WILDFIRE IN THE WEST: EVALUATING WATER QUALITY AND ECOSYSTEM IMPACTS AND THEIR CONTROLLING FACTORS

by

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ABSTRACT

Wildfires are a natural disturbance that are increasing in size and severity in forested landscapes across the Western United States. Forest fires affect water quality in the disrupted watershed, which can significantly alter the aquatic ecosystem, including sensitive trout (*Salmonidae*) and macroinvertebrate species. However, the type and duration of the water quality impact from fire is unpredictable. Previous studies on individual fires have observed an increase in various forms of nutrients, ions, sediments, and metals in stream water for different post-fire time periods. After some western wildfires, a shift in the macroinvertebrate community has been observed, although this response is not consistent either. Fire can also directly impact fish and lead to local extirpations of fish populations, but this has only been observed in a few rare instances. While wildfires clearly are growing in size and severity, less is known about the type of stream water quality and community impacts that should be anticipated from these growing fires.

This dissertation first explores how water quality and the aquatic community were disrupted by a large Colorado fire. The West Fork Fire Complex consumed 110,000 acres of forest in the state of Colorado during the summer of 2013. The recent fire surrounded the Rio Grande, affecting water quality and habitat critical to insects and fish. The water quality of the Rio Grande (above and below the burn) and some of the effected tributaries was monitored for three years after the fire. All water quality parameters remained the same above and below the fire except turbidity and total suspended solids. Steep, severely burned hillslopes experienced erosion and were the source of sediment loading into the surface water. Despite elevated turbidity levels that persisted for three years in close proximity to the fire, the ecosystem showed resiliency and aquatic macroinvertebrate populations and trout populations have recovered.

From the first study on a single fire, the question remained, what is the typical water quality response after fire? To answer this question, data was compiled for over 24,000 fires across the western United States to evaluate post-fire water quality response. Data from 172 of these fires in 153 burned watersheds were used to identify common water quality response during the first five years after a fire. Within this large dataset, a subset of ten fires was examined further to identify trends in water quality response. Change-point analysis was used to identify

moments in the post-fire water quality data where significant shifts in analyte concentrations occurred. Evidence from this analysis reveals significant increases in nutrient flux (different forms of nitrogen and phosphorus), major-ion flux, and metal concentrations are the most common changes in stream water quality within the first five years after fire. Assembling this unique and extensive data set provided the opportunity to determine the most common post-fire water quality changes in the large and diverse Western U.S.

After looking at this large number of fires, the next research objective emerged, which evaluates the physical determinants driving water quality response and recovery. Through the use of the conditional inference tree technique and Spearman correlation analysis, the drivers of post-fire water quality response were exposed. Results show that the geology of the burn area, which influences the soil characteristics, burn severity, and the rate of vegetation recovery are the key determinants of water quality change and resiliency.

Results from this dissertation expand the scientific knowledge of how wildfires impact surface water quality and the communities that depend on it. Through compiling a large dataset, it was possible to identify common post-fire water quality responses and the watershed features that control them. This research also provides critical information on water quality behavior to land and water managers in anticipation of, and following, a wildfire. Empirical data from this study will also help parameterize future post-fire water quality models to further develop and improve our predictive capabilities.

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CHAPTER 1

INTRODUCTION

While wildfires are a natural disturbance in the Western United States, the frequency and area consumed by fire has been increasing in recent decades (Westerling et al. 2006; Miller et al. 2009). The extent of stand replacing, or high severity, fires in the West has also escalated (Keeley 2009; Miller et al. 2009). Climate change has extended the fire season, through earlier spring snowmelt and warmer, drier spring and summer seasons, wildfires have become a more frequent occurrence in western landscapes (Westerling et al. 2006). Although land use and fire suppression in the West have also changed, climate is the ultimate driver of forest fires. When climate conditions are conducive, fires will burn regardless of land use and fire management (Morgan et al. 2008). Wildfires are expected to continue to grow in the western U.S., disturbing 50-100% more area than previous time periods over the next fifty years (McKenzie et al. 2004; Spracklen et al. 2009; Harvey 2016).

Wildfires are known to impact surface water quality, potentially threatening downstream drinking water supplies (Smith et al. 2011; Bladon et al. 2014). The forested mountain areas of the Western United States receive high quantities of precipitation and are the source of high quality drinking water for millions of customers (Bladon et al. 2014). The quality of the irrigation and drinking water is dependent on forest health. Simultaneously, populations in the western United States are growing, placing greater demand on water supplies. Increasing frequency, size, and severity of wildfires alter water quality in an area where water is already over-allocated and scarce (Bladon et al. 2014).

Wildfires change the landscape, reduce vegetation and expose soil, making soil vulnerable to erosion. Not only is soil exposed after fire, but the soil's infiltration depth, porosity and hydraulic conductivity are reduced (DeBano 2000). As a result, following a fire, precipitation quickly becomes overland flow and post-fire runoff during both low and high flow periods is increased (Kinoshita and Hogue 2015; Saxe et al. *in press*).

In addition to changing water quantity, fire can also impact the water quality in receiving streams (Smith et al. 2011). Exposed soils coupled with amplified runoff lead to increased sediment delivery to fire-impacted streams (Lane *et al.* 2006; Silins *et al.* 2009; Ryan *et al.* 2011; Smith *et al.* 2011). Streams below wildfires often have elevated suspended solids (Silins *et al.*, 2009; Noske *et al.*, 2010; Smith *et al.*, 2011; Dahm *et al.*, 2015), nutrients (Burke *et al.* 2005; Mast and Clow, 2008; Rhoades *et al.*, 2011; Writer *et al.*, 2012) and metal concentrations (Burke et al., 2013; Burton *et al.*, 2016). Typical summer rain events over burn areas displace soils and cause an increase in total suspended solids (Moody et al 2013). The particulate nutrients and metals are associated with the displaced sediments from the fire denuded landscape, while dissolved constituents are delivered to the stream via the surface runoff (Riggan *et al.* 1994; Earl and Blinn, 2003; Bladon *et al.* 2008; Rhoades et al 2011).

After wildfires disturb the landscape, alter the quantity and timing of runoff, and diminish water quality, the compromised water quality can impact sensitive aquatic life inhabiting streams below fires. Wildfires have caused short-term declines in the density and diversity of aquatic macroinvertebrates in receiving streams (Minshall et al. 2001; Arkle et al. 2010). Moreover, some post-fire studies have observed an absence of macroinvertebrate species that are sensitive to water quality and instead there is an increase in pollution tolerant species (Verkaik et al. 2015). Fires can also directly affect fish populations, especially sensitive *Salmonid* species that inhabit the coldwater streams of the Western U.S. Minshall and Brock (1991) observed dead salmonid species immediately after the 1988 Greater Yellowstone National Park fires. The cause of fish mortality after fire however are relatively unknown, and often the impact on the fish population is short-lived, and populations can recover within three years (Minshall 1989; Rieman et al. 1997; Howell 2001; Burton 2005).

Any impact on wild fish populations increases anxiety in communities that depend on the outdoor recreation industry. In many parts of the Western U.S., the recreational fishing industry is a world class, multi-million-dollar industry. In Colorado, over 14,000 people are employed by the recreational fishing industry (BBC Research & Consulting, 2016). Wildfires can cause an un-welcomed interruption in the recreation economy, and the uncertainty around the duration of the impact only increases anxieties.

While several studies have examined wildfire's impact on water quality, macroinvertebrate, and fish populations separately, there is a paucity of studies which have taken an integrated systems approach to post-fire impacts and evaluated the interactions and correlations among water quality macroinvertebrate and fish populations after fire. There are significant gaps in our knowledge of how fire disrupts both the abiotic and biotic functions of an aquatic ecosystem. Furthermore, little is known about ecosystem recovery after wildfire.

The 2013 West Fork Complex Fire, which was a large fire in southwestern Colorado, presented an opportunity to help a community monitor long-term water quality and ecosystem impacts. After the fire, a group of stakeholders and agencies came together to form the Rio Grande Watershed Emergency Action Coordination Team (RWEACT). RWEACT needed assistance and expertise in monitoring water quality along the Rio Grande and its tributaries in the burn area. The large fire provoked concern from landowners, ranchers, anglers and farmers dependent on surface water downstream of the fire.

Despite extensive literature on how fires remove vegetation, alter soil properties, and lead to increased post-fire runoff, it is still unclear what type of water quality impact should be expected after a fire. Prior studies have examined individual fires, revealing how a particular fire impacted the water quality downstream. Yet, sometimes findings from individual fire studies are inconsistent or contradict one another (Smith *et al.*, 2011). Observed water quality impacts from individual fires do not reveal how fires generally alter water quality across the western U.S. There is a need to move beyond individual fire studies and evaluate water quality responses over a broader area.

Some individual fire studies have identified a few key characteristics of the fire itself that are correlated to water quality response, specifically: fire size, severity, and proximity to urban areas (Riggan *et al.,* 1994; Townsend and Douglas, 2004; Arkle *et al.*, 2010; Rhoades *et al.*, 2011; Burke *et al*, 2013). These few studies used correlation to identify variables that were associated with water quality response after individual fires. However, a broader view of the drivers of water quality response is still lacking. It is unclear what type of watershed-scale physical and climatological variables determine the type and magnitude of post-fire water quality response.

The primary goal of this dissertation is to improve understanding of how wildfires effect water quality and stream ecosystems in the Western United States and what controls the magnitude and direction of water quality change. The next three chapters of the dissertation evaluate wildfire impacts on receiving waters, where the following science questions are addressed: 1) In what ways did the West Fork Complex fire impact the aquatic ecosystem of the Rio Grande? How did the fire impact water quality, insects and fish? How quickly do water quality, insect and fish populations recover? 2) What type of water quality responses are the most common after forest fires in the Western United States? 3) What physical or climatological characteristics control water quality response after fire? Are some watersheds more prone to recover quickly, where water quality returns to baseline while others are more likely to show impacts from the forest fire for a longer period?

CHAPTER 2

WATER QUALITY IMPACTS FROM THE WEST FORK COMPLEX FIRE: MONITORING ECOSYSTEM HEALTH AND RECOVERY IN THE UPPER RIO GRANDE

Modified from a manuscript in preparation for publication
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Abstract

Forest fires affect water quality in disrupted watersheds, which can significantly impact aquatic ecosystems including sensitive trout (*Salmonidae*) and macroinvertebrate species. The West Fork Complex (WFC) fire consumed 44,360 hectares of forest in the state of Colorado during the summer of 2013. Damage to the soils was of moderate to high severity in the majority of the area (60%). The fire surrounded the Rio Grande, affecting water quality and habitat critical to insects and fish. The current research investigated the impacts of the WFC on the river's ecosystem, and included monitoring the water quality in the mainstem of the Rio Grande (above and below the burn) as well as several effected tributaries for three years after the fire. Parameters important to the survival of aquatic life, such as flow, temperature, dissolved oxygen, pH, conductivity, total dissolved solids, and turbidity were measured using deployed sensors transmitting real-time data for water managers to use. Other water quality characteristics such as metals, nutrients, and total suspended solids were sampled and analyzed bi-monthly. Macroinvertebrate and fish populations were sampled annually at the same water quality monitoring sites in the Rio Grande and tributaries. Precipitation and flow data revealed monsoon rain events delivered sediments into the Rio Grande and its tributaries from steep, severely burned hillslopes. The monsoon events caused acute and dramatic fish kills, where hundreds of trout were reported killed in one tributary in a single day event. Turbidity was observed as high

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as 505 NTU in the impacted stream with fish kill, whereas the turbidity was 25 NTU in a neighboring tributary outside of the burn area. Salmonids can be negatively impacted by relatively low turbidity, with prior studies noting that the turbidity threshold for rainbow trout is 50 NTU. Despite elevated turbidity levels that persisted after three years in severely burned areas in close proximity to the fire, the ecosystem appears to be recovered. Results indicate a resilient ecosystem where insects and fish populations impacted by the fire are similar to pre-fire levels within three years. Results of this research can further inform western land and water resource managers to better manage fire-resilient ecosystems. Empirical data from this study will also help parameterize post-fire water quality models to further develop and improve our predictive capabilities.

2.1 Introduction

Forest fires burned more area in the last three decades than in any previous time (Miller 2009). In the 2017 season, wildfire consumed 44% more area in the Western United States than the average ten years ago (National Interagency Coordination Center, 2017). This is partly due to climate change, which contributes to increasing wildfire frequency and size (Westerling et al. 2006; Morgan et al. 2008; Westerling et al. 2011). Wildfire seasons are longer with warmer spring temperatures and earlier snowmelt leading to inevitable fires (Westerling et al. 2006; Running, 2006). Warmer, drier conditions will continue to drive an increase in fire activity across the globe (Pechony and Shindell, 2010). Climate models predict continued warming in the Western United States, leading to a predicted 200% increase in total area burned in the Southwestern U.S. (Spracklen et al. 2009). Forest fires change vegetation, alter soils, and affect water quality in disrupted watersheds (DeBano 2000; Smith et al., 2011; Burke et al. 2013), which can devastate the aquatic ecosystem including sensitive aquatic macroinvertebrate species and trout (*Salmonidae*; Minshall 1989 and 2001; Howell 2001; Burton 2005).

Wildfires change the landscape, removing vegetation and altering soils, leading to an increase in erosion and sediment delivery to aquatic systems. A fire vaporizes hydrophobic substances (pyrolysis) in the soil and leaf litter causing the formation of a water repellent layer just below the soil surface (DeBano, 2000). This hydrophobic soil layer changes the soil structure, reducing hydraulic conductivity and infiltration depth (DeBano 2000). Fires are classified by their severity; a high severity fire completely consumes vegetation and organic matter on the surface (Keeley 2009). Soil-water repellency is the most severe when areas experience high burn severity and temperatures reach 175-200° C (DeBano et al. 1981).

Between the lack of leaf interception, increased imperviousness due to the hydrophobic soil layer, and reduced hydraulic conductivity from the heat of the fire, the moisture storage in burned watersheds is temporarily diminished. After fire, precipitation becomes runoff quickly as soils are rapidly saturated due to low retention (Doerr et al. 2000; Murphy et al., 2012). A range of studies have documented the change in a watershed's hydrology after fire, where annual discharge, low flows, peak discharge, and storm runoff all increased in burned watersheds (Lane et al. 2006; Kinoshita and Hogue 2011 and 2015; Coombs et al. 2013; Saxe et al. in press).

Bare soils coupled with increased runoff can lead to elevated sediment loading in fire-impacted streams (Lane *et al.* 2006; Silins *et al.* 2009; Ryan *et al.* 2011; Smith *et al.* 20011). Rain physically displaces more sediment than snowmelt runoff (Moody et al 2013). Fine sediments erode more easily by rain events and are delivered to streams while large coarse soil material is left behind (Shakesby et al. 2006). Two Colorado fire studies found that post-fire erosion was most substantial during high-intensity short duration storms (Murphy et al. 2012, Benavides-Solorio and MacDonald, 2005). Another Colorado fire study observed peak turbidity events occurred during the summer season due to convective storm events (Rhoades et al. 2011). High magnitude low-frequency storm events can even lead to shallow landslides (Shakesby et al. 2006). Morris et al., 2012 observed that a 1 in 5-year rain event was enough to remove sediment and cause surface changes of greater than 1 m depth (Morris et al. 2012). Sediment loading is a function of a fire's severity and landscape topography; high burn severity sites yield more sediment than moderate or very high severity sites. After Colorado's Hayman Fire, Rhoades et al. (2011) observed that the stream turbidity from hillside erosion increased linearly with the amount of a basin that burned at high severity. The degree of hillslope also influences the amount of erosion that occurs; whereby erosion will be more severe on a steeper slope (Shakesby et al. 2006, Morris et al. 2012).

Wildfires can also increase the concentration and loading of nutrients and metals in stream water. It is well documented that wildfire can lead to an increase in total particulate phosphorus, total nitrogen, and dissolved nutrients, such as nitrate and phosphate in receiving waters (Riggan et al. 1994; Burke et al. 2005; Mast and Clow 2008; Noske et al. 2010; Rhoades et al. 2011; Writer et al. 2012; Stein et al. 2012; Burke et al. 2013; Coombs and Malack 2013; Emelko et al. 2016). Although the duration of the water quality impact varies, where some have observed water quality returning to baseline conditions within a year after the fire (Earl and Blinn 2003) and others have documented elevated nutrient concentrations in streams for five to seven years after fire (Rhoades et al. 2011; Emelko et al. 2016). A few studies on fires near urban centers detected increases in total and dissolved metal concentrations in stream water following storm events (Burke et al. 2013; Burton et al. 2016; Pinedo-Gonzalez et al. 2016). An increase in sediments, nutrients, and metal concentrations can cause a shift in the aquatic ecosystem and are a concern for water treatment and water providers (Bladon et al. 2014; Emelko et al. 2016).

Forest fires indirectly increase stream temperature through removal of shading in riparian corridors, which can negatively affect ectotherms such as fish (Dunham et al. 2007; Isaak et al. 2010; Beakes et al. 2014). Forests disturbed by fire may cause stream temperatures to double or even triple compared to basin averages, causing a dramatic shift in thermal regimes. Summer weekly average temperature and daily maxima increase (Brock et al. 1989; Isaak et al. 2010; Beakes et al. 2014) and can remain elevated for up to ten years following a fire (Dunham et al. 2007).

Aquatic macroinvertebrate communities are also impacted by wildfires. Direct and immediate effects from fire on macroinvertebrate species can be minor and undetectable (Minshall 2003). But in some cases, community composition shifts can last for five to ten years after fire (Minshall et al. 2001). Impacts on aquatic macroinvertebrates can be a result of increased flow displacing insects, removal of riparian vegetation which are sources of their allochthonous food, and sediment transport and deposition burying insects (Gresswell 1999; Minshall 2003, Arkle et al. 2010). Arkle et al. (2010) documented changes in macroinvertebrate densities in burned catchments; in severely burned areas, they observed a shift in taxa toward sediment tolerant species. Verkaik et al. (2015) compared macroinvertebrate response after fire in three regions, Idaho, USA, Victoria, Australia, and northeastern Spain. This international study found that a decrease in macroinvertebrate species diversity, overall increase in macroinvertebrate abundance and an increase in pollution tolerant species was common among all three regions. The duration of the impact varied and recovery shortly after the fire was also common (Verkaik et al. 2015). Direct and indirect impacts from wildfire on macroinvertebrate populations tend to be short-term and recovery is expected (Minshall 2003).

Fish are indicators of stream health, and *Salmonid* species are especially sensitive to water quality (Fausch, 1990). Forest fires can have direct effects on fish populations, Minshall and Brock (1991) observed dead salmonid species immediately after the 1988 Greater Yellowstone National Park fires. The causes of fish mortality after fire however are relatively unknown, speculation ranges from increased water temperature, toxicity of the smoke and ash during the fire, poor water quality, change in hydrology, and habitat alteration (Minshall 1989; Howell 2001; Burton 2005). Rieman et al. (1997) witnessed extirpation of some salmonid species immediately after fire then observed fish populations rebound within three years, where densities were greater in burned areas than unburned areas. If the stream is well connected and has diverse habitat, *Salmonid* populations can recolonize and rebound within a year following a severe fire (Gresswell 1999; Burton 2005; Lyon and O'Connor 2008). Rieman et al. (2010) stated that in large well-connected fish populations, it may not be possible to control fire-the alternative is to maintain conditions that increase resilience. Fish can return to burn impacted streams and show great resiliency, but this has only been observed in a few studies (Rieman et al. 1997; Burton, 2005). In areas where fish populations and recreational fishing are important to the local economy, the question looms: how will wildfire impact fish populations and for how long.

Previous studies have generally focused on singular effects of wildfire on aquatic systems; where some studies analyze the impacts on water quality while another evaluates the impacts on macroinvertebrates or fish. In this research, we investigate wildfire impacts on water quality, macroinvertebrate, and fish populations in streams directly below the second largest forest fire in Colorado's history. Our goal is to holistically evaluate a wildfire's impacts on an aquatic ecosystem. Data was collected over a 3-year period on all aspects of water quality: sediment, nutrients, metals, dissolved oxygen, and temperature above and below a large wildfire. We also collected macroinvertebrate samples above and below the fire and compared results to pre-fire population data. Finally, we included fish population monitoring in a small fire-impacted tributary and the mainstem of the river below the fire.

2.2 Background and Study Site

The West Fork Complex (WFC) fire consumed 44,360 hectares (444 km^2) in Colorado during the summer of 2013. It was the second largest wildfire in the state's history. The complex of three fires in the upper Rio Grande watershed, located in Southwestern Colorado, were ignited naturally by lightning strikes (Figure 2.1). The majority (88%) of the burn area was comprised of Engelmann spruce (*Picea engelmanii*) trees killed previously by Spruce Beetle (*Ips spp.*). Damage to the soils was of moderate to high severity in the majority of the area (60%; USDA 2013). The Rio Grande provides excellent recreational fishing opportunities from the reservoirs at the headwaters through the miles of Gold Medal Stream as designated by the Colorado Parks and Wildlife (CPW) and supports an abundance of trophy trout (Colo. Wildlife Commission, 2008). Many local fishing businesses take advantage of the extraordinary large stonefly (*Pteronarcydae)* that hatches in early summer, feeding the large trout and creating excellent fishing opportunities for tourists.

2.3 Methods

Streamflow, physical and chemical water quality, macroinvertebrate and fish populations were monitored for three years after the WFC fire employing a before-after-control-impact (BACI) study design at ten sites. A total of 10 sites were monitored during this study. Five study sites were chosen in the Upper Rio Grande, a sixth site was in the South Fork of the Rio Grande and four additional study sites were selected in fire-impacted tributaries of the Upper Rio Grande (Figure 1, Table 2.1). One site, at Thirty Mile Campground, was upstream of the fire and served as the control site for the Rio Grande. Five impact sites were below the fire in the Upper Rio Grande and the South Fork of the Rio Grande. Additionally, four sites were selected in tributaries where sites were paired as control-impact streams, where one tributary in each pair was within the burn area (impact) and one tributary in each pair was in a sub-watershed outside of the fire (control) (Table 2.1).

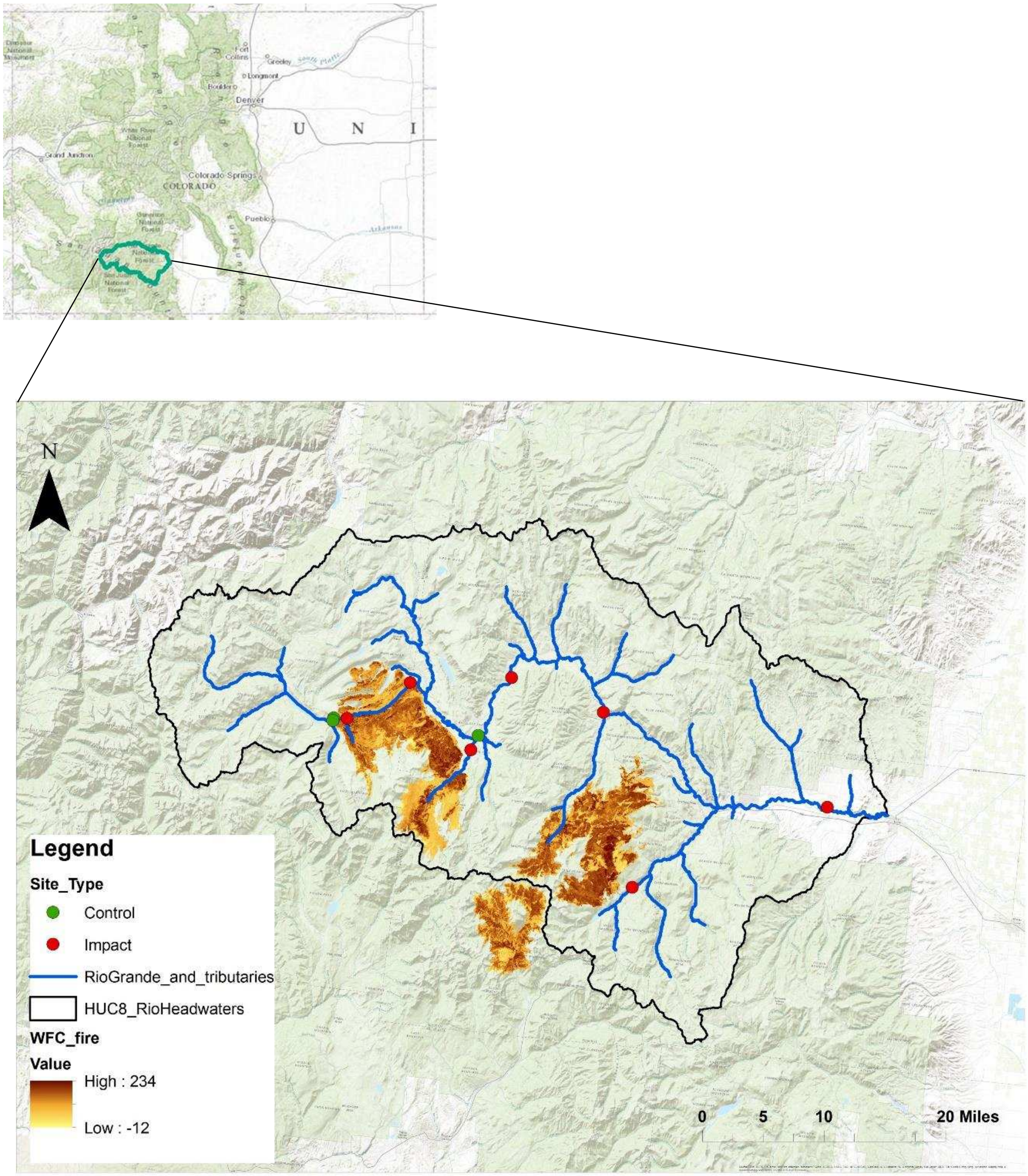


Figure 2.1. The Upper Rio Grande watershed in southwestern Colorado. Colorado is shown above for reference where the watershed is highlighted in green. The West Fork Complex fire shown here, burned 110,000 acres of this steep headwaters watershed.

2.3.1 Streamflow

Streamflow was monitored at each of the ten sites. The Colorado Division of Water Resources (CDWR) operated stream gauges at the six sites on the Upper Rio Grande and South Fork of the Rio Grande. Each gauge recorded stage height which was used to build a rating curve

Table 2.1. Water quality, macroinvertebrate, and fish sampling locations above and below the West Fork Complex Fire.

Site Name	Control/Impact	River	Distance from Fire (km)	Watershed Area (km²)	Area of Watershed Burned (km²)	Percent of Watershed Burned	Percent of Fire Moderate and High Burn Severity	Years of Sampling
Thirty Mile	Control	Rio Grande	0.2	419	0	0	0	2014-2016
Box Canyon	Impact	Rio Grande	0	552	26	2	68	2014-2016
Marshall Park	Impact	Rio Grande	11.13	1,436	63	4	65	2014-2016
Wagon Wheel Gap	Impact	Rio Grande	9.28	1,986	63	3	65	2014-2016
Del Norte	Impact	Rio Grande	22.7	3,575	108	3	2	2014-2016
South Fork at Columbine	Impact	South Fork of Rio Grande	2.28	174	12	7	5	2014-2016
Squaw	Control	Squaw Creek	0.07	55	0	0	0	2015-2016
Little Squaw	Impact	Little Squaw Creek	0	47	7	15	63	2015-2016
Red Mountain Creek	Control	Red Mountain Creek	3	82	0	0	0	2015-2016
Trout Creek	Impact	Trout Creek	1.2	101	17	17	57	2015-2016

to correlate height to flow. To build the rating curve the CDWR took multiple stream flow measurements in the field and recorded height. In the four tributary sites, we installed Hobo Water Level Loggers in stilling wells to record pressure. At each site one Hobo was installed outside of the stream to measure air pressure, to correct water pressure recordings for local air pressure. Each Hobo was deployed from April to October (2014-2016), checked twice per month, where stream height was recorded at the stilling well and streamflow was measured in the field following the US Geological Survey (USGS) mechanical current meter method (Buchanan and Somers, 1969). Recorded pressure levels were correlated to field measurements of flow and stage height to build rating curves.

2.3.2 Water Quality

High resolution temporal water quality monitoring was conducted at each of the ten sites. We deployed a Hydrolab MS5 multiparameter water quality sonde at each site to record stream temperature, pH, specific conductivity, total dissolved solids, and dissolved oxygen. In 2015, two years after the fire, a turbidity sensor was added to each sonde. The multiprobes were installed in stilling wells in conjunction with the Hobo Water Level Loggers from May to October 2014-2016. Data were recorded continuously every 15 minutes at five of the sites where the multiprobes were powered by the CDWR stream gauge. At the other five sites, where the probes were battery powered, data were recorded every three hours. The sondes were checked for debris, cleaned, calibrated, and data downloaded every two weeks. At the end of each season, data recorded by sondes were quality controlled. Upon each site visit, grab samples were collected to later analyze for total suspended solids. Total suspended solids samples were analyzed in our lab following the US Environmental Protection Agency (USEPA) ESS Method 340.2, Total Suspended Solids, Mass Balance methodology (USEPA 1993). Also, upon each site visit every two weeks, a Hach Colorimeter DR 900 was employed at each site to test nitrate, nitrite, and phosphorus (as orthophosphate). The field colorimetric methods used were: Nitrate Cadmium reduction method for low range nitrate, Nitrite USEPA Diazotization Method, Phosphorus USEPA PhosVer3 (Ascorbic Acid) Method (USEPA 1979).

Dissolved and total metals data were collected twice per year during high and low flow events. Dissolved metals samples were collected using a 0.45 µM filter and syringe. Both filtered and unfiltered samples were preserved with 1M nitric acid to a pH below 3 pH units and kept refrigerated until analyzed by inductively coupled plasma mass spectrometry (ICP MS). Deionized water (blank) samples and duplicate field samples were included for every ten samples collected for quality assurance and quality control. High and low flow metals data that were collected by the local mining company, Rio Grande Silver, in some of the same sampling locations provided pre-fire data and were compared to our three years of post-fire sampling.

To determine if there were any detectable changes in analyte concentrations below the fire, non-parametric statistical techniques were used, thereby not assuming normal distribution of the water quality data. Assumptions for the Mann-Whitney *U* test were met (Tamhane and Dunlop, 2000) and the statistical test was performed for each physical and chemical water quality parameter, comparing the difference in median values (or distribution) of concentration above and below the fire. The Mann-Whitney *U* test produces a *U* value and the probabilities of producing a *U* value from one population (above the fire, or control) larger than the other population (below the fire, or impact) have been tabulated and provided as a p-value (Mann and Whitney, 1947). If the p-value is less than the alpha value ($\alpha = 0.05$), a significant difference is detected and the null hypothesis is rejected. Here, the null hypothesis was that the distribution of above- and below-fire analyte concentrations were equivalent, with a difference of zero. The Mann-Whitney *U* test is a rank sum test, reducing the impact of outliers (Helsel and Hirsch, 1992). A non-parametric multiple comparison test, the Kruskal-Wallis test, was conducted to compare the medians of physical and chemical water quality values among all sites, and an approximate p-value for the large sample is calculated. The null hypothesis of the Kruskal-Wallis test states that all the medians among groups are identical (Helsel and Hirsch, 1992).

2.3.3 Macroinvertebrates

In addition to monitoring streamflow and water quality, macroinvertebrate populations were sampled at each of the ten sites once per year, in the first week of August from 2014-2016. The state of Colorado had collected macroinvertebrate data prior to the fire from many of the same sampling sites. To identify wildfire impacts on macroinvertebrate populations, we compared our data collected after the fire to historic data. We followed the Colorado Department of Public Health and Environment (CDPHE) macroinvertebrate sampling protocols and used the same lab for insect identification, and followed the same lab procedures to calculate indices (CDPHE Policy 10-1, Appendix D). Insect collection was done with a semi-quantitative modified kicknet in a single randomly selected location at each water quality monitoring site. The entire sample was preserved in 95% ethyl-alcohol and sent to a lab for identification and metric calculation. Three indices were calculated from the macroinvertebrate samples.

The first metric used in the analysis is the Multi Metric Index (MMI), a metric that scores the macroinvertebrate population diversity and density on a scale from 0-100 where the state has a set threshold of 50 below which water quality is considered to be impaired (CDPHE Policy 10-1). Additional indices calculated from macroinvertebrate samples were the Shannon-Wiener Diversity Index (SDI) which is another metric calculated from lab analysis of relative species abundance where values range from zero to five, a higher value indicates higher species diversity (MacArthur 1965). The third metric calculated from macroinvertebrate sample analysis is the Hilsenhoff Biotic Index (HBI) which reveals the relative abundance of pollution tolerant species, values range from zero to ten, where a higher value indicates more pollution tolerant species are present (Hilsenhoff 1987). The state of Colorado has set water quality standards using the SDI and the HBI, where a Type 1 (based on elevation and temperature) site with a SDI greater than 2.4 and HBI less than 5.4 is in attainment and a SDI value of 3.0 or higher and HBI of 5.1 or less for Type 2 sites is in attainment (CDPHE Policy 10-1). The sampling locations in this study are a combination of Type 1 and Type 2 sites. Calculated metrics from samples taken after the fire are compared to index values before the fire and all calculated metrics are compared to state standards.

2.3.4 Fish Populations

Fish populations were monitored by the Colorado Parks and Wildlife before and after the fire in one location impacted by the WFC fire on the Upper Rio Grande. After observing a fish kill in one of the fire-impacted tributaries, which was a result of a landslide occurring after a monsoon rain event, in the first year after the fire, the fish populations above and below the landslide were also monitored. In both the mainstem of the Rio Grande and in the tributary, fish populations were sampled by electrofishing. On the Upper Rio Grande, a boat mounted electroshocker was pulled through the sampling segment and as fish were stunned they were collected and kept in a live well until they could be weighed, measured, and marked. All fish are given a temporary mark in the first effort and one week later the same segment is sampled and the number of fish caught with and without marks is recorded. German brown trout (*Salmo trutta*) population was estimated following the mark-recapture technique using the Petersen equation:

$$N = \frac{(M + 1)\ (C + 1)}{R + 1} \qquad \text{(Eq. 2.1)}$$

Where, C = total number of fish caught in second sample (including recaptures), M = number of fish caught, marked and released in the first sample, N = population estimate, R = number of recaptures in the second sample (fish marked and released in the first sample) (Lockwood and Schneider 2000). The assumptions made for this method are as follows: marked and unmarked fish have the same mortality rates, are equally vulnerable to capture, marks are retained during the one week sampling period, marked fish mix with unmarked fish, and there is negligible immigration or emigration during the one week recapture period.

A fish kill occurred one year after the fire. Monitoring was subsequently initiated at this fire-impacted tributary. The tributary, Trout Creek, where the fish kill is believed to have occurred because of a landslide in the burn scar. Population estimates above the landslide (control site) are compared to the fish population below the landslide (impact site). The trout populations above and below the landslide were estimated the first and second years after the landslide following the depletion method, also known as the Zippin method (Zippin 1958, Lockwood and Schneider 2000). For this technique, the segment of interest is sampled twice in one day, where all fish stunned and captured in the first pass in the segment are kept in a live well in the stream while a second pass is made through the segment and any remaining fish are captured. Assumptions made for this method are similar: emigration and immigration must be negligible, all fish are equally vulnerable to capture, vulnerability to capture is the same in each pass, water conditions (water clarity) must remain constant between the two passes (Lockwood and Schneider 2000). The population is then estimated using the Seber and Le Cren (1967) method:

$$P = \frac{C_1 - C_2}{C_1} \qquad \text{(Eq. 2.2)}$$

$$N = \frac{C_1^2}{(C_1 - C_2)} \qquad \text{(Eq. 2.3)}$$

$$\text{Variance of } N = \frac{C_1^2 C_2^2 (C_1 + C_2)}{(C_1 - C_2)^4} \qquad \text{(Eq. 2.4)}$$

Where, C_1 is the number of fish removed in the first pass, C_2 is the number of fish removed in the second pass, N is the population estimate, and P is the probability of capture.

2.4 Results

2.4.1 Water Quality

The high resolution water quality monitoring allowed us to identify water quality impacts as they occurred. Most of the observed chemical and physical water quality parameters remained within normal ranges, and water quality in the Rio and tributaries below the fire were the same as the water quality in control sites. The median and ranges of pH, temperature, specific conductivity, dissolved oxygen concentration and percent saturation, and total dissolved solids remained equivalent among all sites, control and impact. The WFC fire did not alter the pH, dissolved oxygen concentration, specific conductivity or total dissolved solids concentration in the water of the Rio Grande and its tributaries.

The concentrations of nutrients, total nitrate, total nitrite, and orthophosphate, were largely un-changed by the fire. All sampling sites on the Rio Grande and South Fork of the Rio Grande had similar concentrations of all three nutrients, and were not significantly different than the control site. In one tributary pair, Squaw Creek (control) and Little Squaw Creek (impact), the fire-impacted creek had significantly higher nitrate concentrations for one monitoring season, 2015 (Table 2.2). While the concentration of nitrate in Little Squaw was significantly higher than Squaw Creek, the elevated nitrate concentrations were still relatively low and much lower than the state standard, the Little Squaw Creek mean concentration of nitrate was 0.06 mg/L the current interim state standard for total nitrogen is 1.25 mg/L (CDPHE Reg. 31). Nitrate typically accounts for the majority of the total nitrogen concentration, and the level of nitrate is much lower than the state standard for total nitrogen. This was the only year that there was a detectable difference in nitrate concentrations among sites. The other tributary pair, Red Mountain and Trout Creek, did not show any significant difference among sites for all three nutrients.

Table 2.2. Results from statistical comparison testing among control (Thirty Mile, Squaw, and Red Mountain) and fire-impact sites (Little Squaw, Trout, Box, Marshall Park, and Wagon Wheel Gap).

Analyte	Site & Year	Mean Control	Mean Impact	Median Control	Median Impact	SD Control	SD Impact	Mann Whitney U, p-value	Kruskal-Wallis p-value
Turbidity 2015	All Sites								2.20E-16
Turb NTU	Squaw & Little Squaw 2015	8.7	12.7	0.0	0.8	102.3	116.4	4.75E-46	
	Squaw & Little Squaw 2016	25.1	60.9	23.1	24.1	127.3	309.1	7.45E-32	
	Red Mountain & Trout 2015	87.3	21.7	3.1	4.9	468.9	136.1	1.75E-14	
	Red Mountain & Trout 2016	146.8	1801.6	0.0	3000.0	507.9	1397.1	2.05E-184	
	Thirty Miley & Box Canyon 2015	12.3	215.9	6.2	6.8	43.9	681.7	2.51E-11	
	Thirty Miley & Box Canyon 2016	257.7	697.7	0.0	8.5	582.1	1073.3	5.26E-25	
	Thirty Mile & Marshall Park 2015	12.3	144.0	6.2	10.6	43.9	534.0	4.45E-57	
	Thirty Mile & Wagon Wheel Gap 2015	12.3	17.6	6.2	7.2	43.9	43.9	5.70E-12	
NO3	Squaw & Little Squaw 2015	0.00	0.06	0.00	0.03	0.01	0.06	0.0172	
TSS 2016	Thirty Mile and Marshall Park	4.0	14.2	2.5	10.0	3.6	11.6	0.04	

At all sites, the total and dissolved metal concentrations did not significantly differ among sites and were not significantly different from historic data. There was only one single instance where a metal concentration exceeded state standards, in the spring of 2014, the first year after the fire, the total iron concentration was observed to be 1.12 mg/L in a segment of the Rio Grande within the fire scar, Box Canyon, where the Colorado state water quality standard for that segment is 1.0 mg/L (Reference for CDPHE Basic Standards document). This was a small exceedance and was only observed that one time in the first episode of snowmelt runoff after the fire. Otherwise, the WFC fire did not have a significant impact on the high and low flow concentrations of total and dissolved metals.

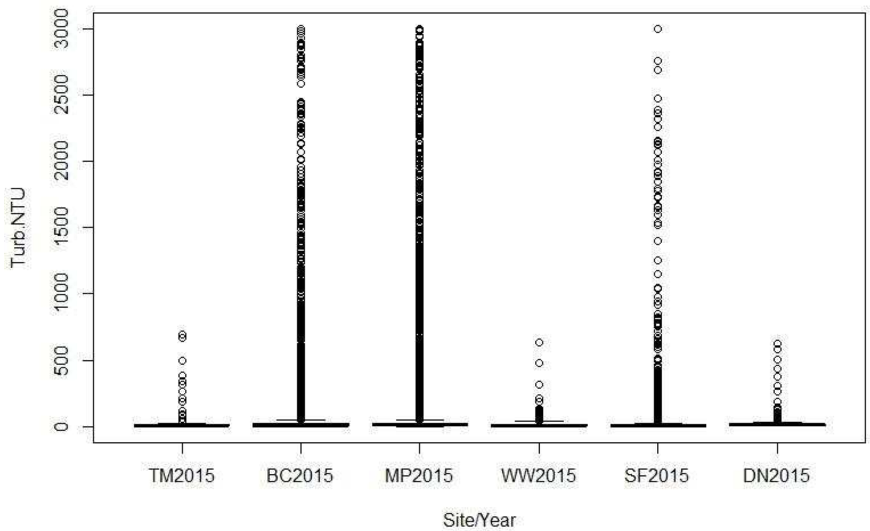
While many chemical and physical water quality characteristics remained unchanged after the fire, the mean and median turbidity of the streams below the fire were significantly higher than control sites (Table 2.2). After the observed fish kill on Trout Creek, one year after the fire in August of 2014, we started monitoring turbidity and total suspended solids. At the time of the fish kill, the concentration of total suspended solids in Trout Creek was 1018 mg/L. The turbidity of Trout Creek at the time of the fish kill was 505 NTU, where an instantaneous threshold of 50 NTU has been set by many western U.S. states because beyond 50 NTU a trout's ability to breathe and hunt is impaired (Rowe et al. 2003).

Elevated turbidity and total suspended solids concentrations and frequency of exceedances were the significant detectable water quality impacts from the WFC fire (Table 2.2, Figures 2.2 a and b, 2.3 a and b, and Figure 2.4). Stream segments within the burn scar and sites outside of the burn scar but less than 12 kilometers from the fire, had significantly higher turbidity than the control sites. This includes sites along the Rio Grande and the two impacted tributaries. We conducted pair-wise comparisons, Mann-Whitney *U* test, between all control and impact sites for all measured water quality parameters (Table 2.2). The majority of these significant differences are the median and distribution of the turbidity levels between control and impact sites, while other water quality parameters remained un-changed.

The control site, Thirty Mile, is compared to the impact sites in the Rio Grande (Figure 2.2 a and b). The impact sites, Box Canyon, Marshall Park, Wagon Wheel Gap and the South Fork of the Rio Grande, all had significantly higher median turbidity levels in 2015 (Figure 2.2a, Table 2.2). Box Canyon, the first site on the Rio Grande below the control site and in the fire scar, had mean turbidity levels that were 18 times greater than the control in 2015 (Table 2.2). Furthermore, the frequency of turbidity exceedances, above the threshold of 50 NTU, in 2015 was eleven time higher below the fire than the control site above the fire (p-value < 0.05 Mann-Whitney *U*). In Box Canyon, the turbidity of the water exceeded the threshold of 100 NTU fifteen times more than the control site (p-value < 0.05 Mann-Whitney *U*). The next site downriver of the fire, Marshall Park, which is over eleven kilometers from the fire perimeter, had nine times more exceedances than the control site at Thirty Mile (p-value < 0.05 Mann-Whitney *U*). However, other down-river sites were less impacted, where the water in the Rio Grande at Wagon Wheel, an impact site, had almost identical number of exceedances as control. The control site, Thirty Mile, had a high mean turbidity level (257 NTU) the third year after the fire in 2016, where turbidity levels were elevated during the monsoon rain season. Where due to these high observed turbidity levels, the control site had significantly higher turbidity than two of the impact sites, Marshall Park and Wagon Wheel (Table 2.2). Despite the high mean and median turbidity in the control site in 2016, Box Canyon, the site within the burn scar, still experienced significantly higher turbidity than the control (p-value <0.05, Mann-Whitney U). Box Canyon exceeded threshold turbidity levels (50 NTU) twice as often as the control, the third year after the fire.

A multiple comparison test, the Kruskal-Wallis, was conducted to identify any significant water quality differences among all sites. The statistical test revealed that the only water quality parameter that was significantly different among sites was the turbidity of the water in 2015. Levels of turbidity were significantly higher at all impact sites on the Rio Grande in 2015 (Figure 2.2a).

Paired comparisons among the tributary sites show a similar pattern, impact sites had significantly higher turbidity levels in 2015 and 2016. Red Mountain Creek, a control tributary site, had a higher mean turbidity level in 2015 but a significantly lower median turbidity level (Table 2.2). More value is placed on the median values as we do not assume that the data are normally distributed we are choosing all non-parametric comparisons. The box plots of the paired tributaries, Red Mountain Creek and Trout Creek in Figures 2.3a and 2.3b, show similar median and distribution in 2015 and in 2016, a higher median and 75th percentile in the impact stream, Trout Creek. Despite the subtle differences in the median and distribution in 2015, Trout Creek still had 2 times as many exceedances above 50 NTU, but it had the same number of exceedances above 100 NTU. While in 2016, Trout Creek had six times more exceedances above 50 and 100 NTU with the mean of 1801 NTU and median of 3000 NTU (p-value <0.05, Mann-Whitney *U*). Similar observations were made in the other paired tributary sites, Squaw Creek, control, and Little Squaw Creek, impact. Where in 2015, Squaw and Little Squaw had the same number of exceedances above 50 and 100, and in 2016 Little Squaw had 3 times more exceedances above 50 but same number of exceedances above 100. Little Squaw had significantly higher median turbidity levels than Squaw Creek in both 2015 and 2016 (p-value < 0.05, Mann-Whitney *U*, Table 2.2).



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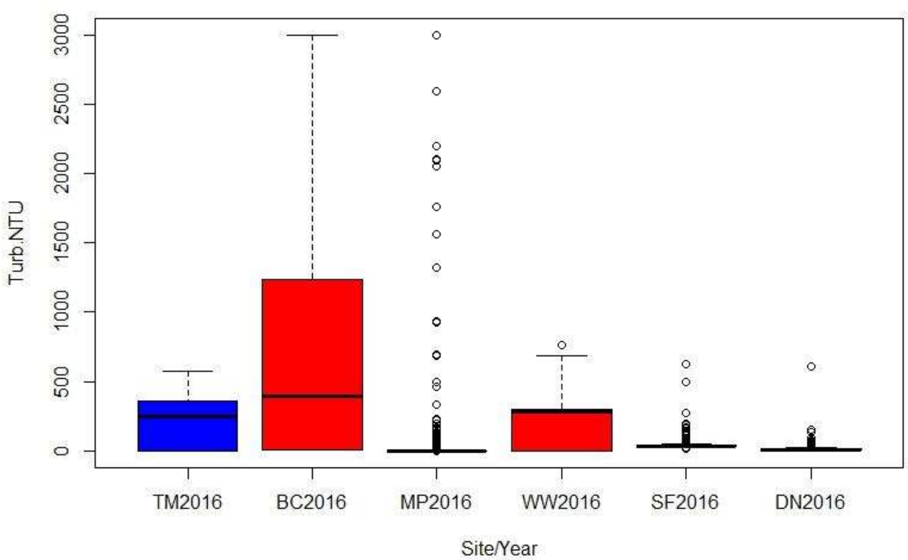
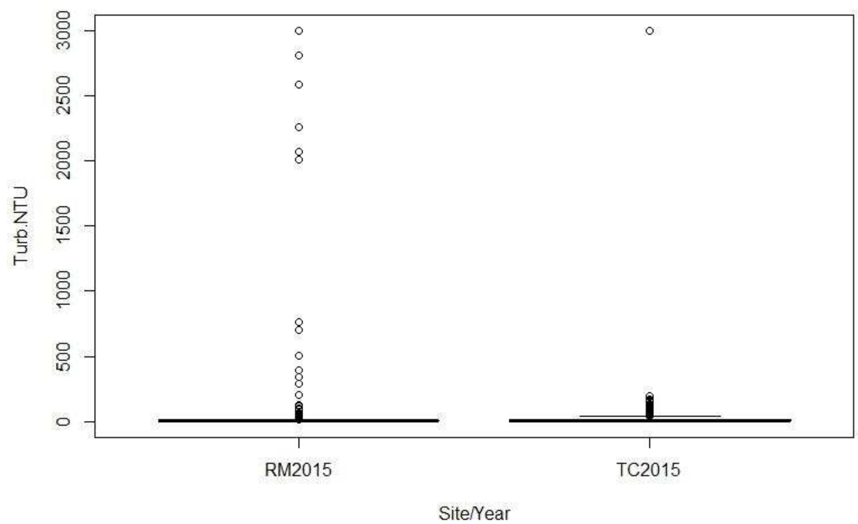


(b)

Figures 2.2 a and b. Boxplots showing turbidity levels measured in 2015 (a) and 2016 (b) in the Rio Grande above the fire in blue, Thirty Mile (TM), and below the fire in red at Box Canyon (BC), Marshall Park (MP), Wagon Wheel Gap (WW), the South Fork of the Rio Grande (SF) and Del Norte (DN).



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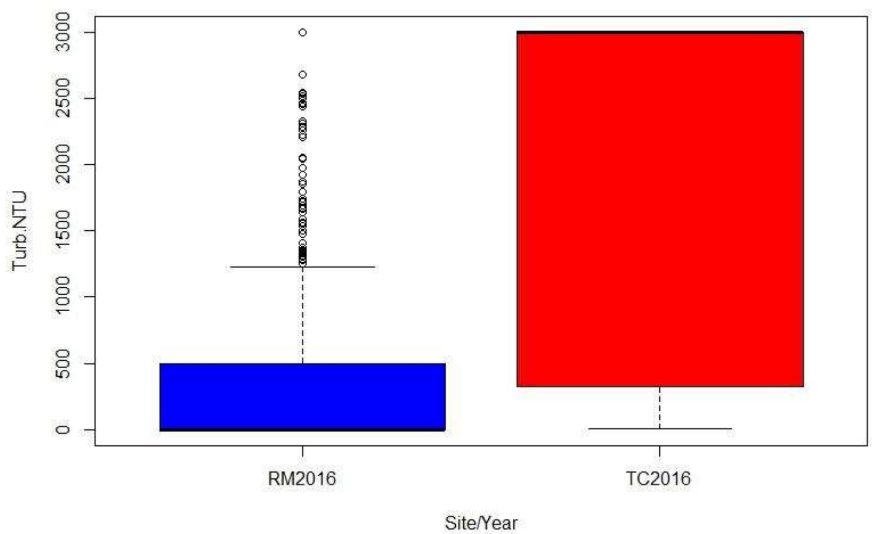


(b)

Figures 2.3 a and b. Paired study tributary sites, Red Mountain, control, and Trout Creek, impact in 2015 (a) and in 2016 (b).

Statistically significant differences in the TSS were only observed between the control site, Thirty Mile, and the second site downstream of the fire, Marshall Park in 2016 (Figure 2.4). Where median TSS values were 2.5 mg/L in the Rio Grande at the control site and were 10 mg/L in the Rio Grande at Marshall Park (p-value < 0.05, Mann-Whitney *U*, Table 2.2). At each site, there were fewer recordings of total suspended solids because samples were only taken when we were on site every two weeks. While the turbidity sensor was left in the stream all season taking frequent daily measures.

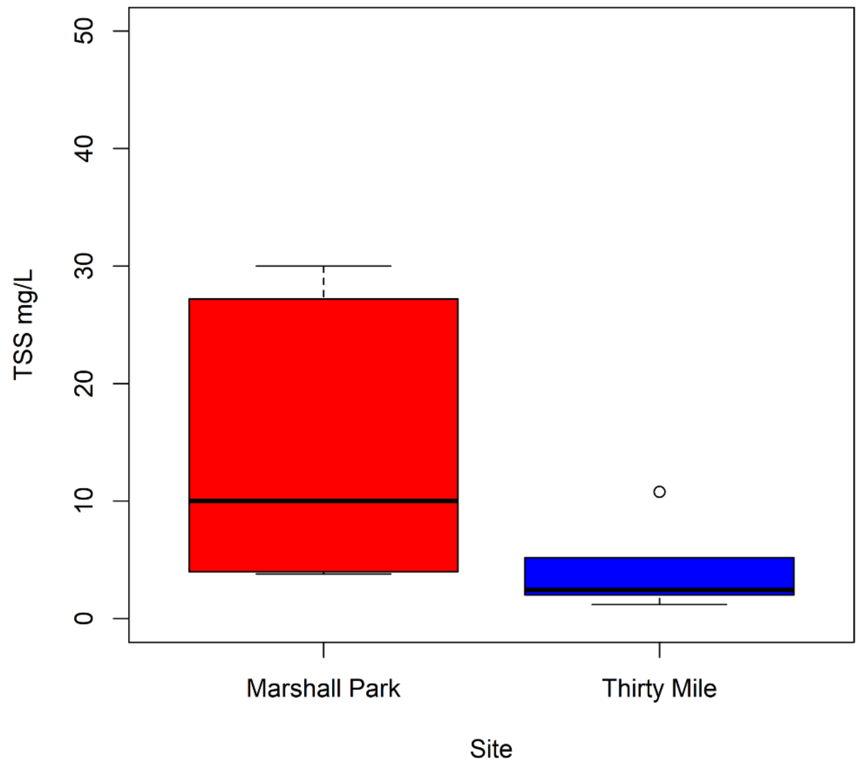
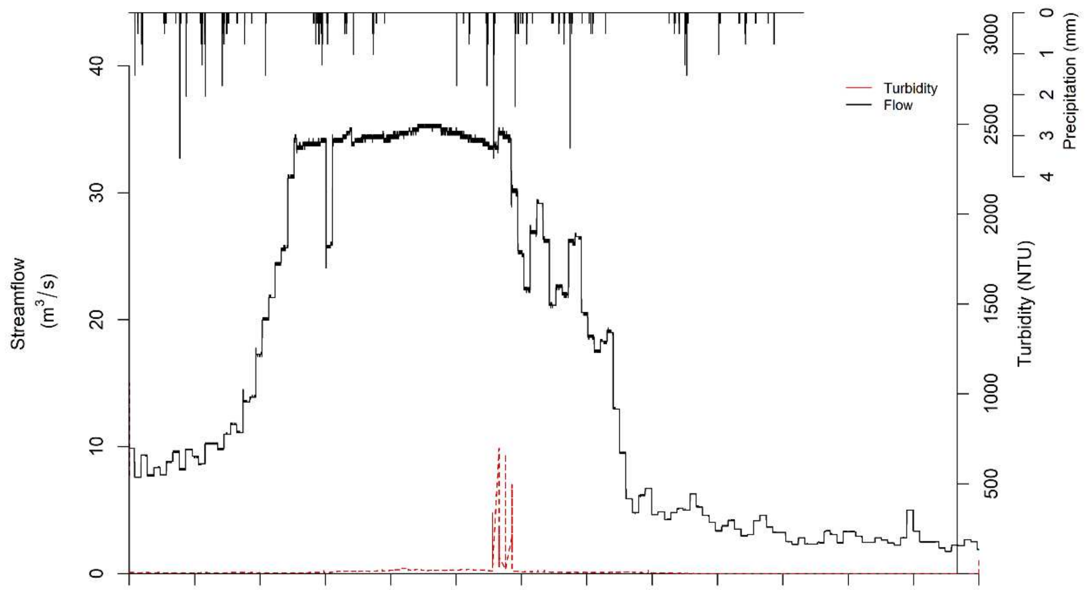


Figure 2.4. Boxplot comparing median total suspended solids (TSS) concentrations among fire-impact site, Marshall Park, and control site, Thirty Mile.

The elevated turbidity levels observed in fire-impacted sites were a result of both the high flow during snowmelt runoff and the late summer monsoon rain events. Measured flow, precipitation, and turbidity are overlain in Figures 2.5 through 2.8. Each represents the hydrograph, hyetograph and turbidity measured by the sondes. Looking at the side-by-side comparison of the control site, Thirty Mile, with the impact site, Box Canyon, the turbidity increases immediately following every small rain event in the fire-impacted site. Meanwhile, the turbidity levels in the control site have fewer spikes (Figures 2.5 and 2.6). In 2016, there were still some short-lived high turbidity events in the Rio Grande above the fire, however, the frequency and duration of high turbidity events are much higher in the Rio Grande in the fire scar at Box Canyon (Figure 2.6). The comparison between the control tributary, Red Mountain Creek, and the fire-impacted site, Trout Creek, is similar (Figures 2.7 and 2.8). In this instance, the turbidity at the control and impact sites are similar in 2015 (Figure 2.7), yet the median turbidity levels of Trout Creek were significantly higher; the turbidity in Trout Creek in 2016 is consistently higher and becomes elevated after every small rain event (Figure 2.8).



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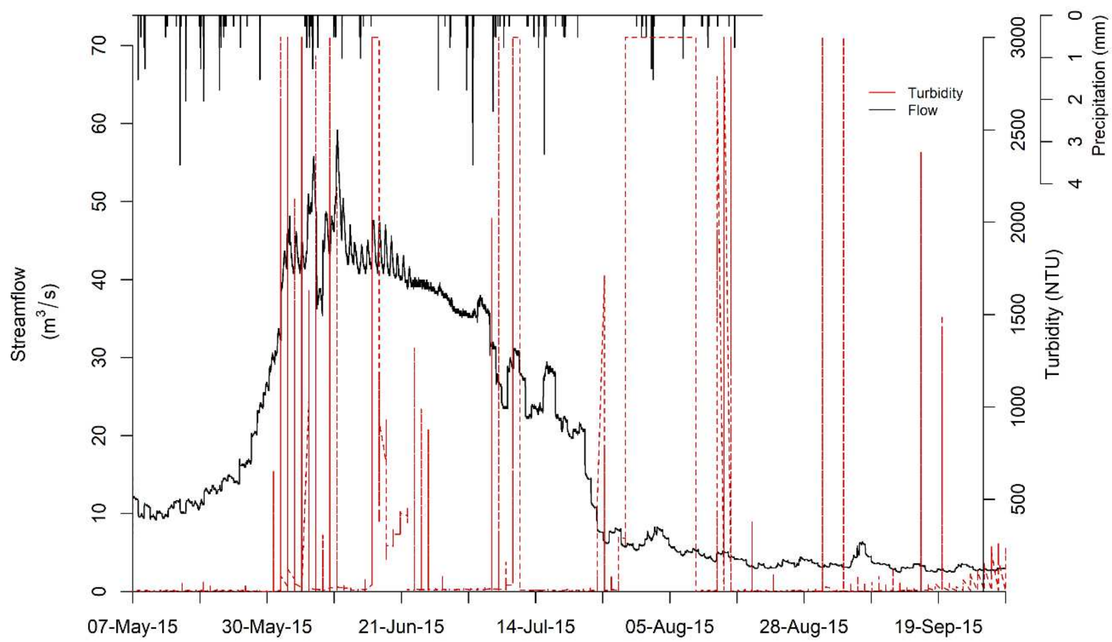
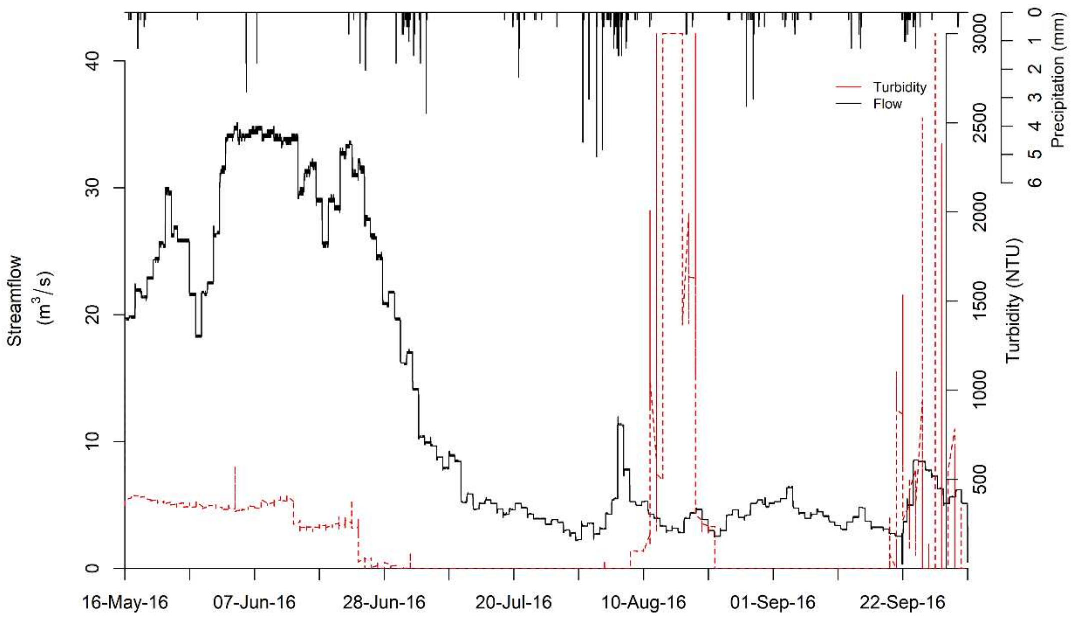
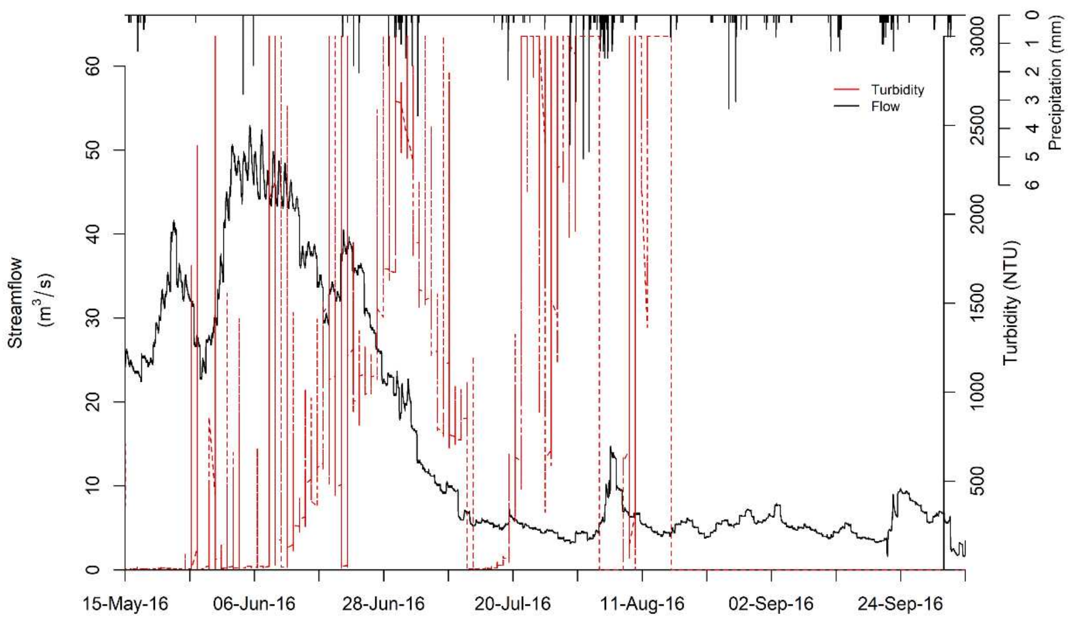


(b)

Figures 2.5 a and b. 2015 Hydrographs, hyetographs, and turbidity levels measured in the Rio Grande above the fire, at Thirty Mile (a), and below the fire, at Box Canyon (b). Below the fire, turbidity levels in red, jump up to high levels above trout thresholds after every precipitation event.



(a)



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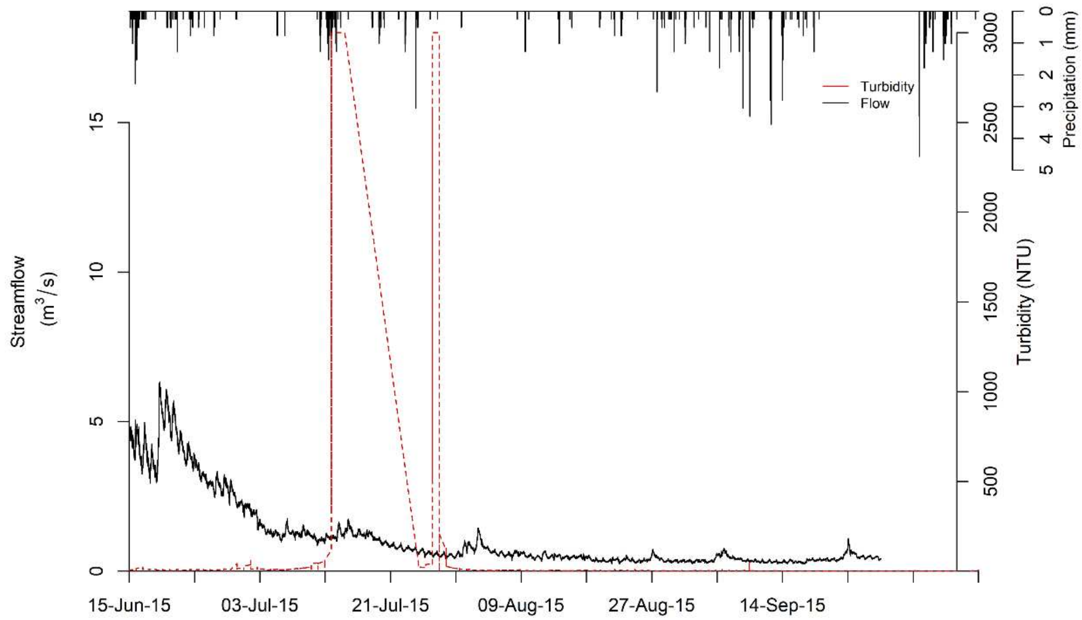
Figures 2.6 a and b. 2016 Hydrographs, hyetographs, and turbidity levels measured in the Rio Grande above the fire, at Thirty Mile (a), and below the fire, at Box Canyon (b).



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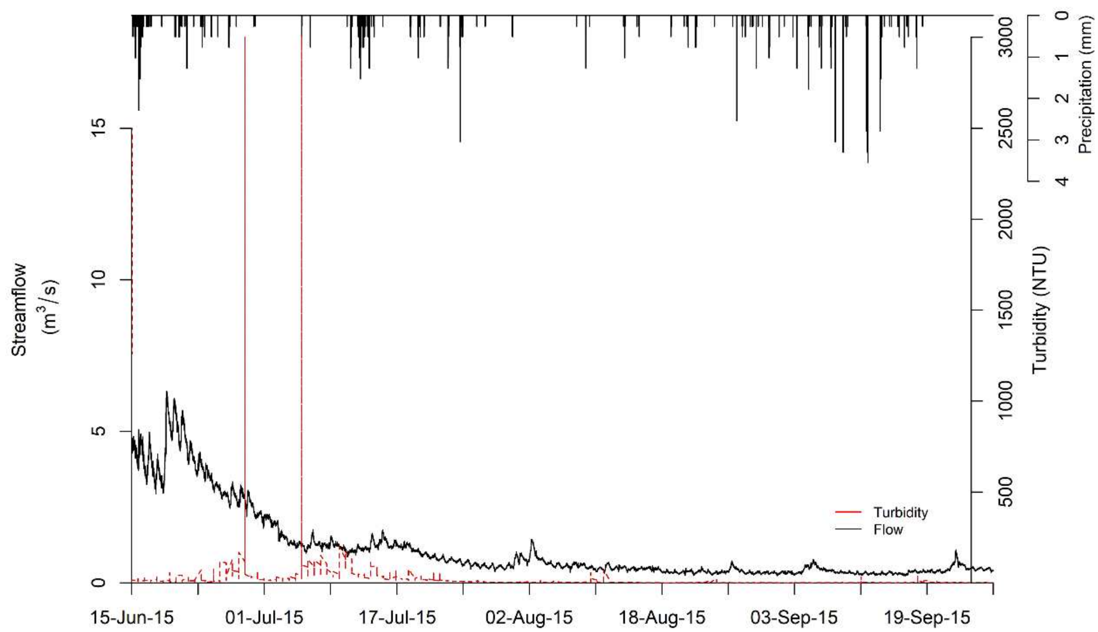
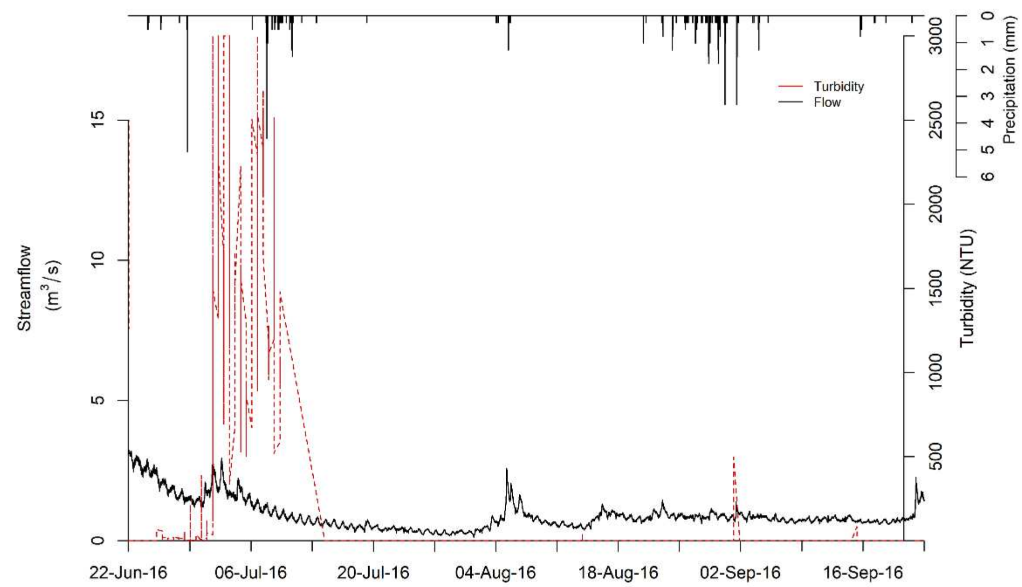
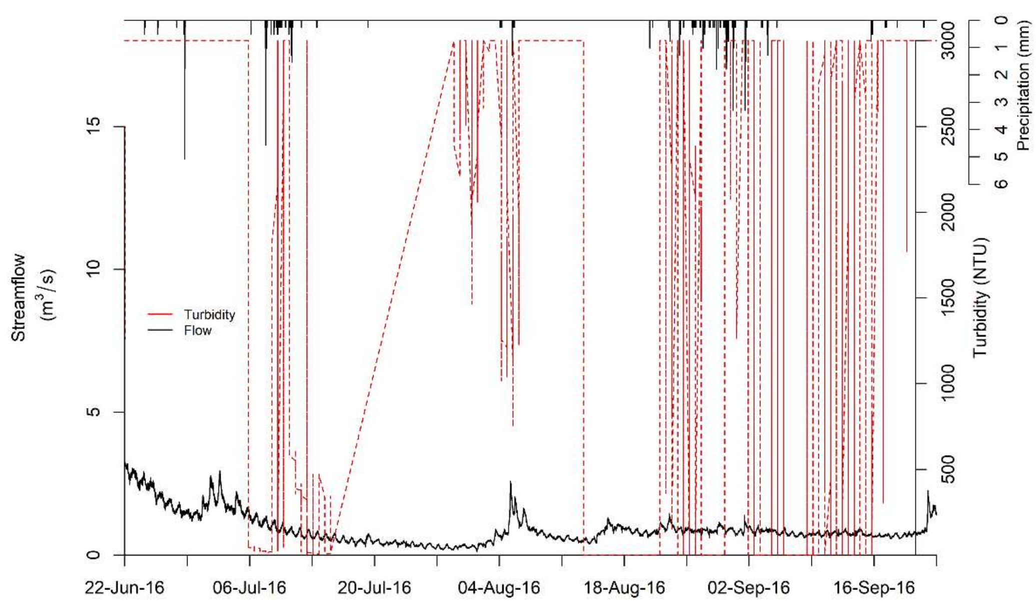


(b)

Figures 2.7 a and b. 2015 Hydrographs, hyetographs, and turbidity levels measured in tributaries outside of the fire, at Red Mountain Creek (a), and within the fire, at Trout Creek (b).



(a)



(b)

Figures 2.8 a and b. 2016 Hydrographs, hyetographs, and turbidity levels measured in tributaries outside of the fire, Red Mountain Creek (a), and within the fire, Trout Creek (b). Unlike 2015, the fire-impacted tributary had higher turbidity.

2.4.2 Macroinvertebrates

We compared collected population data with historic data from before the fire from the CDPHE. Unfortunately, our control site had not been sampled by the state before the fire so there was no historic data at Thirty Mile for comparison However, looking at the patterns in Figures 2.9 through 11, the macroinvertebrates were impacted by the WFC fire whether they lived within the fire area or near the fire. Even in the control sites, populations were impaired after the fire (Figures 2.9 through 11). All sites (control & impact) show a decline in the multi-metric index the first year after fire. The metric assesses macroinvertebrate abundance and diversity. All observed MMI values fall under attainment threshold of MMI 50 (Figure 2.9). The state of Colorado's impairment threshold for MMI is lower, it is an MMI of 42; many sites fall below this threshold the first year after fire.

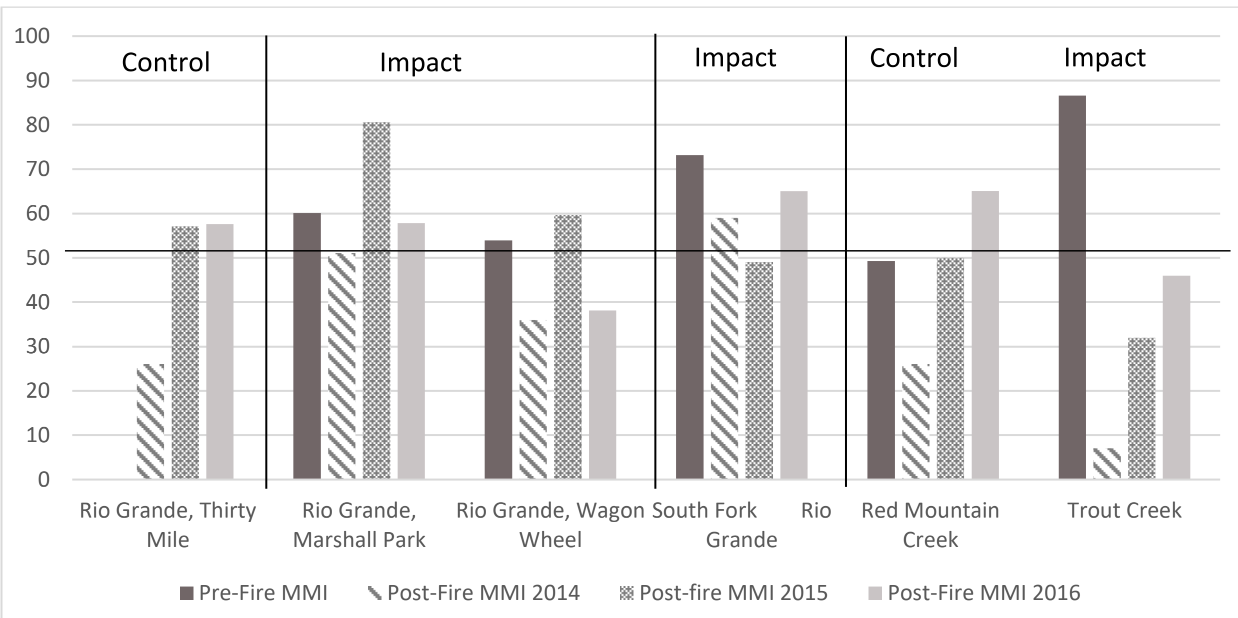


Figure 2.9. Multi Metric Index (MMI) results for control and impact sites from before the fire in 2012 and after the fire in 2014-2016. Vertical bars separate control and impact sites and sites in different streams (Rio Grande, South Fork of Rio Grande, and tributaries, Red Mountain and Trout Creek). The MMI is a measure of insect abundance and diversity, below a MMI of 50, the state of Colorado believes the water quality is impaired.

Trout Creek (impact) and Red Mountain creek (control) both experienced a drop in MMI values of greater than twenty points, which signals an impairment. The state of Colorado would consider these sites to have impaired water quality due to the severe drop in MMI value. Here the impairment is not human caused, but a natural impact from wildfire. All sites except Trout Creek show a recovery in MMI value by 2015, two years after the fire. While Trout Creek shows a slower recovery, almost reaching the minimum MMI threshold by 2016. A second line of evidence, the Shannon-Wiener Diversity Index (SDI), also reveals a reduction in aquatic macroinvertebrate diversity. Any site below a SDI of 3.0 is impaired, it is evident that all sites fell below this metric the first year after the fire (Figure 2.10). Some sites show recovery two years after fire where the SDI value is at or above impairment thresholds, and all sites recover by the third year after fire (Figure 2.10). A third metric, the Hilsenhoff Biotic Index (HBI), is a measure of pollution tolerant insects present, in this sense, a healthy mountain stream would have a low HBI. Figure 2.11 illustrates that none of the sites reached impairment under this metric but in all cases the year after the fire the HBI was elevated, revealing a shift in the macroinvertebrate taxa toward more pollution tolerant species, similar to the Verkaik et al. 2015 results. Again, some sites show a decline in HBI by year two and all sites except the control site, Thirty Mile, show recovery where HBI levels are back to pre-fire conditions by 2016.

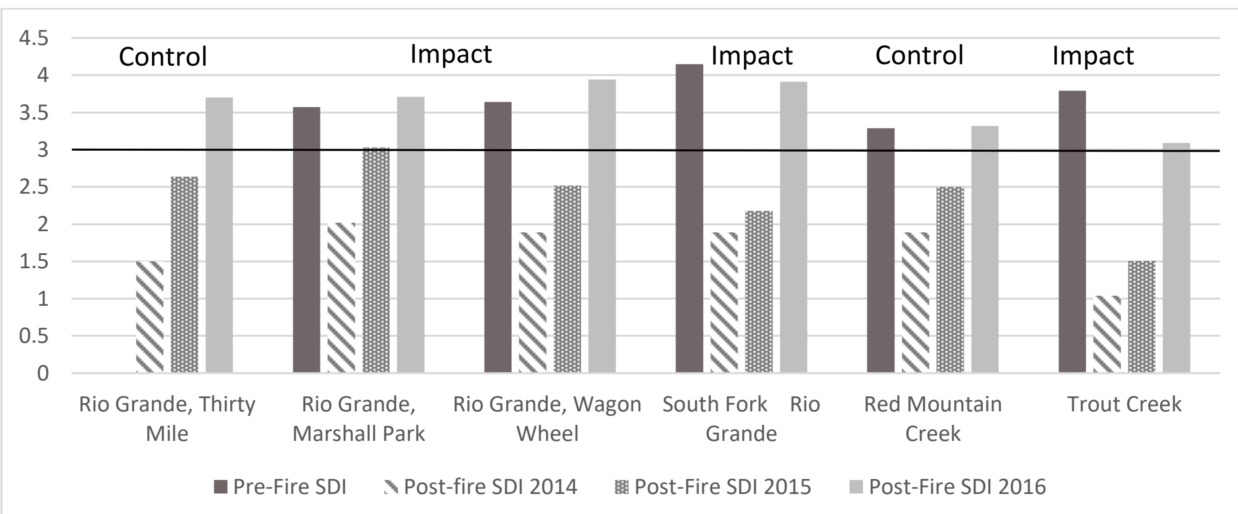


Figure 2.10. The Shannon Diversity Index (SDI) for control and impact sites from before the fire in 2012 and after the fire in 2014-2016. Vertical bars separate control and impact sites and sites in different streams (Rio Grande, South Fork of Rio Grande, and tributaries, Red Mountain and Trout Creek). The Shannon Diversity Index is a measure of macroinvertebrate species diversity. A SDI below 3.0 is evidence of water quality impairment in the state of Colorado.

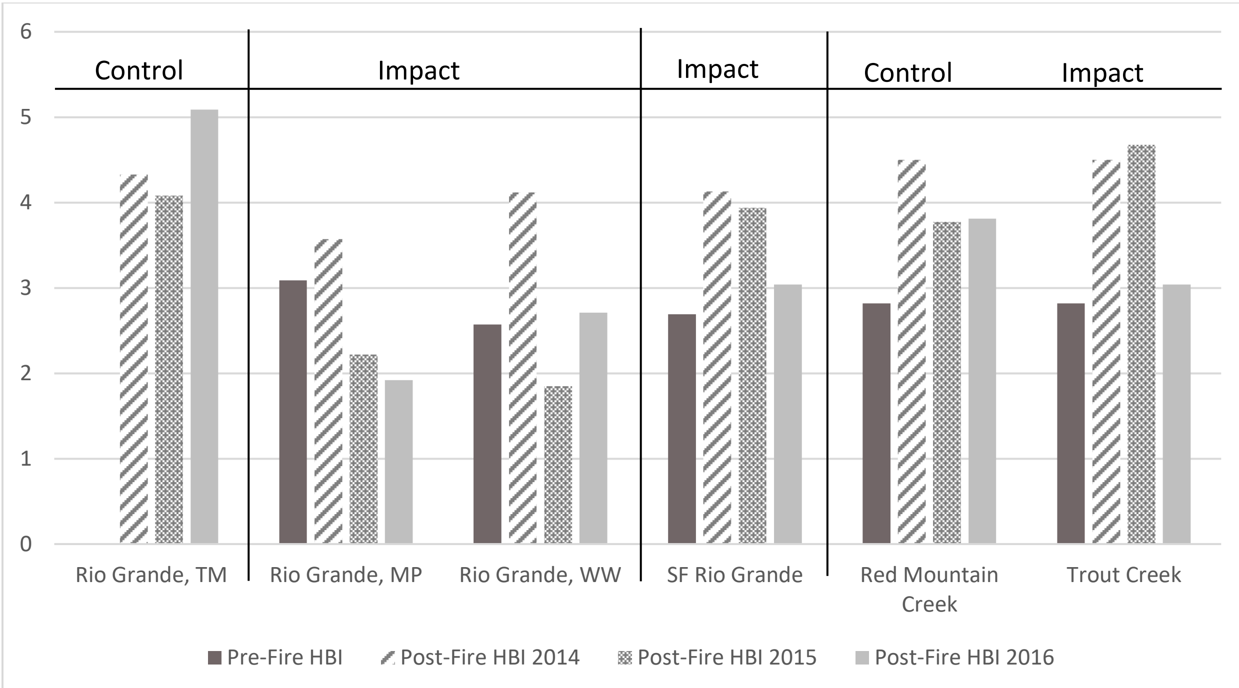


Figure 2.11. The Hilsenhoff Biotic Index (HBI) for control and impact sites from before the fire in 2012 and after the fire in 2014-2016. Vertical bars separate control and impact sites and sites in different streams (Rio Grande, South Fork of Rio Grande, and tributaries, Red Mountain and Trout Creek). The HBI is an index of pollution tolerant macroinvertebrate species. A HBI above 5.4 is evidence of water quality impairment in the state of Colorado.

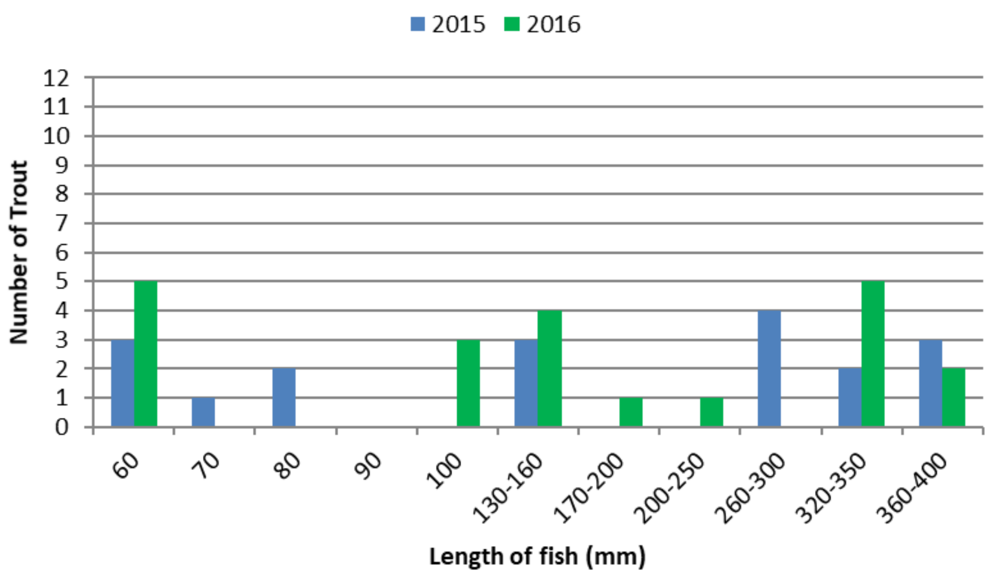
2.4.3 Fish Populations

Trout population sampling in the Rio Grande below the WFC fire revealed that the fish population density and the number of large fish have both increased since the fire. Trout populations are larger in number and mass per acre in the Rio Grande after the fire than before the fire. There was an increase in the density of large fish above 35 cm per acre after the fire, where the density of large fish per acre in 2011 was 16 and had climbed to 18 in 2014 the year after the fire (CPW personal communication 1/20/2015). A density of 12 large trout per acre is the qualifying threshold for Gold Medal trout waters (Colorado Wildlife Commission 2008).

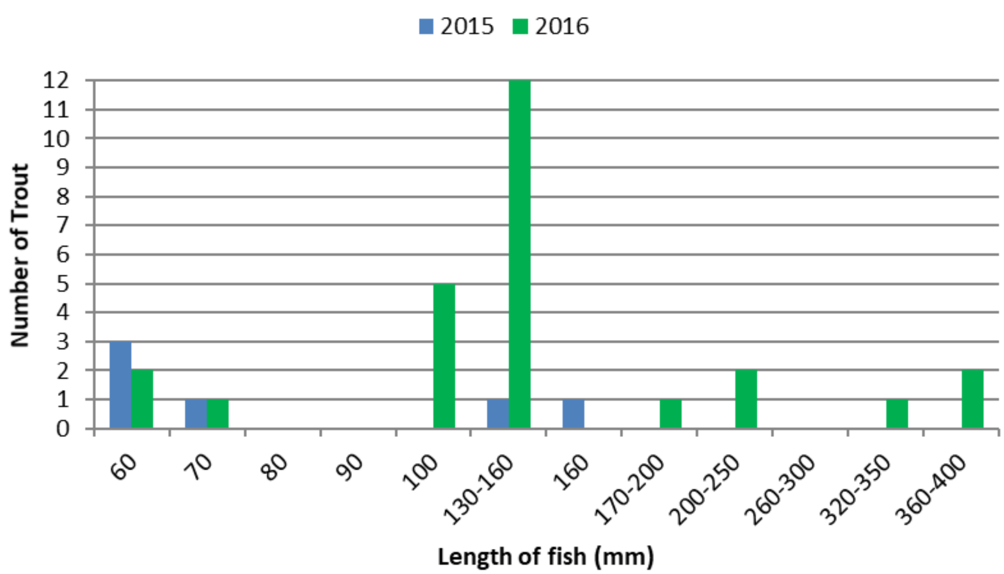
However, the fish populations in the fire-impacted tributary, Trout Creek, experienced a dramatic decline due to an observed fish kill in 2014 caused by a monsoon-driven landslide in the fire scar. Extremely high total suspended solids and turbidity levels were observed during the fish kill event. Fish sampling in 2015 revealed that there were no adults inhabiting the segment of Trout Creek below the landslide two years after the fire (Figure 2.12 a). Trout fry were captured - either they had been spawned in the segment, or above the landslide and young of year were washed down. Either way, the presence of young of year fish demonstrated that the water quality in the segment was suitable enough to support fish life. Looking at the comparison of the estimated population, number of fish per acre, and kilograms per kilometer in 2015 (Table 2.3), it is clear the site above the landslide was supporting a greater number of fish and more kilograms of fish per kilometer. The size distribution of fish above the landslide reveals a wide variety of sizes of fish, typical of a healthy population. While below the landslide, in 2015, there were only young of year fish, adults were completely absent. By 2016, the population shifts and shows dramatic recovery (Figure 2.12 b). At both sites, above and below the landslide on Trout Creek, the estimated number of fish per acre and kilograms per kilometer is larger in 2016 (Table 2.3). The size distribution of fish in 2016 was also a sign of recovery, below the landslide there was a wider range of fish sizes and there were more adults occupying the segment. While the population size and distribution above the landslide remained similar to the year before, 2015.

2.5 Discussion

Results show significant post-fire water quality effect in the form of elevated turbidity and total suspended solids. The water quality perturbance had direct impacts on fish. Macroinvertebrates were affected by the fire in a different way. The fish and insect populations recovered despite continued turbidity exceedances. While the other water quality parameters we monitored were not different upstream or downstream of the fire, turbidity levels below the WFC fire were significantly higher than control sites. The median turbidity and frequency of exceedances over the threshold were higher in stream segments that were closer to the fire and below fire areas where there was moderate and high burn severity. The four sites within 12 kilometers of the fire perimeter and in watersheds where the majority of the fire was moderate to high severity (> 63% of fire moderate to high severity) had significantly higher turbidity levels two years after the fire (Table 2.1 and 2.2). The monitoring site that was 22 kilometers away from the fire and at the outlet of study watershed, where only two percent of the fire was moderate to high severity, had no significant change in turbidity or total suspended solids levels after the fire. Fire severity and proximity drove the turbidity response in 2015. By 2016, elevated turbidity levels and frequency of exceedances were only observed in the fire impacted tributaries and the Rio Grande site within the fire scar. In all three cases, the percent moderate and high severity were above 63% and the stream segments were within 1.5 kilometers of the fire. Rhoades et al. (2011) observed a similar correlation, finding the turbidity in the streams increased linearly with the amount of the basin burned at high severity. In the Rhoades et al. (2011) study the percent of the basin with high burn severity was correlated with turbidity and a greater extent of each basin was burned (> 60%).



(a)



(b)

Figure 2.12 a and b. Distribution of fish sizes captured above (a) and below (b) the landslide in Trout Creek. Fish were sampled one and two years after the landslide.

Table 2.3. Fish population metrics above and below the landslide in Trout Creek one year after the landslide and two and three years after the fire.

		N (pop. est.)	Standard Error of N	Confidence Limits of N	Number per acre	Kilograms per kilometer
Above Landslide	2015	24	8	24 +/- 16	93	5
	2016	24	1	24 +/- 2	142	16
Below Landslide	2015	7	1	7 +/- 2	43	0
	2016	28	3	28 +/ - 6	151	5

The proportion of the watershed burned was smaller (< 17%), but the percent of the fire area that burned moderate and high severity triggered higher turbidity. Although we did not find a correlation between turbidity and total suspended solids, the turbidity was more likely a result of suspended solids than dissolved solids. We monitored dissolved solids and specific conductivity and they were not significantly elevated or different among sites.

Turbidity levels in the WFC fire-impacted streams were elevated during runoff and after every late summer monsoon rain event (Figures 2.5 – 2.8). The frequency of turbidity exceedances above the trout threshold was from two to fifteen times higher in fire-impacted streams. Exceedances were driven by precipitation events (Figures 2.5 – 2.8). While instantaneous spikes in turbidity levels may be benign to trout, turbidity that remains high for extended periods of time inhibits their ability to respire and hunt (Shaw and Richardson 2001). Elevated turbidity and total suspended solids below wildfire, where summer precipitation events cause erosion and sediment delivery, has been observed in other studies (Silins et al. 2009; Dahm et al. 2015). Murphy et al. (2015) studied another Colorado wildfire and observed total suspended solids concentrations that were 10-156 times higher below the fire. The high TSS levels were driven by short-duration monsoon rain events causing erosion and carrying sediment to receiving streams (Murphy et al. 2015). Here, elevated turbidity levels occurred following monsoon rain events.

The year after the fire the macroinvertebrate populations were depressed (Figure 2.9 and 2.10). At every site, control and impact, there was a shift toward pollution tolerant species (Figure 2.11). While the control sites were upstream of the fire, they were also in close proximity to the fire (< 0.3 km from fire perimeter) and were surrounded by smoke and ash during the large fire. Most macroinvertebrates live for one year and their populations can be responsive to disturbance. Macroinvertebrates could have been impaired by air quality during the long fire (June 10- mid August) because this is their time to reproduce. Most aquatic insects are weak flyers when they emerge, and smoke and ash may have inhibited their ability to fly and find mates. As air pollution was temporary, occurring only for the duration of the fire, the populations recovered from the disturbance quickly. In Minshall's (2003) summary of wildfire impacts on macroinvertebrates, he states that direct effects of fire are rarely detected, where in this study, the shift in macroinvertebrate species is likely a direct effect of fire. However, in Minshall's (2003) review, indirect wildfire effects are more common and macroinvertebrate recovery is swift, where populations are back to pre-fire levels within five to ten years. Earl and Blinn (2003) observed an impact on macroinvertebrate densities due to ash from the fire in the water, but population recovery was fast, within one year of the fire. Other studies have observed an increase in the macroinvertebrate abundance but a decrease in diversity, often with more pollution tolerant species present (Roby 1989; Lawrence and Minshall 1994). After the WFC fire, the macroinvertebrate population and diversity indices returned to pre-fire levels within two years at some sites and three years at others (Figures 2.9-2.11).

It is possible that ash and sediment buried insects; however, we did not observe any sedimentation during our regular sampling visits (and did not quantify sedimentation of the streambed). Ash was observed in eddy pools the first year after the fire but was flushed during 2015 runoff. The headwaters of the Rio Grande lie in a steep basin, the stream channels are also steep, giving ash and sediment little opportunity to accumulate. Because the macroinvertebrate populations were depressed in control areas, which were not impacted by ash and sediment, it is more likely air pollution impacted the macroinvertebrates more than any change in water quality or sedimentation in the riverbed.

The WFC fire burned over a headwaters catchment with steep slopes. The average slope of the burn area was 21 degrees, with many slopes at the top of the watershed divide ranging from 40 to 75 degrees. It was towards the top of the Trout Creek watershed where a hillslope in the burn area slid, and this landslide caused direct mortality of resident fish. While the trout population below the landslide was diminished, the creek is well connected to the Rio Grande and fish recolonized. Young of year were observed inhabiting the stretch of Trout Creek directly below the landslide one year later, and adult populations returned two years later. Meanwhile, above the fire and landslide, the trout population remained diverse in size and age and the population remained unchanged (Figure 2.12). Our estimate of the fish population above the landslide had wide confidence limits (Table 2.3). This could be due to using undersized sampling gear; we were using a backpack shocker that was small for the width of the creek and it is likely that we missed a few fish in our sampling. However, the estimate of the population the next year was identical, with a narrow confidence limit, so we believe that despite our undersized sampling gear, our population estimates were accurate. The absence of adult fish below the landslide the first year was most apparent; in this sampling effort we do not believe we missed many fish in that segment. We used a small waterfall structure as our sampling end-point for the site below the landslide, so any fish that we missed would likely move upstream and would be trapped and captured. In 2016, the third year after the fire and second year after the fish kill, the landslide area was covered in new vegetation and the soils were less vulnerable to erosion. While turbidity levels in Trout Creek were still elevated after every rainstorm, the erosion from the landslide was less severe and water quality was good enough for fish of many sizes to inhabit the segment directly below the landslide.

Other studies have observed localized, patchy extirpations of fish populations after fire where the fish recolonized and recovered within a year (Minshall and Brock 1991; Gresswell 1999). In most studies, the cause of fish mortality is unknown; often elevated temperature or depressed dissolved oxygen levels are presumably to blame (Gresswell 1999). In this research, the cause of the fish kill in Trout Creek was obvious, where total suspended solids concentrations were observed to be 1018 mg/L and creek water looked like a mud slurry. This concentration of suspended solids is ten times the threshold level, clogging fish gills and leading to suffocation. Burton (2005) noted the severity of the fire determined the localized impact on fish habitat and populations. The Burton (2005) study also found the Idaho trout populations recovered within five to ten years after fire, often experiencing an increase in population due to increased post-fire flows scouring streambeds, importing nutrients thereby increasing fish productivity (Burton

2005). The Burton (2005) study is among the few studies to observe fish population recovery after fire; where streams were connected, fish recolonized fire-impacted areas. Our observations agree with Burton (2005), fires are a natural disturbance that trout have evolved with, while localized fish kills still occur as a direct effect of fire, the recovery is swift.

Few studies have quantified the impact of wildfire on water quality, macroinvertebrate and fish populations. Even fewer have witnessed recovery. This study was an opportunity to monitor wildfire's impacts on a whole ecosystem and witness recovery in less than four years in most locations. We were encouraged to see that even after a large wildfire that was 60% moderate and high severity, vegetation, mostly Aspen (*Populus tremuloides*) grew back naturally within three years without human intervention. While steeper slopes are slower to recover, and turbidity remains elevated directly below the fire, insect and fish populations have generally recovered.

2.6 Conclusion

While water quality was still impaired, with high turbidity levels three years after the fire in locations directly below the fire scar, aquatic life was generally recovered in the Rio Grande. Fish populations were improving and macroinvertebrate population and diversity had recovered. Insect populations were more perturbed by air pollution than by in stream water quality the year after the fire. Observing a decline in insect diversity and abundance in control sites was an unexpected result. Future studies should investigate the extent of the area around a fire's perimeter that impacts macroinvertebrate populations.

Trout appear able to tolerate elevated turbidity in large connected streams like the Upper Rio Grande, where they can move out of poor water quality and take refuge up and downstream of turbid waters. This is evident in the state's fish population estimates in the Rio Grande. Where fish are trapped in small tributaries water quality impacts can be deadly, but still, the populations rebound quickly. Continued monitoring of water quality, aquatic life and vegetation recovery for the three to five- year period after fire could help identify determinants of system-wide recovery. Ultimately correlating vegetation recovery to water quality and aquatic life abundance could also inform land and water resource managers on how best to support ecosystem resiliency.

It is our hope to use the collected data to test and parameterize water quality models under wildfire conditions. We are continuing to monitor water quality and flow data at some sites to input into a water quality model to test the model's ability to predict post-fire nutrient and suspended solids concentrations. Future work will also include testing a geomorphological model that predicts hillslope failure after fire to see if it would have correctly identified the landslide in Trout Creek. The more often models are tested under a variety of fire conditions the more relevant and useful they will be for managers.

Wildfires are a natural disturbance that are increasing in frequency and consuming more land in the western U.S. While the fires disrupt water quality in downstream systems, our results show the impact is temporary. Aquatic life has evolved to recover from such dramatic disturbances; populations recover quickly and at even greater densities. The West Fork Complex Fire gave us an opportunity to monitor water quality impacts in the Upper Rio Grande and witness ecosystem recovery within three years.

CHAPTER 3

POST-FIRE WATER QUALITY RESPONSE IN THE WESTERN UNITED STATES

Modified from a manuscript submitted for review in the International Journal of Wildland Fire

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Abstract

Wildfires are a natural disturbance and are increasing in size and intensity in forested landscapes across the Western United States. Not only do fires alter land surfaces, but they also impact the surface water quality in downstream systems. Previous studies of individual fires have observed an increase in various forms of nutrients, ions, sediments, and metals in stream water for different post-fire time periods. In this research, data was compiled for over 24,000 fires across the western United States to evaluate post-fire water quality response. The database included millions of water quality data points downstream of these fires, and was synthesized along with geophysical data from each burned watershed. Data from 172 fires in 153 burned watersheds was used to identify common water quality response during the first five years after a fire. Within this large dataset, a subset of ten fires was examined further to identify trends in water quality response. Change-point analysis was used to identify moments in the post-fire water quality data where significant shifts in analyte concentrations occurred. Evaluating individual fires revealed strong initial increases and decreases in concentrations that are masked when averaged over five years. Evidence from this analysis reveals significant increases in nutrient flux (different forms of nitrogen and phosphorus), major-ion flux, and metal concentrations are the most common changes in stream water quality within the first five years after fire. Dissolved constituents of ions and metals tended to decrease in concentration five years after fire while particulate matter concentration continued to increase. Assembling this unique and extensive data set provided the opportunity to determine the most common post-fire water quality changes in the large and diverse Western U.S. Results from this study could inform

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studies in other parts of the world, will help parameterize and validate post-fire water quality models, and assist communities impacted by wildfire to anticipate changes to their water quality.

3.1 Introduction

Large wildfires are increasingly common in the western United States. Wildfires impact water resources, already in high demand in the arid West. Since the mid-1980s, warmer spring and summer temperatures, coupled with lower than average precipitation, produced longer wildfire seasons causing more frequent and larger fires (Westerling et al. 2006; Morgan et al. 2008; Westerling et al. 2011). Consequently, forest fires burned more area in the last three decades than in previous time periods (Miller 2009). As climate models predict continued warming in the Western United States, total area burned by fire is expected to also increase into the future (Spracklen et al. 2009) and may even lead to changes in vegetation types and fuel load (McKenzie and Littell 2016). Models predict that burn areas in the Western United States will increase by 50% to 100% over the next fifty years (McKenzie et al. 2004; Spracklen et al. 2009; Harvey 2016). Given that the Western U.S. is a large region with a variety of ecotones, findings about the growth and extent of fire could be further applied to a broader global scale.

Likewise, wildfires not only disturb vegetation but change soil structure altering hydrological processes and water quality in fire-impacted watersheds. As wildfires consume even more forested areas, water supplies and aquatic ecosystems are increasingly affected. In addition, forested mountain areas of the Western United States receive high quantities of precipitation and are the source of high quality drinking water for millions of customers (Bladon et al. 2014). Wildfires are known to impact water quality and can threaten these downstream drinking water supplies (Smith et al. 2011; Bladon et al. 2014). Furthermore, the surface waters of the arid West are limited and in demand by aquatic ecosystems, irrigated agriculture, and the region's growing human population. Ultimately, studies on individual wildfires demonstrate these impacts on water quality and quantity in a water-scarce region (Runge and Mann 2008; Emelko et al. 2011; Bladon et al. 2014; Kinoshita and Hogue 2015).

This research builds on these earlier observations of how individual fires impact water quality, and looks at a broader spatial scale, examining water quality impacts from 172 fires in 153 burned watersheds in the western U.S. to determine if there is a typical water quality response after fire. Similarly, findings from this study regarding the impacts of fire on water quality in this broad and diverse region could be applied elsewhere.

3.1.1 Fire impacts on soil and erosion

A range of studies, typically focused on individual fires, have observed mobilization of sediments, nutrients, and contaminants into receiving streams (Burke *et al.* 2013; Burton *et al.* 2016; Mast and Clow, 2008; Moody and Martin, 2001; Murphy *et al.* 2012; Rhoades *et al.* 2011; Silins *et al.* 2009). Once wildfires kill trees and vegetation that keep soils intact, soils are exposed and susceptible to erosion. Additionally, the heat from fires can alter soils and induce the formation of a hydrophobic layer. This hydrophobic soil layer reduces infiltration depth and hydraulic conductivity (DeBano, 2000). After altering vegetation and soils, precipitation becomes runoff quickly as soils are rapidly saturated with their low retention (Murphy *et al.*, 2012).

Exposed soils and increased runoff can lead to increased sediment loading in fire-impacted streams. Sediment exports are particularly elevated the first two years after fire before vegetation recovers (Lane *et al.* 2006; Silins *et al.* 2009; Ryan *et al.* 2011; Smith *et al.* 2011). Fires are classified by severity: a high-severity fire is one that removes all vegetation, kills root systems, and consumes most of the organic matter on the surface, whereas a low-severity fire leaves some trees with live needles or leaves, the soil organic layer intact and only burned a few millimeters in depth (Keeley 2009). In the end, high burn severity sites yield more sediment than moderate or low severity sites. As evidence of this, after Colorado's Hayman Fire in 2002, Rhoades *et al.* (2011) observed the stream turbidity from hillside erosion increased linearly with the amount of a basin that burned at high severity. Ultimately, the slope angle, slope aspect, and burn severity within a fire-impacted watershed influence the rate of vegetation recovery and therefore the duration of soil exposure and erosion (Shakesby *et al.* 2006; Kinoshita and Hogue, 2011; Morris *et al.* 2012).

3.1.2 Fire impacts on water quality

When soils are exposed and erosion occurs, nutrients associated with sediments, like total particulate phosphorus, may increase in streams impacted by fire. Studies on individual fires have observed an increase in total phosphorus associated with higher suspended sediments lasting for three to ten years after fire (Pinder *et al.* 2005; Mast and Clow 2008; Noske *et al.*

2010; Emelko *et al.* 2016). Along with the particulate forms, concentrations of dissolved nutrients, including phosphate and the different forms of nitrogen: nitrate, dissolved organic nitrogen, ammonium, and total nitrogen, also increased in streams below wildfire. Hence, studies on individual forest fires throughout the Western United States, from the Canadian Rockies to the arid mountains of New Mexico and coastal mountains of California observed increases in nitrogen concentrations in receiving waters after forest fire (Riggan *et al.* 1994; Earl and Blinn, 2003; Bladon *et al.* 2008; Rhoades et al 2011). The current study moves beyond examining the impacts of single forest fires. Instead, this research evaluates a multitude of fires (172) over a broad spatial and temporal scale (1984-2015) to determine if an increase in dissolved and particulate nutrient concentrations are a common water quality response after fire.

After a wildfire, elevated heavy metal concentrations in post-fire runoff can also impair water quality. Although few studies have focused on metal concentrations after fire, two studies on the 2009 Station Fire outside of Los Angeles, California, observed increases in total and dissolved metal concentrations (Burke *et al.* 2013; Burton *et al.* 2016). Findings indicated rainstorms after the fire carried dissolved metals (Al, As, Cd, Cu, Fe, Mn, Ni, Pb, and Zn) and metals bound to ash and suspended sediment into the Arroyo Seco (Burke *et al.* 2013; Burton *et al.* 2016). Additionally, a study on another southern California fire near an urban area observed a significant increase in total dissolved concentrations of metals (Cu, Fe, Zn, Pb, and V) in the storm runoff from the burned area (Pinedo-Gonzalez *et al.* 2016). After the Cerro Grande Fire in New Mexico, Gallahar *et al.* (2002) observed an increase in particulate metals in runoff but no increase in dissolved metals. Outside of this limited number of studies, the impact of wildfire on metal loadings remains largely unknown.

With an understanding of how fires remove vegetation, alter soil properties, and lead to increased post-fire runoff, it is still unclear what type of water quality impact should be expected after a fire. Each study on an individual fire reveals how that fire impacted the water quality downstream. However, sometimes findings from individual fires are inconsistent or contradict one another (Smith *et al.*, 2011). Individual fires do not reveal the general impact of fires on water quality across the western U.S. Utilizing public databases with historic water quality data, information was assembled to evaluate changes for a range of 89 water quality parameters in 153 rivers before and after 172 wildfires. Our goal is to better understand how fires generally alter physical and chemical water quality in receiving streams and the duration of the water quality impact. Through using a large sample size, we investigate how forest fires impact concentrations of suspended and dissolved materials in receiving streams, beyond the localized stream impacts that are the focus of the current literature.

3.2 Methods

3.2.1 Fire data

Information on recent wildfires in the Western United States was acquired from the Monitoring Trends in Burn Severity (MTBS) website (www.mtbs.gov). The MTBS website provided burn perimeters, burn area, and burn severity data for each of this study's 24,042 fires from 1984 to 2015, from available satellite data. Fire perimeters were used to locate USGS stream gauges below fires. All USGS gauge stations and water quality sampling sites were mapped in ArcMap 10.1 to check their locations relative to fires, verifying all water quality samples were downstream of a fire. The USGS stream gauge locations were used to define the watershed boundaries and watershed area for each of the study's burn-impacted watersheds across the Western U.S. (Figure 3.1). A summary of the general physical attributes of each of the fires and watersheds that are the focus of this study is presented in Table 3.1.

3.2.2 Water quality data assembly

Pre- and post-fire water quality data were acquired by data mining federal and state water quality databases. The federal sites used were the USGS National Water Information System: Web Interface (NWIS) and EPA STORET (EPA Storage and retrieval and water quality exchange). In addition to searching national websites for available water quality data, western U.S. state public health and environmental department websites were also searched for chemical and physical water quality data; when available, data from before and after a fire were obtained. Data from the USGS NWIS were collected in the field, as grab samples, and by continuous data loggers. All data collected by the USGS follows their accepted methodology and has been finalized by the USGS. The water quality data available on EPA's STORET site and from individual state agencies has been collected in the field following accepted methodology and protocols. Water quality sampling methodologies from different agencies varied in some cases.

Sampling frequency also varied for each site, some sites had semi-annual or quarterly data while other sites collected samples more frequently. Chemical and physical water quality data from different sources was then formatted in R and assembled into a database (using Microsoft Access). All dates for water quality sampling events were synchronized to water years (WY) (October 1- September 30). Water quality data for each sampling station were plotted over time to check data for obvious typographic errors, to confirm that sampling occurred throughout the five years before and after fire, and to verify results were not reported below instrument detection limits before inclusion in the database. The resultant database included fire, chemical, and physical water quality data from multiple sources with two million water quality data-points for western wildfires from 1984-2015. A total of 89 separate chemical and physical water quality parameters were included in the database and evaluated in this analysis.

Mean daily flow from the USGS gauge stations was used along with concentration data to calculate the loading rate or flux for each of the water quality parameters. Less than 10% of the USGS gauges had missing flow data for the pre- and post-fire periods of interest. To calculate flux, mean daily flow was multiplied by the concentration of each analyte measured on the same day to give an estimate of the instantaneous rate at which the load was passing the water quality sampling station. Flux was calculated to address seasonality, climate differences, and the variability of flow that alters analyte concentrations among sites across broad regions and time periods. Flux is equally as important as concentration data when evaluating the potential impacts of water quality on an aquatic system (Helsel and Hirsch, 1991). Both changes in analyte concentration and flux are analyzed in this study.

3.2.3 Data selection

The assembled fire-water quality database was screened to meet minimum pre- and post-fire sampling frequency for analysis. The pre-fire, or baseline, period was defined as the five years prior to fire. All watersheds included in this study had a minimum of ten water quality data points over the five- year period preceding fire to define baseline water quality conditions. Post-fire water quality was defined as the first five years after a fire. To evaluate water quality impact from fire, a minimum of ten records five years after fire were used to define post-fire water quality conditions. Pre-fire water quality conditions were compared to post-fire conditions; mean and median analyte concentration and mean and median flux values were compared and used to calculate the percent change in analyte concentration and flux after fire (Table 3.2). Synopsis data for all analytes evaluated in this study are included in Table 3.2; the number of fire impacted watersheds that had adequate data (according to screening criteria) for each analyte is also listed.

If there were multiple fires in a watershed in a single year, the burned areas were summed together and the total area was used to calculate the percent of the watershed area burned. If a fire consumed less than 5% of the watershed area, the fire would likely be too small to show meaningful effect on the water quality, therefore, these fires were eliminated from the analysis. When there was more than one fire in a watershed over a five-year period, a clear baseline pre-fire condition could not be established; therefore, those fire-events were also eliminated from this analysis. Consequently, a minimum of five years between fire events was required for the wildfire event to be included. Between the minimum criteria for water quality data points and the required minimum five years between fires, just 172 out of the 24,042 fires were deemed acceptable. In summary, 172 fires which impacted 153 western U.S. watersheds are utilized for analysis in this study.

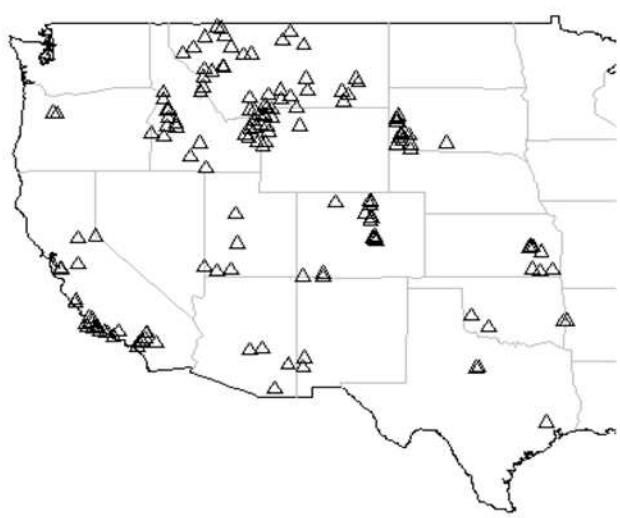


Figure 3.1. Map of 153 fire-impacted watersheds in the western United States with sufficient pre- and post-fire water quality data used in this analysis.

Table 3.1. Physical characteristics of watersheds used in this study. Aridity Index is defined as the ratio of mean annual precipitation to mean annual potential evapotranspiration.

	Watershed Latitude, (deg.)	Watershed Elevation, (m)	Aridity Index	Burn Area (ha)	Percent of Watershed Burned
mean	40.5	1642	0.63	204,453	18
median	40.6	1773	0.56	71,225	11
standard dev.	5.1	737	0.28	350,750	17.5

3.2.4 Subset analysis

In water quality monitoring, a higher frequency of sampling is generally required to detect trends in physical and chemical measures (Kirchner et al, 2004). In the current study, fire-impacted watersheds with higher frequency of sampling, such as monthly sampling, were selected from the fire-water quality database. Ten fire-impacted watersheds with monthly water quality sampling data for five years preceding and following a fire were sub-selected for analysis, Table 3.3 lists information on four of these fires highlighted in this study. Data from this group was evaluated to look at the impact of hydrology and climate on water quality response. Daily gridded precipitation data for each selected watershed was retrieved from the PRISM Climate Group website (www.prism.oregonstate.edu). Precipitation data was paired with daily flow data from the USGS to assess how rain and flow events influence analyte concentrations. Analyte concentration data from each of the subset watersheds were plotted over time to detect changes in analyte concentrations, providing a means to evaluate trends over time and shifts in mean analyte concentration after fire.

3.2.5 Statistical analysis of entire dataset

Using water quality data from all 172 fires, mean, median, and standard deviations were calculated for each analyte concentration and flux at each water sampling station over the five years pre-fire and compared to the same statistics five years post-fire. The average of the mean and standard deviations from all water sampling stations for each analyte was calculated to summarize general response to fire (Table 3.2). The percent change in concentration and flux was then calculated for each analyte for each of the 172 fires.

To determine if there were any detectable changes in analyte concentration and flux after fire, non-parametric statistical techniques were used, thereby not assuming normal distribution of the water quality data. Assumptions for the Mann-Whitney *U* test were met (Tamhane and Dunlop, 2000) and the statistical test was performed for each analyte, comparing the difference in median values (or distribution) of concentration and loading rate, or flux, before/after the fire. The Mann-Whitney *U* test produces a *U* value and the probabilities of producing a *U* value from one population (pre-fire) larger than the other population (post-fire) have been tabulated and provided as a p-value (Mann and Whitney, 1947). If the p-value is less than the alpha value ($\alpha = 0.05$), a significant difference is detected and the null hypothesis is rejected. Here, the null hypothesis was that the distribution of pre- and post-fire analyte concentrations and fluxes were equivalent, with a difference of zero. The Mann-Whitney *U* test is a rank sum test, reducing the impact of outliers (Helsel and Hirsch, 1992).

3.2.6 Statistical analysis for selected subset fires

To evaluate trends in water quality after fire, change-point analysis was conducted on the subset of ten burned watersheds with regular and frequent monitoring data. Change-point analysis detects a shift in mean concentration over time while controlling the change-wise error rate and provides a confidence interval for the time of change (Taylor, 2000). This method was applied to identify abrupt changes in the mean concentration of various analytes. Change-point analysis is also considered robust to outliers and is better at identifying abrupt changes in data than trend analysis (Taylor, 2000). Though, linear trend tests were also conducted in this study, by computing Kendall's Tau and Spearman's Rho (Helsel and Hirsch, 1992), no general linear trend in analyte concentration was detected over time, so no further results or discussion are shown on linear trends.

Within the subset of ten fires with high frequency sampling, time series data was sometimes irregular (monthly samples were not always taken exactly 30 days apart); however, a change-point detection technique was applied for use with irregular interval data (Gurarie *et al.*, 2009) known as behavioral change-point analysis (BCPA). This technique sweeps a "window" over the dataset and identifies the most likely change-points (Gurarie *et al.* 2009). BCPA analyzes indiscriminate time series data that by its nature is assumed to be autocorrelated and Gaussian with a mean, a standard deviation and an autocorrelation coefficient. When autocorrelation coefficient values are low ($\rho < 0.05$) the analyte concentration data points in the time series are considered not autocorrelated and are significantly different from one another. A sub-sampling window size of 40 sequential analyte concentration data points was used; this exceeded the minimum sample size required for the Bayesian Information Criterion (BIC) used in model selection. This technique detects shifts in the concentration values at a small temporal scale within BCPA (Gurarie *et al.* 2009). The BIC model was fitted through the analyte concentration data, as shifts in mean concentration were detected a change-point was identified. Change-point dates were compared to the fire date for each watershed in the subset analysis. Assumptions of the model were verified by inspecting the Q-Q plot and the autocorrelation function. The technique was implemented in R with code provided by Gurarie *et al.* (2009).

3.3 Results

Results show that water quality changes of one form or another are common after forest fire across the 172 fires that were analyzed. Concentrations of most dissolved nutrients, major ions, and metals significantly decreased in about one third of the fires. While the concentrations of dissolved constituents decreased, their flux or loading rate over the five years following fire still significantly increased due to increased flows. In contrast to the dissolved constituents, post-fire concentrations of suspended nutrients and metals increased after roughly one quarter to one half of the fires; loading rates also increased. Table 3.2 presents a comparison of average concentrations and fluxes for each analyte before and after fire. For each analyte, the number of watersheds that met data criteria is listed in the second column and the percentage of fires where there was a statistically significant response (where p-value < 0.05 from the Mann-Whitney U test) is listed in the last column. To detect the most common, significant changes in water quality after fire, analysis for this study focused on analytes where there was sufficient data for 20 or more watersheds. There were fewer watersheds with sufficient total and dissolved metals data, nevertheless they will be reviewed to expose some interesting results.

3.3.1 Nutrients

Post-fire nutrient concentrations showed the most consistent response. Concentrations of all particulate forms of nitrogen (organic nitrogen, organic nitrogen plus ammonia, nitrate plus nitrite, and total nitrogen) significantly increased in concentration and flux in the five years following fire in one-third of the burned watersheds. Meanwhile, concentrations of some dissolved forms of nitrogen tended to decrease significantly in 30-45% of the burned watersheds (Table 3.2). The particular results are as follows: dissolved organic nitrogen and total nitrogen concentrations decreased after 37% of fires. Concentrations of dissolved ammonia also decreased in 45% of the fire-impacted watersheds. Notably, some dissolved nutrients tended to increase in concentration five years after fire, they were nitrite, nitrate and the combination of soluble nitrate

and nitrite. Dissolved nitrite and nitrate increased in concentration and loading rate in response to 24-33% of the fires.

Furthermore, both dissolved and particulate forms of phosphorus increased significantly in one out of three fires. Orthophosphate, total phosphate, dissolved phosphorus and particulate phosphorus each increased in concentration and loading rate in the five years following a fire. On average, in 46 burned watersheds, orthophosphate and total phosphate concentrations increased by 30 percent and flux increased by almost 4000 percent (Figure 3.2). The percent change in flux includes the increased flow after fire and increased concentration, resulting in a more obvious response after fire. From this analysis, it is possible to conclude that while some forms of dissolved nutrients concentrations decreased, concentrations of nitrate, nitrite, phosphate and phosphorus all increased and loading rates of all dissolved and particulate nutrients significantly increased after one-third of the studied fires.

3.3.2 Major ions

Changes in major ion concentrations after fire demonstrate a similar pattern: in one out of three fires, most dissolved ions decrease in concentration but increase in flux. Specifically, concentrations of inorganic major ions such as sulfate, calcium carbonate, dissolved calcium, dissolved magnesium, and hardness all tend to decrease in concentration while still increasing in

Table 3.2. Summary of average analyte concentration and flux data for all fires. The first column lists number of watersheds with sampling events that meet minimum data criteria. Rows where twenty five percent or more watersheds changed significantly after fire are highlighted in light gray, fifty percent or more are highlighted in dark gray.

Analyte	Number of Fires with WQ Data	Mean Conc. Pre-fire	Mean SD Pre-fire	Mean Flux Pre-fire	Mean Flux SD Pre-fire	Mean Conc. Post-fire	Mean Conc SD Post-fire	Mean Flux Post-fire	Mean Flux SD Post-fire	Perc. Change Mean Conc.	Perc. Change Mean Flux	% Fires Sig. Change MWU
SpC.uS/cm	155	646.7	188.9	NA	NA	701.7	433.4	NA	NA	22.8	NA	43.3
Temp_Water	171	12.5	6	NA	NA	12.2	5.8	NA	NA	-0.9	NA	16.3
pH	66	8	0.3	NA	NA	8	0.3	NA	NA	0.3	NA	26.3
ANC.mg/L	358	146.8	26.5	270098	164756	130.6	24.2	250550	225227	-5.9	338.4	20
Alkalinity.mg/L	2	94.5	23	584	1741	69.6	10.2	5366	10543	-26.4	818.1	100
DO.mg/L	53	9.6	1.5	NA	NA	9.8	1.6	NA	NA	3	NA	15.6
DO.%.Sat	53	101.1	11.8	NA	NA	101.8	11.9	NA	NA	1.3	NA	21.3
Hardness.mg/L	47	265.4	68.5	165194	107420	255.1	71.5	157303	141016	-3.4	171	31.4
HCO3.mg/L	5	112.6	16.7	726130	474442	104.4	17.6	337041	236497	-11.4	69.4	28.6
CO3.mg/L	2	1.4	2.5	13297	3064	1.2	1.1	1376	2465	-36.9	-32.5	0
CO2.unfilt.mg/L	40	3.2	2.4	6047	16686	2.6	1.5	3023	4066	-7.5	460	20.7
Turb.NTU	13	12.8	22.4	NA	NA	702.9	1354.4	NA	NA	2350.4	NA	15.8
TSS mg/L	3	2.8	4	4216	21891	12.9	20.4	78831	366109	244.6	89.3	25
TDS	3	257.9	56.9	NA	NA	226.8	51.5	NA	NA	-7	NA	0
Ca.filt.mg/L	47	60.3	15	44238	28944	56.3	15.1	42016	37436	-3.5	193.6	26.1
Cl.filt.mg/L	46	67.9	29	36631	25888	67.8	34	39821	43998	-1.3	176.8	20
Mg.mg/L	46	57.1	21.1	22020	20032	33.7	12	19010	14095	-25.8	358.6	28.2
Na.filt.mg/L	40	82.1	34.5	35526	24357	83	34.6	43403	38476	5.9	225.8	28.8
Na.fraction.cations%	36	30.8	3	NA	NA	30.1	3	NA	NA	-2.2	NA	24.1
SAR	39	1.8	0.5	NA	NA	1.8	0.6	NA	NA	-6	NA	24.2
K.filt.mg/L	36	4.1	1	3723	2618	4.1	1	4617	3568	2.4	924.7	28.8
SO4.filt.mg/L	46	176.6	63.7	53560	34249	173.9	66.3	53302	44445	0.4	187.8	30.4
OC.filt.mg/L	4	5.2	2.2	11276	17813	5.3	3.7	13698	18042	23.3	32.3	16.7
OC.unfilt.mg/L	5	6.6	2.7	13883	20749	8.4	3.2	13022	19121	42.9	164.8	28.6
Org.N.filt.mg/L	7	0.4	0.1	629	980	0.3	0.1	405	697	-6.7	75.6	37.5
Org.N.unfilt.mg/L	40	0.6	0.5	1423	2154	1.6	2.8	4583	7040	150.2	1056.2	31.9
NH3.filt.mg/L.N	46	0.1	0.1	111	148	0.1	0.1	84	109	2.9	216.9	45.2
NH3.unfilt.mg/L	14	0.1	0.1	131	161	0.1	0.1	154	263	66.9	438.8	22.2
NH3plusOrgN.filt.mg/L	7	0.4	0.2	676	1050	0.3	0.1	460	813	-4.3	97.8	25
NH3plusOrg.N.unfilt.m	31	0.6	0.4	1836	2655	0.8	1	2220	3669	43.2	182.7	35.3
NO2filt.mg/L.N	31	0	0	23	23	0	0	26	29	1	176.4	23.4
NO3.filt.mg/L.N	32	0.5	0.3	508	345	0.5	0.4	579	559	75.5	406.9	24.5
NO3plusNO2.filt.mg/L	47	0.6	0.3	525	427	0.7	0.4	735	702	63.6	3554.4	33.3
NO3plusNO2.unfilt.mg	4	0.7	0.3	850	699	0.6	0.5	228	381	502.3	451.8	28.6
TN.filt.mg/L	7	1.7	0.6	1454	1682	1.2	0.5	730	1146	-5.8	75.4	37.5
TN.unfilt.mg/L	32	1.4	1.3	3072	5022	1.4	1.1	2922	3880	35.8	150	34.2
OP.filt.mg/L	47	0.3	0.2	248	242	0.3	0.2	294	301	32.2	3872.4	34.8
PO4.unfilt.mg/L.P	46	0.1	0.1	82	80	0.1	0.1	98	100	35.1	3974.4	33.3
P. filt.mg/L	20	0.1	0.1	155	179	0.1	0.1	165	200	5.9	197.1	18.2
P.unfilt.mg/L	46	0.3	0.4	397	716	0.5	1.1	601	1706	265.8	257.6	34.6
Ag.filt.ug/L	13	0.8	0.1	3	3	0.7	0	3	3	-26.6	33.4	15.8
Ag.unfilt.ug/L	4	0.6	0.2	1	1	0.4	0.1	0	0	-43.8	-55.6	50
Al.filt.ug/L	14	36	69.5	109	197	24	25.7	69	103	69.4	266.1	27.3
As.filt.ug/L	21	4.2	1.6	11	7	4.9	2.5	11	7	0.1	83.9	28.1
As.unfilt.ug/L	20	29	12.3	46	21	30.4	18.3	47	27	50.9	149.1	29.2
Ba.filt.ug/L	13	44.4	12.9	142	108	43.7	12.1	98	80	1.7	99.5	30
Ba.unfilt.ug/L	7	102.7	90	112	118	158	235.3	158	383	100.5	281.3	33.3
Be.filt.ug/L	13	0.7	0.5	2	3	0.2	0.1	1	2	-48.6	-2.4	62.5
Bo.filt.ug/L	17	223	59.4	30	31	199	55.1	34	39	-4.9	428.2	21.9
Bo.unfilt.ug/L	8	102.1	39.9	78	56	102.6	62.2	109	130	17.5	208.4	18.2
Cd.filt.ug/L	16	0.6	0.2	3	3	0.4	0	2	3	-15	53.2	33.3
Cd.unfilt.ug/L	11	0.5	0.2	1	2	0.2	0.3	0	1	-28.2	72.8	60
Cr.filt.ug/L	14	2.1	1.2	4	5	1	0.7	6	11	-3.5	90.5	20.8
Cr.unfilt/ug/L	12	4.5	4.5	6	8	4.1	7.3	4	8	40.4	9.9	46.7
Co.filt.ug/L	8	3	0.5	15	13	2.3	0.2	14	14	-22.7	-8.5	20
Cu.filt.ug/L	27	2.9	2.2	10	19	2.2	1.4	7	11	-1.9	68.4	30.6
Cu.unfilt.ug/L	18	10.8	12.9	16	24	15.6	31.1	21	53	79	228.3	36.4
F.filt.mg/L	38	0.5	0.1	557	366	0.5	0.1	602	583	-0.2	257.1	23.8
Fe.filt.ug/L	28	42	74.1	64	120	41.5	27.8	47	60	77	9076.7	19.1
Fe.unfilt.ug/L	11	2222.6	2604.6	2360	3739	3165.1	5773.2	2685	5100	179.2	244	23.8
Li.filt.ug/L	8	32.5	8.4	148	66	34.6	8.4	108	56	-8.4	150.4	15.4
Mo.filt.ug/L	9	8.4	1.2	47	40	7.3	0.7	43	42	-10.5	-6.9	18.2
Mn.filt.ug/L	32	40.2	28	21	23	28.8	31.4	13	15	37.1	182	32.7
Mn.unfilt.ug/L	19	209.8	290.3	191	301	1463.4	3653.9	966	2331	497.1	5089.4	20
Ni.filt.ug/L	19	6.4	6.3	10	18	3.3	2.6	5	7	-14.7	63.6	11.1
Ni.unfilt.ug/L	16	7.2	7.9	6	9	12.1	19.2	11	24	117.1	203.5	22.2
Pb.filt.ug/L	18	2.5	2.3	8	12	0.7	0.5	3	4	-26.7	26	50
Pb.unfilt.ug/L	16	17.8	25.1	13	25	50.3	127.7	40	121	227.6	633.4	32
Se.filt.ug/L	22	1.3	0.5	2	2	1.2	0.5	2	2	-2.8	130	28.6
Se.unfilt.ug/L	19	1.4	0.7	1	1	1.3	0.7	1	1	8	86.6	38.1
Si.filt.mg/L	32	16	6.7	22467	19473	15.1	2.8	22133	21209	-1.6	354.5	14.5
Sr.filt.ug/L	7	442.6	98	1173	641	561.4	105.3	922	606	-2.5	234.4	27.3
Zn.filt.ug/L	25	10.3	9	36	53	6.5	4.8	20	29	-2.1	Inf	17.1
Zn.unfilt.ug/L	17	40.5	55.5	44	62	88.6	229	72	199	85.1	262.8	24

flux in the five years following fire 28-31% of the time (Table 3.2). However, dissolved sodium and potassium significantly increase in both concentration and flux in 28% of more than 36 burned watersheds.

3.3.3 Metals

The final grouping of analytes to examine are the dissolved and particulate metals. Fewer watersheds had adequate data for total and dissolved metals. In many cases fewer than 20 fire-impacted watersheds had enough data for metals before and after a fire. When there was a statistically significant change, dissolved metals decreased in concentration after fire while particulate forms increased, and loading rates of both types increased (Table 3.2). The specific results include: dissolved copper concentrations decreased in the five years following fire by 1.9% on average 30% of the time while particulate forms of copper increased in concentration by 79% after 36% of fires. Selenium followed the same pattern, concentration of dissolved selenium decreased in greater than 28% of the burned watersheds while particulate selenium concentrations increased after 38% of the fires. Dissolved zinc also slightly decreased in concentration by 2% after 17% of fires while total zinc concentrations increased by 85% in 24% of the fire-impacted watersheds. Arsenic, manganese, and iron were exceptions to this pattern, both total and dissolved arsenic significantly increased in concentration after more than 28% of fires. Dissolved manganese increased by 38% concentration in 32% of the fires while particulate manganese dramatically increased by 497% after 20% of the fires. Similarly, dissolved and total concentrations of iron increased after fire. In evaluating the other metal species where there were data from fewer than 20 watersheds, the pattern continues, dissolved concentrations decrease while particulate forms increase. In almost all cases, whether dissolved or particulate, flux increased regardless of the direction of change in metal concentration.

3.3.4 Subset analysis

To evaluate post-fire water quality response for trends, a more in-depth analysis was needed. A subset of fires was selected where water quality sampling before and after fire was more frequent, such as monthly water quality monitoring. This provided an opportunity to compare changes in analyte concentrations and loading rate, with the watershed's hydrology. Information on four of the ten selected subset fires, name, date, size and total water quality samples taken five years before and after fire are presented in Table 3.3.

Table 3.3. Summary of characteristics for selected subset of fires and the receiving streams where there was high frequency water quality sampling.

River	U.S. State	Fire Name	Fire Date	Fire Size (ha)	Distance from Fire to WQ Sampling Station (km)	# Samples 5 years Pre-Fire	# Samples 5 years Post-Fire
Boise	Idaho	Foothills	8/19/1992	104004	24.9	75	135
Santa Ana	California	Old	10/25/2003	36940	14.3	56	67
South Platte	Colorado	Hayman	6/8/2002	55750	42.2	118	58
Cache La Poudre	Colorado	High Park	6/9/2012	35323	27.3	60	37

Evaluating the subset watersheds revealed a more immediate short-term water quality response after fire that was masked when data was averaged over the five years.

The first fire highlighted in this study is the 1992 Foothills fire which impacted the Boise River. The Boise River is a snow driven hydrologic system where there was monthly water quality monitoring for a long period before and after the fire. Immediately following the Foothills fire, dissolved nitrate and nitrite and total phosphorus increased dramatically, raising the mean concentration of those analytes for three years. Runoff from snowmelt events after the fire resulted in a large flow response and corresponding elevated nutrient concentrations (Figure 3.2). Change-point analysis identified a significant decrease in the mean nitrate and nitrite, and mean total phosphorus concentrations at the end of the third year (Figure 3.2). Overall, when this watershed was evaluated with the larger data set, the nitrate and nitrite concentrations declined by 39% percent in the five years following fire, the immediate increase in concentrations the first three years are masked by the overall five-year trend. Similarly, total phosphorus showed a spike in concentration in the first months after fire following runoff events, but an average 41% decrease in concentration over five years. Dissolved potassium also exhibited the same pattern, immediate increase after the Foothills fire with an overall five-year decrease in concentration of 28%. Despite the overall decrease in concentrations over five years, loading rates of nitrate and nitrite, total phosphorus, and potassium all increased after the fire due to increased discharge of the Boise River.

The fire-altered landscape led to higher short-term concentrations of dissolved nitrate and nitrite, total phosphorus, and dissolved potassium. In the water year that followed the large fire, peak and annual total flows were 30 and 50 percent below average (total annual flow WY 1993 5694 m^3, 13,816 m^3 average annual flow). The elevated concentrations of nutrients and ions were not due to large precipitation and flow events but instead due to the fire-altered landscape's reduced infiltration, altered vegetation and large volume of ash. As a result, below average annual precipitation (470 mm in WY 1993 compared to 830 mm average) delivered concentrations of ions and nutrients 2-5 times larger than pre-fire periods, which was detected in the change-point analysis. In the 2003 Old Fire in southern California, mean chloride and sulfate concentrations significantly decreased immediately before and following the fire, resulting in a statistically significant change of 9%, until the change-point technique identifies a shift in the mean concentrations three years after the fire, where concentrations increased (Figure 3.3). The Santa Ana River during WY 2004 had the highest peak and annual total precipitation (104 mm and 1012 mm compared to the average peak of 48 mm and average total of 430 mm). The wet year flushed chloride and sulfate from the burned watershed and concentrations increased during precipitation events. In this semi-arid, precipitation driven watershed in southern California, flows and salt concentrations increased quickly after precipitation events.

The South Platte River in Colorado was downstream of the largest fire in Colorado's history, the 2002 Hayman fire. Here, the change-point analysis identifies an increase in total phosphorus concentrations over a three-year period caused by an immediate spike in phosphorus concentration after the fire, which raised the overall mean concentration for the three-year period (Figure 3.4). The dramatic increase in total phosphorus following the fire occurred on a lower than average annual flow year, where total annual flows were 30% of average the year of fire (WY 2002) and 50% of average the following water year. The spike in total phosphorus corresponded with a large precipitation event the year after fire. Despite lower than average flows, due to the large precipitation event, concentrations of phosphorus were three times larger than average. Again, the total phosphorus spike was due to flushing of ash, sediment, and organic material after the fire. The change-point model demonstrates a decrease in phosphorus concentration a year and a half after the fire. Once more, the higher temporal resolution water quality sampling allowed detection of higher resolution temporal changes in phosphorus concentrations after fire.

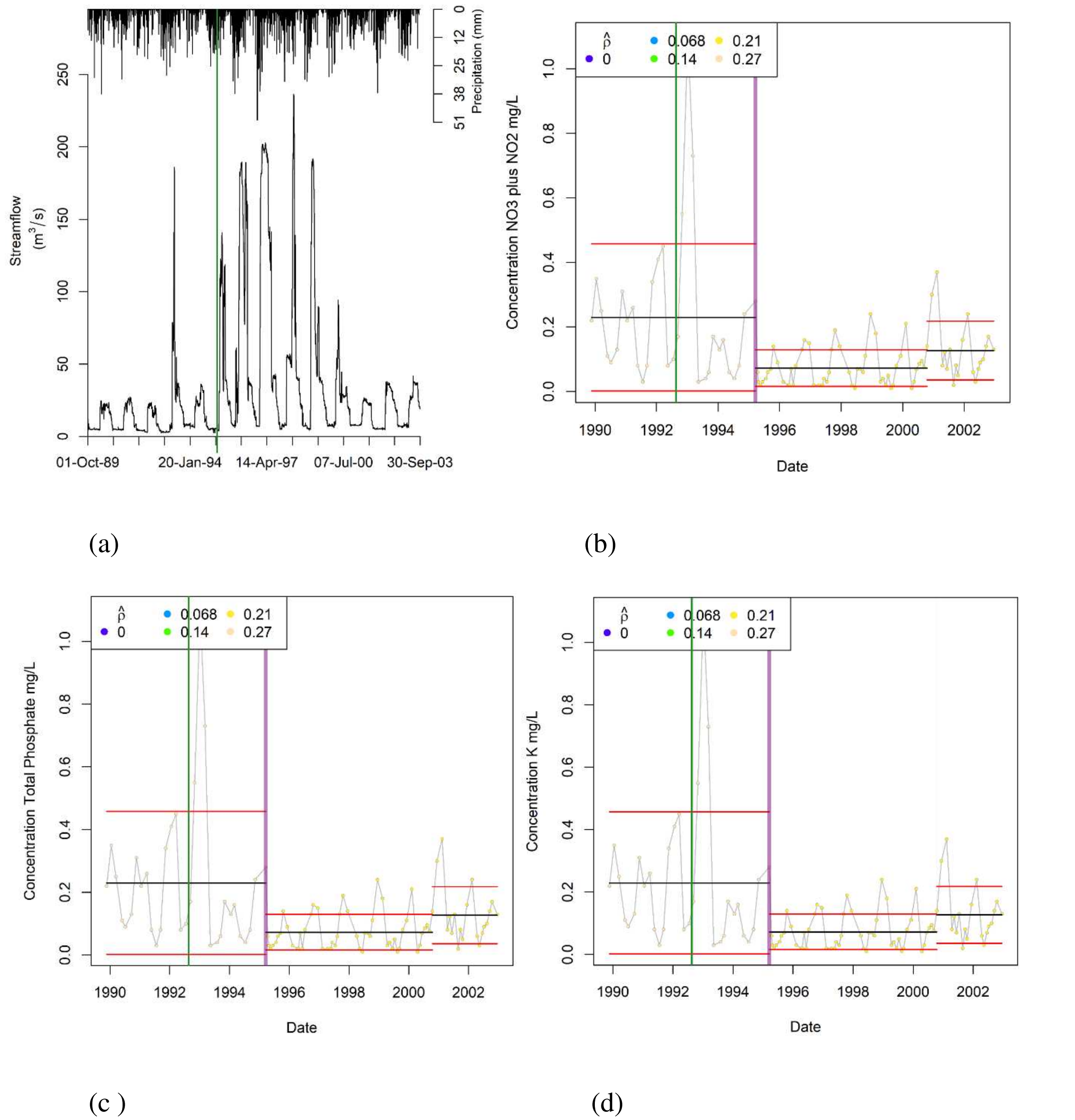
The High Park fire of 2012, also in Colorado, impacted the Cache La Poudre River. Total iron concentrations increased dramatically six months following the fire, from below 500 µg/L to 1,500 µg/L (Figure 3.5). A change-point is detected almost exactly on the fire date, and another change-point occurred one year later as the peak total suspended iron concentrations increased to 4,000 µg/L. The first year after fire, WY 2012, had peak and annual flows that were 50 percent below average; despite low peak and annual flow years, peak concentration of particulate iron was five times larger than in previous years. The elevated iron concentration was not a result of increased flow volume in the Cache La Poudre the first year after fire, however, iron concentrations remained elevated in the five years post-fire and coincided with high peak flows. The higher suspended iron concentrations immediately after the fire and in the years that follow could be flushing events, where iron bound to sediment from steep slopes was washed down during snowmelt runoff events.

3.4 Discussion

Our analysis utilized public data that enabled us to assemble a fire-water quality database for a large number of fires. This made our study unique in that we evaluated water quality response after 172 fires, with a more in-depth analysis for a select subset of these fires.

3.4.1 Nutrients

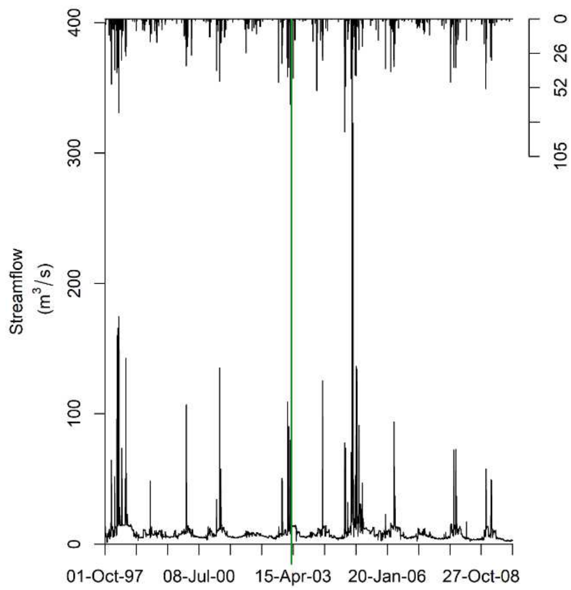
Evidence from this study suggests that concentrations of dissolved nitrate, nitrite, orthophosphate, and phosphorus increased after about 33 percent of fires. Additionally,



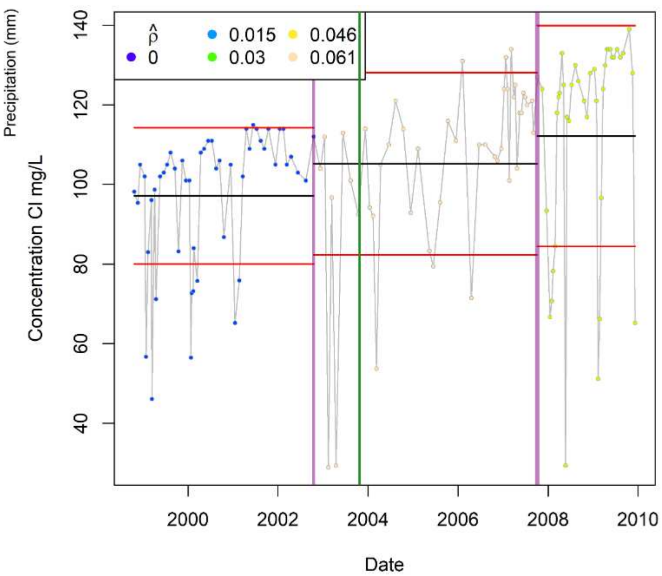
(a) (b)

(c) (d)

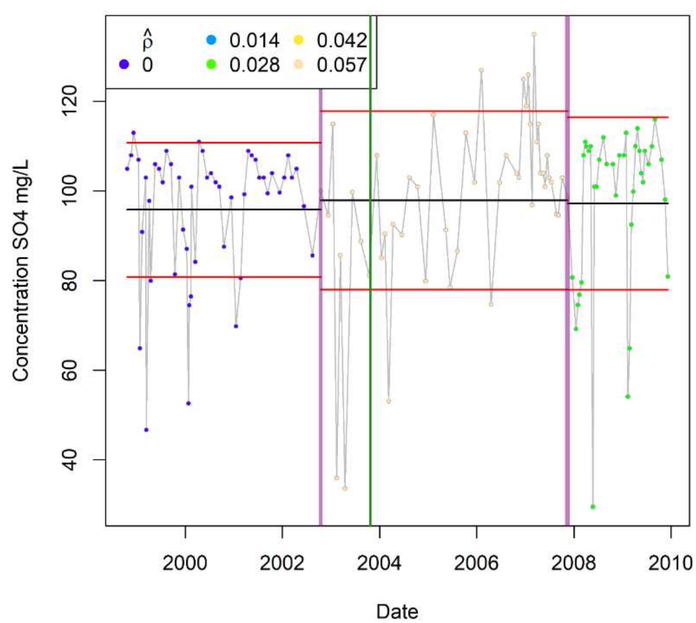
Figure 3.2. Hydrograph and and hyetograph (a), and flat model change-point graphs for nitrate plus nitrite (b), total phosphate (c), and dissolved potassium concentrations (d) with data from the Boise River downstream of the Foothills Fire. Fire date is indicated with the vertical green line, purple lines show change-points, red lines show confidence intervals for the BIC model for measured concentration data. The autocorrelation coefficients for the time series analyte concentration data are displayed in the top left corner of each changepoint graph.



(a)

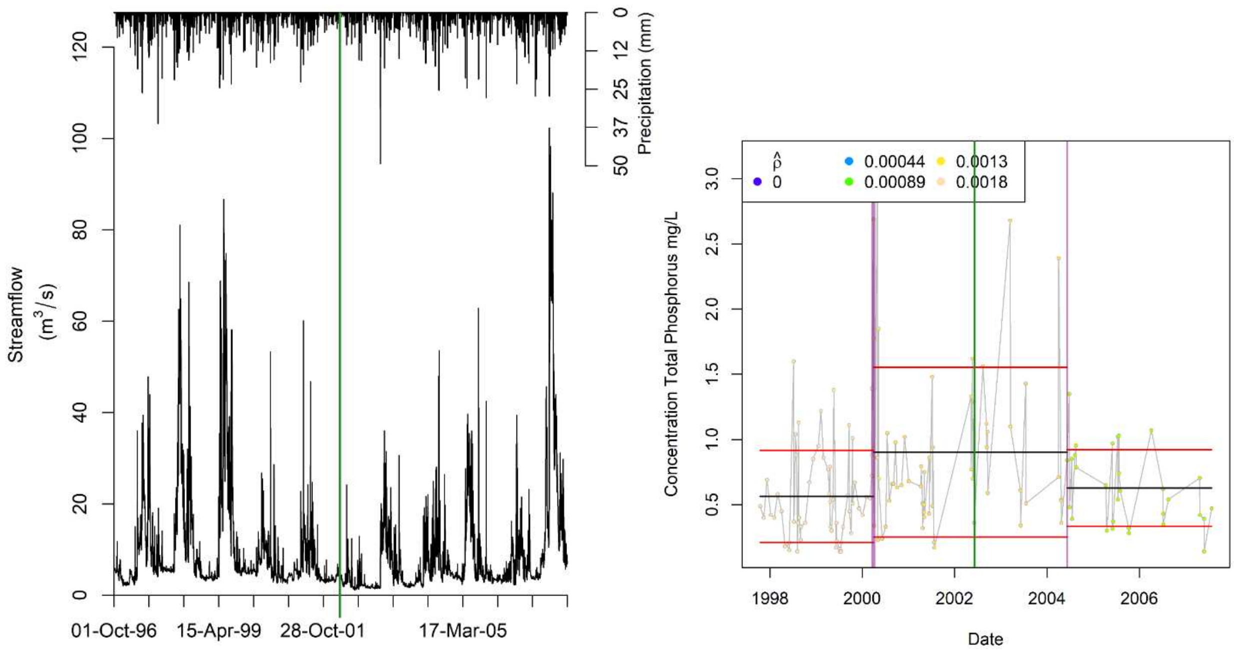


(b)



(c)

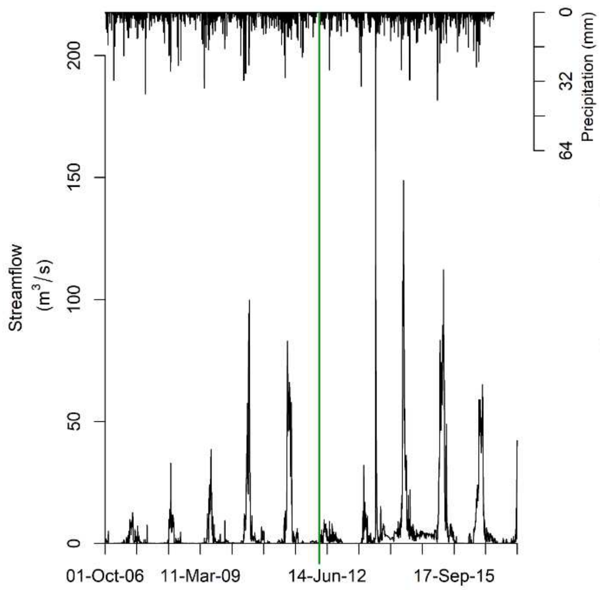
Figure 3.3. Hydrograph and hyetograph (a), and flat model change-point graphs for chloride (b) and sulfate (c) concentrations with data from the Santa Ana River downstream of the Old Fire. Fire date is indicated with the vertical green line, purple lines show change-points, red lines show confidence intervals for the BIC model for measured concentration data. The autocorrelation coefficients for the time series analyte concentration data are displayed in the top left corner of each changepoint graph.

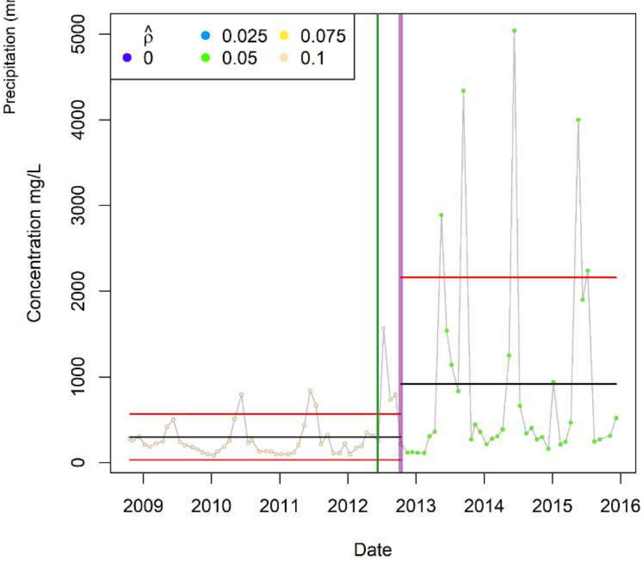


(a) (b)

Figure 3.4. Hydrograph and hyetograph (a), and a flat model change-point graph for total phosphorus (b) concentrations with data from the South Platte River downstream of the Hayman Fire. Fire date is indicated with the vertical green line, purple lines show change-points, red lines show confidence intervals for the BIC model for measured concentration data. The autocorrelation coefficients for the time series analyte concentration data are displayed in the top left corner of each changepoint graph.

concentrations of particulate nitrogen, phosphate and phosphorus also increased in the five years after fire. When taken with the increases in stream discharge after average rain and snow events, the loading rate of all forms of nitrogen and phosphorus significantly increased one third of the time. However, it is clear from the subset analysis that averaging analyte concentration data over the five-year period could be muting even more dramatic spikes in dissolved and total nitrogen and phosphorus concentrations so that the frequency of nutrient response immediately after fire could be even higher.





(a) (b)

Figure 3.5. Hydrograph and hyetograph (a) and a flat change-point model (b) for total iron concentrations with data from the Cache La Poudre River downstream of the High Park Fire. Fire date is indicated with the vertical green line, purple lines show change-points, red lines show confidence intervals for the BIC model for measured concentration data. The autocorrelation coefficients for the time series analyte concentration data are displayed in the top left corner of each changepoint graph.

Higher loadings of nitrogen and phosphorus species can impact the level of treatment necessary by water providers. Surface water used for agricultural irrigation may be loaded with nutrients, unbeknownst to the irrigator, adding excessive nitrogen and phosphorus to crops. An increase in nutrient loading can also alter the aquatic ecosystem by accelerating eutrophication, which is important information for recreational users and resource managers. Deposition of particulate matter, including the nutrients discussed here associated with sediments, also alters the aquatic habitat and food web for macroinvertebrates (Minshall, 2003).

Other studies on individual fires agree with our findings and have also observed an increase in nutrient concentrations and loadings after fire (Belilas and Rodá 1993; Bladon *et al.* 2008; Earl and Blinn 2003; Mast and Clow 2008; Ranalli 2004; Rhoades *et al.* 2011; Riggan *et al.* 1994). However, our study utilizes data from hundreds of fires to investigate systematic impacts on water quality across the western U.S. Nutrients may be volatilized during the combustion of the fire, stay in particulate form and travel to the waterway via erosion, or they may be leached by precipitation following fire (Ranalli 2004). In some studies, the increase in nitrate concentrations were short-lived and were elevated only during the first year following a fire (Earl and Blinn 2003) while others observed elevated concentrations for 3-6 years after fire (Bladon *et al.* 2008; Rhoades *et al.* 2011). Two studies did not observe an increase in any form of nitrogen concentrations but instead observed increased exports of dissolved and particulate phosphorus for four to seven years following fire (Burke *et al.* 2005; Emelko *et al.* 2016). In the Emelko *et al.* study, the phosphorus was associated with sediments that were mobilized from the denuded landscape after the fire for seven years (2016). Our findings echo what other studies have found on individual fires and extrapolate them out to a broader region.

Leaching of dissolved nutrients through subsurface and delivery of nutrients in the particulate form from erosion after precipitation events was illustrated in our results both on a broader scale and in the highlighted, subset fires – specifically the Foothills fire in Idaho and the Hayman fire of Colorado. In these examples, upon closer examination, where water sampling occurred at a higher frequency the post-fire spikes in nutrient concentrations occur even in watersheds where the overall five-year average concentrations of those nutrients declined. Many of this study's burned watersheds are located in high elevation mountainous regions with steep topographic relief, where rain driven erosion and a steep hydrologic gradient deliver nutrients and sediment directly to a stream. Elevated total and dissolved nitrogen species have been observed in studies in similar regions (Bladon *et al.* 2008; Rhoades *et al.* 2011). Elevated dissolved nitrate and nitrite indicate that nitrification mediated by bacteria is still occurring in the soils after fire. Additionally, total and dissolved ammonia concentrations and loading rates increased in 30-45 percent of fires in this study, likely due to the augmented dead organic material available after fire and reduced plant up-take. Ammonia and phosphorus are ingredients in some fire retardants aerially applied to forest fires, it is also possible the retardants are the source of the elevated nutrients in receiving streams after fire. Ammonia is delivered to the stream through leaching in the subsurface and erosion. After looking at hundreds of fires, our analysis shows concentrations of dissolved, readily available nutrients, such as nitrate, nitrite, and orthophosphate, increase significantly one third of the time.

It should be noted that two out of three fires evaluated in this study did not have a detectable and significant change in nutrient concentrations the five years after fire. Hence, a post-fire nutrient response is not guaranteed. Other studies have observed elevated nutrients over a shorter duration (Earl and Blinn 2003; Burke *et al.* 2005) than the five years evaluated in this study. Some have observed a delayed surge of nitrate export, occurring the third year post-fire (Rhoades *et al.* 2011; Murphy *et al.* 2015). Here, concentrations have been averaged over five years, during which vegetation re-growth can occur, reducing erosion and dampening the nutrient response. For instance, within one to three years following the 1988 Yellowstone fire, the same species of plants re-appeared, sprouting from belowground structures. The anticipated leaching of nutrients did not occur in Yellowstone, and concentrations of nitrogen in stream water were only detectable the first three years after in first and second order streams and were negligible in the larger order streams (Romme *et al.* 2011). The hydrophobicity of soils also deteriorates over time and has been observed to decrease and allow infiltration six years following fire (DeBano 2000), further reducing delivery of particulate material. Detecting a change in water quality five years after a fire may not be possible because vegetation and soil is recovering in a shorter time span.

Hence, the subset analysis demonstrates how high frequency sampling is more effective at revealing short-term dramatic increases in dissolved and particulate nutrient concentration driven by precipitation events. Elevated nutrient concentrations after fire could be more common than one third of fires when water quality is sampled on a finer temporal scale, and such increases may be short-lived.

Initial post-fire surges of particulate phosphorus in receiving streams could also be a result of widespread application of fire suppressants. Fire retardants contain fertilizers and increase in total phosphorus has been observed immediately following a fire (Boulton *et al.* 2003). Dramatic post-fire increases in phosphorus observed in the Boise River after the Foothills fire and in the South Platte after the Hayman fire could be related to the amount of fire retardant flushed into the system following precipitation.

3.4.2 Major ions

The increase in loading rate of the four major ions (Ca^{2+}, Mg^{2+}, Cl^{-}, and $SO4^{2-}$) and hardness in the 3-5-year period after wildfire is similar to observations made by Earl and Blinn (2003) and Belilas and Rodá (1993) where an increase in calcium and potassium were measured in overland flow on individual fires. While the loading rate of the major ions increased, the concentrations of these ions tended to decrease on a five-year average. Again, this is likely a result of coarse temporal sampling, Rhoades *et al.* (2011) observed elevated ANC and Ca^{2+} concentrations for two years following the Hayman fire, before concentrations returned to pre-fire levels. Following a fire, the pH of the soil can increase due to ash deposited on soil. Elevated pH has been attributed to alkaline compounds such as calcium carbonate, magnesium oxide and magnesium carbonate (Ranalli 2004). Soto and Diaz-Fierros observed a mobilization of the divalent cations (Ca^{2+} and Mg^{2+}) following higher temperature fires (1993). In our analysis of the Old Fire in southern California, concentrations of calcium and sulfate fluctuated greatly; concentrations surged after precipitation events when the ions were flushed from the watershed into the receiving river. Our study looked at a larger collection of fires, some more severe than others, yet loading rates of both monovalent and divalent ions significantly increased five years after fire in about 25 percent of watersheds.

3.4.3 Metals

Limited studies have analyzed and found an increase in total and dissolved metal concentrations in surface water following fire (Burke *et al.* 2013; Burton *et al.* 2016; Pinedo-Gonzalez *et al.* 2016). While the database we assembled has millions of water quality data points available, the quantity of total and dissolved metals data was the lowest. In many instances, there was not enough data to establish baseline water quality conditions for metal concentrations. Most water quality data were taken over a short-term, likely a collection effort centered on a specific project. Although fewer fire-impacted watersheds provided sufficient data, our study found evidence of elevated metal concentrations associated with sediments and some elevated dissolved metal concentrations five years after fire. Dissolved metals can inhibit osmoregulation of fish, reducing their ability to uptake important macronutrients such as Ca^{2+} and Na^{+} (Hogstrand *et al.* 1994). The change-point from the High Park fire in Colorado (Figure

3.5) illustrates a dramatic increase in total iron concentrations in the Cache La Poudre river immediately following and for four years after the fire.

Increased erosion and sediment loading has been widely observed after fire (Lane *et al.* 2006; Shakesby *et al.* 2006; Silins *et al.* 2009; and Rhoades *et al.* 2011). Our research suggests that suspended nutrients and metal concentrations are the most common water quality response after fire. Metals and nutrients bound to sediments are more easily removed and transported with runoff from the denuded landscapes.

3.5 Conclusion

The overarching goal of this study was to determine the most common water quality response after wildfire across broad regions of the western U.S. Studies on individual fires do not reveal general changes in post-fire water quality; evaluating a larger dataset was necessary to identify the general response from fire. Results show that nutrient concentrations and mass loading rates of nitrate, nitrite, orthophosphate, phosphate, total and dissolved phosphorus increased after 22-38% of fires. Concentrations and flux of some ions (calcium, magnesium, chloride and sulfate) each increased significantly after 20-30% of fires. Total metal concentrations and flux also significantly increased after 25-50% of fires. Analytes bound to sediments, such as phosphorus and metals, tended to increase after fire, while their dissolved counterparts did not. Comparing time series data for analyte concentrations to daily precipitation and flow data demonstrates that average precipitation events after fire resulted in large flow responses, increased particulate concentrations and analyte loading rates.

Relying on publicly available data provided an excellent opportunity to investigate water quality change after fire. Yet, the dataset was coarse, where sampling was infrequent and often from a distance downstream of the fire. In this sense, studies on individual fires can conduct more focused sampling.

Through a large sample size, the current work moves beyond localized impacts and provides water managers and downstream communities critical information on water quality impacts after fire. There is a paucity of long-term water quality monitoring data in the U.S. and many parts of the world. We advocate that researchers and water managers alike continue to monitor water quality regularly, at least six times per year and ideally twelve times per year, and share data after wildfire and other land disturbances. As fires in the West continue, the developed database will expand and can be utilized for future studies on water quality trends and to parameterize hydrologic models for prediction of post-fire impacts on downstream communities. We also hope that studies like this one promote more sharing of data in public databases to further our knowledge of disturbance impacts on water quality.

CHAPTER 4

DRIVERS OF POST-FIRE WATER QUALITY RESPONSE

Modified from a manuscript in preparation for publication

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Abstract

Wildfires continue to grow in size and severity across the western United States and the world. While fires are a natural disturbance, they can have dramatic impacts on water quality in receiving streams. Previous studies on individual fires have observed an increase in various forms of nutrients, ions, sediments, and metals in stream water for different post-fire time periods. Our prior work using 160 wildfires across the western United States from 1984-2012 revealed significant increases in nutrient flux (different forms of nitrogen and phosphorus), major-ion flux, and metal concentrations are the most common changes in stream water quality within the first five years after fire. The current research follows up on this extensive survey of water quality response and evaluates the physical determinants driving the observed behavior for these watersheds. Physical, climatological, geochemical and biological data for each burned watershed were gathered to identify which variables drive water quality response after fire. The conditional inference tree technique was used to create a tree-branch model that identifies the most significant variables determining the post-fire water quality response. Results show that the geology of the burn area, which influences the soil characteristics, burn severity, and the rate of vegetation recovery are the key determinants of the type and duration of water quality response following wildfire. Results also provide critical information on water quality behavior to land and water managers in anticipation of, and following, a wildfire.

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4.1 Introduction

Climate change has lengthened the wildfire season, therefore the fire frequency and forested areas consumed by fire have also increased (Westerling et al. 2006; Morgan et al. 2008; Miller 2009; Westerling et al. 2011). As climate change continues, fire will disturb more forested land in the western United States and across the globe (Spracklen et al. 2009; Pechony and Schindell, 2010; Harvey 2016). Forest fires alter the landscape and as a result, degrade the water quality of streams within the fire perimeters and below them.

Wildfires engulf forested areas that are the source of high quality drinking and irrigation water across the globe (Bladon et al. 2014). The forested mountain areas of the Western United States receive high quantities of precipitation which becomes runoff with low levels of contamination, providing a source of high quality drinking water for millions of customers (Kaushal et al. 2006; Bladon et al. 2014). Wildfires can alter stream water quality, diminishing the quality of water for downstream consumers and disrupting water supplies (Writer et al. 2013). Given the increasing frequency of wildfires and their impact on important water supplies, a broader study evaluating multiple fires over a large, diverse region is essential.

While other studies have investigated forest fire impact on water quality, most have focused on individual fires. Results have shown that fires caused an increase in total suspended solids (Silins *et al.*, 2009; Noske *et al.*, 2010; Smith *et al.*, 2011; Dahm *et al.*, 2015), nutrients (Burke *et al.* 2005; Mast and Clow, 2008; Rhoades *et al.*, 2011; Writer *et al.*, 2012) and metal concentrations and exports in receiving streams (Burke et al., 2013; Burton *et al.*, 2016). In many cases, the post-fire water quality response from one study may disagree with the water quality response from another study. Earl and Blinn (2003) observed significant increases in nutrient concentrations from wildfire ash over a short period, with water quality in streams returning to pre-fire conditions after only four months. However, Burke et al. (2005) found dissolved and particulate forms of phosphorus were elevated for four years after fire and Rhoades et al. (2011) observed elevated nitrate and turbidity in fire impacted streams for five years. Furthermore, Emelko et al. (2016) observed a different magnitude and type of water quality change among neighboring tributaries six and seven years after a fire; in fact, some tributaries experienced a significant increase in sediment-associated metal concentrations and different forms of phosphorus while a proximate burned tributary did not. Clearly, wildfires impact water quality, but the type and duration of the water quality change varies depending on a range of factors.

The direction, magnitude and longevity of fire impacts on water quality can vary due to physical, geographical, geological, and climatological differences among burned watersheds. Some studies on individual fires have found correlations between physical factors, such as the size and severity of the fire and water quality response (Riggan *et al.,* 1994; Townsend and Douglas, 2004; Arkle *et al*., 2010; Rhoades *et al*., 2011). While other studies observed post-fire mobilization of sediment and dissolved nutrients, altering water quality, after summer convective storms (Burke *et al*., 2005; Lane *et al*., 2006; Bladon *et al*., 2008; Smith *et al*., 2011; Murphy *et al*., 2012; Writer *et al*., 2012). Some studies attribute the dramatic nitrate mobilization and metals loading observed after fire to the proximity of urban areas and input of urban air pollution (Riggan *et al*., 1994; Burke *et al*, 2013). In the current study, we expand the focus by looking at a larger number of fires across a broader region to identify physical characteristics that drive post-fire water quality response.

In our previous work evaluating response from 160 fire-impacted watersheds (Rust et al. 2017 under review), water quality was significantly changed in 33% of studied fires up to five years after the event when compared to pre-burn conditions. For basins with higher frequency data, it was evident that water quality changes were significant in the first three years following a fire. In both the initial years following fire, and five years after fire, concentrations and loading rates of dissolved nutrients, such as nitrite, nitrate and orthophosphate and particulate forms of nutrients, total nitrogen, total phosphate, and total phosphorus increased thirty percent of the time. Concentrations of some major dissolved ions and metals decreased, with increased post-fire flows, while total particulate concentrations increased; the flux of both dissolved and particulate forms increased in thirty percent of the fires over five years. However, in sixty percent of streams, there was no detectable water quality impact from fire.

In the aftermath of fire, a range of factors control response and recovery from landscape disturbance. Wildfires reduce and remove vegetation, modifying nutrient dynamics. Available soil nutrients are no longer taken up by trees. Trees consumed by the fire become nutrient rich ash and dead organic material. Surface runoff tends to increase as hydrophobic soil forms and soil porosity, infiltration depth, and hydraulic conductivity are reduced, in addition to the fact there are fewer trees to absorb water (DeBano 2000; Arkle *et al*., 2010; Murphy *et al*., 2012; Kinoshita and Hogue, 2011 and 2015). Furthermore, due to the loss of vegetation, especially living trees, hillslopes become vulnerable to erosion (Brock *et al*., 1989; Moody and Martin, 2001; Mast and Clow, 2008; Silins *et al*., 2009; Ryan *et al*., 2011). The increase in surface runoff leads to higher flows and increases export and loading of particulate materials from the denuded landscape into the stream (Lane et al. 2006; Ryan *et al*., 2011; Smith et al. 2011). The degree of water quality change depends on the variety and severity of these factors.

Likewise, water quality response is likely a function of vegetation recovery (Smith *et al*., 2011). Climate, burn severity, soil, and local geographic conditions govern how quickly vegetation recovers. The length of time that soils are exposed, with no vegetation, determines duration of water quality response. Local climate conditions influence vegetation re-growth after a fire which may govern the direction and magnitude of downstream water quality response. All these factors come together in a complex, and varied, recovery process.

The goal of the current study is to identify watershed characteristics driving water quality response and recovery. Specifically, we investigate the physical, geochemical, and climatological parameters that control the type, direction, and magnitude of water quality change. Characterizing water quality responses and their causes is exactly what other researchers have called for (Moody *et al*. 2013). Ultimately, improved understanding of post-fire response and related drivers will advance potential mitigation and treatment strategies as well as aid in the parametrization of post-fire models of water quality.

4.2 Methods

4.2.1 Fire and Water Quality Data

The current study utilizes data from 160 fires in the Western United States that occurred between 1984 and 2012 (Figure 4.1). Information about each fire, including the date, location, size and severity, was acquired from the Monitoring Trends in Burn Severity (MTBS) database (Eidenshink *et al*. 2007). In addition, the MTBS database provided fire perimeters, burn area and burn severity data (Eidenshink *et al*. 2007). Table 4.1 summarizes the average size and severity of the 160 fires that are the focus of this study.

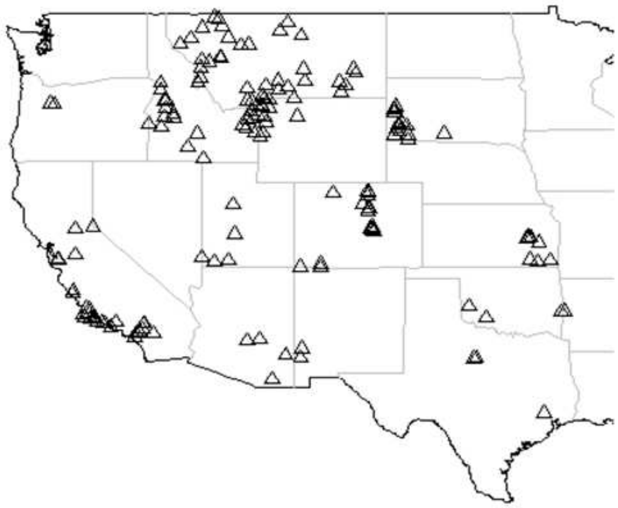

Figure 4.1. Map of 148 fire-impacted watersheds in the western United States with sufficient pre- and post-fire water quality data for inclusion in this analysis.

Table 4.1. Information on the western United States forest fires that are the subject of this study.

Number of Fires	Mean Size (ha)	Mean % of Watershed Burned	Maximum Size (ha)	Minimum Size (ha)	Mean % Moderate Severity	Mean % High Severity
160	10,476	18	228,791	0.64	21	24

Fire perimeter and location data were imported into GIS using ArcMap 10.1 to identify rivers downstream of the fires. Identified USGS stream gauge locations on these rivers became the outflow point which was used to define the watershed's perimeter. In total, there were 148 watersheds impacted by 160 individual fires. Table 4.2 summarizes the average size and landcover type for each of the 148 watersheds. Data for this table was retrieved from the USGS GAGES-II attributes (Falcone, 2011).

Table 4.2. General landcover data for all of the 148 study watersheds. Landcover data are from the USGS GAGES II database (Falcone, 2011).

Number of Watersheds	Mean Size (ha)	Mean % Forest	Mean % Evergreen	Mean % Deciduous	Mean % Mixed Forest	Mean % Developed	Mean % Grassland	Mean % Shrubland
148	204,453	42	37	2	3	4	21	26

All water quality data were acquired from United States federal and western state government databases. Federal databases used for this study included water quality data associated with USGS stream gauges from the USGS National Water Information System: Web Interface (NWIS) as well as other federal data from Environmental Protection Agency's STORET (EPA Storage and retrieval and water quality exchange) database. Public databases belonging to state government agencies, which collect water quality, were also searched and data from these sources were included.

Water quality data included five years of pre-fire observations to establish baseline conditions, as well as data from ten years post-fire. For each time period, before and after fire, at least two water quality samples per year per site were required to establish sufficient baseline water quality and estimate impact from fire, resulting in a total of 160 fires impacting 148 watersheds (6 watersheds had two fires over the study period, fires were 11 or more years apart). If a watershed burned more than once in the study period, only the second fire was included if there was a gap of ten years or more between fires in order to establish baseline water quality. Also, fires that consumed less than 5% of the watershed were eliminated from the analysis.

The percent change in median concentration and median loading rate (or flux) from pre- to post-fire was calculated for each water quality parameter in the database. Mean daily flow data from USGS gages associated with water quality sampling points were used to calculate loading rates in order to eliminate seasonality in flow-concentration. Flux is equally as important as concentration data when evaluating the potential impacts of water quality on an aquatic system (Helsel and Hirsch, 1991). Using percent change in median concentration and flux from pre-to post-fire also provided a way to compare water quality parameters with concentrations on different scales. Both changes in analyte concentration and flux are utilized as response variables in this study.

For most of the analysis, the water quality response time period was a composite of the water quality data the first five years after fire. Percent change was calculated comparing five years of pre-fire data to five years post-fire data, each with ten or more data points recorded. Additionally, where there was sufficient data, different post-fire response periods were defined to evaluate water quality recovery over time. Each time period was defined by water year (October 1- September 30). Post-fire water quality response time periods were defined as the first year, second year, third through fifth year, and sixth through tenth year after the fire.

Physical, geochemical, climatological, and biological recovery data were gathered for each fire-impacted watersheds (Table 4.3).

4.2.2 Physical Determinant Variables

Burn parameters considered as determinants included percent of watershed burned and percent moderate and high burn severity. The percent of the watershed burned was calculated using fire perimeters from the MTBS database overlaid with our watersheds defined by the USGS gauge locations. The percentage of each fire that was moderate and high severity was estimated using the Differenced Normalized Burn Ratio (DNBR) from Landsat imagery on a 30m spatial resolution (Eidenshink *et al*. 2007). The percentage of the fire that was moderate and high severity were considered both as individual variables and summed together as a third variable: percent of fire moderate and high severity.

Physical attributes included latitude, slope and aspect as they may govern vegetation recovery and the potential for erosion. The elevation of each burned watershed and burn area was found using a digital elevation model in ArcMap 10.1. The average elevation of each burn area was then calculated. The mean latitude of each fire was calculated in ArcMap 10.1 using the burn perimeter locations. The average percent hillslope for each burn area and burn aspect degree were also estimated using the digital elevation models in ArcMap 10.1.

In many cases, the water quality sampling station was a significant distance downstream from the fire, therefore we included the effect of sampling distanceas a predictor variable. The Euclidean distance from each burn area perimeter to the water quality sampling station was measured in ArcMap 10.1 and reported in meters.

Urban areas are sources of air pollutants that can be retained by trees and released after fires, pollutants can be a source of heavy metals (Burke et al., 2013). The proximity of the nearest urban area to the fire burn perimeter was estimated in ArcMap 10.1. Any city with a population of 100,000 people or more was considered as a source of air pollution. A forty-kilometer buffer was created around the center point of each city in the Western United States, then the Euclidean distance from this buffer to the fire perimeter was measured.

4.2.3 Climatological Determinant Variables

Climate drivers that were considered included solar radiation and aridity index. To estimate solar energy available to vegetation for re-growth, the solar radiation was calculated for each burn area (McCune and Keon, 2002). The solar radiation calculation combines slope, latitude, and aspect to estimate potential annual direct solar radiation that would be available to plants in the top of the canopy. The formula is as follows:

$$\begin{aligned} SolarRad &= -1.467+1.582*\cos(latitude)*\cos(slope) \\ &1.5*\cos(aspect)*\sin(slope)*\sin(latitude) - \\ &0.262*\sin(latitude)*\sin(slope) + 0.607*\sin(aspect)*\sin(slope) \end{aligned} \qquad \text{(Eq. 4.1)}$$

The aridity index (AI), defined as the ratio of mean annual precipitation to mean annual potential evapotranspiration, was used to describe the climate conditions of the different regions across the western U.S. in this study. The aridity index for each watershed came from the Consortium of Spatial Information CGIAR-CSI website for Global Aridity and Global PET database (www.cgiar-csi.org). The aridity index raster was clipped by each watershed boundary in ArcMap 10.1 to get an average AI for each fire-impacted watershed.

4.2.4 Geochemical Determinant Variables

Geochemical data from soil surveys that describe the nature and composition of the top-soils in each burn area were included as determinant variables. Soils data for each burn area were retrieved from the U.S Natural Resource Conservation Service (NRCS) State Soil Geographic (STATSGO) database (Schwarz and Alexander, 1995; Soil Survey Staff). Data were averaged over the burn area to summarize the general soil conditions before the fire. Variables collected from STATSGO include the available water capacity of the soil, expressed as both a high and low estimate of the inches of water per inch depth of soil. The available water capacity estimates reflect the water available to plants' roots as well as the oxidation-reduction conditions of the soil. The low estimate of available water capacity is an estimate of field capacity. The composition of the soil was included as the average percent of the top soil layer that was silt, clay, organic matter, and the average percent of calcium carbonate in the soil. Each of the composition variables may contribute to sediment and nutrient transport after a fire. The soil's k-factor which is an erodibility factor, is an input in the Universal Soil Loss Equation and reveals the soil's susceptibility to erosion. The k-factor from STATSGO selected for this study was the adjusted k-factor that includes the effect of rock fragments in the surface layer of the soil which may serve as protection from erosion.

4.2.5 Biological Determinant Variables

Because vegetation recovery may mitigate water quality response, data to measure vegetation growth were gathered. Satellite data was used to quantify vegetation recovery after each fire. The Normalized Difference Vegetation Index (NDVI) is defined as the ratio of the near-infrared (NIR) and visible radiances (VIS): (NIR-VIS)/NIR + VIS) (Deering et al. 1975). NDVI can be interpreted as a greenness signal, and is a good estimate of primary production and canopy coverage during the growing season (Prince 1991; Prince and Goward 1995). NDVI values range from negative one to positive one, where a high NDVI close to one indicates healthy, green vegetation, and a value close to zero (0.05) indicates bare soil; a negative one value reveals an error in the data (Goward and Huemmrich, 1992). Data to calculate the NDVI for each burn area before and after the fires was retrieved from the Landsat 5 and Landsat 7 satellites, which record images at a thirty meter spatial resolution every 16 days. Landsat satellite images were downloaded through the USGS Earth Explorer website and processed in Google Earth Engine (Google, Inc.). Images where greater than thirty percent of the image was covered in clouds were excluded, remaining images were then processed using a cloud-masking algorithm. The maximum NDVI values for each pixel in the burn area over the growing season from May 1 to September 30 were composited. The Maximum Value Composite technique minimizes problems commonly encountered in remote sensing data: cloud contamination, atmospheric attenuation, sun angle, directional surface reflectance and topographic surface shading (Holben 1986). For each burn area, the baseline pre-fire NDVI was calculated using the annual maximum NDVI for each growing season averaged over the five years before the fire. The post-fire NDVI was calculated using maximum composited NDVI values for each post-fire time period (1 year after fire, 2 years after fire, 3-5 years after fire, and 6-10 years after fire).

Predictor variables are listed in Table 4.3, including variable name, abbreviation, units of measure and range of values in the dataset.

4.2.6 Statistical methods

Statistical analysis was conducted in R using the ‘party’ package (R Core Development Team 2010). Spearman’s rank correlation coefficient, or Spearman’s rho (r_s), is a non-parametric measure of correlation among two variables (Tamhane and Dunlop 2000). The correlation coefficient ranges from negative one to positive one and measures the strength and direction of the association. Spearman’s rho was used to test statistical independence of each predictor variable to each other. The correlation between each water quality response variable (percent change in concentration and flux of each parameter) and predictor variable was also evaluated using Spearman’s rho. Results were compared to a chart of critical values for Spearman’s rho, for a two-tailed hypothesis, where correlations were statistically significant if $p < 0.05$. Some of the predictor variables were highly correlated and redundant, while all predictor variables were tested and included in the analysis, not all correlations are shown in the figures. Relationships among predictor variables and the water quality response variables are complex and many of the predictor variables interact with one another. Multiple linear regression and principle components analysis techniques were initially tested but did not provide meaningful results. Ultimately a conditional inference tree (CI tree) method was applied (Hothorn et al., 2006a; Hothorn et al., 2006b). The CI tree technique was chosen in part because of the potential correlation among predictor variables; the technique does not assume independence among predictor variables. The CI tree method is a non-parametric class of regression trees where the branches of the tree are separated based on the probability of the outcome of the water quality response variable. An *a priori p-value* is set (p-value <0.05) to identify when splitting is no longer possible; the outcome is a tree-branch model with only statistically significant variables. CI tree is noted to be an improvement over traditional linear regression models because the method does not rely on underlying assumptions of linearity. The number of predictor

Table 4.3. Predictor variables utilized in this analysis to identify drivers of post-fire water quality response.

Parameter	Abbreviation	Definition	Range in Dataset	Unit of Measure
Aridity Index	AI	ratio of mean annual precipitation to mean annual evapotranspiration	0.3-2.5	unitless ratio
Calcium Carbonate	CaCO3	average percent calcium carbonate in soil	0-29	percent
Clay	Clay_total	average percent clay in soil	0.3-40	percent
Elevation	Elev	average elevation of burn area	13-2925	meters
Latitude	Lat.Rad.	average latitude of burn area	0.5-0.85	radians
NDVI one year	NDVI_1y	ratio of the near-infrared (NIR) and visible radiances (VIS), maximum averaged 1 year post-fire	0.21-0.79	unitless ratio
NDVI two year	NDVI_2y	maximum averaged 2 years post-fire	0.17-0.81	
NDVI three to five years	NDVI_3.5y	maximum averaged 3-5 years post fire	0.28-0.77	
NDVI pre-fire	NDVI_pre	maximum averaged 5 years pre-fire	0.14-0.76	
Percent Burn	Perc.Burn	percent of watershed burned	5-100	percent
Percent Moderate Burn Severity		percent of fire that was moderate severity	0-65	percent
Percent Hi Burn Severity	Perc.Hi.Burn	percent of fire that was hi severity	0-84	percent
Percent Moderate and Hi Burn Severity	Perc.Mod.Hi.Burn	percent of fire that was moderate and hi severity summed	0-93	percent
Proximity to City	Prox.to.City.m	Distance from fire perimeter to nearest city buffer	0-351,352	meters
Proximity to Fire	Prox.to.Fire.m	Distance from fire perimeter to water quality sampling site	0-256,874	meters
Silt	Silt.total	average percent silt in soil	1.5-52	percent
Slope	Slope.Perc	average slope angle of burn area	0.3-17.6	percent
Soil K Factor	Soil.K.Fact	erodibility factor	1 to 13	
Solar Radiation	Solar.Rad.	potential annual direct solar radiation	0.86 - 1.05	MJ cm^{-2} yr^{-1}
Water Content Low	Water.Cont.L	lowest available water capacity	0-0.16	inch of water/inch of soil
Water Content High	Water.Cont.H	highest available water capacity	0-0.20	inch of water/inch of soil

variables used in CI tree is unlimited, and the model is not invalidated by multi-collinearity of predictor variables (Quinn and Keough, 2002). CI tree selects case weights with the strongest association with the response variable, displaying the relative importance of predictor variables, and does not overfit the sample (Hothorn et al. 2006a; Hothorn et al. 2006b).

4.3 Results

Results of the Spearman's rho between each predictor variable demonstrate the correlation among predictor variables, many of which are related and not independent. Some of the correlations were anticipated, such as the positive correlation between the soil's water content and the percent clay and silt as they help retain moisture in the soil. Additional logical correlations among predictor variables include solar radiation, which is negatively correlated with AI, solar radiation drives evapotranspiration and would therefore decrease the AI ratio. Solar radiation is also negatively correlated with elevation and latitude, which are variables in the solar radiation equation. Other correlations among predictor variables provide insight into a fire's impact on vegetation recovery. The percent of the fire that was classified as moderate or high severity was negatively correlated with post-fire NDVI values ($r_s = - 0.47$, $n = 160$, $p<0.001$); indicating slower vegetation recovery after more severe fires. Pre-fire NDVI values are also weakly negatively correlated with moderate and high severity fires ($r_s = -0.28$), which indicates the presence of drier vegetation before fire.

The entire list of predictor variables (Table 4.3) was used in the CI tree analysis to further identify the key drivers of water quality response (measured as the percent change in median concentration and loading rate). Conditional inference trees are shown using the loading rate of particulate constituents, such as particulate nutrients and metals as the response variable, while CI trees with concentration as the response variable were used for dissolved constituents that are highly soluble in water. The loading rate for particulate substances is more relevant to downstream water users while absolute concentration of dissolved substances at any given time affect the quality of water for aquatic life and water users.

4.3.1 Determinants of nitrogen and phosphorus response

4.3.1.1 Dissolved nitrogen

The amount of calcium carbonate in the soil within the burn area determines the percent change in median concentration of ammonia, nitrate, and organic nitrogen after fire (p = 0.017; Figure 4.2). The more calcium carbonate available (>10%) the greater the increase in concentration of dissolved nitrogen species, specifically dissolved nitrate (Figure 4.3) in the receiving water five years after fire. When there is less calcium carbonate available (Node 2) there was a median of zero percent change in dissolved nitrogen concentrations. The percentage of calcium carbonate in the soil reflects the watershed's geology; revealing that geologic differences do control the type and magnitude of dissolved nitrogen response.

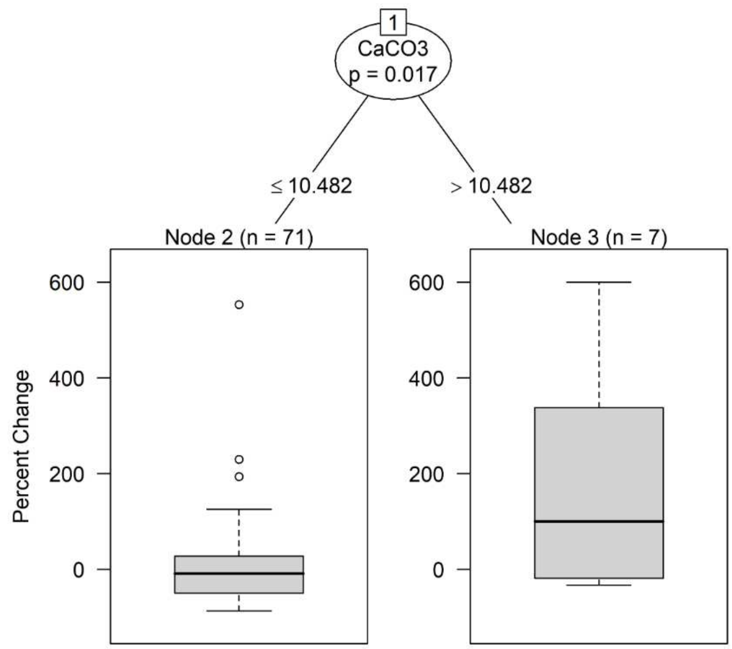

Figure 4.2. Conditional inference tree results illustrating the significant relationship between the percent of calcium carbonate in a burn area and the percent change in dissolved ammonia, nitrate, and organic nitrogen concentrations.

The developed heatmap (Figure 4.3) illustrates individual predictor variable correlations with the response variable, in this case, the median percent change in dissolved ammonia, nitrate, and organic nitrogen concentrations. The scale of correlation coefficient values is narrow and many correlations are weak (r_s <0.5). A significant positive correlation between the percent of calcium carbonate in the soil and the percent change in dissolved nitrate concentration ($r_s = 0.66$, n = 78, p<0.001), mirrors the same significant relationship identified by CI tree. When calcium carbonate is available in the soil, nitrification continues and ammonia is converted to nitrate.

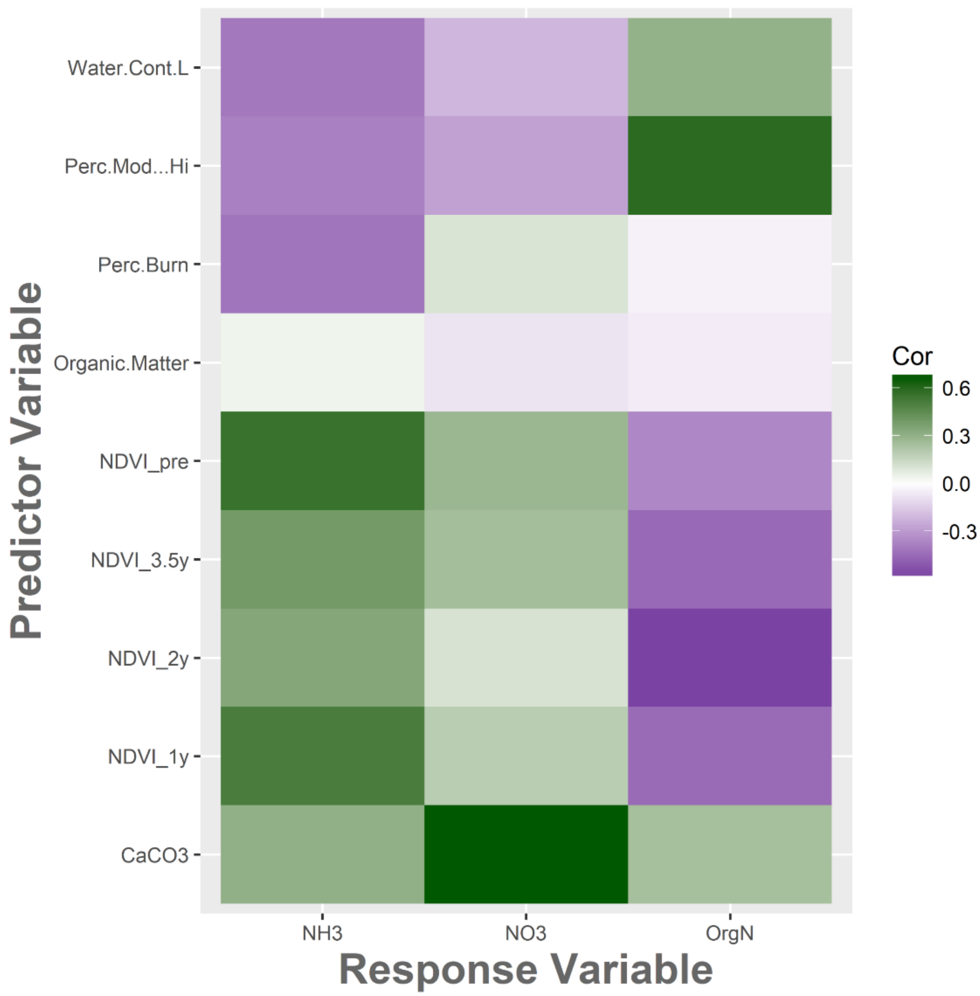

Figure 4.3. Heatmap illustrating Spearman correlation coefficients among selected physical predictor variables and median change in dissolved ammonia, nitrate, and organic nitrogen concentration the first five years after fire.

Nitrification is also a pH sensitive process, the presence of calcium carbonate indicates a neutral or high pH environment, conditions which allow nitrification to continue. The percent of the fire that is high severity shows a significant positive correlation with the percent change in concentration of dissolved organic nitrogen ($r_s = 0.59$, n = 78, p <0.001). A more severe fire is correlated with a positive increase in post-fire organic nitrogen concentrations. The first year

NDVI, or greenness value, is positively correlated with the concentration of dissolved ammonia (r_s = 0.51, n = 78, p <0.001) and weakly correlated with nitrate but the NDVI values show a significant negative correlation with dissolved organic nitrogen (r_s = -0.58, n = 78, p <0.01). When vegetative growth is strong before and after fire, it is correlated with an increase in dissolved ammonia and nitrate in receiving streams and a significant decrease in dissolved organic nitrogen concentrations. The amount of calcium carbonate in the soil, the severity of the fire, and the vegetation recovery are all correlated with the change in dissolved nitrogen concentrations.

4.3.1.2 Total nitrogen

Conditional inference tree did not identify a single driver of total nitrogen concentration or flux. We further examined the change in total nitrogen response and recovery over time. The individual correlations among each independent variable and the total nitrogen flux over each post-fire time period is displayed as a heatmap (Figure 4.4). The loading rates of total nitrogen the first year and six to ten years after fire are significant and positively correlated with the percent of the fire that was moderate and high burn severity (r_s = 0.53 and r_s = 0.56, n =32, p <0.01). Meaning, the more severe the fire, the greater the loading rate of particulate nitrogen in receiving streams the first year after the fire. This increased loading rate is still significant up to six to ten years later. The fire's proximity to a large urban center was also significantly correlated with the total nitrogen loading rate six to ten years after fire (r_s = 0.56, n = 32, p <0.01); signifying that urban centers are correlated with an increase in particulate nitrogen loading rates in nearby streams. The geology of the burned area is also associated with the total nitrogen flux, both the percent of calcium carbonate in the soil and the percent clay are negatively correlated with nitrogen flux the first two years after fire (r_s = - 0.62 to r_s = -0.69, n = 32, p < 0.001). The severity of the fire, proximity to a city, and nature of the soils are correlated with the loading rate of total nitrogen in receiving waters after fire.

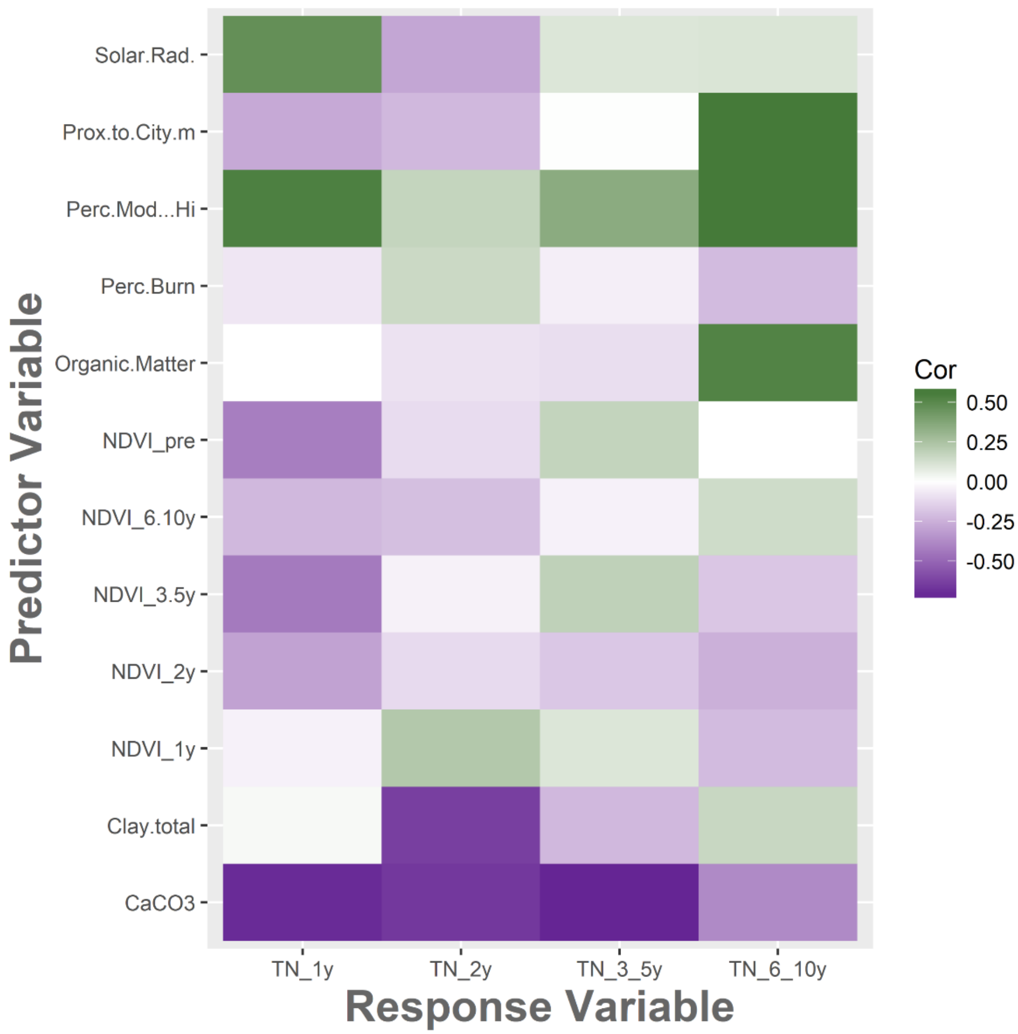

Figure 4.4. Heatmap illustrating Spearman correlation coefficients among selected physical predictor variables and median change in total nitrogen concentrations the first, second, third through fifth, and sixth through tenth year after fire.

4.3.1.3 Total phosphorus

The conditional inference tree technique once again, did not identify a single driver of total phosphorus loading rate after fire. Correlations among each individual predictor variable and total phosphorus flux over the ten years post-fire are shown in Figure 4.5. The correlations reveal that the main association between the physical characteristics of a watershed and post-fire phosphorus flux is the severity of the fire. The percent of fire that was moderate and high severity is significantly correlated with total phosphorus loading rate during all post-fire time periods (r_s = 0.53 to r_s = 0.70, n = 21, p <0.05). The more severe the fire, the more particulate phosphorus will be eroded from the watershed and enter the receiving streams. NDVI values over almost all time periods show weak negative correlations with total phosphorus loading rate. Although, there is a weak positive correlation with total phosphorus flux in the three to five year post-fire time period. Overall, where vegetative recovery is weak, there is an increase in flux in particulate nutrients, in this case, total phosphorus. The geology of the burn area also contributes to the phosphorus flux, there is a significant negative correlation between calcium carbonate and total phosphorus the first year after fire (r_s = -0.58, n = 21, p <0.01).

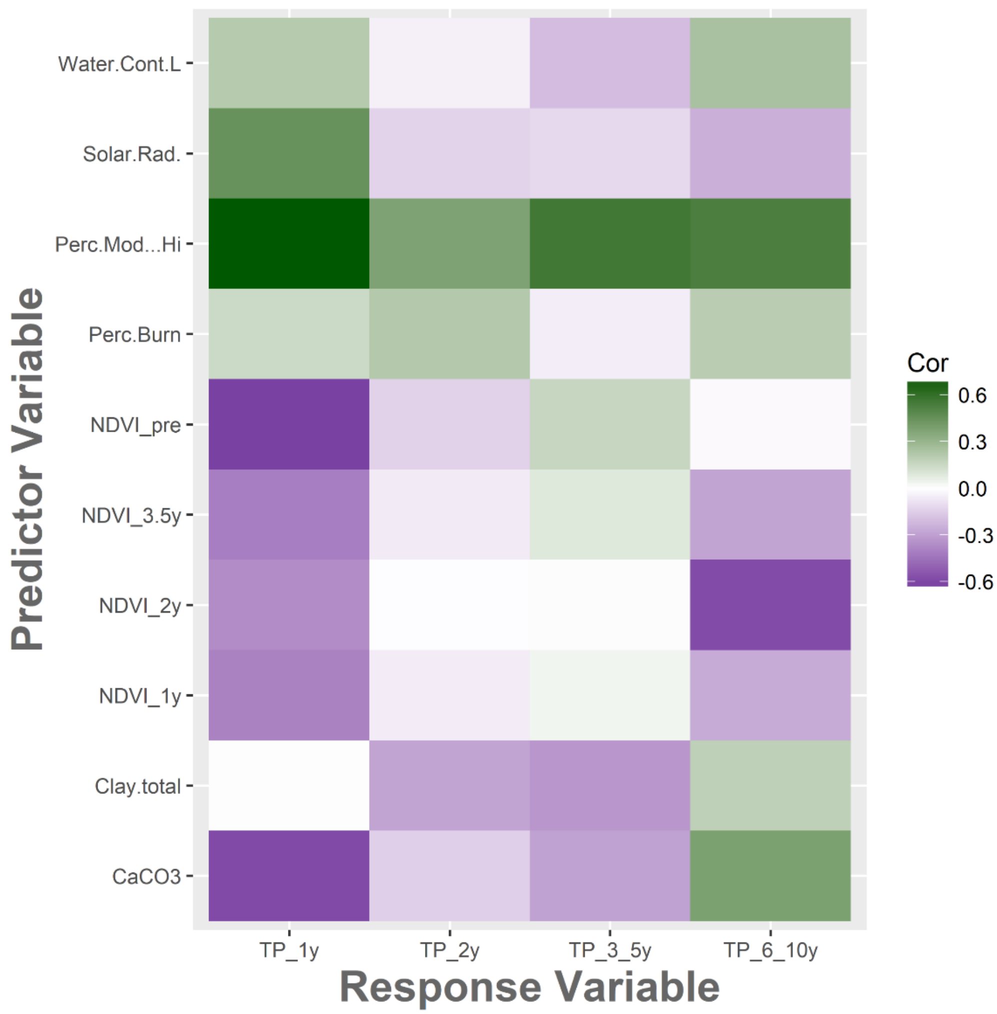

Figure 4.5. Heatmap illustrating Spearman correlation coefficients among selected physical predictor variables and median change in total phosphorus concentrations the first, second, third through fifth, and sixth through tenth year after fire.

4.3.2 Determinants of major ion response

The cations, calcium, potassium, magnesium, and sodium, were analyzed together, assuming they would behave similarly in the environment. The direction and magnitude of change in concentration of these cations is significantly related to distance from the fire ($p = 0.002$; Figure 4.6). When stream sampling sites were less than 42 kilometers to the fire perimeter there was a median decrease in cation concentrations. When water sampling sites are farther than 42 kilometers from the fire perimeter there is a median increase in concentration of the cations. Fire impacts are dampened and undetectable farther away from the fire. A decrease in median ion concentrations could be a result of dilution from higher post-fire flows.

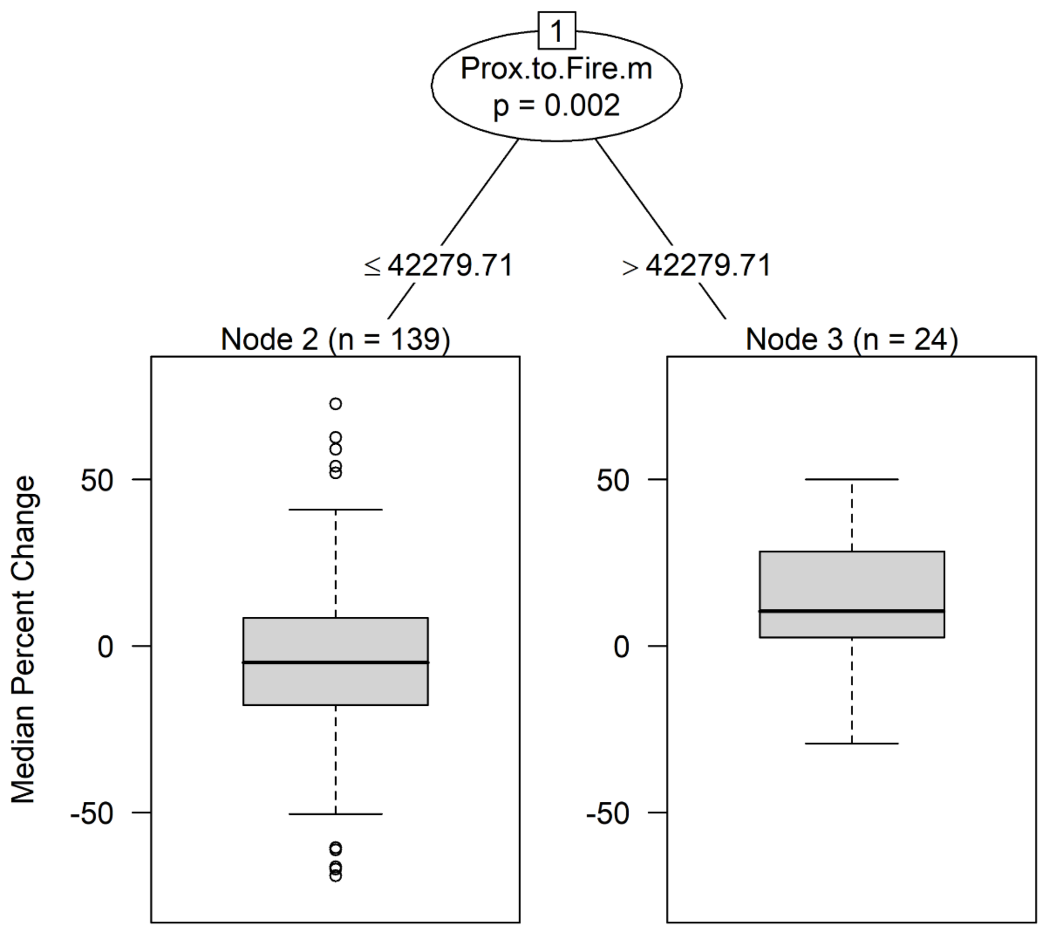

Figure 4.6. Conditional inference tree results illustrating the significant relationship between the water quality sampling location’s proximity to the fire perimeter and the median percent change in calcium, magnesium, potassium, and sodium concentration five years after fire.

While the CI tree method identified the distance from the fire as the main determinant of post-fire ion concentrations, there were no statistically significant correlations among individual predictor and response variables. The low estimate of water content in the soil, or field capacity, and percent clay in the soil are positively correlated with post-fire ion concentrations ($r_s = 0.04$ to $r_s = 0.49$; Figure 4.7). A higher field capacity of the soil in the burn area before the fire, is correlated with an increase in ion concentration in receiving waters. The water in the soil helps to dissolve and transport ions to receiving streams. Overall, the vegetation indices have weak negative correlations with the change in ion concentration in water, except for the two year NDVI. The weaker the vegetative recovery the greater the increase in post-fire ion concentration.

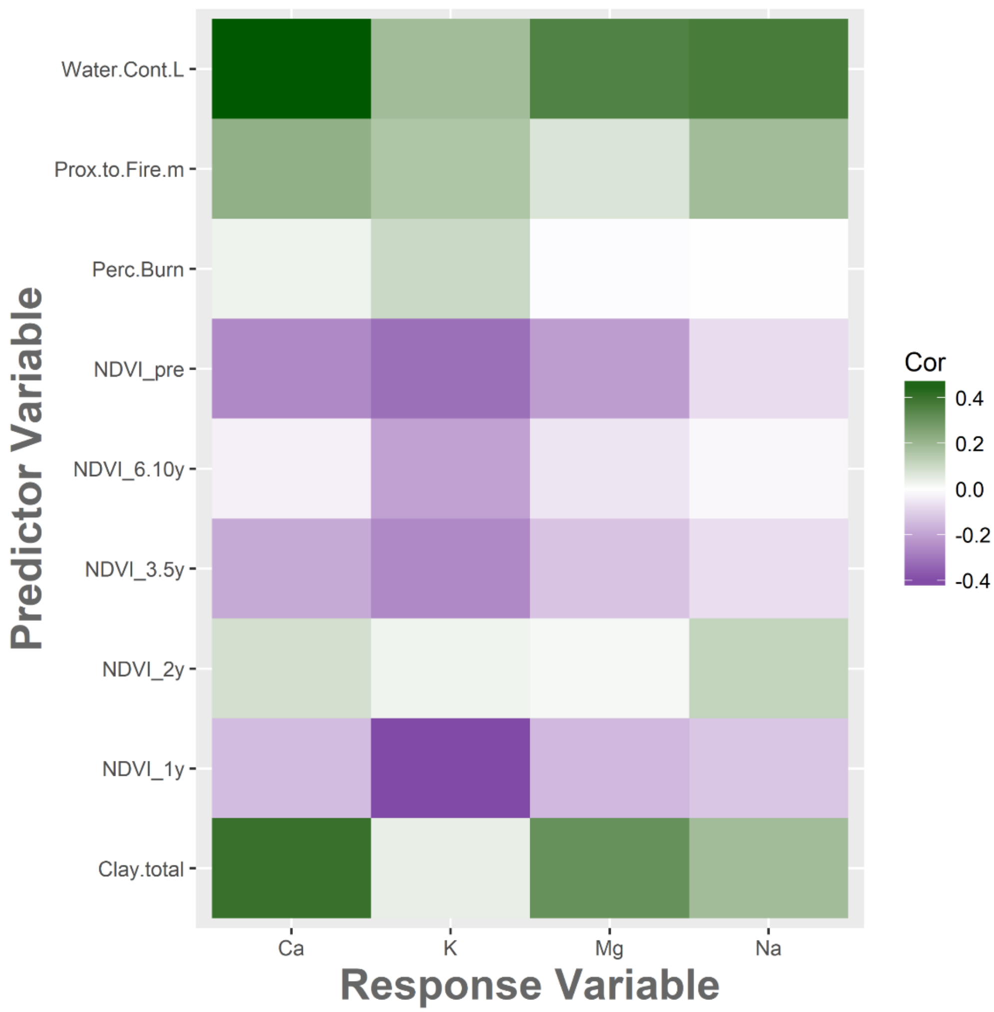

Figure 4.7. Heatmap illustrating Spearman correlation coefficients among selected physical predictor variables and median change in calcium, potassium, magnesium, and sodium concentrations the five years after fire.

Evaluating chloride alone can be used as a surrogate for flow; chloride tends to be conserved in an aquatic system and is not consumed by aquatic life. For this reason, correlations between predictor variables and chloride follow observations of post-fire flow response from other studies. Moderate and high severity fires tend to have higher flow responses. Here, the percent of the fire that is moderate and high severity is significantly correlated with a decrease in post-fire chloride concentration ($r_s = -0.51$, $n = 47$, $p < 0.001$; Figure 4.8). Again, the higher flows dilute the concentration of chloride after fire and this effect continues for ten years after the fire. The percent of clay in the soil is also negatively correlated with chloride concentrations; when soil conditions are drier the concentration of chloride in receiving streams is lower after fire ($r_s = -0.59$, $n = 47$, $p < 0.001$). There is less water available to transport the chloride. The soil's k-factor, a measure of the soil's erodibility, is negatively correlated with the change in chloride concentration in the last time period, six to ten years after fire ($r_s = -0.522$, $n = 47$, $p < 0.001$). The presence of more erodible soil decreases the concentration of chloride. Fire severity, the percent of clay in soil, and the erodibility of the soil are correlated with the change in post-fire chloride concentrations.

4.3.3 Determinants of heavy metal response

4.3.3.1 Dissolved metals

Dissolved metals are analyzed together and so are the total suspended metals. The change in median concentration of dissolved metals after wildfire is determined by the amount of organic matter in soil ($p = 0.016$; Figure 4.9). When there is more organic matter present in soil (> 2.146%) it is more likely that there will be a negative percent change in dissolved metal concentrations in the five years after fire. When the percent of organic matter in the soil is lower than two percent there is a median of zero percent change in dissolved metal concentrations. Though the median is zero, the fourth quartile shows a 100 percent change in concentration with several outliers that are even higher. If there is little organic material in the soil there is a chance that there will be a large positive change in dissolved metal concentrations.

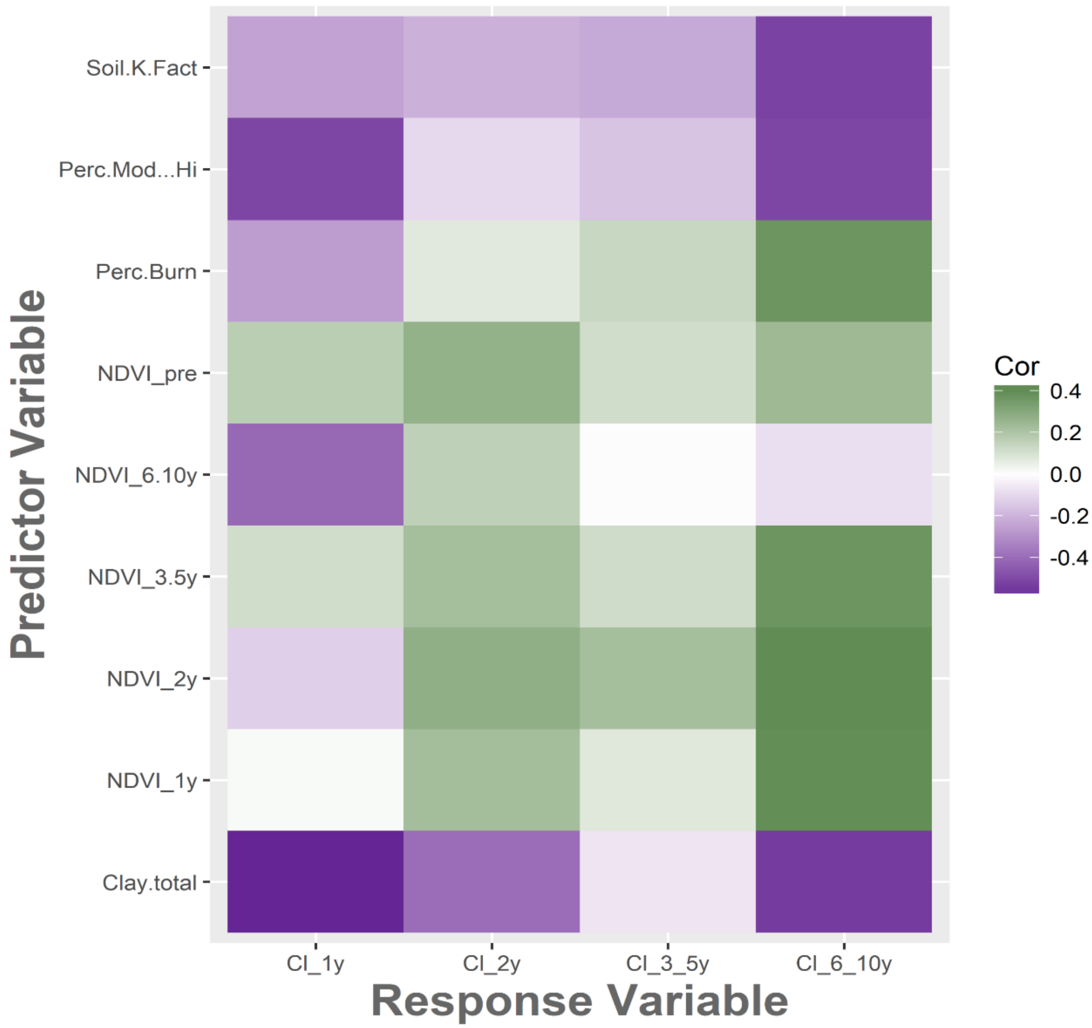

Figure 4.8. Heatmap illustrating Spearman correlation coefficients among selected physical predictor variables and median change in chloride concentrations the first, second, third through fifth, and sixth through tenth year after fire.

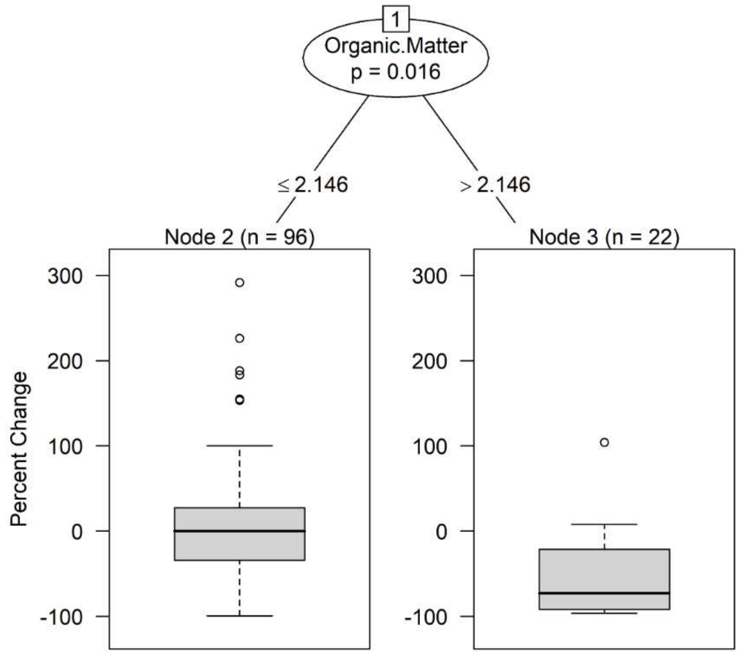

Figure 4.9. Conditional inference tree results illustrating the significant relationship between the percent of organic matter in the burn area soil and the median percent change in dissolved metals concentration (Al, Ba, Cd, Cu, Mn, Pb, and Zn) five years after fire.

The correlations among the predictor variables and the dependent variables, the dissolved metal concentrations are also illustrated (Figure 4.10). Water content of the soil is included to determine if saturated soils, which create anoxic conditions, influence dissolved metal concentrations after fire. In this analysis, the water content of the soil is not significantly correlated with concentration of dissolved metals in receiving water. The amount of solar radiation available is positively correlated with concentrations of all dissolved metals in this analysis ($p < 0.001$). An increase in solar radiation, coincides with an increase in dissolved metal concentrations. The NDVI values are weakly positively correlated with dissolved metal concentrations, except cadmium and barium which are both weakly negatively correlated with vegetation recovery. While solar radiation is closely related to vegetation recovery (measured as NDVI), in this instance the correlations among dissolved metals and these two variables are inconsistent.

The negative correlation between the percent organic matter in the soil and dissolved metal concentrations in receiving water is consistent with the CI tree result, and is a significant negative correlation with dissolved aluminum and cadmium (r_s = -0.61 and r_s = -0.87, n = 122, p < 0.001). Both the aridity index and the percent of the fire that is moderate to high severity are only significantly correlated with dissolved barium. Aridity index is negatively correlated (r_s = -0.57, n = 122, p<0.001) while the percent of the fire that is moderate to high severity is positively correlated with the change in barium concentrations after fire (r_s = 0.57, n =122, p < 0.001). The burn area aspect is significantly negatively correlated with dissolved aluminum (r_s = -0.8, n =122, p <0.001), and weakly correlated with the other dissolved metals, again the correlations vary by metal. While the amount of organic matter in the soil is the main determinant of dissolved metals response, solar radiation, aridity index, and fire severity are correlated with the change in some of the post-fire dissolved metal concentrations.

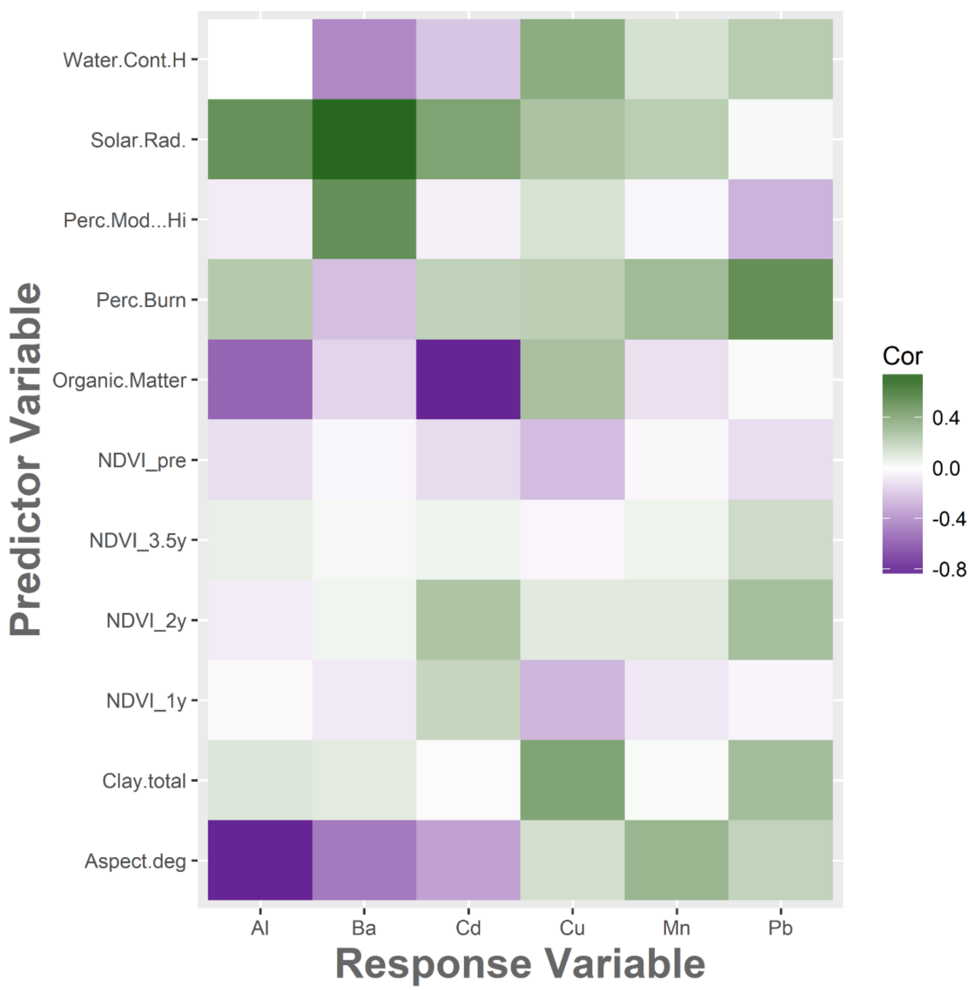

Figure 4.10. Heatmap illustrating Spearman correlation coefficients among selected physical predictor variables and median change in dissolved metal concentrations the first five years after fire.

4.3.3.2 Total metals

The CI tree identified drivers of both the change in concentration and loading rate of particulate metals in the five years after fire. First, evaluating results from the CI tree where the change in concentration of total metals is the response variable, the change in total metals concentration is determined by the vegetation recovery (NDVI) the first year after the fire ($p = 0.013$; Figure 4.11). When vegetation recovery is weaker (≤0.386) there is a positive increase in the concentration of total metals (node 2). When the vegetation recovery is strong (>0.386) the amount of solar radiation (node 3) available determines the direction of the total metals response ($p = 0.016$). When the solar radiation available is high (>0.983) there is a median concentration change of zero. Solar radiation is closely related to vegetation growth, when there is high solar radiation, the vegetation recovery is also likely to be high. If the vegetation recovery is weak and there is less solar radiation available (≤0.983), there is a negative change in median total metals concentrations.

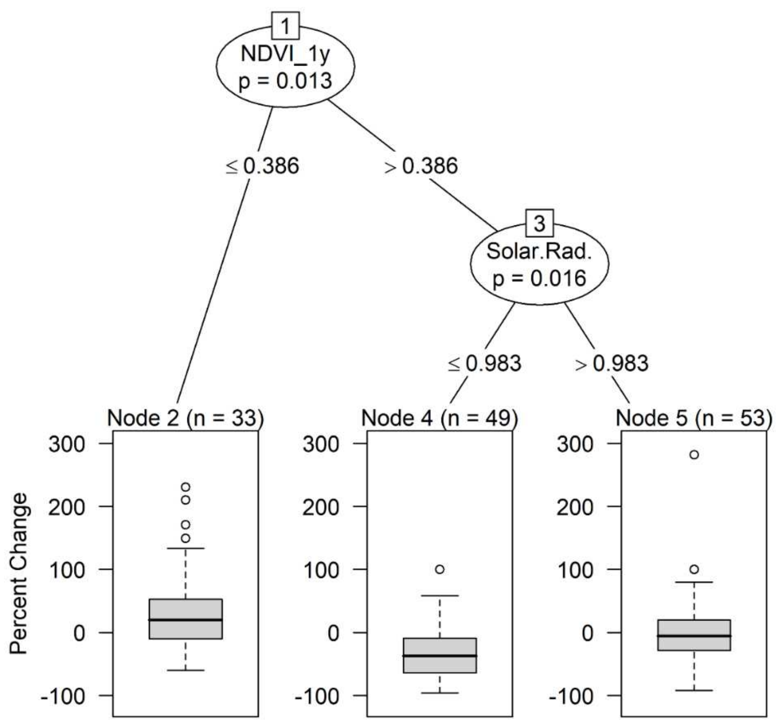

Figure 4.11. Conditional inference tree results illustrating the significant relationship between the first year vegetation growth (NDVI_1y), solar radiation and the median percent change in total metals concentration (Bo, Cd, Cu, Fe, Mn, Ni, Pb, Se, Zn) five years after fire.

Evaluating the change in total metals loading rate, once again, the vegetation recovery determines the magnitude and direction of total metals flux (Figure 4.12). If the NDVI value the second year after fire is low ($\leq$0.281) there is an increase in median loading rate (200% change, node 2, p = 0.024) of total metals in receiving water. When the vegetation recovery the second year after fire is stronger (>0.281) the NDVI value the first year after fire once again determines the direction of change (p = 0.004). When the vegetation recovery is strong the second year but weaker (<0.405) the first year, the overall five year average flux of total metals increases. The final determinant of total metals loading is the soil's water content before the fire (p = 0.003). If the vegetation recovery is strong both the first and second year after fire, and if the soil has a high water content (>0.126) there is a decrease in total metals loading rate the five years after fire (node 7).

There are multiple significant correlations among the loading rate of total metals and the predictor variables, the direction of the correlations depend on the metal being considered (Figure 4.13). The water content of the soil before the fire and the amount of solar radiation available are correlated with one another, and they have the opposite relationship among the response variables. Water content is significant and positively correlated with the loading rate of nickel and significantly correlated in the opposite direction with iron and lead (r_s = -0.8 and r_s = -0.5, n = 135, p < 0.001). Solar radiation has significant positive correlations with cadmium, copper, iron and manganese, but is negatively correlated with nickel (p < 0.001 in all cases). The soil's clay content is positively correlated with cadmium, copper, iron and zinc (p < 0.001 in all cases). The percent of the watershed burned and all vegetation recovery values over all time periods also illustrate this mixed response. The percent of the watershed burned and the NDVI over time are significantly negatively correlated with copper and selenium (p <0.001 in all cases). The severity of the fire displays a significant positive correlation with cadmium, copper, iron and manganese (p <0.001 in all cases). Evaluating the large number of significant correlations among predictor variables and the particulate metals individually, it is difficult to make a broad conclusion. However, using CI tree, the broader conclusion is clear, the strength of the vegetation recovery drives the particulate metals response.

4.4 Discussion

Compiling a large dataset allowed a more systematic and expansive investigation of water quality response and drivers than previously reported. Results of this study illuminate key drivers to a watershed's recovery.

However, working with data at this scale was challenging. Broad assumptions were necessary when assigning mean values across a watershed for characteristics that are usually measured at smaller scales. Assigning one value for the percent of clay or calcium carbonate in soil, or the erodibility factor across a region that may be hundreds of square kilometers is an assumption that was necessary to assess broader trends in water quality response on the

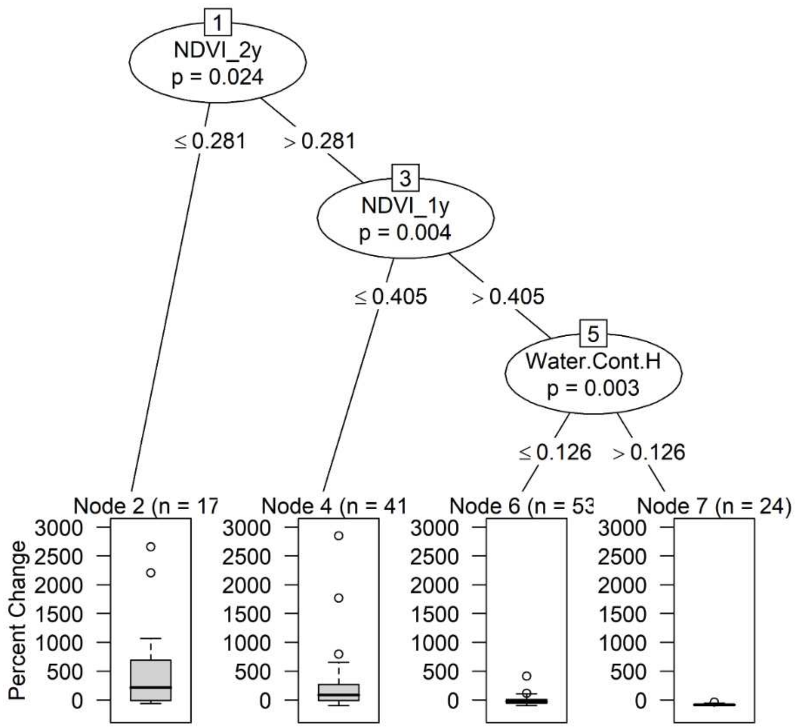

Figure 4.12. Conditional inference tree results illustrating the significant relationship between the second year vegetation growth (NDVI_2y), the first year vegetation growth (NDVI_1y) and the water content of the burn area soil and the median percent change in total metals loading rate (Bo, Cd, Cu, Fe, Mn, Ni, Pb, Se, Zn) five years after fire.

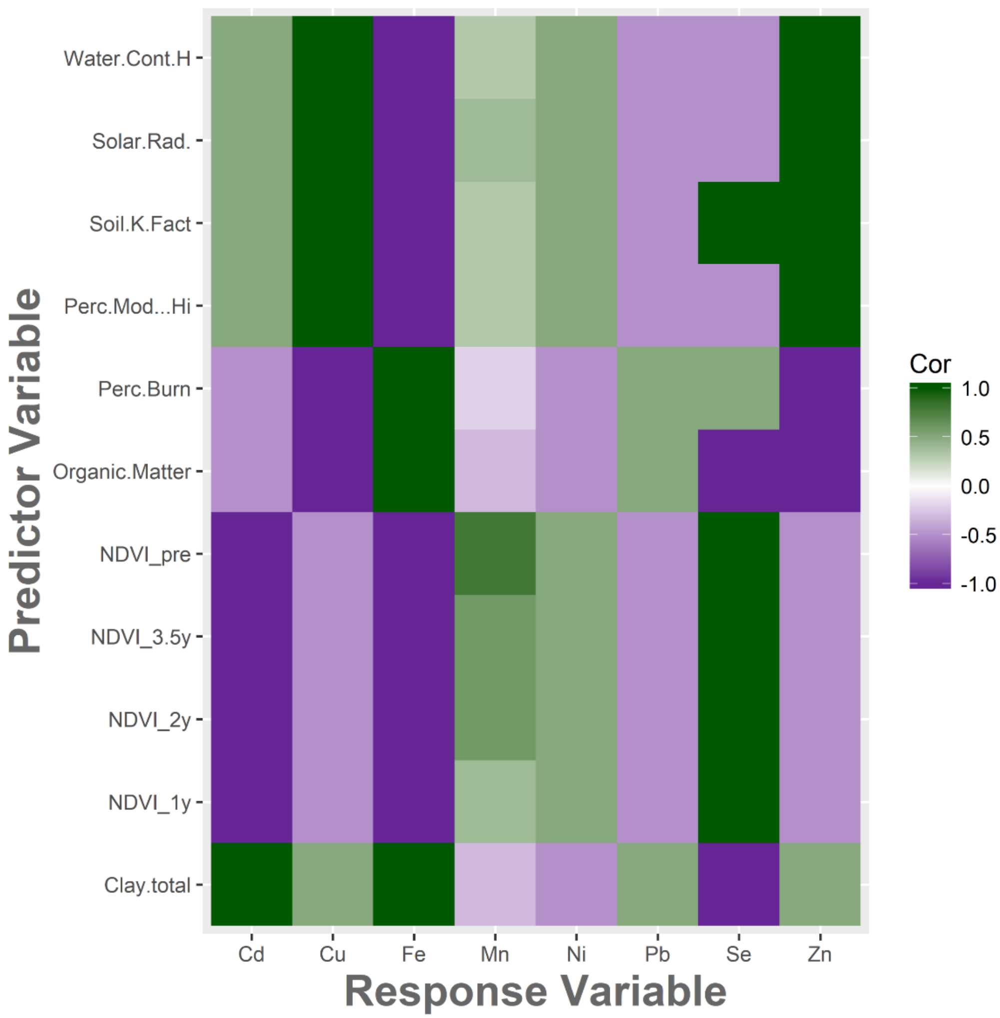

Figure 4.13. Heatmap illustrating Spearman correlation coefficients among selected physical predictor variables and median change in total metal concentrations the first five years after fire.

watershed scale. For some of the watersheds, we assumed that two water samples per year characterizes a stream's water quality. In other studies on individual fires, more intensive and frequent sampling has detected dramatic changes on shorter time scales. Murphy et al. (2015) determined short-duration water quality impairments were driven by high intensity convective storms following a wildfire, an inference made possible through high frequency water quality sampling. Because we relied on public data before and after fire, our sample size of fires is large but the sampling frequency is low. It is likely that forest fires cause more frequent short-term water quality impacts that we did not detect in this dataset. In our other study, (Rust *et al.* in review), we observed watersheds with higher frequency, monthly sampling, had more extreme and frequent short-term water quality responses after fire that were not evident when the data was averaged over the five year post-fire time period. To this extent, we may have missed some short-term water quality responses through the methodology in this study. But we have not over stated the response, instead we have erred on the conservative side. Through averaging the water quality change over five years, more frequent short-term water quality responses are likely overlooked.

However, broad themes are still observed in our results, the composition of the soils, the fire's burn severity, and the vegetation recovery all govern the direction and magnitude of the water quality response. Interactions in soil chemistry determine if material from wildfire and from soil end up in streams. There are sorption reactions between soil and organic material from fire. Moody et al. (2013) summarized numerous studies where soil composition and the physical soil properties altered by wildfire are linked to post-fire erosion and sediment transport. Our exploration of vast water quality data agrees, soil properties and burn severity, which modifies soil properties, drive water quality response. Using conditional inference trees, we identified the key soil properties that drive this response: percent calcium carbonate, organic matter, and water content.

Burn severity is significantly correlated with an increase in both dissolved and particulate matter entering streams. Larger and more severe fires have a greater detectable change in water quality. This supposition agrees with previous research (Riggan *et al.*, 1994; Ranalli, 2004; Townsend and Douglas, 2004; Arkle *et al.*, 2010; Rhoades *et al.*, 2011). Severe fires alter soils, but the affect can be patchy and complex. Holden et al. (2009) used remote sensing of burn severity to infer the elevation, slope and aspect of a fire influence a wildfire's burn severity. Higher severity fires have been correlated with an increase in runoff, erosion, nitrate concentrations and delivery of sediments to streams (Lane *et al.* 2006; Rhoades *et al.*, 2011). Kinoshita and Hogue (2011) also found high severity fires had the slowest vegetation recovery.

Severe wildfires affect soil properties and can cause erosion and leaching of nutrients; however if vegetation recovers, soil properties can recover and even improve (Certini, 2005). In this research, we postulate that vegetation regrowth also heavily controls water quality response. Vegetation recovery governed total metals concentration and flux after fire. Weak vegetation recovery leads to an increase in total metals concentration and loading rate after fire. When vegetation grows back, trees absorb dissolved nutrients, soils are stabilized and less dissolved and particulate matter are delivered to the receiving stream. In this analysis, the erodibility of the original soil before the fire (the k-factor) did not come up as the significant variable, instead, the burn severity and vegetation recovery governed erosion, so it is less important how prone the original soils are to erosion, the loading rate of particulate matter after wildfire is governed by burn severity and vegetation presence.

4.5 Conclusion

Wildfires can cause a wide range of water quality impacts. Identifying determinants of post-fire water quality response, on a broad spatial and temporal scale, as we have done here, facilitates anticipation of water quality impacts following fire. In order to draw comprehensive inferences and correlations, similarly broad assumptions were made.

Overall, we conclude that the natural geology, which dictates soil composition, in addition to fire severity and vegetation recovery drive the post-fire water quality response in receiving waters. Specifically, the amount of calcium carbonate in the soil controls the magnitude and type of dissolved nitrogen response. When organic matter is present in the soils, fewer dissolved metals are released into the water, and the water quality change is mitigated. The severity of the fire controls both the vegetation recovery (as measured in this study with NDVI) and the nutrient concentration and loading in receiving streams. When fires are moderate and high severity, there is an increase in concentration of organic nitrogen, total nitrogen loading, and total phosphorus in streams below the fire. Fire severity was also negatively correlated with vegetation greenness after the fire, when the vegetation is slow to recover, organic nitrogen and total metal concentrations and flux are higher below the fire. When solar radiation is available and vegetation recovery is strong, there is more ammonia and nitrate in the receiving water but no detectable change in total metals concentrations and flux. After evaluating almost thirty different physical watershed and fire characteristics, it is evident that the rate of vegetation recovery mitigates water quality change after fire. While this conclusion may be implicit in general assumptions about fire recovery, utilizing a broad dataset provided evidence to validate this assumption.

Our understanding of the impacts that fire has on water quality is not complete, testing our conditional inference tree models in other regions would further inform our empirical models. Research to further understand mechanisms of wildfire recovery, specifically vegetation recovery, are necessary. Results from this study will help parameterize water quality models for post-fire conditions. Soil characteristics, burn severity, and vegetation recovery should be considered relevant parameters in any post-fire water quality modeling effort.

CHAPTER 5

CONCLUDING REMARKS

The objective of this research was to better understand how wildfires impact water quality and aquatic ecosystems in receiving streams. The aim was to increase knowledge of wildfire's effects by identifying fire's general impacts on water quality and determine the watershed features that govern recovery. This work also sought to improve understanding of the duration of wildfire's impacts and the corresponding recovery of the ecosystem.

Understanding wildfire's impacts on surface water quality and ecosystems was expanded through three studies. In the first study, detailed in CHAPTER 2, water quality, streamflow, macroinvertebrate, and fish populations were monoitored above and below a single large fire, the West Fork Complex Fire in the Upper Rio Grande watershed in southwestern Colorado. Physical, chemical and biological impacts from the WFC fire were observed for three years. While some studies have looked at singular impacts from other wildfires, by examining the changes in water quality parameters or macroinvertebrate or fish populations; this research evaluated the system holistically. In the three years post-fire, water quality, macroinvertebrate, and fish population impacts were detected, and by the third year there were signs of recovery. The second study attempted to broaden the scale of wildfire studies, by looking at 172 burned watersheds to identify common water quality responses to fire and the duration of such changes. Prior to this study, understanding of wildfire's impacts on water quality was from observations of single fires, which, when evaluated together confounds any detectable trends. Examining a large number of fires revealed typical water quality responses in streams. However, there was no obvious pattern to determine which watersheds were likely to have a pronounced water quality impact. The third study scrutinized watershed characteristics to identify which variables control the type, direction, magnitude and duration of the water quality response. Identifying determinants of post-fire water quality response also reveals the characteristics that govern recovery after the fire. The results from each study answered the following research questions:

In what ways did the West Fork Complex fire impact the aquatic ecosystem of the Rio Grande? How did the fire impact water quality, insects and fish? How quickly do water quality, insect populations and fish populations recover?

In CHAPTER 2, the complete impacts of the West Fork Complex fire on water quality, insect, and fish populations are detailed. While most water quality parameters remained identical to un-burned sites, there was a large increase in the median turbidity and the frequency of turbidity exceedances above trout thresholds. The second year after fire, the Rio Grande below the fire exhibited high levels of turbidity after runoff and every summer precipitation event. The fire burned a steep, headwaters watershed, at moderate and high severity, creating ideal conditions for erosion. In the second year after fire, the turbidity impact is limited to the stream segments directly below the fire scar while segments farther downstream had recovered. The macroinvertebrate populations were impaired by the fire, but not necessarily the water quality. All indices of macroinvertebrate population abundance and diversity show a population impaired by fire that recovered quickly. Even macroinvertebrates in river segments above the fire were impacted, although the post-fire water quality did not disrupt the macroinvertebrates as much as smoke and ash during the fire did. After the severe wildfire, a dramatic trout fishkill was observed. A landslide introduced a large volume of sediment into the stream, suffocating resident trout. As new vegetation began to grow over the landslide, sediment concentrations were reduced. One year after the landslide, young-of-year trout were observed inhabiting the impacted stream stretch, two years later a healthy population of adults re-inhabited the segment. Still, three years after the fire, turbidity remained elevated, frequently exceeding thresholds. Despite the continued water quality impairment, the fish population began to recover. Adults were present again, population numbers rebounded within three years of the fire and two years of the landslide.   The wildfire’s effect on water quality endured the longest, and water quality was the slowest to recover. Despite continued turbidity exceedances above threshold, fish returned to fire-impacted segments within three years, recovering only two years after the fishkill. Macroinvertebrate populations were less impacted by the water quality and instead impaired due to conditions during the fire itself. Macroinvertebrate populations recovered within two to three years. Wildfire’s effects on water quality, macroinvertebrates and fish populations were consistent with other studies on singular effects, however this research took the unique approach of linking key aspects of the disturbed ecosystem together.

What type of water quality responses are the most common after forest fires in the Western United States?

In CHAPTER 3 water quality data from a large number of fires was collected and the percent change in concentration and flux of each water quality parameter was evaluated by comparing concentrations and flux at baseline (five years before the fire) to the five years after the fire. Results revealed that the dissolved constituents, such as dissolved nutrients, ions and metals, decreased in roughly thirty percent of the fires while the flux of the same constituents increased. The increased post-fire flow carried more dissolved matter to the stream and concurrently the higher post-fire flows diluted concentrations of dissolved matter. In thirty percent of the fires, it was the particulate matter that increased in both concentration and flux. Total nutrients and metals increased in concentration below thirty percent of the fires. In this study, the impact on water quality was assessed through measuring the difference in the five-year average baseline water quality and the five -year average post-fire water quality. Change-point analysis was utilized to evaluate watersheds where there was a high frequency of data available in order to identify whether the five -year average after fire missed more frequently occurring water quality impacts. The change-point analysis reveals that water quality impacts occur on a shorter time frame, and when results are averaged over five years the change may not be detectable and short-term impacts are missed. These changes are only detectable when frequent water quality sampling data are available.

Still, not every watershed experienced a change in water quality after fire, in fact, many remained unchanged. While commonalities among watersheds were found, it was unclear why some watersheds were more resistant to change than others. This led to the final study in CHAPTER 4, where the final research questions are addressed.

What physical or climatological characteristics control water quality response after fire? Are some watersheds more prone to recover quickly, where water quality returns to baseline while others are more likely to show impacts from the forest fire for a longer period?

In CHAPTER 4 the large data set of 172 fires was further investigated. In this chapter, the drivers of the post-fire water quality responses were evaluated. Utilizing the Conditional

Inference Tree statistical technique, a type of decision tree is produced. Here, broad themes are observed in the results, where the composition of the soils, the fire's burn severity, and the vegetation recovery all govern the direction and magnitude of the water quality response. Interactions in soil chemistry determine if material from wildfire and from soil end up in streams. The percent of calcium carbonate and organic matter in the soil, the fire severity, and the vegetation recovery control what amount and type of dissolved and particulate constituents are transported to surface water. Burn severity had been correlated to water quality response in other studies. However, this is the first study to correlate both soil properties and vegetation growth to the water quality in streams below fire. Clearly the type and direction of water quality response depends on more than burn area and burn severity, as other studies have postulated. There are additional drivers of post-fire water quality response, including geochemical properties of the soil, available solar radiation and the vegetation recovery. The results illustrate how the rate of vegetation recovery governs the water quality response, so that after six to ten years, when some vegetation has grown back, the water quality response is diminished, and the watershed has recovered.

Overall the research compiled in this dissertation highlights the complex nature of wildfire impacts on water quality. Utilizing a broader dataset, including both abiotic and biotic changes from fire, it was possible to evaluate the ecosystem as a whole. The insect and fish populations showed dramatic impairment and recovery. If we had only examined water quality, we would have missed the biotic recovery that was occurring despite the continued water quality impairment from high turbidity. Broadening the spatial scale through looking at multiple fires exposed common responses in water quality. Compiling the physical and climatological characteristics of all fire impacted watersheds provided the chance to discover the key factors driving water quality response. Land and water managers in areas prone to wildfire can use these broader-scale results to anticipate changes in water quality after the next fire. Identifying general water quality responses and the key watershed and climatological characteristics that control them provides the opportunity to optimize the potential for recovery.

Several future research directions arise from this work. After observing the trends in post-fire water quality response in Objective 2, the next step is to develop better tools to help quantify and predict sediment and nutrient loading. Models that simulate hydrological response on the watershed scale are important tools in the field of hydrology. Modeling improves our understanding of the natural world and helps water managers anticipate change. Currently, there are no published studies that attempt to model nutrient response after fire. Through accomplishing the second objective in this proposal, we have demonstrated that elevated nutrient loading (primarily nitrogen and phosphorus) is the most common post-fire water quality response, where the nutrient loading remains elevated for three to five years after the fire. We propose to use a hydrological model capable of modeling water quality, to quantify wildfire's impact on nutrient loading. The ultimate goal is to evaluate how well watershed models can simulate post-fire water quality on the watershed scale to better parameterize models. Future work will also include utilizing in-situ data from the West Fork Complex fire study to test the applicability of a model that predicts post-fire hillslope failure. This is an opportunity to use empirical data to test and improve a model's predictive ability. The decision trees employed in CHAPTER 4 should be tested against data from the most recent fires. Utilizing the conditional inference trees and testing their predictions could reveal whether water quality impacts can be anticipated based on the decision trees presented here.

This research also revealed the value of publicly shared data. In the future, building on the data already collected in the fire-water quality database will help managers to ask and answer even more questions related to post-fire water quality. Data on dissolved and total metals concentrations were limited, and agencies should be encouraged to conduct more regular water quality monitoring, including metals data. With more data available, even more trends in post-fire water quality would be exposed. Understanding the degree and duration of water quality impacts from fire will help support land and water resource managers as they face expanding wildfires and their impacts in the western U.S.

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