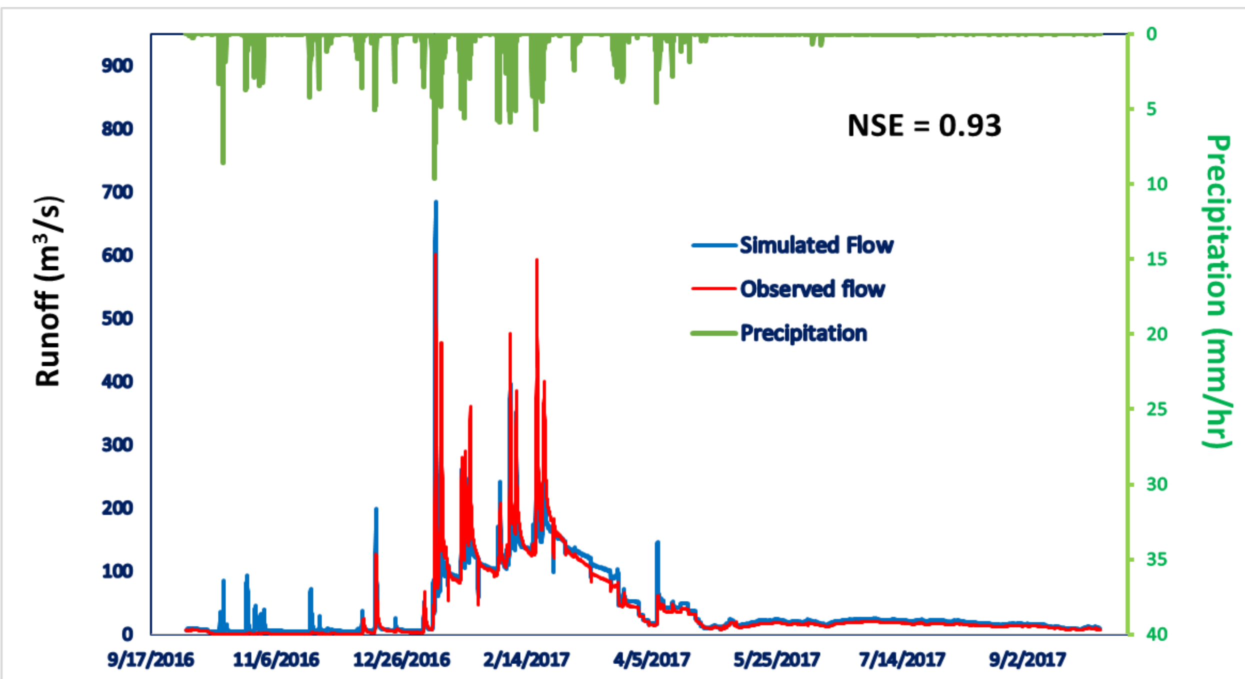

Figure 3-4. Hydrograph showing a comparison of observed and simulated post-fire runoff at upper Cache Creek watershed (Rumsey outlet) for WY 2017. From Wang et al. (2019).

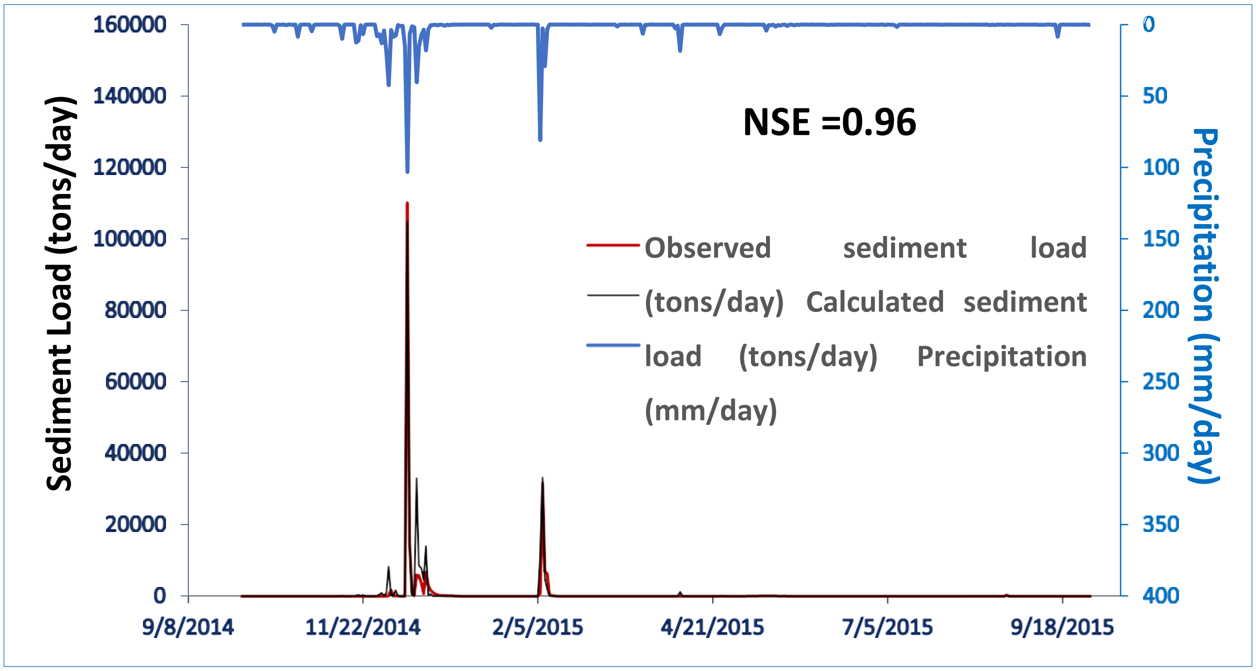

Figure 3-5. Daily time series sediment load at upper Cache Creek Watershed (Rumsey outlet) in WY 2015. From Wang et al. (2019).

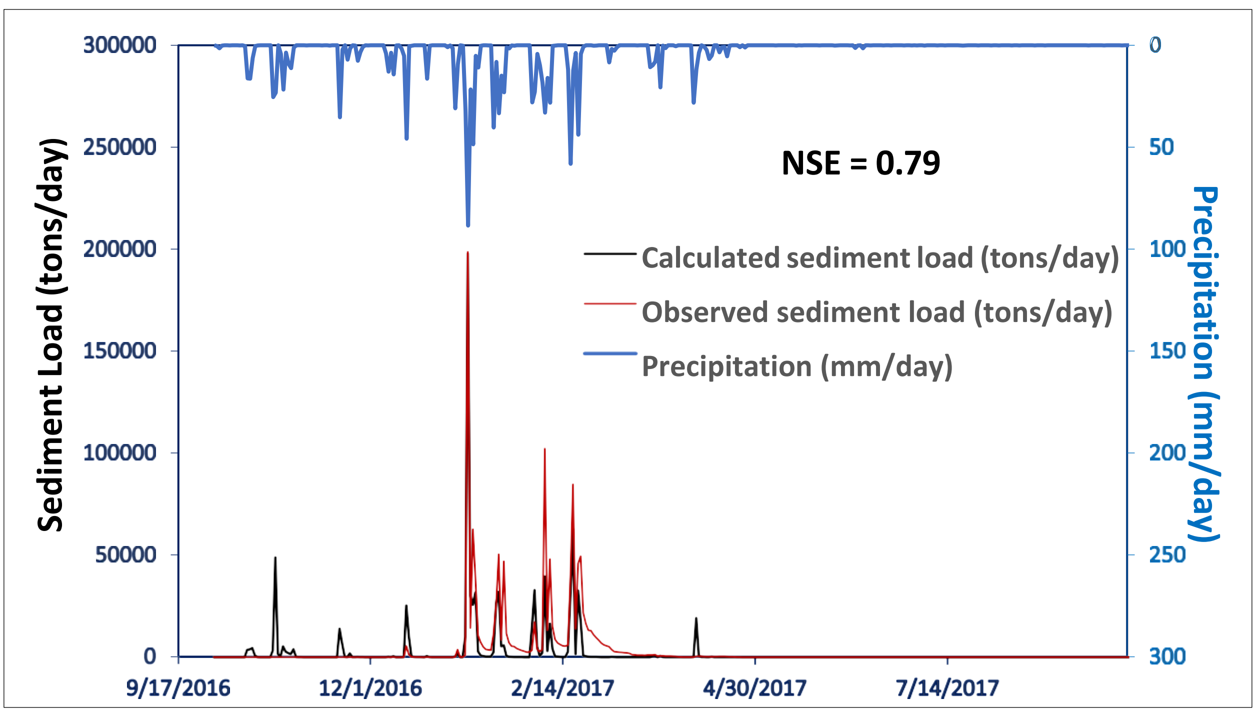

Figure 3-6. Daily time series sediment load at upper Cache Creek Watershed (Rumsey outlet) in WY 2017. From Wang et al. (2019).

3.1.2. Available Data for Modeling of Nutrient Loads in Cache Creek

Based on previous work, there is a daily time series of SS load (kg/day) simulated by the PFHydro, model and calibrated for WY 2015 (pre-fire) and WY 2017 (post-fire) at the Cache Creek at Rumsey station (Wang et al., 2019). However, prior to the present study there were no observed or estimated data for TN and TP loads during WY 2015 or WY 2017 at the Rumsey station, which is the outlet of the watershed where the PFHydro model was applied.

Data for the water years of interest to this study are available at three monitoring stations (continuous-flow stream gages) in the Cache Creek watershed (Fig. 3-1). USGS data are available at https://waterdata.usgs.gov; California Department of Water Resources (DWR) data are available at https://cdec.water.ca.gov/queryTools.html.
Rumsey (USGS station 11451800; DWR station RUM):

- Continuous flow, WY 2015 (DWR) and WYs 2016–19 (USGS)
- Discrete samples for SS, WYs 2015–19 (USGS)
- Discrete samples for TN, TP, and other N and P species, WY 2019 (USGS)

Yolo (USGS station 11452500):

- Continuous flow, WYs 2010–19 (USGS)
- Discrete samples for SS (WY 2017–19) (USGS)
- Discrete samples for TN, TP, and other N and P species (WY 2019) (USGS)

Road 102 (USGS station 11452600):

- Continuous flow, WY 2010–17 (USGS)
- Discrete samples for SS, WY 2010–17 (USGS)
- Discrete samples for TN, TP, and other N and P species (WY 2011)

Several of the N and P species had sufficient data for load calculations, including the following forms of N:

- Nitrate (NO_3^-) plus nitrite (NO_2^-) in filtered water (f_NO3+NO2), analyzed by reduction using cadmium with diazotization (Fishman, 1993a)
- Ammonium plus organic nitrogen in filtered water (f_NH3+N-org) and in unfiltered water (u_NH3+N-org), analyzed by Kjeldahl digestion followed by the colorimetric salicylate- hypochlorite method (Patton and Truitt, 1992)
- Total nitrogen (TN), computed as the sum of u_NH3+N-org and f_NO3+NO2

and the following forms of P:

- Orthophosphate (PO_4^{3-}) in filtered water (f_OP), analyzed by a phosphomolybdate blue method with antimony added to increase the reduction rate (Patton and Truitt, 1992; Fishman, 1993b)
- Phosphorus in filtered water (f_PO4) determined with phosphomolybdate blue after Kjeldahl digestion (O'Dell, 1993)

- Phosphate in unfiltered water (u_PO4) determined with phosphomolybdate blue after Kjeldahl digestion (Patton and Kryskalla, 2003). The Kjeldahl digestion captures both organic and inorganic forms of phosphorus. The parameter u_PO4 is considered to represent total phosphorus (TP).

3.1.3. Methods for Computing Nutrient Loads at Rumsey, Water Years 2015 and 2017

Data collected by USGS at the Rumsey, Yolo, and Road 102 monitoring stations (Figure 3-1) during WY 2011 and WY 2019 are used in this study to develop calibrated estimates of pre-fire and post- fire TN and TP loads for Cache Creek at Rumsey for WY 2011 and WY 2019. Then, the relationships between daily SS and daily TN load, and between daily SS and daily TP load, during WY 2011 and WY 2019, respectively, are assumed to be the same as the relationships in pre-fire WY 2015 and post-fire WY 2017, respectively, so that the data can be used to compute calibrated daily TN and TP loads at Rumsey during WYs 2015 and 2017.

We assume that downstream phosphorus and nitrogen loads are derived mostly from upstream soil erosion. Furthermore, we assume that TN and TP loads are related to suspended sediment (SS) loads by the following equations:

$$\text{TN Load (kg/day)} = \text{SS Load (tons/day)} \times \text{TN Concentration (g/kg)} \times 917.185 \text{ (kg/ton)} / 1000 \text{ (g/kg)} \quad \text{(3-1a)}$$
$$\text{TP Load (kg/day)} = \text{SS Load (tons/day)} \times \text{TP Concentration (g/kg)} \times 917.185 \text{ (kg/ton)} / 1000 \text{ (g/kg)} \quad \text{(3-1b)}$$

where TN Concentration is concentration of total nitrogen in the suspended sediment, and TP Concentration is concentration of total phosphorus in the suspended sediment. It is recognized that filter-passing forms of nitrogen (typically dissolved nitrate, ammonium, and organic N) and phosphorus (typically dissolved orthophosphate and filter-passing organic P) are present, which make equations 3-1a and 3-1b approximations. The proportions of filter-passing N and P species for sites in Cache Creek are evaluated below in section 3-2.

The methods used to compute daily loads of TN and TP for WY 2015 and WY 2019 at Rumsey from available data are shown in Figures 3-7a and 3-7b, respectively. The following additional assumptions were used:

1. The relationship between TN and u_NH3+N-org (Kjeldahl N) at Yolo and Rumsey stations in WY 2019 is the same as the relationship between TN and u_NH3+N-org (Kjeldahl N) at Road 102 in WY 2011. (This assumption is needed because TN loads at Road 102 could not be computed for WY 2011, but loads of u_NH3+N-org (Kjeldahl N) could be computed, as explained in Section 3-2.)
2. The TN and TP loads for Cache Creek at Road 102 during WY 2011 were the same as the TN and TP loads for Cache Creek at Yolo during WY 2011 because these two stations are about 9 km apart and there are no tributary inputs in between the two stations. (Discharge and sediment loads for WY 2017 were within 1% for these two stations.)

3. The relationships between TN and TP loads at Yolo and TN and TP loads at Rumsey during WY 2019 are the same as the relationships between these loads and stations during WY 2011. The daily TN and TP loads at Yolo during WY 2011 can be converted to daily TN and TP loads at Rumsey during WY 2011 by applying a spatial conversion factor.
4. The relationships between TN and SS and between TP and SS at the Rumsey station for WY 2011 can be used to estimate TN and TP loads at the Rumsey station for pre-fire WY 2015 (pre-fire conditions).
5. The relationships between TN and SS and between TP and SS at the Rumsey station for WY 2019 can be used to estimate TN and TP loads at the Rumsey station for post-fire WY 2017 (post-fire conditions).

Additional discussion below explains why the above assumptions are considered to be reasonable.

The conversion factors in Figure 3-7a are discussed in section 3-2. The equations describing the TN- SS and TP-SS relationships are discussed in section 3-3. Some additional information about pre- and post-fire conditions in the Cache Creek watershed are provided in the remainder of this section.
Water-quality data for WY 2011 are considered pre-fire with respect to the Rocky Fire and Jerusalem Fire which affected the Cache Creek watershed in summer 2015. Water-quality data for WY 2019 are considered post-fire with respect to the Rocky and Jerusalem fires and also with respect to various fires that affected the upper Cache Creek watershed in summer and fall 2018, including the Mendocino Complex Fire (River Fire and Ranch Fire) and the Pawnee Fire. WY 2017 had above- average rainfall and runoff, including nine storm events with discharge above 10,000 cfs at Rumsey. The effect of fire on sediment transport for Cache Creek at Rumsey was evident in early WY 2017 storms but less evident toward the end of WY 2017, based on analysis of plots of suspended sediment concentration versus discharge; by the time of storm events later in WY 2017 (February and March 2017), suspended sediment concentrations for a given discharge were similar to pre-fire conditions in WY 2015.

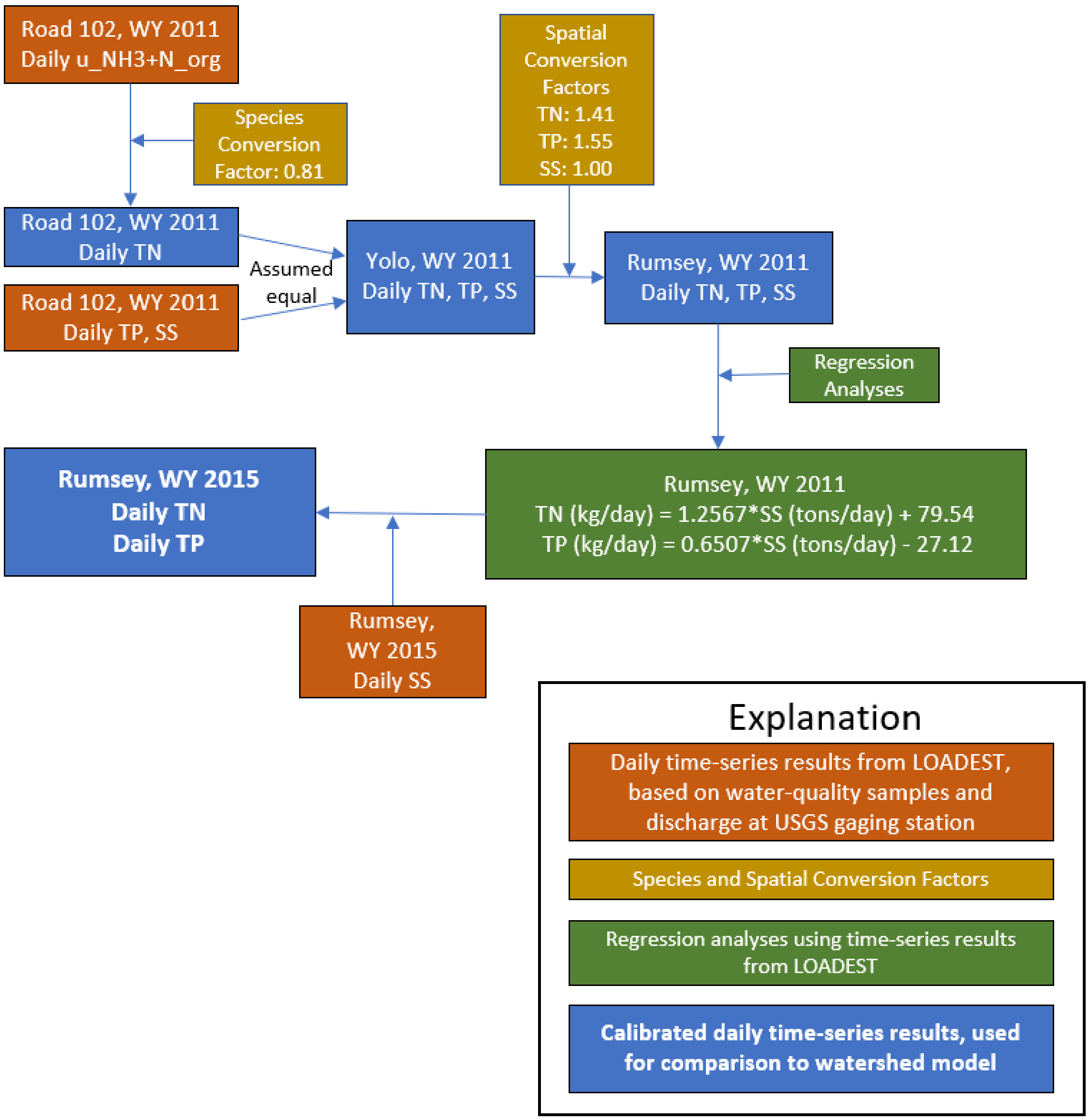

Figure 3-7a. Method used to compute calibrated daily TN and TP loads for WY 2015 (pre-fire) for Cache Creek at Rumsey.

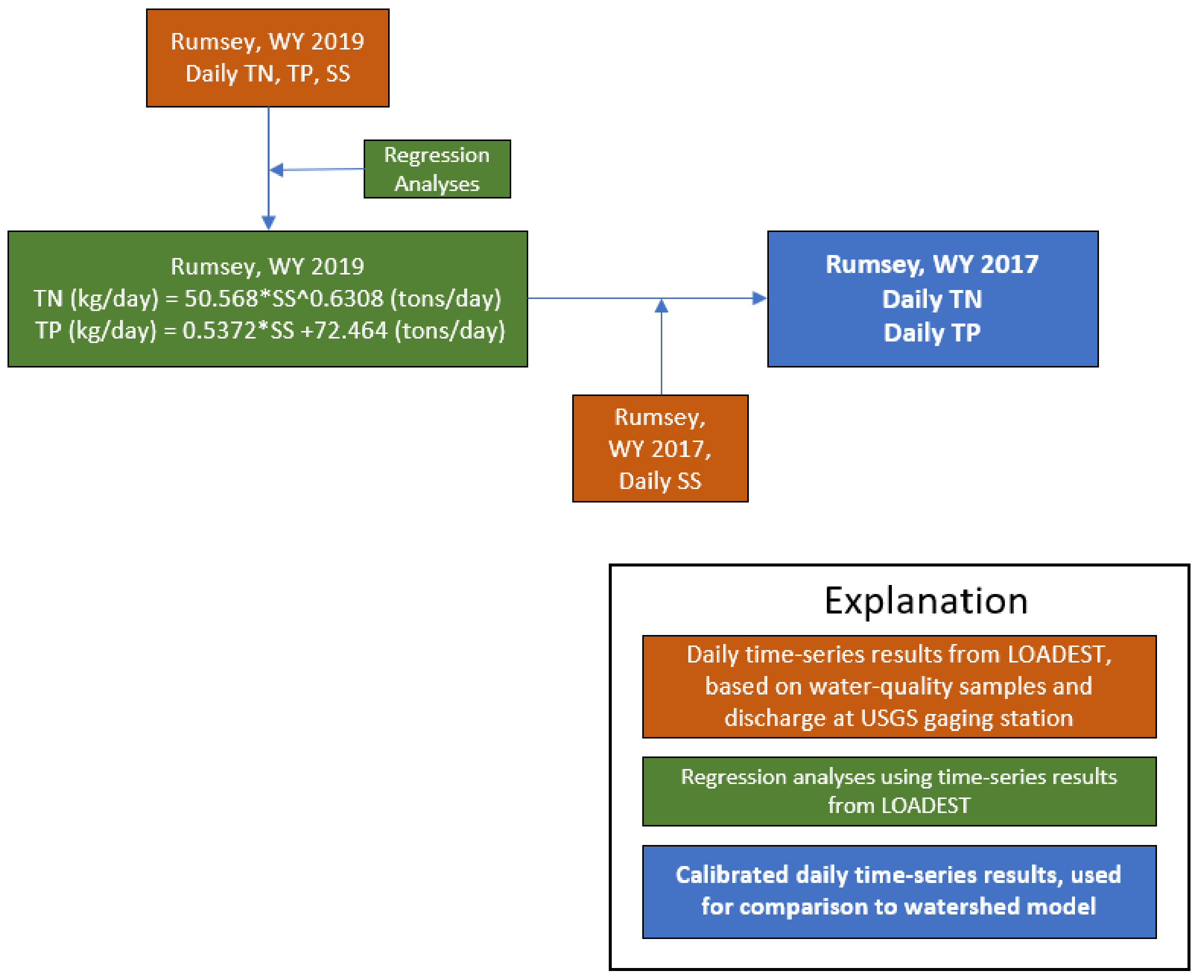

Figure 3-7b. Method used to compute calibrated daily TN and TP loads for WY 2017 (post-fire) for Cache Creek at Rumsey.

Conditions during WY 2019 for Cache Creek at Rumsey are considered to be post-fire, similar in some ways to WY 2017. The 2018 Mendocino Complex Fire and Pawnee Fire affected large parts of the upper Cache Creek watershed, however much of the fire-affected area drains into Indian Valley Reservoir and Clear Lake, which is presumed to have trapped a majority of the suspended sediment. Some areas affected by these two fires drain directly into the North Fork of Cache Creek downstream of Indian Valley Reservoir, likely causing some impact of the 2018 fires on water quality and sediment transport in Cache Creek upstream of Rumsey. WY 2019 was also above-average with regard to rainfall and runoff. Similar to WY 2017, fire effects during WY 2019 are expected to have been more pronounced in early-season storms, including the “first flush” in December 2018, and to have diminished as the water year progressed.

3.2. LOADEST Introduction and Results for Water Years 2011 and 2019

LOADEST regression models (Runkel et al., 2004) calibrated by observed samples were used to estimate a daily time series of SS, TN and TP loads in WY 2019 at the Rumsey and Yolo stations and in WY 2011 at the Road 102 station on Cache Creek (see Figure 3-1).

3.2.1. LOADEST Introduction

LOADEST is a Fortran program originally developed by Runkel et al. (2004) to estimate constituent loads. This model was updated by Lorenz et al. (2017) in the R programming language. The R version, called rloadest, is available on the USGS–R GitHub pages (https://github.com/USGS- R/rloadest). The models are developed in rloadest by first creating a calibration model by regressing discrete concentration values against concurrent instantaneous streamflow values. For each calibration load model, the rloadest program computes regression coefficients by means of the Adjusted Maximum Likelihood Estimation method. Model coefficients are considered significant if they have a p-value less or equal to 0.05. A p-value less than 0.05 indicates less than a 5 percent chance that the relationship being tested is a false positive effect.

In rloadest, for each constituent being considered, nine predefined models (Runkel and others, 2004) are tested (**Table 3-1**), and the models are automatically ranked based on Akaike information criterion scores (Helsel and Hirsch, 2020). Additionally, rloadest computes bias diagnostics that compare estimated loads to observed loads. Bias diagnostics for loads include the load bias percent (Bp) (the percentage the model over- [negative number] or under- [positive number] estimates the sum of the estimated loads compared with the sum of the observed loads). The rloadest program also computes the Nash-Sutcliffe Efficiency index (NSE) which provides a measure of model fit that ranges from 1 (perfect fit) to negative infinity. To select the best model for load calculation from the nine models in **Table 3-1**, the user evaluates the significance of model coefficients (based on p- values), along with the Bp and NSE results. Typically, model selection criteria require p values less than 0.05 for each parameter, Bp values less than 25 percent, and NSE values greater than 0.5 (Runkel et al., 2004). After selecting the best model, diagnostic plots are created to assess the variance and distribution of the model residuals as a function of (1) predicted values, (2) normal quantiles, and (3) time (Runkel et al., 2004; Lorenz et al., 2017).

Table 3-1. LOADEST model equations (from Runkel et al., 2004)

Model number	Regression model
1	$a_0 + a_1 \ln Q$
2	$a_0 + a_1 \ln Q + a_2 \ln Q^2$
3	$a_0 + a_1 \ln Q + a_2 dtime$
4	$a_0 + a_1 \ln Q + a_2 \sin(2\pi dtime) + a_3 \cos(2\pi dtime)$
5	$a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 dtime$

6	$a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi dtime) + a_4 \cos(2\pi dtime)$
7	$a_0 + a_1 \ln Q + a_2 \sin(2\pi dtime) + a_3 \cos(2\pi dtime) + a_4 dtime$
8	$a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi dtime) + a_4 \cos(2\pi dtime) + a_5 dtime$
9	$a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi dtime) + a_4 \cos(2\pi dtime) + a_5 dtime + a_6 dtime^2$

lnQ = ln(streamflow) - center of ln(streamflow); dtime = decimal time - center of decimal time.

3.2.2. Application of LOADEST

We used rloadest models to compute daily and annual loads of nutrients and suspended sediment for WY 2011 for Cache Creek at Road 102, and for WY 2019 for Cache Creek at Rumsey, and Cache Creek at Yolo, using the available continuous streamflow data and nutrient concentrations from discrete water samples for each site (https://waterdata.usgs.gov). Because nutrient data are available only for WYs 2011 and 2019 at those locations, the indicated water years were run individually in rloadest. Data from multiple water years are available for suspended sediment (SS), so multi-year models were developed. For Rumsey, the SS load model was developed for WYs 2015– 19; for Yolo the SS load model was developed for WYs 2017–19, and for Road 102, the SS load model was developed for WYs 2010–17.

Regression models from rloadest for load estimation and bias statistics for each model are shown in **Tables 3-2, 3-3, and 3-4**. Overall, the adjusted coefficient of determination (R^2) indicated that the models predicted a large portion of the variance in the nutrient and suspended-sediment concentration dataset (with R^2 values ranging from 0.86 to 0.99). Bp and NSE values vary for each constituent and greatly depend on the goodness of model fit used for load calculation. All models used for load calculations had a Bp values < 25 except for model number 2 used to calculate loads of u_PO4+P-org at Cache Creek at Road 102 in WY 2011. However, this model was considered acceptable because the NSE value was greater than 0.5.

Table 3-2. Statistics from LOADEST model runs for nutrients and suspended sediment load calculations for the Cache Creek at Rumsey station during WY 2019 (post-fire)

Constituent and water years	Overall p-value	Overall R^2	Bp	NSE	LOADEST model used (Table 3-1)	Number of days with samples used for calibration	Total number of samples
TP	<0.0001	0.97	-0.4	0.981	3	18	27
f_PO4+P-org	<0.0001	0.89	17.9	0.751	3	18	27
f_OP	<0.0001	0.96	2.9	0.896	5	18	27
TN	<0.0001	0.98	-3.0	0.955	3	18	27
u_NH3+N-org	<0.0001	0.98	-4.9	0.949	3	18	27
f_NH3+N-org	No model worked	-	-	-	-	18	27
f_NO3+NO2	<0.0001	0.86	17.3	0.810	3	18	27
SS	0.0001	0.88	2.5	0.775	1	277	678

[f_NH3+N-org, filtered ammonia and organic nitrogen (Kjeldahl N); f_OP, filtered orthophosphate; f_PO4+P-org, filtered phosphate and organic phosphorus; (Kjeldahl P); NSE, Nash-Sutcliffe Efficiency; ppm, part per million; SE, standard error; SS, suspended sediment; TN, total nitrogen; TP, total phosphorus; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N); WY, water year].

Note that SS model is for WYs 2015-19; models for other parameters are only for WY 2019.

Table 3-3. Statistics from LOADEST model runs for nutrients and suspended sediment load calculations for Cache Creek at Yolo station including WY 2019 (post-fire)

Constituent and water years	Overall p-value	Overall R^2	Bp	NSE	LOADEST model used (Table 3-1)	Number of days with samples used for calibration	Total number of samples
TP	<0.0001	0.98	12.9	0.906	3	14	16
f_PO4+P-org	<0.0001	0.94	3.0	0.791	5	14	16
f_OP	<0.0001	0.97	-2.1	0.864	5	14	16
TN	<0.0001	0.96	-5.1	0.923	3	14	16
u_NH3+N-org	<0.0001	0.97	1.9	0.934	3	14	16
f_NH3+N-org	<0.0001	0.89	-18.8	0.582	1	14	16
f_NO3+NO2	<0.0001	0.88	1.3	0.700	1	14	16
SS	<0.0001	0.98	19.4	0.864	6	139	229

[f_NH3+N-org, filtered ammonia and organic nitrogen (Kjeldahl N); f_OP, filtered orthophosphate; f_PO4+P-org, filtered phosphate and organic phosphorus; (Kjeldahl P); NSE, Nash-Sutcliffe Efficiency; ppm, part per million; SE, standard error; SS, suspended sediment; TN, total nitrogen; TP, total phosphorus; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N); WY, water year]
Note that SS model is for WYs 2017-19; models for other parameters are only for WY 2019.

Table 3-4. Statistics from LOADEST model runs for nutrients and suspended sediment load calculations for Cache Creek at Road 102 station including WY 2011 (pre-fire)

Constituent and water years	Overall p-value	Overall R^2	Bp	NSE	LOADEST model used (Table 3-1)	Number of days with samples used for calibration	Total number of samples
TP	<0.0001	0.95	25.7	0.780	2	19	21
f_PO4+P-org	<0.0001	0.93	5.5	0.925	2	19	21
f_OP	<0.0001	0.93	5.5	0.925	2	19	21
TN	No model worked	-	-	-	-	-	-
u_NH3+N-org	<0.0001	0.98	2.0	0.985	1	19	21
f_NH3+N-org	<0.0001	0.99	5.4	0.929	2	19	21
f_NO3+NO2	No model worked	-	-	-	-	19	21
SS	<0.0001	0.89	-14.5	0.776	4	249	336

[f_NH3+N-org, filtered ammonia and organic nitrogen (Kjeldahl N); f_OP, filtered orthophosphate; f_PO4+P-org, filtered phosphate and organic phosphorus; (Kjeldahl P); NSE, Nash-Sutcliffe Efficiency; ppm, part per million; SE, standard error; SS, suspended sediment; TN, total nitrogen; TP, total phosphorus; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N); WY, water year]
Note that SS model is for WYs 2010-17; models for other parameters are only for WY 2011.

Tables 3-2, 3-3 and 3-4 also show the number of samples used for load calculations. These numbers differ from the number of samples available for each site. This is because multiple samples were taken on some days, and rloadest only allows one concentration measurement per day. In those situations, a flow-weighted mean concentration was used.

3.2.2. LOADEST Results for Water Years 2011 (pre-fire) and 2019 (post-fire)

Annual loads of nutrient (nitrogen and phosphorus) species and suspended sediment and the corresponding standard errors for Cache Creek at Rumsey in WY 2019, Cache Creek at Yolo in WY 2019, and Cache Creek at Road 102 in WY 2011 are summarized in **Tables 3-5, 3-6, and 3-7** respectively.

Table 3-5. Results from LOADEST model runs for nutrients and suspended sediment load calculations for the Cache Creek at Rumsey station for WY 2019 (post-fire)

Constituent	Annual Load (kg/yr)	Standard error (SE) (kg/yr)	%SE*	Apparent concentration (nutrient load normalized to SS load)* (mg/kg)	Uncertainty of apparent concentration (%SE)*
TP	610,000	80,000	13%	570	20%
f_PO4+P-org	33,000	7,000	21%	-	-
f_OP	31,000	4,000	13%	-	-
TN	1,200,000	100,000	8%	1100	17%
u_NH3+N-org	1,000,000	90,000	9%	960	17%
f_NH3+N-org	No model worked	-	-	-	-
f_NO3+NO2	190,000	34,803	19%	-	-
SS	1,100,000,000	200,000,000	15%	-	-

[f_NH3+N-org, filtered ammonia and organic nitrogen (Kjeldahl N); f_OP, filtered orthophosphate; f_PO4+P-org, filtered phosphate and organic phosphorus; (Kjeldahl P); kg/yr, kilogram per year; mg/kg, milligram per kilogram; ppm, part per million; SE, standard error; SS, suspended sediment; TN, total nitrogen; TP, total phosphorus; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N)]
* %SE values and apparent concentrations computed before rounding. Propagated uncertainty computed using square- root of the sum of the squares method.

At Cache Creek at Rumsey, during the WY 2019 post-fire conditions, total phosphorus (TP) load was estimated at 650,000 kg/yr using rloadest (**Table 3-5**). Standard error of each load estimate is derived from the rloadest regression model uncertainty and represents the variability that may be attributed to the model calibration (parameter uncertainty). Other sources of error such as uncertainty in discharge measurements or laboratory data are not included. When combining load results (e.g. ratios between constituents or between locations) uncertainty is propagated using the square-root of the sum of the squares method. Filtered orthophosphate (f_OP) accounts for about 5% of the TP load (**Table 3-5**), with a propagated Standard Error of 18%, indicating that about 95% of the TP load is carried by suspended particulates.

At Cache Creek at Rumsey during WY 2019, total nitrogen (TN) load was estimated using rloadest at 1,200,000 kg/yr with 8% SE (**Table 3-5**). Unfiltered ammonia and organic nitrogen (u_NH3+N- org) accounts for about 84% of TN loads (**Table 3-5**). No rloadest model was suitable to calculate filtered ammonia and organic nitrogen (f_NH3+N-org) loads at this site for WY 2019. Suspended sediment (SS) load during WY 2019 was estimated at 1,100,000,000 kg/yr with 15% SE (Table 3-5).

At Cache Creek at Yolo, during the 2019 post-fire conditions (**Table 3-6**), total phosphorus (TP) load was estimated at 1,000,000 kg/yr with 15% SE. Filtered orthophosphate (f_OP) accounts for about 5% of the TP load (**Table 3-6**), implying that about 95% of the TP load is carried by suspended particulates, similar to the results for Rumsey (**Table 3-5**).

Table 3-6. Results from LOADEST model runs for nutrients and suspended sediment load calculations for the Cache Creek at Yolo station for WY 2019 (post-fire)

Constituent	Annual Load (kg/yr)	Standard error (SE) (kg/yr)	%SE*	Apparent concentration (nutrient load normalized to SS load)* (mg/kg)	Uncertainty of apparent concentration (%SE)*
TP	950,000	100,000	15%	930	19%
f_PO4+P-org	46,000	9,000	21%	-	-
f_OP	47,000	6,000	13%	-	-
TN	1,800,000	200,000	14%	1800	21%
u_NH3+N-org	1,400,000	200,000	15%	1400	19%
f_NH3+N-org	440,000	100,000	22%	-	-
f_NO3+NO2	420,000	60,000	14%	-	-
SS	1,000,000,000	100,000,000	11%	-	-

[f_NH3+N-org, filtered ammonia and organic nitrogen (Kjeldahl N); f_OP, filtered orthophosphate; f_PO4+P-org, filtered phosphate and organic phosphorus; (Kjeldahl P); kg/yr, kilogram per year; mg/kg, milligram per kilogram; ppm, part per million; SE, standard error; SS, suspended sediment; TN, total nitrogen; TP, total phosphorus; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N)]
* %SE values and apparent concentrations computed before rounding. Propagated uncertainty computed using square- root of the sum of the squares method.

Total nitrogen (TN) load for Cache Creek at Yolo during WY 2019 was estimated at 1,800,000 kg/yr with 14% SE. Unfiltered ammonia and organic nitrogen (u_ NH3+N-org) accounts for about 78% of TN loads (**Table 3-6**). Suspended sediment (SS) load was estimated at 1,000,000,000 kg/yr with 11% SE.

Nutrient and suspended sediment loads at Cache Creek at Road 102 were estimated for WY 2011 to evaluate pre-fire conditions in the basin (**Table 3-7**). Total phosphorus (TP) load was estimated at 330,000 kg/yr with 25% SE. Unfiltered phosphate plus organic phosphorus (u_PO4+P-org) accounts for 100% of the TP load (**Table 3-7**). None of the rloadest models were suitable to estimate either Total Nitrogen (TN) or filtered Nitrate + Nitrite (f_NO3+NO2) loads at this site during the 2011 water year. An alternative method was used to estimate TN load by comparing annual loads of TN and unfiltered ammonia plus organic nitrogen (u_NH3+N-org) at two stations on Cache Creek (Cache Creek at Rumsey, and Cache Creek at Yolo) during Water Year 2019 and estimating a ratio value of u_NH3+N-org to TN that can be used to estimate TN loads at Cache Creek at Road 102 during WY 2011 (**Table 3-8**). By using that ratio, TN load at Cache Creek at Road 102 in WY 2011 is estimated at 660,000 kg/yr with a standard error of 21% (**Table 3-7**).

Suspended sediment (SS) load at Cache Creek at Road 102 was estimated at 320,000,000 kg/yr with a 16% SE.

Table 3-7. Results from LOADEST model runs for nutrients and suspended sediment load calculations for Cache Creek at Road 102 station including WY 2011 (pre-fire)

Constituent	Annual Load (kg/yr)	Standard error (SE) (kg/yr)	%SE*	Apparent concentration (nutrient load normalized to SS load)* (mg/kg)	Uncertainty of apparent concentration (%SE)*
TP	350,000	100,000	31%	1100	35%
f_PO4+P-org	18,000	4,000	21%	-	-
f_OP	13,000	3,000	22%	-	-
TN	660,000 #	-	21%	2100	26%
u_NH3+N-org	530,000	70,000	14%	1700	21%
f_NH3+N-org	140,000	10,000	7%	-	-
f_NO3+NO2	No model worked	-	-	-	-
SS	320,000,000	50,000,000	16%	-	-

[f_NH3+N-org, filtered ammonia and organic nitrogen (Kjeldahl N); f_OP, filtered orthophosphate; f_PO4+P-org, filtered phosphate and organic phosphorus; (Kjeldahl P); kg/yr, kilogram per year; mg/kg, milligram per kilogram; ppm, part per million; SE, standard error; SS, suspended sediment; TN, total nitrogen; TP, total phosphorus; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N); WY, water year; %, percentage]

Estimated TN load at Road 102 in WY 2011 calculated dividing u_NH3+N-org load by 0.81, the average ratio of u_NH3+N-org to TN at Rumsey and Yolo in WY 2019 (see **Table 3-8**).

* %SE values and apparent concentrations computed before rounding. Propagated uncertainty computed using square- root of the sum of the squares method.

*

Table 3-8. Comparison of annual loads of total nitrogen and unfiltered ammonia plus organic nitrogen at two monitoring stations on Cache Creek during WY 2019

Constituent	WY 2019 Cache Creek at Rumsey	WY 2019 Standard Error (SE)	WY 2019 Cache Creek at Yolo	WY 2019 SE	WY 2019 Average ratio of u_NH3+N-org / TN *	SE
	kg/yr	%	kg/yr	%		%
TN	1,200,000	8%	1,800,000	14%		
u_NH3+N-org	1,000,000	9%	1,400,000	15%		
Ratio of u_NH3+N-org / TN *	0.84	12%	0.78	21%	**0.81**	16%

[TN, total nitrogen; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N); WY, water year]

* Ratios computed before rounding. Propagated uncertainty computed using square-root of the sum of the squares method.

Table 3-9. Comparison of annual loads of selected phosphorus and nitrogen constituents in Cache Creek at Rumsey and Yolo stations during WY 2019

Constituent	WY 2019 Cache Creek at Rumsey	WY 2019 Standard Error (SE)	WY 2019 Cache Creek at Yolo	WY 2019 Standard Error (SE)	WY 2019 Load ratio: Yolo to Rumsey *	WY 2019 Standard Error (SE)	WY 2019 Average Load ratio: Yolo to Rumsey *
	kg/yr	%	kg/yr	%		%	
TP	610,000	13%	950,000	15%	1.55	20%	**1.55**
TN	1,200,000	8%	1,800,000	14%	1.46	16%	
u_NH3+N-org	1,000,000	9%	1,400,000	15%	1.36	17%	**1.41**

[kg/yr, kilogram per year; TN, total nitrogen; TP, total phosphorus; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N); WY, water year; %, percentage]

* Ratios computed before rounding. Propagated uncertainty computed using square-root of the sum of the

squares method.

Pre-fire conditions for nutrient loads in Cache Creek at Rumsey during WY 2011 or WY 2015 are not available. So, we use the estimated loads for Cache Creek at Road 102 in WY 2011 and loads for Cache Creek at Rumsey and Yolo in WY 2019 to estimate loads at Rumsey in WY 2011, as explained below. Comparing TN and TP loads for Cache Creek at Rumsey and Cache Creek at Yolo during WY 2019 (**Tables 3-5 and 3-6**), a substantial increase in TN and TP loads was observed between the stations, which are separated by a distance of about 70 km (about 43.5 miles) along the creek. The increase in nutrients from Rumsey to Yolo is likely caused by agricultural drainage. **Table 3-9** shows the ratio of loads of TP, u-PO4+p-org, TN, and u_NH3+N-org estimated at Cache Creek at Yolo to their respective loads estimated at Cache Creek at Rumsey. The average ratio between sites (downstream divided by upstream) for the P species is 1.54 and for the N species is 1.41. Nutrient load data are available for WY 2011 for Cache Creek at Road 102, located about 9 km (5.5 miles) downstream from Cache Creek at Yolo. As explained above (section 3.1.1) there are no tributaries entering Cache Creek between Yolo and Road 102 and discharge and sediment loads are very similar at these two stations. Therefore, pre-fire loads for Cache Creek at Rumsey during 2011 were calculated by dividing the loads for Cache Creek at Road 102 by the average ratios shown in **Table 3-9**. Using this approach, the estimated loads for Cache Creek at Rumsey during WY 2011 are 230,000 kg/yr for TP and 470,000 kg/yr for TN, with propagated uncertainties of 31% and 21%, respectively (**Table 3-10**).

Table 3-10. Estimated annual loads for Cache Creek at Rumsey and Road 102 for selected phosphorus and nitrogen parameters during WY 2011

Constituent	WY 2011 Cache Creek at Rumsey	WY 2011 Standard Error (SE)	WY 2011 Cache Creek at Road 102	WY 2011 Standard Error (SE)	WY 2019 Load ratio: Yolo to Rumsey *	WY 2019 Standard Error (SE)	WY 2019 Average Load ratio: Yolo to Rumsey *
	kg/yr	%	kg/yr	%		%	
TP *	230,000	37%	350,000	31%	1.55	20%	1.55
TN **#	470,000	26%	660,000	21%	1.46	16%	
u_NH3+N-org #	380,000	22%	530,000	14%	1.36	17%	1.41

[kg/yr, kilogram per year; TN, total nitrogen; TP, total phosphorus; u_NH3+N-org, unfiltered ammonia and organic nitrogen (Kjeldahl N); u_PO4+P-org, unfiltered phosphate and organic phosphorus (Kjeldahl P); WY, water year; %, percentage] Propagated uncertainty computed using square-root of the sum of the squares method.
* Estimated annual TP load at Rumsey in WY 2011 calculated by dividing loads at Road 102 for WY 2011 by 1.55, based on ratio of TP load at Yolo to TP load at Rumsey in WY 2019 (see **Table 3-9**)
** Estimated annual loads at Rumsey for unfiltered and particulate N species in WY 2011 calculated by dividing loads at Road 102 for WY 2011 by 1.41, based on ratio of loads at Yolo to loads at Rumsey in WY 2019 (see **Table 3-9**).
# TN of 470,000 = 380,000/0.81 based on relationship in **Table 3-8**.

Here is a summary of the assumptions that were made to compute the estimated loads for P and N constituents for Cache Creek at Rumsey during WY 2011, as given in **Table 3-10**.

1. The average load ratios for Yolo to Rumsey for unfiltered (i.e. whole-water) P and N constituents during WY 2019 (**Table 3-9**) are the same as the load ratios for Road 102 to Rumsey during WY 2011.
2. The annual TN load (kg/yr) at Rumsey for WY 2011 can be obtained from the annual u_NH3+N-org load (kg/yr) at Road 102 divided by 0.81 (to convert to annual TN load

at Yolo) and further divided by 1.41 (to convert from TN at Road 102 to TN at Rumsey).

3. The annual TP load (kg/yr) at Rumsey for WY 2011 can be obtained from the annual TP load (kg/yr) at Yolo, which is the same as at Road 102 for WY 2011, divided by 1.55 (to convert from TP at Road 102 to TP at Rumsey).

3.3. Daily Loads of Total Nitrogen and Total Phosphorus at Rumsey for Water Years 2015 and 2017

The daily loads of TN, TP, and SS at Rumsey for WY 2011 were obtained using methods described in section 3-2 and summarized in Figure 3-7a. Daily loads of TN and TP at Rumsey for WY 2019 were obtained directly from USGS sampling data using LOADEST, as described in section 3-2. The daily SS load at Rumsey was previously available (Wang et al., 2019, 2020) for WY 2015 (pre-fire) and WY 2017 (post-fire), the years of interest for evaluating the effects of fire retardant using the PFHydro model.

In this section, it is explained how the loads for WY 2011 are used to estimate loads for WY 2015 (pre-fire) and how the loads for WY 2019 are used to estimate the loads for WY 2017 (post-fire).

3.3.1. Nutrient Loads at Rumsey for WY 2015 (pre-fire)

A strong linear relationship ($R^2 = 0.9942$) (Figure 3-8a and Eq. 3-2) exists between TN (kg/day) and SS (tons/day) for WY 2011 at Rumsey:

$$TN_{Rumsey}\ (kg/day) = 1.2567 \times SS_{Rumsey}\ (tons/day) + 79.54\ (kg/day) \qquad (3\text{-}2)$$

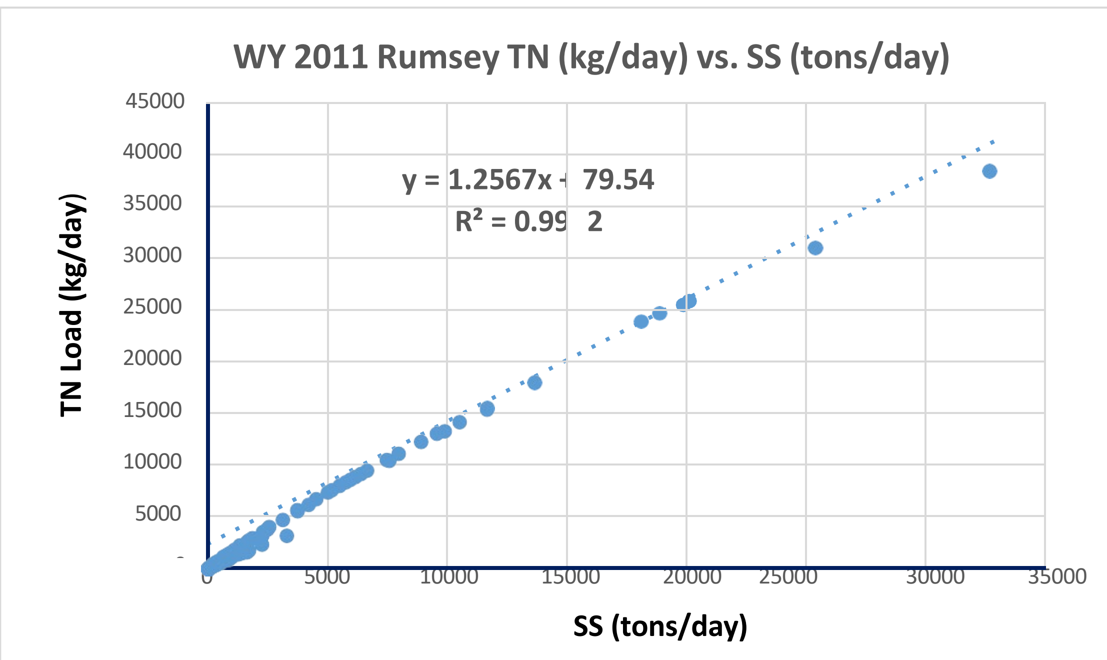

Figure 3-8a. The relation between TN (kg/day) and SS (tons/day) for WY 2011 at Rumsey.

The relationships between TN and SS and between TP and SS at the Rumsey station for WY 2011 can be used to estimate TN and TP loads at the Rumsey station for pre-fire WY 2015 (pre-fire conditions) using the assumption 4 in section 3-1-3 and that was discussed in section 3-2.

The daily TN load (kg/day) at Rumsey (TN_{Rumsey}) of WY 2015 was obtained by applying Eq. 3-2 with daily SS load (SS_{Rumsey}) (tons/day) at Rumsey of WY 2015 from a previous research project (Wang et al., 2019).

A strong linear relationship (R^2 = 0.9983) (Figure 3-8b and Eq. 3-3) exists between TP (kg/day) and SS (tons/day) of WY 2011 at Rumsey:

TP_{Rumsey} (kg/day) = 0.6507 × SS_{Rumsey} (tons/day) - 27.121 (kg/day) (3-3)

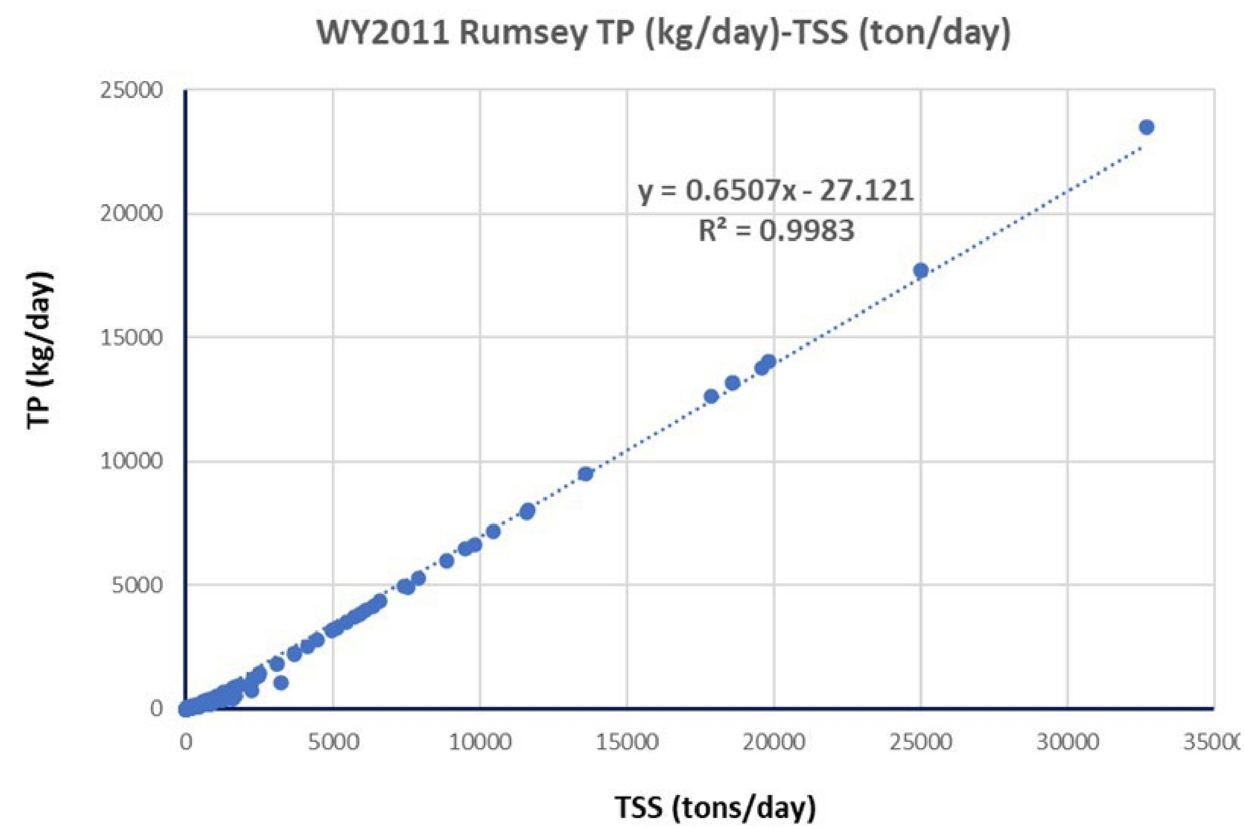

Figure 3-8b. The relation between TP (kg/day) and SS (tons/day) for WY 2011 at Rumsey.

The daily TP load (kg/day) at Rumsey (TP_{Rumsey}) for WY 2015 was obtained by applying Eq. 3-3 to the daily SS load (SS_{Rumsey}) (tons/day) at Rumsey for WY 2015 from a previous research project (Wang et al., 2019).

3.3.2. Nutrient Loads at Rumsey for WY 2017 (post-fire)

The relationships between TN and SS and between TP and SS at the Rumsey station for WY 2019 can be used to estimate TN and TP loads at the Rumsey station for WY 2017 (post-fire conditions), using assumption 5 explained in section 3-1-3 and discussed in section 3-2.

A strong power relationship (R^2 = 0.955) (Figure 3-9 and Eq. 3-4) was established between TN (kg/day) and SS (tons/day) for WY 2019 at Rumsey:

TN_{Rumsey} (kg/day) = 50.568 × SS_{Rumsey} (tons/day) $^{0.6308}$ (3-4)

The daily TN load (kg/day) (TN_{Rumsey})at Rumsey for WY 2017 was obtained by applying Eq. 3-4 with daily SS (tons/day) (SS_{Rumsey}) load for WY 2017 at Rumsey from a previous research project (Wang et al., 2019).

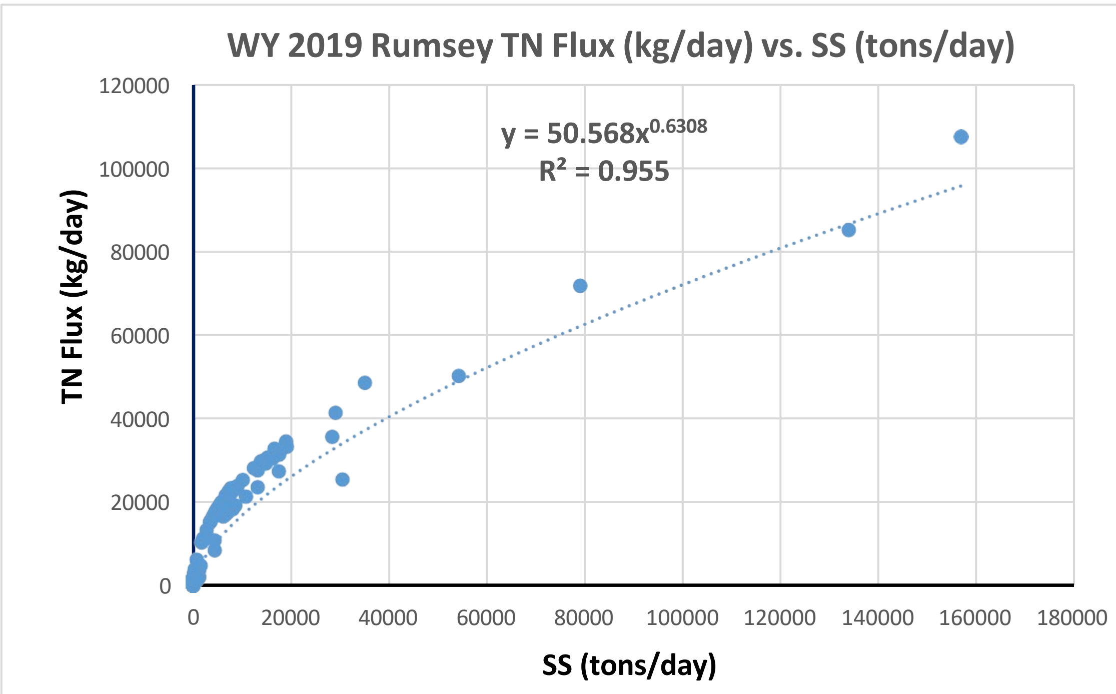

Figure 3-9. The relation between TN (kg/day) and SS (tons/day) in WY 2019 at Rumsey.

A strong linear relationship (R^2 = 0.985) (Figure 3-10 and Eq. 3-5) was established between TP (kg/day) and SS (tons/day) of WY 2019 at Rumsey:

TP_{Rumsey} (kg/day) = 0.53723 × SS_{Rumsey} (tons/day) + 72.464 (3-5)

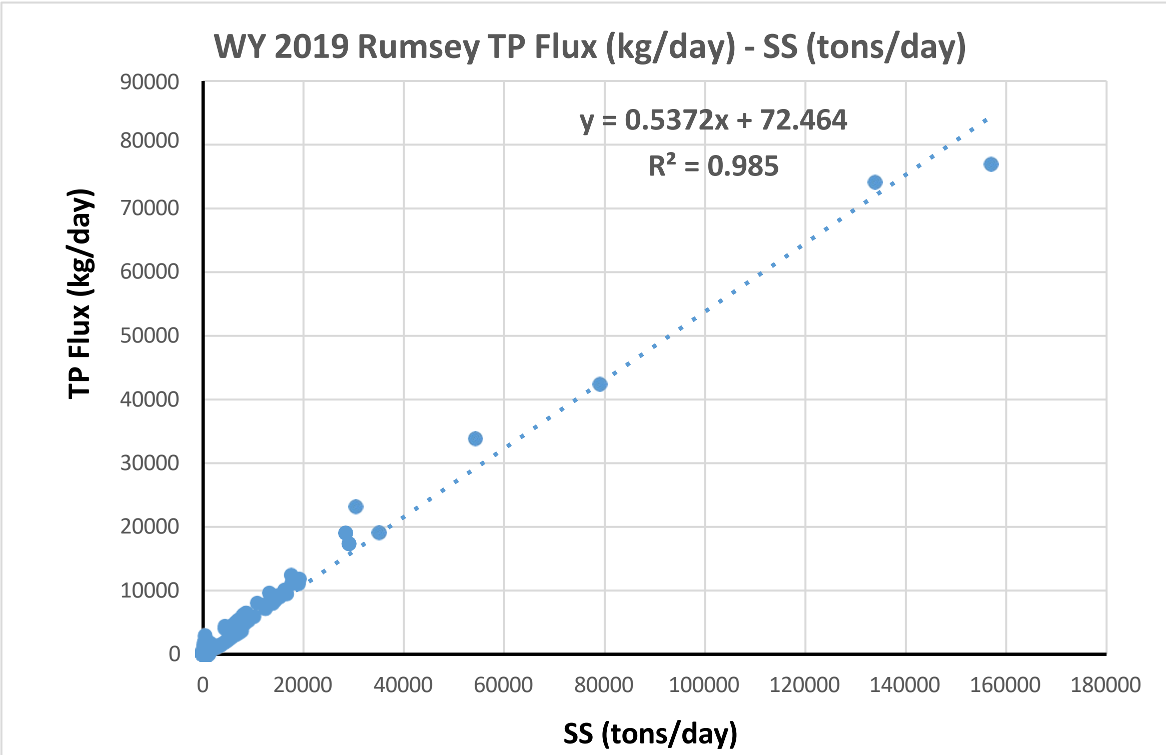

Figure 3-10. The relation between TP (kg/day) and SS (tons/day) for WY 2019 at Rumsey.
The daily TP load (TP_{Rumsey})(kg/day) at Rumsey for WY 2017 was obtained by applying Eq. 3-5 with daily SS (tons/day)(SS_{Rumsey}) load for WY 2017 at Rumsey from a previous research project (Wang et al., 2019).

The daily TN and TP loads for WY 2015 (pre-fire) and WY 2017 (post-fire) at Rumsey are considered to be calibrated daily TN and TP loads that are used to verify and calibrate the updated PFHydro water-quality module. The application of the PFHydro water-quality module to the watershed, with the Rumsey station as the outlet, for simulating daily TN and TP loads during WY 2015 and WY 2017 is described in section 3-4.

3.4. Simulation of Pre- and Post-fire TN and TP Loads at Rumsey using PFHydro Water-quality Module

The updated PFHydro water-quality module was used to simulate daily TN and TP loads from the watershed modeling area for WY 2015 (pre-fire) and WY 2017 (post-fire) using data for soil N and P concentrations from Chapter 2 of this report. The daily TN and TP loads from the PFHydro water- quality module are compared to the calibrated daily TN and TP loads at Rumsey obtained in section 3-3.

The information in **Tables 3-11 and 3-12** on runoff and sediment loads in the study area (Figure 3- 1) are from previous research using the PFHydro model (Wang et al., 2019; Wang et al., 2020). It can be seen from **Table 3-11** that the post-fire impacts in WY 2017 (2nd year post-fire) on runoff and sediment erosion were still significant because annual surface runoff and sediment load for WY 2017 were about 3 times those for WY 2015, whereas the annual total runoff during WY 2017 was only about 1.5 times the annual total runoff during WY 2015. Saturation overland flow dominated the surface runoff in the modeling area (Wang et al., 2020). Therefore, the increased surface runoff should mostly have been due to soil water repellence which caused less infiltration and more surface runoff (Wang et al., 2020).

Table 3-11. Annual runoff and sediment load for WYs 2015 and 2017 (from Wang et al., 2019)

Water Year	Annual precipitation (mm)	Simulated annual runoff (cfs)	Simulated annual surface runoff (cfs)	Percentage of surface flow in total flow	Simulated annual sediment load (tons/day)
WY 2015 (pre-fire)	580 (22.9 in)	1,450,397	221,205	8.6%	255,398
WY 2017 (post-fire)	1118 (44.0 in)	2,664,382	652,991	15.2%	810,907

[cfs, cubic foot per second; in, inch; mm, millimeter; WY, water year; %, percentage]

It can be seen from **Table 3-12** that 24.4% and 3.9% of the annual sediment load at Rumsey was from upstream of the PFHydro modeling area watershed (Figure 3-1) as boundary inputs in WYs 2015 and 2017, respectively.

Table 3-12. Contributions of boundary sediment load in the total sediment load at Rumsey (from Wang et al., 2019)

Water Year	Annual observed sediment load at Rumsey (tons)	Annual boundary sediment load (tons)	Annual boundary sediment load /Annual Rumsey sediment load
WY 2015	272,704	66,637	0.244
WY 2017	1,242,738	48,738	0.039

In the following mass balance equations for the PFHydro modeling area (Figure 3-1), "TN/TP" refers to "either total nitrogen or total phosphorus" (note that this does not refer to the ratio of TN and TP).

$$SS_{Rumsey}\ (kg/day) = SS_{Modeling\ Area}\ (kg/day) + SS_{Boundary}\ (kg/day)$$

$$TN/TP_{Rumsey}\ (kg/day) = TN/TP_{Modeling\ Area}\ (kg/day) + TN/TP_{Boundary}\ (kg/day)$$

So,

$$SS_{Modeling\ Area}\ (kg/day) = SS_{Rumsey}\ (kg/day) - SS_{Boundary}\ (kg/day) = SS_{Rumsey\text{-}Boundary} \qquad (3\text{-}6)$$

$$TN/TP_{Modeling\ Area}\ (kg/day) = TN/TP_{Rumsey}\ (kg/day) - TN/TP_{Boundary}\ (kg/day) = TN/TP_{Rumsey\text{-}Boundary}\ (kg/day) \qquad (3\text{-}7)$$

Where $TN/TP_{Modeling\ Area}$ (kg/day) is daily TN or TP load simulated by the PFHydro model for the watershed modeling area upstream of Rumsey; TN/TP_{Rumsey} (kg/day) is the calibrated daily TN or TP load at Rumsey obtained in section 3-3; $TN/TP_{Boundary}$ (kg/day) is the total daily TN or TP boundary input to the modeling area. $TN/TP_{Rumsey-Boundary}$ (kg/day) is the difference between calibrated daily TN or TP load at Rumsey and the boundary inputs. The latter quantity is used to compare to and calibrate $TN/TP_{Modeling\ Area}$ (kg/day).

As described in Equations 3-1a and 3-1b, TN and TP loads can be derived from SS loads by multiplying SS loads by the concentration of either TN or TP.

For WY 2015 (pre-fire) at Rumsey:

$TN/TP_{Boundary}$ (kg/day) = SS $_{Boundary}$ (tons/day) * $C_{TN/TP\ Boundary\_Pre\text{-}Fire}$ (kg/day) (3-

8) $TN/TP_{Modeling\ Area}$ (kg/day) = $SS_{Modeling\ area}$ (tons/day) * $C_{TN/TP\ Modeling\ area\_Pre\text{-}Fire}$ (kg/day) (3-9)

where $C_{TN/TP\ Boundary\_Pre\text{-}Fire}$ and $C_{TN/TP\ Modeling\ area\_Pre\text{-}Fire}$ are the average concentrations of TN or TP at the boundary and within the modeling watershed, respectively, during pre-fire conditions. These values were estimated in Chapter 2 of this report.

For WY 2017 (post-fire) at Rumsey:

$TN/TP_{Boundary}$ (kg/day) = SS $_{Boundary}$ (tons/day) * $C_{TN/TP\ Boundary\_Post\text{-}Fire}$ (kg/day) (3-

10) $TN/TP_{Modeling\ Area}$ (kg/day) = $SS_{Modeling\ area}$ (tons/day) * $C_{TN/TP\ Modeling\ area\_Post\text{-}fire}$ (kg/day)

(3-11)

where CTN/TP Boundary_Post-Fire and CTN/TP Modeling area_Post-Fire are the average concentrations of TN or TP at the boundary and within the modeling watershed, respectively, during post-fire conditions. These values were estimated in Chapter 2 of this report. $C_{TN/TP\ Boundary\_Pre\text{-}Fire}$ is the same as $C_{TN/TP\ Boundary\_Post\text{-}Fire}$ because no wildfires happened between WY 2015 and WY 2017 upstream of the modeling watershed area.

Daily SS loads for WY 2015 and WY 2017 are available from previous S&T research (Wang et al., 2019) for SS the model boundary ($SS_{Boundary}$), the modeling watershed area ($SS_{Modeling\ area}$), and Rumsey station (SS_{Rumsey}).

3.4.1. Pre-fire WY 2015 Total Nitrogen (TN) and Total Phosphorus (TP) modeling results

The concentration of TN assumed for the model boundary is 1.4 g/kg in WY 2015 (pre-fire; see Chapter 2). The average concentration of TN assumed in the modeling area is 1.4 g/kg (Chapter 2, **Table 2-5**).

The boundary daily TN load (kg/day) of WY 2015 was calculated by using Eq. 3-12.

$$TN_{Boundary}\ (kg/day) = SS_{Boundary}\ (tons/day) \times 1.4\ (g/kg) \times 917.185\ (kg/ton)\ /1000\ (g/kg) \quad (3\text{-}12)$$

The daily TN (kg/day) in the modeling area (watershed where Rumsey is the outlet) ($TN_{modeling\ area}$) for WY 2015 is calculated using Eq. 3-13.

$$TN_{modeling\ area}\ (kg/day) = SS_{modeling\ area}\ (tons/day) \times 1.4\ (g/kg) \times 917.185\ (kg/ton)\ /1000\ (g/kg) \quad (3\text{-}13)$$

Daily TN load (kg/day) at Rumsey (TN_{Rumsey}) WY 2015 was calculated using Eq. 3-2, as discussed in section 3-3:

$$TN_{Rumsey}\ (kg/day) = 1.2567 \times SS_{Rumsey}\ (tons/day) + 79.54\ (kg/day) \quad (3\text{-}2)$$

Where SS_{Rumsey} (tons/day) is daily SS load at Rumsey of WY 2015.

Eq. 3-14 was obtained by applying the above results in Eq. 3-7,

$$\begin{aligned} TN_{Modeling\ Area}\ (kg/day) &= TN_{Rumsey}\ (kg/day) - TN_{Boundary}\ (kg/day) \\ &= TN_{Rumsey\text{-}Boundary}\ (kg/day) \end{aligned} \quad (3\text{-}14)$$

The daily TN load (kg/day) from PFHydro water quality module ($TN_{Modeling\ Area}$) is compared to the daily calibrated TN load ($TN_{Rumsey\text{-}Boundary}$) (kg/day).

Figures 3-11 and 3-12 show the comparison between daily TN load (kg/day) from PFHydro modeling area ($TN_{Modeling\ Area}$) to the daily calibrated TN load ($TN_{Rumsey\text{-}Boundary}$) (kg/day) at Rumsey. The NSE of 0.92 indicates a good match.

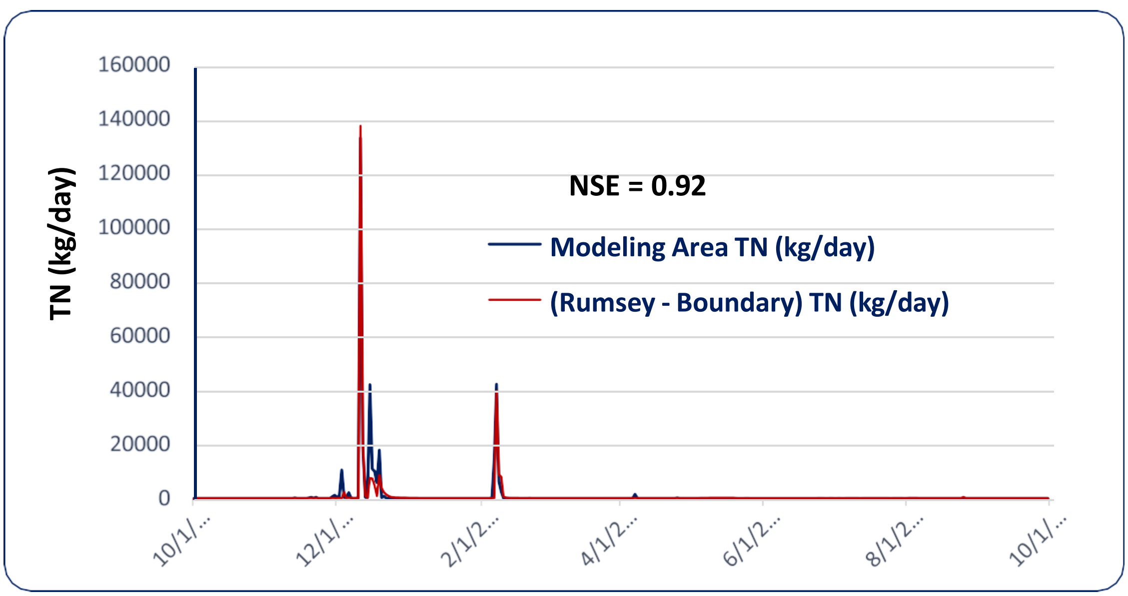

Figure 3-11. Daily time series TN load (kg/day) at Cache Creek Watershed (Rumsey outlet) in WY 2015

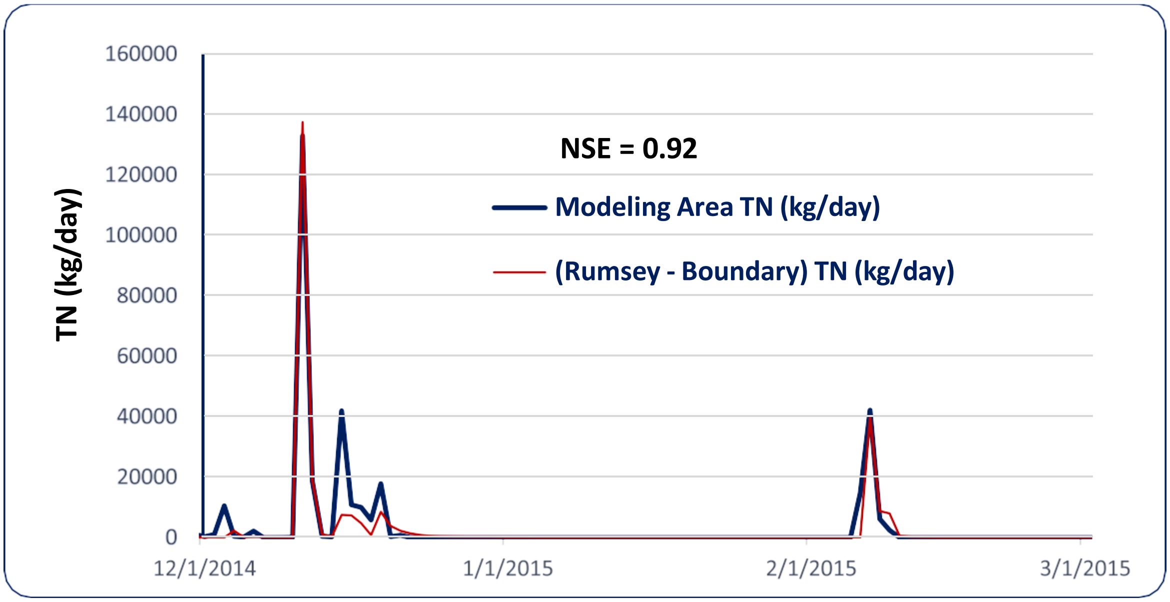

Figure 3-12. Daily time series TN load (kg/day) at Cache Creek Watershed (Rumsey outlet) in WY 2015 — major storms for WY 2015.

The boundary TP concentration is 0.65 g/kg in WY 2015 (See Chapter 2). The average concentration of TP is 0.6 g/kg (Chapter 2, **Table 2-5**) in the modeling area for WY 2015 (pre-fire). The boundary daily TP load (kg/day) for WY 2015 was calculated using Eq. 3-15.

$TP_{Boundary}$ (kg/day) = $SS_{Boundary}$ (tons/day) × 0.65 (g/kg) × 917.185 (kg/tons) /1000 (g/kg) (3-15)

The daily TP (kg/day) in the modeling area (watershed where Rumsey is the outlet) ($TP_{modeling\ area}$) of WY 2015 is calculated using Eq. 3-16.

$TP_{modeling\ area}$ (kg/day) = $SS_{modeling\ area}$ (tons/day) × 0.6 (g/kg) ×917.185 (kg/tons) /1000 (g/kg) (3-16)

Daily TP load (kg/day) at Rumsey (TP_{Rumsey}) WY 2015 was calculated using Eq. 3-3 discussed in section 3-3

TP_{Rumsey} (kg/day) = 0.6507×SS_{Rumsey} (tons/day) -27.121 (kg/day) (3-3)

Where SS_{Rumsey} (tons/day) is daily SS load at Rumsey of WY2015.

Eq. 3-17 was obtained by applying the above results in Eq. 3-7,
$TP_{Modeling\ Area}$ (kg/day) = TP_{Rumsey} (kg/day) - $TP_{Boundary}$ (kg/day)
= $TP_{Rumsey-Boundary}$ (kg/Day) (3-17)

The daily TP load (kg/day) from PFHydro ($TP_{Modeling\ Area}$) is compared to the daily calibrated TP load ($TP_{Rumsey-Boundary}$)(kg/day).

Figures 3-13 and 3-14 show the comparison between daily TP load (kg/day) from PFHydro modeling area ($TP_{Modeling\ Area}$) to the daily calibrated TP load ($TP_{Rumsey-Boundary}$)(kg/day) at Rumsey. The NSE of 0.90 indicates a good match.

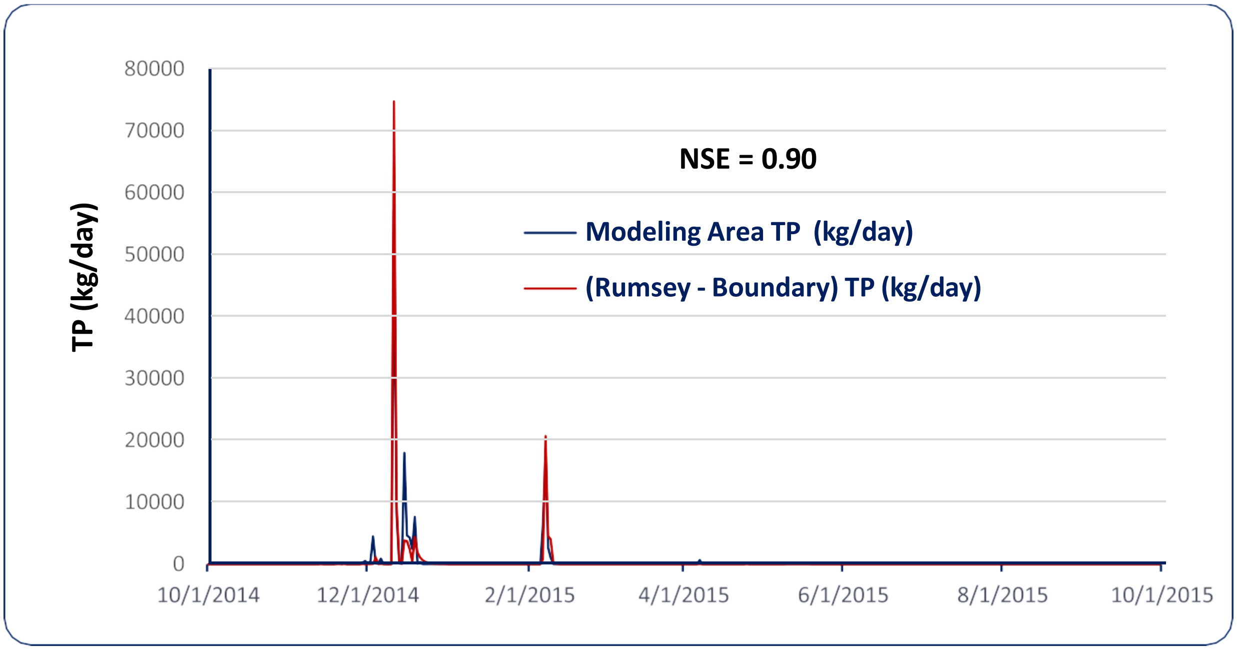

Figure 3-13. Daily time series TP load (kg/day) at Cache Creek Watershed (Rumsey outlet) in WY 2015

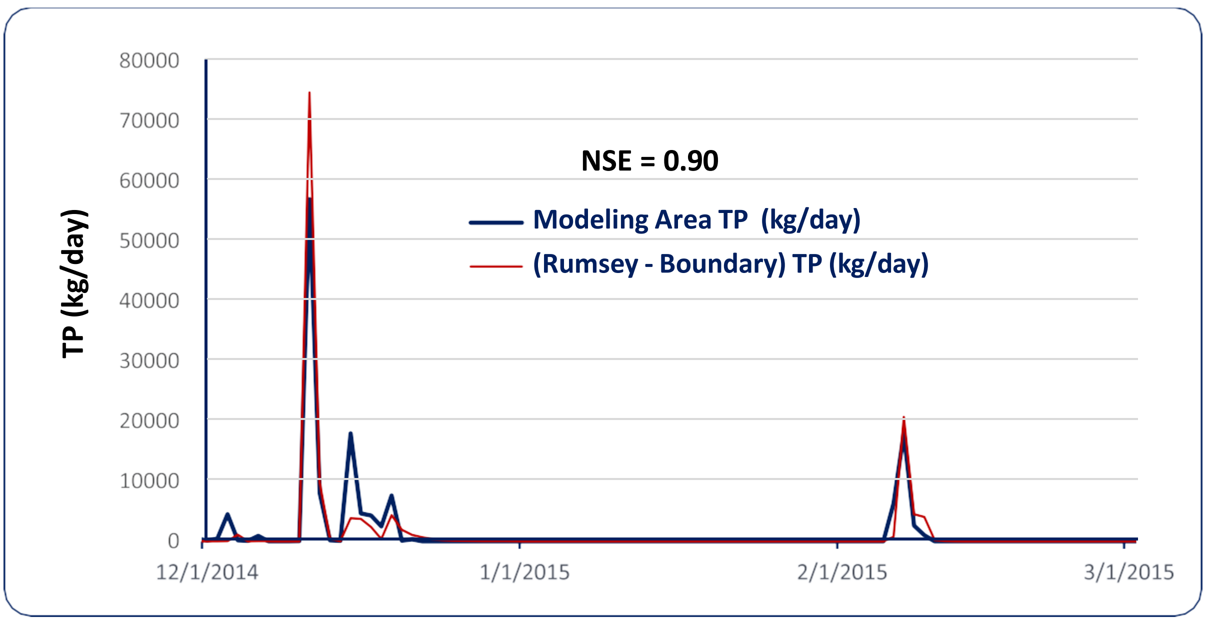

Figure 3-14. Daily time series TP load (kg/day) at Cache Creek Watershed (Rumsey outlet) in WY 2015 — major storms for WY 2015.

3.4.2. Post-fire WY 2017 Total Nitrogen (TN) and Total Phosphorus (TP) Modeling Results

The concentration of TN assumed for the model boundary is 1.4 g/kg in WY 2017 (the same as the TN concentration in WY 2015 because no wildfires happened between WY 2015 and WY 2017 upstream of the modeling watershed area).

The average modeling concentration of TN is calibrated to be 1.841707 g/kg with mixing depth of 0–2 cm (Chapter 2, **Table 2-5**) in the modeling area for post-fire conditions of WY 2017 in the Cache Creek watershed. For storm events, TN concentration of 1.2 g/kg is used because of expected dilution.

The boundary daily TN load (kg/day) of WY 2017 was calculated by using Eq. 3-18.

$$TN_{Boundary}\ (kg/day) = SS_{Boundary}\ (tons/day) \times 1.4\ (g/kg) \times 917.185\ (kg/ton)\ /1000\ (g/kg) \quad (3\text{-}18)$$

The daily TN (kg/day) in the modeling area (watershed where Rumsey is the outlet) ($TN_{modeling\ area}$) for WY 2017 is calculated using Eq. 3-19.

$$TN_{modeling\ area}\ (kg/day) = SS_{modeling\ area}\ (tons/day) \times 1.841707\ (g/kg) \times 917.185\ (kg/ton)/1000\ (g/kg) \quad (3\text{-}19)$$

Daily TN load (kg/day) at Rumsey (TN_{Rumsey}) WY 2017 was calculated using Eq. 3-4, as discussed in section 3-3:

$$TN_{Rumsey}\ (kg/day) = 50.568 \times SS_{Rumsey}\ (tons/day)^{0.6308} \quad (3\text{-}4)$$

Where SS_{Rumsey} (tons/day) is daily SS load at Rumsey of WY 2017.

Eq. 3-20 was obtained by applying the above results in Eq. 3-7,

$$\begin{aligned} TN_{Modeling\ Area}\ (kg/day) &= TN_{Rumsey}\ (kg/day) - TN_{Boundary}\ (kg/day) \\ &= TN_{Rumsey\text{-}Boundary}\ (kg/day) \quad (3\text{-}20) \end{aligned}$$

The daily TN load (kg/day) from PFHydro water quality module ($TN_{Modeling\ Area}$) is compared to the daily calibrated TN load ($TN_{Rumsey\text{-}Boundary}$) (kg/day).

Figures 3-15 and 3-16 show the comparison between daily TN load (kg/day) from PFHydro modeling area ($TN_{Modeling\ Area}$) to the daily calibrated TN load ($TN_{Rumsey\text{-}Boundary}$)(kg/day) at Rumsey. The NSE of 0.78 indicates a good match.

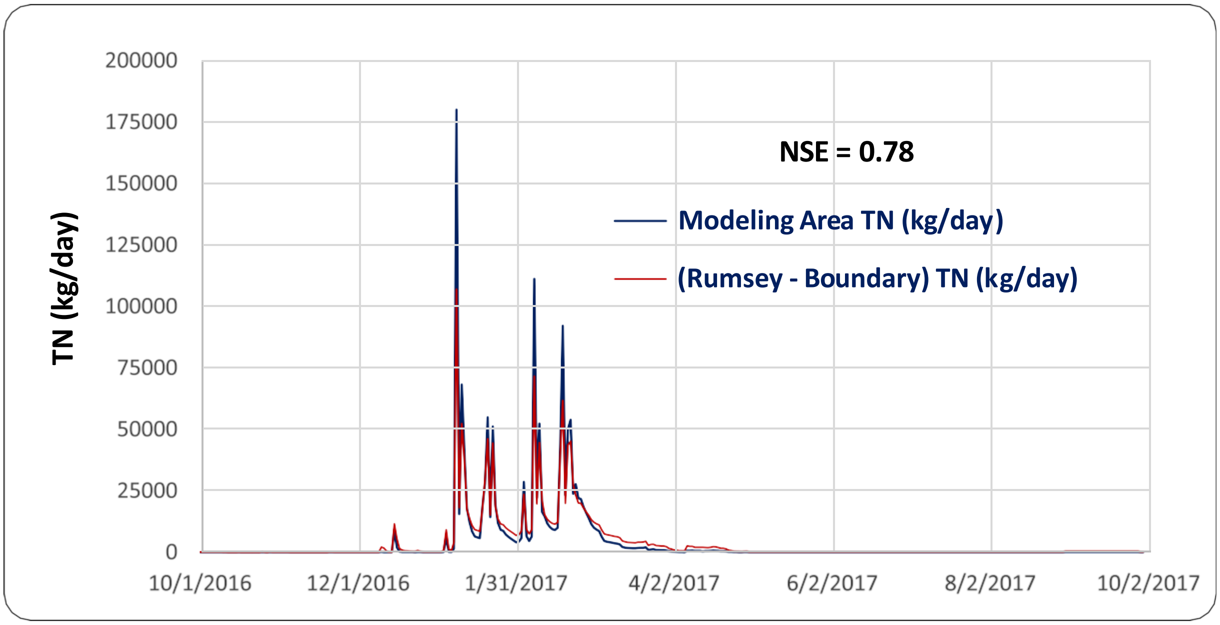

Figure 3-15. Daily time series TN load (kg/day) at Cache Creek Watershed (Rumsey outlet) in WY 2017

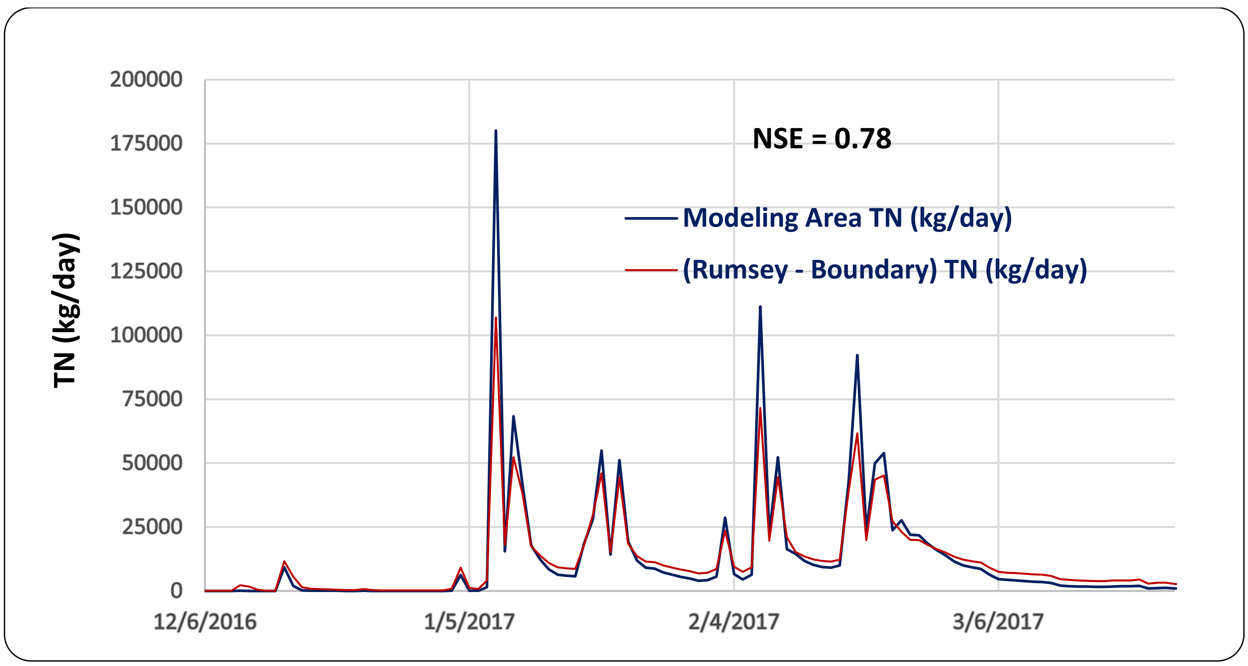

Figure 3-16. Daily time series TN load (kg/day) at Cache Creek Watershed (Rumsey outlet) in WY 2017 — major storms for WY 2017.

The concentration of TP assumed for the model boundary is 0.65 g/kg in WY 2017 (the same as the TP concentration in WY 2015, see Chapter 2).

The average modeling area concentration of TP is calibrated to be 0.626968 g/kg with mixing depth of 0-2 cm (Chapter 2, **Table 2-5**) in the modeling area for the WY 2017 (post-fire).

The boundary daily TP load (kg/day) of WY 2017 was calculated by using Eq. 3-21.

$$TP_{Boundary}\ (kg/day) = SS_{Boundary}\ (tons/day) \times 1.4\ (g/kg) \times 917.185\ (kg/ton)\ /1000\ (g/kg) \quad (3\text{-}21)$$

The daily TP (kg/day) in the modeling area (watershed where Rumsey is the outlet) ($TP_{modeling\ area}$) for WY 2017 is calculated using Eq. 3-22.

$$TP_{modeling\ area}(kg/day) = SS_{modeling\ area}(tons/day) \times 0.626968\ (g/kg) \times 917.185(kg/ton)/1000\ (g/kg) \quad (3\text{-}22)$$

Daily TP load (kg/day) at Rumsey (TP_{Rumsey}) WY 2017 was calculated using Eq. 3-5, as discussed in section 3-3:

$$TP_{Rumsey}\ (kg/day) = 5.5333 \times SS_{Rumsey}\ (tons/day)^{0.795} \quad (3\text{-}5)$$

Where SS_{Rumsey} (tons/day) is daily SS load at Rumsey of WY2017.

Eq. 3-23 was obtained by applying the above results in Eq. 3-7,

$$\begin{aligned} TP_{Modeling\ Area}\ (kg/day) &= TP_{Rumsey}\ (kg/day) - TP_{Boundary}\ (kg/day) \\ &= TP_{Rumsey\text{-}Boundary}\ (kg/day) \end{aligned} \quad (3\text{-}23)$$

The daily TP load (kg/day) from PFHydro water quality module ($TP_{Modeling\ Area}$) is compared to the daily calibrated TP load ($TP_{Rumsey\text{-}Boundary}$) (kg/day).

Figures 3-17 and 3-18 show the comparison between daily TP load (kg/day) from PFHydro modeling area ($TP_{Modeling\ Area}$) to the daily calibrated TP load ($TP_{Rumsey\text{-}Boundary}$)(kg/day) at Rumsey. The NSE of 0.97 indicates a good match.

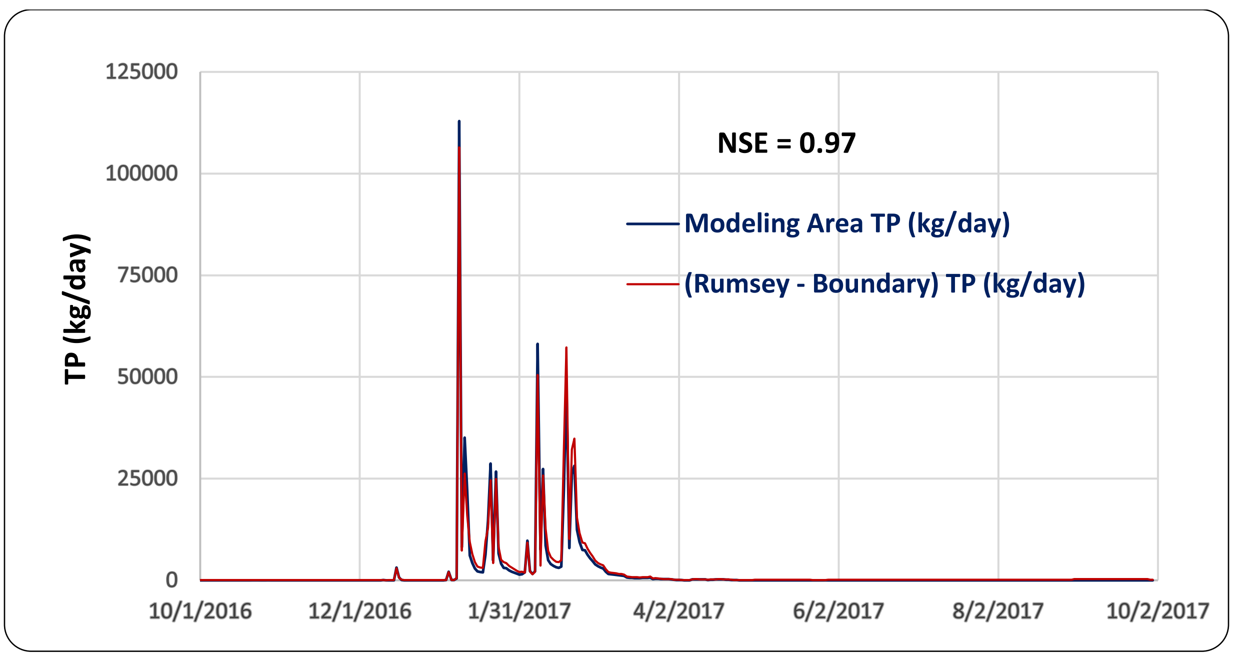

Figure 3-17. Daily time series TP load (kg/day) at Cache Creek Watershed (Rumsey outlet) in WY 2017

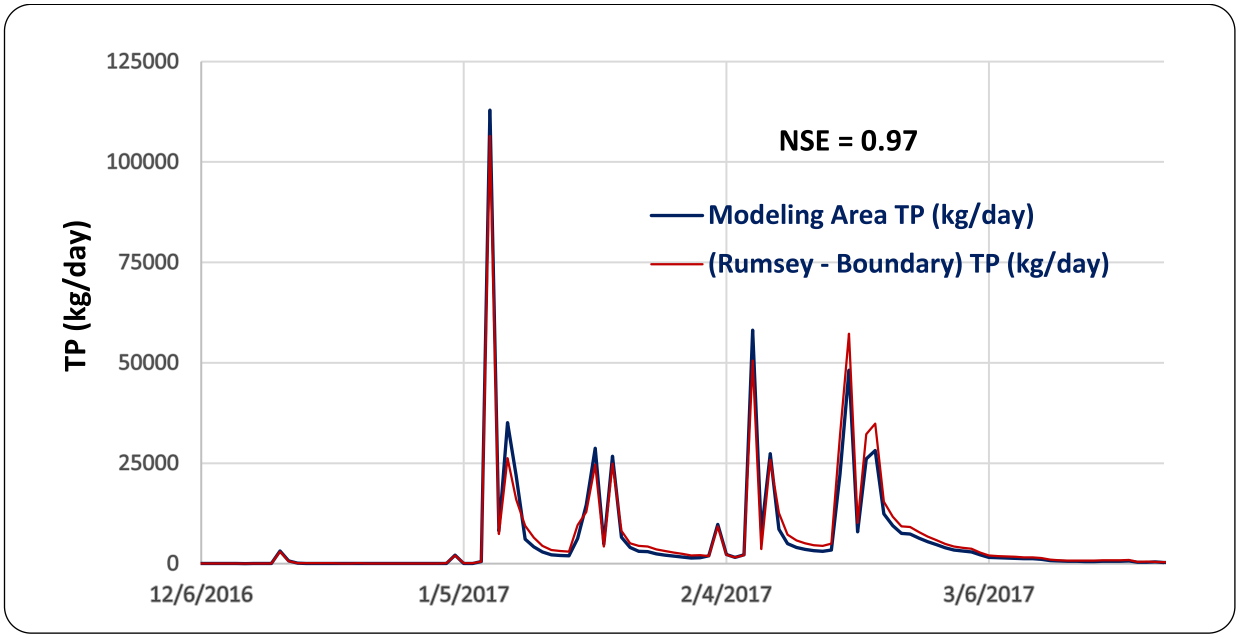

Figure 3-18. Daily time series TP load (kg/day) at Cache Creek Watershed (Rumsey outlet) in WY 2017 - major storms for WY 2017.

Table 3-13. Summary and comparison of annual TN and TP loads between WY 2015 (pre-fire) and WY 2017 (post-fire), for the modeling area and calibrated loads at Rumsey minus boundary inputs

	Modeling Area TN load	(Rumsey - Boundary) TN load	Modeling Area TP load	(Rumsey - Boundary) TP load
Annual load (kg) of WY 2015	324,370	282,838	139,016	136,216
Annual load (kg) of WY 2017	1,467,935	1,485,273	678,595	756,473
Modeling Area load / (Rumsey - Boundary) load of WY 2105	1.15		1.02	
Modeling Area load / (Rumsey - Boundary) load of WY 2107	0.99		0.90	
Load WY 2017/Load of WY 2015	4.5	5.2	4.9	5.6

As can be seen from **Table 3-13**, the annual TN and TP loads from the modeling area is 115% and 102% of the calibrated annual TN and TP loads, respectively, for WY 2015 (pre-fire); the annual TN and TP loads from the modeling area is 99%, 90% of the calibrated annual TN and TP loads for WY 2017 (post-fire), respectively. The annual TN and TP loads for WY 2017 (post-fire) are about 4.5 to 6 times the TN and TP loads of WY 2015 (pre-fire). As mentioned earlier in Section 3-4, for comparison, sediment load for WY 2017 was about 3 times than that for WY 2015, and annual total runoff during WY 2017 was only about 1.5 times than that during WY 2015. One reason for the higher ratios of WY 2017 to WY 2015 nutrient loads compared with sediment loads is that the assumed post-fire TN and TP concentrations are higher than the pre-fire concentrations.

The modeling results indicate that the updated PFHydro water-quality module did a satisfactory simulation based on previous good NSE numbers (focusing on daily values and short-term storm events) as well as the comparison of annual loads between the modeling area and the calibrated loads at Rumsey minus the contributions from the boundary (**Table 3-13**).

Now that we have confidence in the nutrient loads for Cache Creek at Rumsey for WYs 2015-17, the PFHydro water-quality model is ready to be used to simulate fire retardants impacts on downstream nutrients.

3.5. Potential Fire-retardant Impacts on Loads of Total Nitrogen (TN) and Total Phosphorus (TP)

To understand the potential for fire-retardant impacts on loads of TN and TP in watersheds affected by wildfire, we have started with nutrient loads for the modeled area on Cache Creek watershed above Rumsey for WY 2015-17 (as described in Section 3-3) and then added hypothetical amounts of fire retardant based on a range of application rates. As described in Chapter 2, each aerial retardant drop of mono-ammonium phosphate (MAP) from a Type 1 aircraft deposits about 380 kg of TP and 175 kg of TN. (Use of di-ammonium phosphate, DAP, would double this N amount.)

Records from the Rocky Fire indicate that at least seven drops of fire retardant were made during two days in summer 2015. This corresponds to an increase in TP in the watershed by about 2,700 kg and an increase of TN by about 1,200 kg. To evaluate a higher application, it might be assumed that a similar number of drops occurred for the next two days of the fire (this is the point forward progress was stopped). If this was the case, the expected number of drops would be around 20, corresponding to a total watershed load from fire retardants of about 7700 kg TP and about 3500 kg TN.

Table 2-5. Summary of applied retardant within 28,200 ha the modeling boundary. The effects of assumed mixing depth on average soil nutrient concentrations are included. Note the decimal places do not signify the precision of estimated (spurious digits) but were included to give a sense of the calculated effects of fire retardant and show the calculated effects are small (From Chapter 2).

	Post Fire TN (g/kg)	Post Fire TN (g/kg)	Post Fire TN (g/kg)	Post Fire TN (g/kg)	Post Fire TP (g/kg)	Post Fire TP (g/kg)	Post Fire TP (g/kg)	Post Fire TP (g/kg)
Pre-fire	1.4	1.4	1.4	1.4	0.6	0.6	0.6	0.6
Mixing depth (cm)	**0-0.5 cm**	**0-1 cm**	**0-1.5 cm**	**0-2 cm**	**0-0.5 cm**	**0-1 cm**	**0-1.5 cm**	**0-2 cm**
No Retardant	3.16683	2.283415	1.988943	1.841707	0.708824	0.654412	0.636275	0.627206
20 drops	3.167606	2.283803	1.989202	1.841902	0.710571	0.655286	0.636857	0.627643
30 drops	3.168406	2.284203	1.989469	1.842102	0.711871	0.655936	0.63729	0.627968
100 drops	3.172506	2.286253	1.990835	1.843127	0.721371	0.660686	0.640457	0.630343

To assess even more intensive use of fire retardant, we assess scenarios assuming that 20, 30, or 100 drops of fire retardant were applied in the modeling area. Mixing depth was calibrated to be 0–2 cm. The annual total TN loads were calculated using the flowing equations and are shown in **Table 3-14**.

$$\text{Annual } TN_{\text{No retardant}} \text{ load (kg)} = \sum (SS_{\text{modeling area}} \times C_{\text{No retardant}} \times 917.185 / 1000) \quad (3\text{-}24)$$

$$\text{Annual } TN_{\text{20 drops}} \text{ load (kg)} = \sum (SS_{\text{modeling area}} \times C_{\text{20 drops}} \times 917.185 / 1000) \quad (3\text{-}25)$$

$$\text{Annual } TN_{\text{30 drops}} \text{ load (kg)} = \sum (SS_{\text{modeling area}} \times C_{\text{30 drops}} \times 917.185 / 1000) \quad (3\text{-}26)$$

$$\text{Annual } TN_{\text{100 drops}} \text{ load (kg)} = (SS_{\text{modeling area}} \times C_{\text{100 drops}} \times 917.185 / 1000) \quad (3\text{-}27)$$

where $SS_{\text{modeling area}}$ is Daily SS load (kg/day) from the watershed modeling area, $\sum$ is sum from day 1 to day 365;

where $C_{\text{No retardant}}$ is 1.841707, 1.2 (g/kg), for base flow and storm events, respectively;
where $C_{\text{20 drops}}$ is 1.841902, 1.2 (g/kg), for base flow and storm events, respectively;
where $C_{\text{No retardant}}$ is 1.842052, 1.2 (g/kg), for base flow and storm events, respectively;
snd where $C_{\text{No retardant}}$ is 1.843127, 1.2 (g/kg), for base flow and storm events, respectively.

Table 3-14. Comparison of fire-retardant impacts on TN load

	TN_{No} Retardants	$TN_{\text{20 drops}}$ Retardants	$TN_{\text{30 drops}}$ Retardants	$TN_{\text{100 drops}}$ Retardants
Annual TN Load (kg)	1,467,935	1,467,997	1,468,061	1,468,387
TN Retardants/ TN no retardants		1.00004	1.00009	1.00031

It can be seen from **Table 3-14** that the addition of annual TN load is not significant (about 0.031% increase) even with 100 drops of retardants.

The annual total TP loads were calculated using the following equations and are shown in **Table 3- 15**.

$$\text{Annual } TP_{\text{No retardant}} \text{ load (kg)} = \sum(SS_{\text{modeling area}} \times 0.626968 \times 917.185 / 1000) \quad (3\text{-}28)$$

$$\text{Annual } TP_{\text{20 drops}} \text{ load (kg)} = \sum(SS_{\text{modeling area}} \times 0.627668 \times 917.185 / 1000) \quad (3\text{-}29)$$

$$\text{Annual } TP_{\text{30 drops}} \text{ load (kg)} = \sum(SS_{\text{modeling area}} \times 0.627968 \times 917.185 / 1000) \quad (3\text{-}30)$$

$$\text{Annual } TP_{\text{100 drops}} \text{ load (kg)} = \sum(SS_{\text{modeling area}} \times 0.630343 \times 917.185 / 1000) \quad (3\text{-}31)$$

where $SS_{\text{modeling area}}$ is Daily SS load (kg/day) from the watershed modeling area, $\sum$ is sum from day 1 to day 365.

Table 3-15. Comparison of fire-retardant impacts on TP load

	TP_{No} Retardants	$TP_{20\ drops}$ Retardants	$TP_{30\ drops}$ Retardants	$TP_{100\ drops}$ Retardants
Annual TP Load (kg)	679,164	679,922	680,247	682,830
TP Retardants/TP no retardants		1.00112	1.00159	1.00540

It can be seen from **Table 3-15** that the addition of annual TP load is not significant (about 0.54% increase) even with 100 drops of retardants.

The modeling results indicate that annual TN and TP loads would have increased by less than 1% even if 100 aerial drops of fire retardant had been made on the Rocky and Jerusalem fires in 2015.

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4. Summary of Findings, Challenges and Limitations, and Directions for Future Research

In this chapter, the major findings of the project are summarized, along with summaries and synthesis of the first three chapters. Challenges and limitations of the approach are discussed and guidance for future work is provided.

4.1. Major Findings

1. The outcomes of this project, both in the literature review and modeling efforts, clearly demonstrate that that the additive effect of aerial ammonium-phosphate (AP) fire retardant on nutrient loading to downstream waterbodies are small compared to nutrient load increases from burning alone. (Chapters 1, 2, 3)

2. Detailed modeling of the Cache Creek watershed indicates that annual TN and TP loads would have increased by less than 1% even if 100 aerial drops of fire retardant had been made on the Rocky and Jerusalem fires in 2015. (The known number of drops was 7 over a 2-day period, and the likely total was probably less than 20.) (Chapter 3)

3. After wildfire there may be large increases in nutrient loads (N and P) from transport of ash and erosion of bare soil. (Chapters 1, 3)

4. In response to the Rocky and Jerusalem Fires in summer 2015, TN and TP loads increased by 4.5-6 times from WY 2015 to WY 2017 in Cache Creek at Rumsey whereas the sediment loads increased only by ~3 times. Increases in water-borne nutrient concentrations and loads are consistent with the conceptual model that nutrients are added to soil post-fire, and the transported soil particles are enriched in nutrients compared to pre-fire levels.

5. According to a comprehensive literature review, there is no empirical evidence that fire retardant applied to the landscape is transported to streams in runoff. (Chapter 1)

6. For both N and P, there is a possibility that under specific circumstances and in specific watersheds, the added fire retardant could increase the nutrient export, particularly small watershed with low post-fire nutrient exports. (Chapter 1)

4.2. Chapter Summaries

4.2.1. Summary of Chapter 1 — Literature Review

In Chapter 1, the literature review of post-fire nutrient concentrations in soils, spatial nutrient loads, and nutrient export loads provides ranges of likely nutrient characteristics that can be applied to most watersheds. Fuel residue, although being somewhat depleted in pre-fire organic N, can deposit large quantities of N and P to the forest floor (NH_4^+, NO_3^-, organic N, PO_4^{3-}, organic P) causing net increases of nutrient forms that are likely to be transported to the surficial watershed soil. These estimates are highly variable across watersheds with different fuels and antecedent conditions. In the case of P, organic forms are converted to inorganic forms, but no major spatial loss is likely. For both N and P, the processes are quite complex as the cycling of both involves multiple forms, chemical and biological processes, and physical transport processes linked to hydrology and watershed characteristics.

Complexities notwithstanding, the general result of fire residue in burned watersheds is the increased transport of nutrients. Export yields of N can range from 1.2 to 69 kg ha^{-1} in the first year following fire and total P yields can range from 0.25 to 1.7 kg ha^{-1}. For both nutrients, the effect of fire on export decreases greatly each year following the fire and returns to pre-fire conditions within about 3 years, depending on climatic conditions, especially rainfall.

The addition of aerially applied ammonium-phosphate (AP) retardant does result in areas of high nutrient loading, but these are limited to the approximate 0.5 ha drop zone in which about 175 kg N (assuming monoammonium phosphate) and 380 kg P are deposited. The total nutrient addition to the landscape would depend greatly on the fire specifics and the total number of aerial drops made during fire suppression. There are practical limits to the application of retardant, such as how many aircraft participate, how many times per day a drop can be made (distance to refilling), how many days the fire burns, etc. Considering practical limits, there is likely an upper bound to the typical application of retardant and associated spatial nutrient loading.

A review of the literature suggests only a small fraction of nutrients is likely to mobilize in the watershed, further limiting possible effects of the practice. Indeed, to date, no studies examining transport have identified retardant-attributed transport of nutrients in direct overland runoff from the watershed into surface water. The lack of observed transport from lands receiving AP fire retardant is likely due to soil retention, biological uptake, dilution effects, or specific hydrologic conditions that make it unlikely to see transport (e.g., too little rain, too much rain, lack of hydraulic connectivity).

Net increases in nutrients originating form burned lands typically far outweigh the effects of AP fire retardant. For instance, using the stream order classification system (stream order equals the number of upstream drainage confluences), the average 1st order drainage is about 315 ha in area. The range of yield with burning alone is from 0.02 to 68.8 kg ha^{-1} and even 20 drops in this small watershed would not theoretically elevate the total N yield beyond about 20 kg ha^{-1}. It is possible that total nutrient loads could increase in a watershed with low N export and high retardant application, but this would require efficient transport of the retardant from the land into the surface water, something that has not been documented. Phosphorus export is generally lower than that of N and there is greater P mass in retardant, so theoretically P export could be increased after application of a small number of aerial drops in a 1^{st} order watershed, but this too would require a large proportion of the applied P to be transported, which has not been demonstrated to occur.

For both N and P there is a possibility that under specific circumstances and in specific watersheds, the added AP fire retardant could increase the nutrient export, particularly in a small watershed with low post-fire nutrient exports. The potential for increased nutrient export from retardants diminishes greatly with increased watershed area, stream order, and landscape nutrient inputs. The practical result is that fire retardants will not have major, or even discernable, impacts on downstream water quality unless applied directly to a water body or in a small watershed, with an area less than the typical 2^{nd} order stream's watershed (~1800 ha) under the most conservative estimates (conservative assumptions could lead to observations of an effect).

4.2.2. Summary of Chapter 2 — Nutrient Concentrations

To support watershed modeling efforts in the Cache Creek watershed, estimates were made of post- fire nutrient (N and P) concentrations in soils. No data are available for soil N in the Cache Creek model domain, but some data exist adjacent to the domain, lower in the watershed. The median 0–2 cm soil N concentration in adjacent sites was 1.4 g kg^{-1} (1400 ppm). Due to the lack of additional information and uncertainly, an approximate background concentration of 1 g kg^{-1} was used. This assumption provided a good model fit (see Chapter 3). For comparison, observed N concentrations from the Rumsey gage in 2019 (post fire) were obtained by dividing the annual total nitrogen load by the suspended sediment load, resulting in a flow-weighted average annual relationship of 1.2 g-TN kg-$sediment^{-1}$. (Chapter 3). Data from the Rumsey gage shows that during storm flow sediment- normalized TN decreased by 25%, which is attributed to water-column sediment dilution by resuspended bed sediment or erosion of streambanks during high flows.

The effects of fire on spatial nutrient loads are highly dependent on fuel and fire severity. Generally, for a given fire severity class the nutrient export followed the order herbaceous < mixed oak < evergreen < chaparral . Chaparral is predicted to generate the most nutrient export as it is nearly all combusted during fire. Fire can cause ash exports ranging from 28 kg per hectare (herbaceous) to 350 kg N per hectare (chaparral) and 2 kg P per hectare (Herb) to 20 kg P per hectare (chaparral). The spatial nutrient loads were converted to a concentration by assuming a mixing depth and soil density and adding ash nutrient inputs to an assumed background value. Both the mixing depth and density of ash are assumed based on a simple conceptual model that ash falls to the forest floor, and ash particles are integrated into the soil structure. A specific mechanism is not proposed; it is possible precipitation would cause this. The bulk density of soil is unknown, but it is very likely to be between 1.5 g cm^{-3} and 2.0 g cm^{-3}; the higher value was assumed. Based on the assumption of a 2-cm mixing depth, the increase in soil N is estimated to range from 0 g kg^{-1} to 0.8 g kg^{-1}. The effect on total P in soil is about a $4x10^{-2}$ g kg^{-1} increase considering a 2-cm mixing depth. Applying the ash nutrient inputs to the pre-fire soil nutrient concentrations provides post-fire soil nutrient concentrations for the model domain. In the case of nitrogen, the average post-fire total soil concentrations are found to be 1.7 g kg^{-1} TN and 0.62 g kg^{-1} TP.

4.2.3. Summary of Chapter 3 — Watershed Modeling

One major challenge in applying a watershed model to study and evaluate wildfire and AP fire-retardant impacts on the loads of nutrients in downstream waters is the lack of observed nutrient load data for calibrating the model. In the Cache Creek watershed, wanted to leverage a previous model of pre- and post-fire sediment transport using Rumsey as the watershed outlet, but there were no TN and TP load for pre-fire (WY 2015) and post-fire (WY 2017) conditions. This research uses available data for nutrients in Cache Creek at Rumsey and Yolo during WY 2019 and at Road 102 (near Yolo during WY 2011 to estimate daily time series TN and TP loads during WY 2015 and WY 2017 at Rumsey. It was demonstrated that the estimations are reasonable and reliable to be used to calibrate the PFHydro model.

A simplified approach was used to simulate the daily TN and TP loads that were derived from the watershed modeling area. Relationships between daily TN and TP loads with daily SS loads during WY 2019 at Rumsey and during WY 2011 at Road 102 were used to derive TN and TP loads from time series SS loads during WYs 2015 and 2017 at Rumsey. Concentrations of TN and TP in suspended sediment were consistent with observed values from the literature (Chapters 1 and 2) and with observed concentrations of TN and TP normalized by SS at Rumsey (Chapter 3). Modeled TN concentrations on particles were in the range of 1.0 to 1.8 g/kg, with an increase of 0.42 g/kg post-fire in the modeling area. Modeled TP concentrations on particles were in the range of 0.60 to 0.65 g/kg.

The modeling results indicate the above assumptions are reasonable and the updated PFHydro water-quality module did a satisfactory daily TN and TP simulation based on good NSE numbers (focusing on daily values and storm events) as well as the comparison of annual loads between the modeling area and the calibrated loads at Rumsey minus the contributions from the boundary The analyses from the calibrated PFHydro modeling results indicate that annual TN and TP load increased from fire retardants application in the 163 km^2 within the 282 km^2 watershed area will be less than 1.0% even with 100 drops of fire retardant applied.

4.2.4. Synthesis of Chapters 2 and 3 — Theoretical Effects of Fire-retardant Additions to Cache Creek Watershed

To confirm the accuracy of the PFHydro model inputs for nutrient concentrations on sediment particles, a comparison of the N and P loads was made with the outcomes of the LOADEST model (Chapter 3) and the regional soil concentrations (Chapter 2). Agreement between the two approaches for both TN and TP were good, indicating the approach of using the soil concentrations of N and P as an estimate for concentrations of particle-associated N and P. Using the LOADEST results for annual N and P loads, as well as estimated annual sediment loads, the concentrations of particle-associated nutrients (g N L^{-1} water; g P L^{-1} water) were normalized by the suspended sediment concentration (g sediment L^{-1} water) to obtain average N and P concentrations on the sediment particles (g N g^{-1} sediment; g P g^{-1} sediment). At the Rumsey gage, the particle associated P was found to be in near perfect agreement with regional soil P at 600 mg kg^{-1} (or ppm). Nitrogen concentrations were also in fairly close agreement; the estimated particle nitrogen concentration was around 1300 ppm (compared to 1400 ppm, the regional soil median) but decreased to 1000 ppm during storm flow. It was assumed that resuspension of streambed sediment dilutes the N concentration in newly eroded sediments and lowers the effective N concentration on the particles by about 30%. Based on this observation, the PFHydro model applied two N concentrations to account for dilution during post-fire storm flow.

Soil N concentrations are assumed to apply throughout the modeling domain and the greater Cache Creek watershed and are used for boundary concentration on the sediment particles. Similarly, soil P concentrations are used for boundary concentrations on the sediment particles but were adjusted slightly higher to account for an increase in soil P in the watershed.

In the Water Year 2017 (post-fire), TN and TP loads increased by a factor of 5-fold compared to Water Year 2015 (pre-fire). The increase in N and P is attributed to the combination of increased sediment originating in the burn area and increased N and P concentrations on the sediment particles. The predicted increase based on ash inputs (Chapter 2) suggested concentrations of particle-associated N increased to around 1800 ppm, about 30% higher than the assumed pre-fire of 1400 ppm. Particle-associated P concentrations remained relative unchanged. Again, these estimates provide good agreement with the LOADEST results (Chapter 3), providing confidence in the approach to estimating how landscape changes and nutrient inputs influenced downstream loads and particulate associated N and P.

Modeling application results in the upper Cache Creek watershed (Chapter 3), using estimated post- fire soil N and P concentrations (Chapter 2), confirms the conclusions of the literature review (Chapter 1). The theoretical increase in nutrient export due to addition of AP fire retardant is exceedingly small compared to the measured post-fire nutrient loads, indicating that the contribution from fire retardants from the 2015 Rocky Fire and Jerusalem Fire was likely insignificant in this watershed.

The assessment of AP fire-retardant additions to the Cache Creek watershed considers the net effect on annual nutrient export load and assumes that all the retardant applied was mobilized into Cache Creek. Reporting on the 2015 Rocky Fire indicates at least seven aerial drops of AP fire retardant were made, which would have added around 1,200 kg TN and 2,700 kg TP. To evaluate possible effects beyond this, the PFHydro model was run using 20 drops (3,500 kg TN and 7,700 kg TP) and using 30 drops (5,200 kg N and 11,400 kg P) as a "worst case" scenario. The model was also evaluated under the unrealistic scenario that 100 drops (17,272 kg of N and 38,200 kg P) were made. Within the model domain (28,000 ha), the calculated spatial load under the 100-drop scenario remains effectively unchanged for N (less than 0.1 % increase) and minimally changed for P (increase of less than 0.6%).

In comparison with the theoretical yield (Figure 4-1 and Figure 4-2), the modeled transport of fire retardant does not increase the yield by the amount expected had all the retardant entered the watershed. The model uses a distributed retardant load across the model domain, and transport increases are only associated with mobilized sediment.

Figure 4-1 illustrates the maximum additional yield of TN based on the drop scenarios for the Cache Creek watershed. In the case of the TN load at Rumsey, even 100 drops of retardant would contribute only 0.05 kg/ha (normalized over the entire watershed), adding a very small amount to the average yield of 3.6 kg/ha annually.

Figure 4-2 illustrates the maximum additional yield of TP based on the drop scenarios for the Cache Creek watershed. In the case of the TP load at Rumsey, even 100 drops of retardant would contribute < 0.1 kg/ha (normalized over the entire watershed), adding a small amount to the average yield of 2 kg/ha annually.

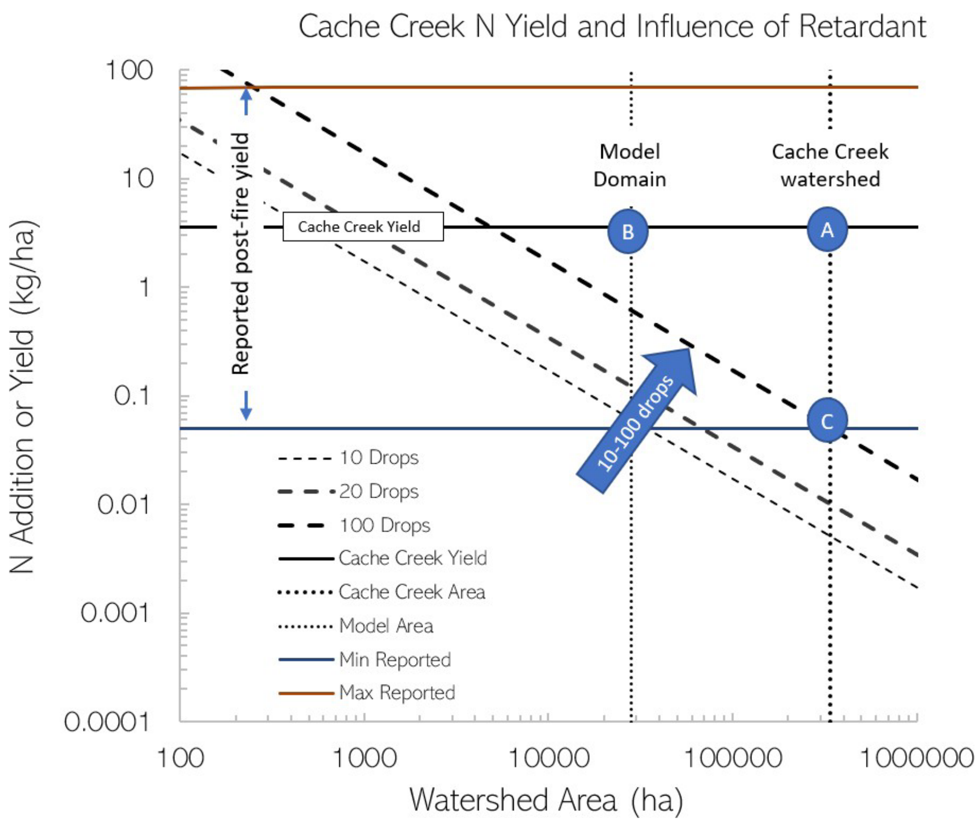

Figure 4-1: The relationship between watershed TN yields and watershed area as a function of aerial drops of ammonium-phosphate (AP) fire retardant are shown. Intersections "A" and "B" show the yield and area of Cache Creek and the modeling area respectively, diagonal lines indicate the Maximum additional yield from retardant for the indicated number of aerial drops. Intersection "C" represents the impact of 100 drops of AP fire retardant on the Cache Creek watershed.

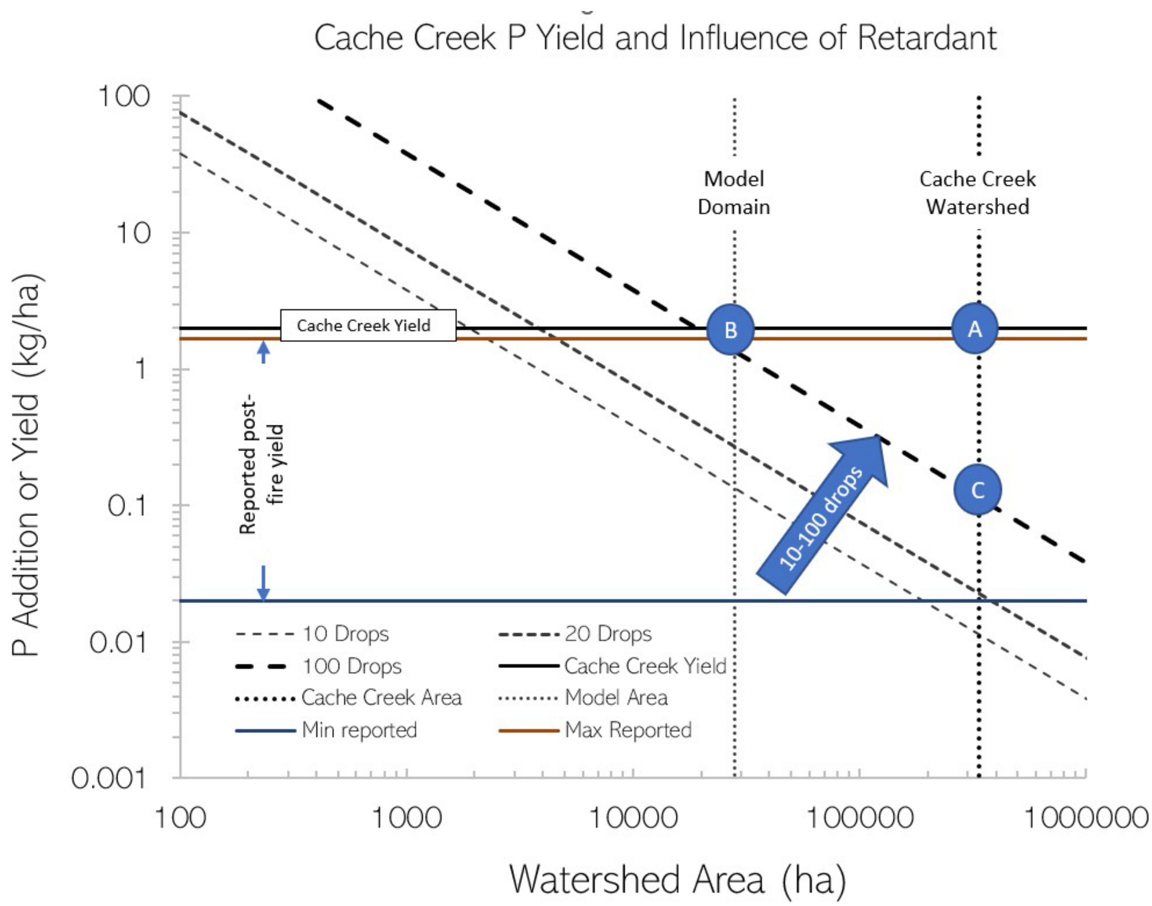

Figure 4-2: The relationship between watershed TP yields and watershed area as a function of aerial drops of ammonium-phosphate fire retardant are shown. Intersections "A" and "B" show the yield and area of Cache Creek and the modeling area, respectively; diagonal lines indicate the maximum additional yield from retardant for the indicated number of aerial drops. Intersection "C" represents the impact of 100 drops of AP fire retardant on the Cache Creek watershed.

4.3. Challenges and Limitations

1. Nutrient cycles are complex, involving several forms of N and P that vary temporally and spatially, as well as with soil, vegetation, climate, and biogeochemical activity. Wildfire disrupts nutrient cycling, compounding the complexity.
2. In most areas, there is a lack of comprehensive data for concentrations of nutrients (N and P) in soil and water. There are few data sets available relating to the same location before and after fire.
3. Data on the use of fire retardants during specific fires is difficult to obtain. Exact usage, drops, aircraft capacity, and specific watershed location information for retardant application does not seem to be available, but is necessary for detailed modeling.
4. Changes in nutrient loading due to burning remains a very difficult component to predict due to its complexity, but the simplified modeling approach developed and applied here provides a clear outcome to the research.

4.4. Directions for Future Work

1. Initiate better record keeping of fire-retardant use in major fires. Data needed includes specific fire retardants used, number and volume of applications, and spatial and temporal information.
2. Better data are needed for N and P concentrations and forms in soils, both pre-fire and post- fire.
3. More data are needed for nutrient and sediment concentrations and loads in watersheds prone to wildfire. Watersheds could be prioritized for monitoring base on probability of burning in the next 20 years as well as potential impacts on downstream water quality including drinking water source and recreation in reservoirs prone to harmful algal blooms that could be triggered by post-fire nutrient inputs.
4. Careful assessment is needed of nutrient mass balances during wildfire. Based on the literature review, obvious discrepancies exist in emissions data for NO_x. Losses of particulate phosphorus and nitrogen to the atmosphere should be examined.

Impacts of wildfire on physical watershed hydrology need better quantification. Effects on water infiltration through ash and fire-affected soils need evaluation. Transfer of nutrients into the subsurface by post-fire infiltration should also be evaluated.