

THESIS

THE EFFECTS OF THE HIGH PARK FIRE ON CACHE LA POUUDRE DRINKING WATER
QUALITY

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ABSTRACT

THE EFFECTS OF THE HIGH PARK FIRE ON CACHE LA POUFRE RIVER WATER QUALITY

The High Park fire burned 87,284 acres in the Cache la Poudre River watershed from June 9th to June 30th, 2012. The effects of the fire not only impacted the vegetation and soil within the watershed, but also induced changes in the water quality of the Poudre River, a local drinking water source. The utilities and food and beverage producers were concerned that the fire might impart a smoky flavor and other taste and odor issues to the water that may negatively impact their products. Sensory tests were performed on a variety of Poudre River water samples using Flavor Profile Analysis (FPA) according to Standard Method 2170. Results showed that there was not a smoky flavor in Poudre River water several months after the fire compared to smoky flavor just after the fire. In testing samples with a standard smoky compound, guaiacol, FPA results indicated that neither the conventional treatment nor the powdered activated carbon (PAC) treatments removed the smoky smell. However, the benchscale tests conducted with charred riverbed sediment indicated that the smoky flavor is removed by conventional and powdered activated treatments. In the second part of this study, nutrients, organics, and metals were shown to leach into the river water in a laboratory setting. The potential of these parameters to increase in the Cache la Poudre River may cause algal blooms and result in increases in taste and odor issues, as well as metallic tastes caused by the leaching of metals such as iron and manganese. So although the High Park fire did not impart a smoky flavor to the Poudre River water, there remains a potential for taste and odor issues in the future.

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INTRODUCTION

With increasing drought conditions around the world, dry conditions lead to wildfires, many of which are large and destructive. The High Park fire just west of Fort Collins, CO in the summer of 2012 was one of these large fires, burning 87,284 acres. The High Park Fire, the second largest and second most destructive fire in Colorado history in terms of home destruction, started on June 9th, 2012, after a lightning strike and was contained on June 30th. The fire burned 25% of the Cache la Poudre River watershed, which was concerning to local drinking water utilities as the river serves as a primary source. Alterations within the watershed could have a large impact on water quality and treatment of the drinking water source.

Runoff from fires carries burnt plants and ash into water bodies. Introduction of sediments and ash that are high in nutrients, metals and incompletely combusted organics can affect water quality and safety. Increases in nitrogen and phosphorus in water bodies cause algae and bacteria to grow, thus producing taste and odor issues as well as the potential for formation of disinfection-by-products after chlorination. The taste and odor issues are significant to the palatability of drinking water. The earthy, musty odors caused by algal metabolites (e.g. geosmin and 2-methylisoborneol) as well as the smoky flavor after fires are not aesthetically pleasing and cause a perception of unsafe water to the consumers. Since the Poudre River is a major drinking water source for the cities of Fort Collins and Greeley, the implications of the fire on water quality are important to investigate to adequately treat the water to satisfactory safety and aesthetic levels. This study focuses on: 1) sensory perceptions and treatment of smoky flavors the fire may have imparted in the water; and 2) the levels of nutrients, organics and metals that may leach from the river sediments.

This thesis is comprised of two main chapters. The first chapter gives the background information obtained through a literature review on the impacts of wildfires on water quality including sediment transport, nutrient and metal concentration changes, and taste and odor issues. The second chapter is prepared in a manuscript format for an academic journal submission. In the second chapter, materials, experimental methods, results, analyses, and conclusions obtained from this research are presented. The raw data is also presented in the appendix.

1.0 LITERATURE REVIEW

1.1 IMPACTS OF WILDFIRES

Forest fires greatly affect countries all over the world. Each year forests are affected by fire especially in Australia, western United States, and the Mediterranean (FAO, 2010). Colorado has experienced large scale wildfires, specifically the Hayman fire in 2002 that burned 137,760 acres, the Missionary Ridge fire in 2002 that burned 71,739 acres, the High Park fire of 2012 that burned 87,284 acres (Makings, 2012) and the Waldo Canyon fire in 2012, which was the most financially destructive fire in Colorado history burning 346 residences (Coffman, 2012). The effects of the fires change the watershed characteristics and thus the water quality of the surface water bodies and groundwater. It is important to understand the after effects of fires on watersheds to anticipate changes in the water quality in the future, especially for drinking water utilities.

1.1.1 EROSION AND SEDIMENT TRANSPORT

Wildfires, which may be caused by nature (lightning) or by humans, change the landscape and in most cases increase erosion. Erosion rates depend on: 1) the fire intensity, 2) the slope steepness, and 3) the post-fire vegetation re-growth (Tiedemann et al., 1979; Wondzell and King, 2003). A higher severity fire destroys more vegetation than lower severity fires and can also change the composition of the soil by creating a hydrophobic layer (DeBano, 2000). Decomposed humic organic matter in the lower layers of soil is volatilized during a fire and is then adsorbed to the cooler particles deeper in the soil to create the hydrophobic layer that limits water flow through the pores in the soil (Goforth et al., 2005). This layer decreases infiltration

capacity, so instead, precipitation is converted to runoff causing a higher sediment transport than in unburned areas (Gottfried et al., 2003; Gallaher and Koch, 2004; Neary, 2004; Kolka, 2012).

A lack of vegetation post-fire destabilizes the hillside because the canopy, forest floor litter, and roots do not anchor the soil anymore, so introduction of water or wind displaces the soil more easily than in unburned areas (Morris and Moses, 1987; Wondzell and King, 2003). In addition, the ash deposited on the forest floor is easily eroded by the surface runoff. In many cases, the ash and particulates are washed away with the storm events within the first year after a fire (Gallaher and Koch, 2004; Stein et al., 2012). As Minshall et al. (2001) found in central Idaho after the Mortar Creek Fire, there was a higher concentration of suspended sediment persisting in the affected streams during the first year after the fire and the suspended sediment loads were correlated with storm events, with the largest peak occurring during the spring runoff in April.

There are long-term implications of increased sediment transport. Nutrients and metals adsorbed on to the sediment settle in the to the stream and may be reintroduced into the water body through leaching (Dissmeyer, 2000). This may pose potential problems for drinking water utilities due to increased levels of metals, promotion of algal and bacterial growth, which may cause treatment concerns, filtration problems as well as taste and odor issues.

1.1.2 TOTAL ORGANIC CARBON

The organic matter in a watershed can change significantly after fires. In the organic (O) horizon of the soil, the amount of organic matter can be reduced by 9 to 21% after a fire, mostly due to volatilization (Smith, 1970). However, the reduction of organics in the soil greatly depends on the type of vegetation and fire severity. In some cases, the topsoil on the forest floor may have a >90% reduction of organics due to burning of vegetation and litter. However, some of the organic carbon may return to the top soil in atmospheric deposited ash (Smith et al., 2011).

When organic material is combusted completely, white ash is formed. However, in most fires, the combustion is incomplete so the ash is grey or black as it still contains organic material (Ranalli, 2004). Goforth et al. (2005) found that darker pigmented ash contained higher amounts of organic carbon (C) than white ash, so if the dark ash is re-deposited on the soil, the organic C levels would increase.

The most common form of organic C depositing into the streams is the particulate form due to runoff events (Smith et al., 2011). Gill (2004) found an increase in average total organic carbon (TOC) concentrations in the Salt River one year after the Rodeo-Chediski fire in Arizona. The TOC increased from 5.42 mg/L-C pre-fire to 228 mg/L-C after the fire in the Salt River. Although the concentrations were elevated for the first year, the TOC concentration decreased significantly after 2 years, so the impacts of the fire were only concerning for one year post-fire.

Organic C may also be in dissolved form (dissolved organic carbon; DOC). The DOC concentration in 5 Idaho streams after the Mortar Creek fire was directly correlated with storm events, although the DOC concentrations did not exceed 4 mg/L (Minshall et al., 2001). After the Lost Creek fire in the Canadian Rocky Mountains in 2003, the median DOC concentration in

the discharge from the watersheds affected by wildfire for the first two years post-fire was 3 mg/L whereas the reference stream had only 1-2 mg/L DOC. The DOC in the burned South York, Lynx and Drum Creeks remained elevated for at least 4 more years (Emelko et al., 2011). Lakes are more resilient to spikes in DOC concentrations because of dilution so the impact on lakes is less than in streams (Kolka, 2012).

1.1.3 NUTRIENTS

Nutrients, i.e. nitrogen and phosphorus, are essential to the lifecycle of aquatic ecosystems. However from the standpoint of a potable water utility, nutrients can cause issues in the source and treated waters. Presence of nitrogen and phosphorus in surface waters promote algal blooms that result in release of algal metabolites, and lower dissolved oxygen levels when algae die and decompose (Dodds, 2002). Algal metabolites impart off-flavors to the surface and drinking waters (Omur-Ozbek and Dietrich, 2005; Ho et al., 2009; Zuo et al., 2009). Sabater et al. (2003) found that presence of nitrogen and high levels of phosphorus, at 0.39 mg/L-NO₃⁻ and 0.4 mg/L-P respectively, create ideal circumstances for blue-green algae growth. Smith (1983) determined that the epilimnetic N:P ratios of less than 29:1 have a tendency for blue-green algae. Saadoun et al. (2001) confirmed that in freshwater, phosphorus is the limiting factor for the growth of cyanobacteria and an increase in phosphorus can produce taste and odor issues, specifically geosmin production. Changes in nutrient concentrations in the surface waters due to ash and sediments carried with runoff and leaching from the river sediments after fire may promote algal blooms in the short and long term and thus may make source waters unusable for drinking water purposes.

The original soil and vegetation, fire severity and fire intensity affect the post-fire concentrations of nutrients (Stein et al., 2012). Kolka (2012) found that after prescribed, low-severity fires in the Southeastern United States, there is little influence of fire on nitrogen or phosphorus availability. Also in prescribed burns of low severity in the Southern Appalachian Mountains, Elliott and Vose (2005) showed that there were no significant differences in nitrate (mg/L NO_3^- -N), ammonium (mg/L NH_4^+ -N) or phosphate (mg/L PO_4^{3-}) concentrations in the burned versus control watersheds.

However, other studies that investigated the conditions after higher severity wildfires showed that there is a significant difference in nitrogen and phosphorus levels post-fire. After the Lost Creek fire, Emelko et al. (2011) found that the dissolved organic nitrogen (DON) increased from a mean of 120 to 205.3 $\mu\text{g/L-N}$ and total phosphorus increased from a mean of 5.7 to 42.2 $\mu\text{g/L-P}$ in the four years after the fire occurred. The reported nutrient concentrations in the burned creeks (South York, Lynx, and Drum Creeks) were highest within the first year of the fire, but all returned to unburned reference levels within four years (Emelko et al., 2011).

The Cerro Grande fire near Los Alamos, New Mexico, also showed increases in nutrient levels for months after the fire (Gallaher and Koch, 2004). The phosphate concentration was historically at a maximum at 1.74 mg/L-P but peaked to 14.5 mg/L-P two months after the fire. However, the nutrient levels returned to background levels within a year.

Minshall et al. (2001) also reported elevated levels of nitrates and orthophosphates within the first year after the Mortar Creek fire. In another study that focused on fires in southern California, Stein et al. (2012) discovered an increase in mean concentrations of nitrate+nitrite from 0.5 mg/L-N preburn to 2.45 mg/L-N post-fire as well as a 921 fold increase in total

phosphorus (from 0.15 mg/L-P to 15 mg/L-P). It was also noted that the background conditions returned back to normal within one to three years after the fire due to ash and burned soils being removed in runoff.

Hauer and Spencer (1998) investigated the short and long term impacts of the Red Bench fire in Montana in 1988, focusing on nutrients. The researchers determined that the highest concentrations of soluble reactive phosphorus (SRP) and ammonium occurred during the fire itself, and then decreased to baseline levels within a few weeks (Spencer and Hauer, 1991). However, continued testing showed that the first spring runoff after the fire demonstrated increased concentrations of SRP, ammonium and nitrate. SRP peaked at 69.2 $\mu\text{g/L-P}$ compared to $<10 \mu\text{g/L-P}$ in control streams, while ammonium peaked at 220 $\mu\text{g/L-N}$, 2 orders of magnitude higher than control streams. Nitrate peaked at 427 $\mu\text{g/L-N}$ compared to $<75 \mu\text{g/L-N}$ in the first post-fire spring runoff. For the five following years, similar trends were observed for spring runoff. All nutrients returned to almost control stream levels after five years, except for SRP, which remained high.

These studies show that the first year is the most critical time period for nutrient and mineral introduction into streams. The spike in nutrients within the first year post-fire has several causes. Initially, some of the original nitrogen present in the soil is often lost in volatilization during the fire (Grier, 1975; MacDonald and Stednick, 2003; Smith et al., 2011), and a portion is returned to the soil in ash deposition. Partial burning of plant material increases mineralization of organic nitrogen, followed by increased nitrification producing more nitrate, which is an easily transportable anion (Hauer and Spencer, 1998). Ash is also rich in oxidized cations like calcium, magnesium, potassium, and sodium that can also be dissolved in water and transported to streams. Phosphorus typically adsorbs onto sediment particles and is also

transported through erosion (Novotny, 2011). As discussed previously, there is an increase in erosion after fires, so there is also a surge in the nutrient and mineral concentrations in streams. Kutiel and Naveh (1987) found that in the first two months after a fire in Israel, the phosphorus concentration in the soil increased 300%, but was washed away as the ash was removed by wind and water. The concentrations of nutrients transported in streams depend on the fire intensity, the rainfall intensity and duration, as well as the cumulative precipitation after the fire (Stein et al., 2012). After a year, watersheds begin to recover and vegetation and other biota begin to grow on the slopes again. This growth reduces sediment mobilization and thus less nutrient transport (Gill, 2004). Vegetation also takes up the nutrients from the ash and soil for use in biological processes. Over time, the turbidity in the river decreases as the particulates settle, but the transported nutrients remain attached to the particles at the bottom of the stream and thus have the potential to leach back into the water column (Dissmeyer, 2000).

1.1.3.1 RELEASE OF NUTRIENTS FROM SEDIMENT

Wu et al. (2012) studied the release of nutrients from the heavily contaminated Haihe River in northern China that is being considered for restoration to a drinking water source. Laboratory experiments were conducted with varying dredging depths, salinity conditions and light exposure. The ammonia-nitrogen in the leachate reached a maximum of 1.4 mg/L after 3 days for 5 cm dredging depth samples and a maximum of about 2.5 mg/L after 12 days for undredged samples. Total phosphorus and total dissolved phosphorus (TDP) were low for the first 3 days of the test and then linearly increased until day 20 (approximately 0.30 mg/L-P). The results also indicate that the release of nutrients and organic matter is significantly higher from riverbed sediment than from bank sediment. Under different salinity concentrations, the TP and

TDP increased with time, while the ammonia-nitrogen increased until day 12, and then decreased.

In a study of nutrient fluxes in an artificial lake in South Korea, Lim et al. (2011) reported that nitrogen and phosphorus species moved into the overlying water in a benthic chamber. The phosphate concentrations increased from 0.014 to 0.276 mg/L-P over a period of 20 days and ammonia-nitrogen peaked on day 11 with a concentration of approximately 5 mg/L-N. The release rates of TN and TP from the sediment were 61.97 kg/d and 6.55 kg/d, respectively, which are greater than the inflow of nutrients of 4.32 and 0.22 kg/d, respectively. Since the release rates from the sediment were greater than the inflow rates to the lake, the sediment was considered a source of nutrients entering the lake water.

In shallow lakes, internal phosphorus sources are influenced by several factors (Søndergaard et al., 2003). Resuspension of sediment allows for more contact of particles with the water, so there is a higher potential for phosphorus release. Temperature is also influential to phosphorus release into the water body because the oxygen content in the sediment decreases due to increases in microbial activity. This causes the redox potential and the pH to change and the iron (III) compounds with adsorbed phosphorus reduce to iron (II) and the phosphorus is then released into the overlying water (Søndergaard et al., 2003).

1.1.4 METALS

Fires also impact the mobility and concentration of metals. Fires induce a release of metals from the litter and soil. Fire changes the composition of the soil, and alters the pH, which modifies the form metals are present in the soil (Smith, 1970). After fires, total and soluble manganese is increased in soils due to the breakdown of complex organics containing manganese that are present in ash (Smith et al., 2011). Iron, zinc, and copper follow the same trend as manganese (Certini, 2005). The increase in these soluble metals in soils support an increase in streams once precipitation occurs. In Lithuania after drought and fires, copper, lead and zinc increased to 21-74% of the background levels (Sakalauskiene and Ignatavicius, 2003). Ballard (2000) reported mobilization of manganese, zinc, and aluminum in soils after fire. A short flush of metals occurred after the Israeli fires, but concentrations returned to background levels as soon as ash was removed from the watershed (Kutiel and Naveh, 1987). Stein et al. (2012) reported a 3 fold increase in the event mean concentrations of copper, lead, nickel and zinc in rivers after southern California fires. After the Cerro Grande fire, the copper concentration ranged from 4-290 $\mu\text{g/L-Cu}$, which is higher than concentrations prior to the fire (Gallaher and Koch, 2004).

MacDonald and Stednick (2003) note that metals, especially iron and manganese, can become soluble and cause aesthetic issues in drinking water. Iron can produce an orange color to the water and manganese imparts a metallic taste to the water, both of which are unappealing to consumers. The presence of toxic metals could negatively affect the aquatic ecosystem and reduce the algae, diatom, and periphyton populations within the water bodies (Genter and Lehman, 2000). Although this might reduce aesthetic issues in drinking water, it may impact the long-term sustainability of the water body and require further treatment for a safe drinking water.

1.1.5 TASTE AND ODOR ISSUES

Taste and odor (T&O) in drinking water is important to many drinking water utilities around the world. Millions of dollars are spent each year to handle T&O problems in drinking water, by updating treatment systems, managing source waters and for handling customer complaints. If there is a negative aesthetic perception of water, many people believe that there is a problem with the water and will not consume it (McGuire, 1995).

Many compounds present in source and treated waters contribute to off-flavors. For example, geosmin and 2-methylisoborneol (2-MIB) are the most common cyanobacterial compounds detected in source waters around the world, and impart an earthy/musty odor to the drinking water at low ng/L concentrations (Suffet et al., 1995; Dietrich, 2006; Zaitlin and Watson, 2006). While these compounds are still a concern to water utilities, the potential for smoky flavor in drinking water as well as in other food and beverage products is a concern post-fire. Drinking water utilities, breweries, wineries and other members of the food industry are concerned that the smoky flavor might change the flavor of their products and thus cause a negative financial impact.

Parker et al. (2012) studied the effect of smoke on vineyard grapes and the smoky flavors of the resulting wines. It was found that phenols (specifically o-cresol, p-cresol, and m-cresol) and guaiacol were found in smoke treated wines resulting in a smoky/ashy flavor. Since guaiacol and m-cresol have low odor thresholds, 27 and 20 µg/L respectively, they are important indicators of smoke affected wines.

Another study of wine grapes that were exposed to smoke was conducted by Kennison et al. (2008). Among the many volatile phenols found in the smoke affected wine, guaiacol and 4-

methylguaiacol were the most prevalent. Although the juice from smoke-treated grapes only had trace amounts of guaiacol, the fermentation process intensified the concentration to 388 µg/L compared to 4 µg/L in the control wine made from non-smoked grapes.

Fudge et al. (2012) examined the effectiveness of various fining agents to remove the smoke taint from smoke-affected grapes. Activated carbon was most effective at removing the smoky and cold ash aromas from Pinot Noir, Cabernet Sauvignon and Merlot wines. In the Pinot Noir, guaiacol was reduced to 10 µg/L from 24 µg/L and in the cabernet sauvignon and merlot guaiacol was reduced from 18 and 17 µg/L to 8 and 8 µg/L, respectively (Fudge et al., 2012). Since activated carbon can remove guaiacol from tainted wines, there could be potential applications in drinking water uses.

In the food industry, smoke flavoring is sometimes desired in products such as meats, cheeses, and nuts. Smoke flavorings, like liquid smoke solutions, mimic the flavors obtained by traditional food smoking techniques. Fiddler et al. (1970) tried to isolate the single compound that created a smoky flavor. However, no one compound alone produced the desired flavor so it was concluded that the smoky flavor is not limited to one compound, but rather is a combination of several compounds. Analysis of smoke flavorings was also conducted by Kostyra and Barylko-Pikielna (2006) and the main compounds determined were phenols, guaiacols, carbonyls, syringols and their derivatives. Phenols are predominantly responsible for the intensity of the smoky flavoring and carbonyls contribute to the sweet/caramel notes. The combination of many compounds produce the charred or smoky tastes in smoke flavorings. Since the smoke flavorings are made from thermal decomposition of wood, it can be inferred that these compounds can also be found in the watersheds and surface waters after a wildfire. Presence of smoky flavor compounds in source waters may be problematic to drinking water utilities as the

persistence of smoky flavors through conventional drinking water treatment is not well studied or understood.

2.2.2.2 FLAVOR PROFILE ANALYSIS

An easy, quick and reliable way to monitor off-odors drinking water is through sensory analysis. The human nose is very sensitive, sometimes more sensitive than available instrumentation. Although there are many sensory methods available for describing water samples, the Flavor Profile Analysis (FPA) (Standard Method 2170) is a very reliable and robust method (APHA, AWWA, WEF, 2005).

FPA panelists are trained in a daylong event in which they are introduced to basic flavors, mouthfeels and common drinking water odors at varying intensities. Basic tastes include sour, sweet, salty, and bitter at four intensities: very weak (2), weak (4), moderate (8), and strong (12). The different intensity gradations allow for panelists to calibrate themselves. Next, the panelists are given mouthfeel samples that are examples of the physical sensations in one's mouth. These samples do not relate to odor or taste, but rather are physical sensations including tingling, drying, or astringent. The final step of the training is to become familiar with typical drinking water odors. Panelists are given 500 mL Erlenmeyer flasks containing 200 mL of a typical drinking water odor such as geosmin, 2-MIB, or hexanal (Omur-Ozbek and Dietrich, 2008). The panelists are asked to smell the samples and utilizing the taste standards and relate the odors into intensity gradations. This permits panelists to depict the odor of a sample using descriptive adjectives and odor intensities.

In regular FPA panel sessions, the samples are presented in cups or glass 500 mL Erlenmeyer flasks at 45 °C. After each panelist tastes and/or smells a sample, they rate the

descriptor(s) and intensity(ies). Once all samples are tested, the panel leader records each panelist's results and the panel comes to a consensus on descriptors and intensities for each sample. The results provide a quantitative and qualitative way to describe the off-odors in the samples, which is important for utilities so they can modify the treatment processes in order to prevent customer complaints.

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2.0 THE EFFECTS OF THE HIGH PARK FIRE ON CACHE LA POUDRE DRINKING WATER QUALITY

2.1 INTRODUCTION

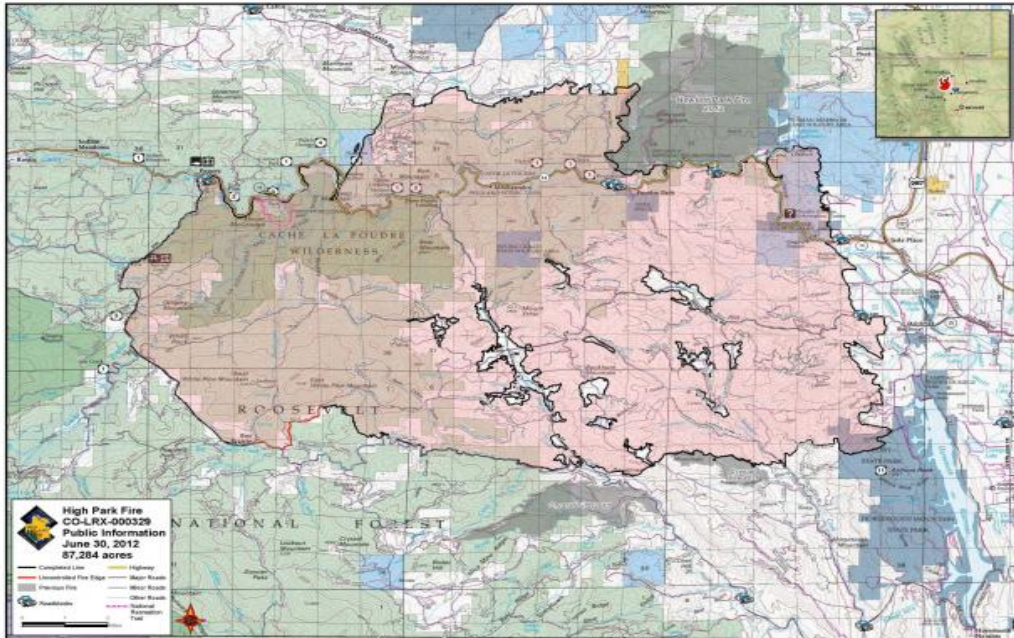
Wildfires are an important part of the natural lifecycle of a forest ecosystem. However, wildfires also pose threats to human safety, property and potable water sources. Many cities throughout the world obtain water for potable uses from forested areas. When forest fire occurs in a watershed, the water may be polluted with contaminants or off-flavor (smoky) compounds.

Colorado, with its mountainous and heavily forested terrain, has seen many large wildfires. In 2002, the Hayman fire grew to over 137,000 burned acres and impacted water supplies of the cities of Denver and Colorado Springs (Graham Technical Editor, 2003). Many other fires occurred in the summer of 2002 due to the severe drought conditions including the Missionary Ridge fire that burned 71,739 acres, the Bircher fire near Mesa Verde National Park, which burned 23,607 acres, and the Trinidad Complex fire that burned approximately 33,000 acres. The summer of 2012 had similar conditions as existed in 2002 including severe drought conditions and several large, intense fires that caused a state of emergency to be declared by State Forest, fire, and government officials (Longmont-Times-Call, 2012). The Waldo Canyon fire near Colorado Springs became the most destructive fire destroying 346 homes (Makings, 2012). The High Park fire near Fort Collins was a large, intense fire and is the focus of this study.

The High Park Fire was ignited on June 9th, 2012, from a lightning strike and was contained on June 30th. Figure 2.1 shows the burn area. In the summer of 2012, most of Colorado was under severe or worse drought conditions due to lack of precipitation the preceding winter and warm temperatures occurring early in the spring (WWA and NIDIS, 2012).

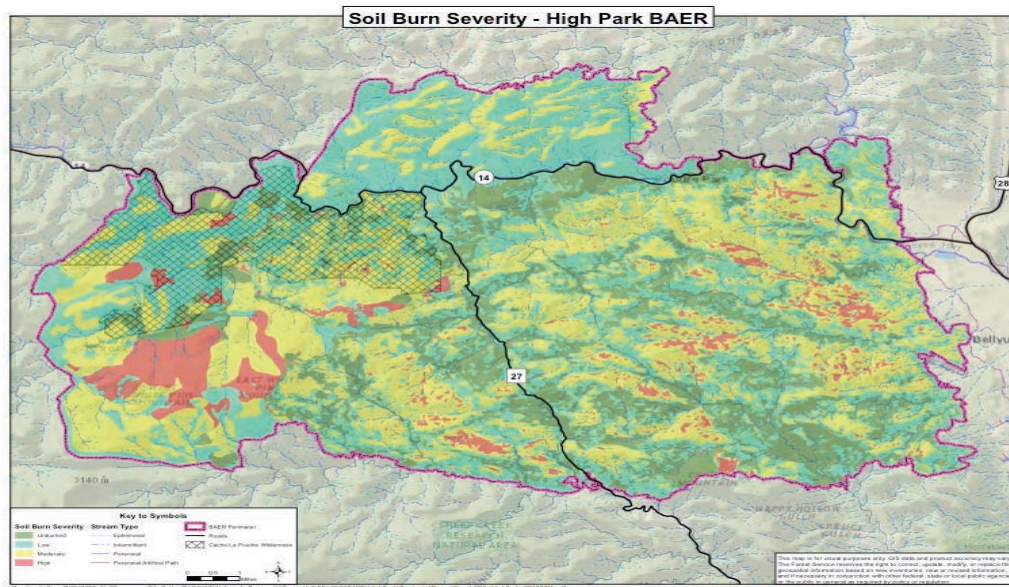
Due to the dry fuel and drought conditions, the fire grew to be 87,284 acres and is the second largest and second most destructive fire to occur in Colorado destroying 259 homes (*High Park Fire Burned Area Emergency Response Report*, 2012). Different areas burned with different severities, but 40.4% of the area had moderate severity and 6.5% with high severity (*High Park Fire Burned Area Emergency Response Report*, 2012). Figure 2.2 shows the burn severity within the watershed.

The High Park Fire was located predominately in the Cache la Poudre watershed, which is a source for drinking water utilities in Fort Collins and Greeley, CO. Along with the Hewlett fire Northeast of the High Park fire (May 14-22, 2012), 25% of the Poudre watershed was affected (Oropeza and Dustin, 2012). The other primary water source for Fort Collins utilities is Horsetooth Reservoir, which is a part of the Colorado-Big Thompson Project. Only 4% of the watershed draining into Horsetooth Reservoir was affected by the High Park fire (Oropeza and Dustin, 2012).



(courtesy of US Forest Service)

Figure 2.1 - High Park fire Burn Area map



(High Park Fire Burned Area Emergency Response Report, 2012)

Figure 2.2 - Burn Severity map of High Park Fire

Wildfires induce changes within watersheds that affect soil composition, hydrology, and water quality. A higher severity fire destroys more vegetation than lower severity fires and can also change the composition of the soil by creating a hydrophobic layer (DeBano, 2000). This layer decreases infiltration capacity, so instead, precipitation is converted to runoff causing a higher sediment transport than in unburned areas (Gottfried et al., 2003; Gallaher and Koch, 2004; Neary, 2004; Kolka, 2012). The lack of vegetation post-fire destabilizes the hillside so erosion potential increases leading to runoff and turbidity (Morris and Moses, 1987; Wondzell and King, 2003). Increased turbidity poses difficulties for water treatment process, for example, the clogging of filters.

Total Organic Carbon

Total organic carbon (TOC) in rivers within fire affected watersheds increases due mobilization of organics from the soil and ash. Gill (2004) found an increase in average total organic carbon (TOC) concentrations from 5.42 mg/L-C pre-fire to 228 mg/L-C after the fire in the Salt River one year after the Rodeo-Chediski fire in Arizona. Although the most prevalent form of organic C depositing into streams is the particulate form, increases in dissolved organic carbon (DOC) after fires also occur. DOC in the streams after the Mortar Creek fire correlated with storm events although the concentrations did not exceed 4 mg/L-C in the first year after the fire (Minshall et al., 2001). After the Lost Creek fire in the Canadian Rocky Mountains in 2003, the median DOC concentration in the discharge from the watersheds affected by wildfire for the first two years post-fire was 3 mg/L whereas the reference stream had only 1-2 mg/L DOC (Emelko et al., 2011). The increase in TOC or DOC could cause issues for drinking water utilities because of increased potential for taste and odor issues and the formation of disinfection-by-products.

Nutrients - Nitrogen and Phosphorus

Fires also change nutrient concentration, specifically nitrogen and phosphorus, in receiving water bodies. After the Lost Creek fire, Emelko et al. (2011) found that the dissolved organic nitrogen (DON) increased from a mean of 120 to 205.3 $\mu\text{g/L-N}$ and total phosphorus increased from a mean of 5.7 to 42.2 $\mu\text{g/L-P}$ in the four years after the fire occurred. The Cerro Grande fire near Los Alamos, New Mexico, also showed increases in nutrient levels within months of the fire (Gallaher and Koch, 2004). The phosphate concentration was historically at a maximum at 1.74 mg/L-P but peaked to 14.5 mg/L-P two months after the fire. In another study that focused on fires in southern California, a region with a history of frequent fires, Stein et al. (2012) discovered an increase in mean concentrations of nitrate+nitrite from 0.5 mg/L-N preburn to 2.45 mg/L-N post-fire as well as a 921 fold increase in total phosphorus (from 0.15 mg/L-P to 15 mg/L-P). In the first spring runoff after the Red Bench fire in Montana in 1988, the soluble reactive phosphorus (SRP) peaked at 69.2 $\mu\text{g/L-P}$ compared to <10 $\mu\text{g/L-P}$ in control streams, while ammonium peaked at 220 $\mu\text{g/L-N}$, 2 orders of magnitude higher than control streams. Nitrate peaked at 427 $\mu\text{g/L-N}$ compared to <75 $\mu\text{g/L-N}$ in the first post-fire spring runoff (Hauer and Spencer, 1998). Leaching of nutrients from riverbed sediments unaffected by fire from the Haihe River in China was studied by (Wu et al., 2012). The ammonia-nitrogen reached a maximum of 1.4 mg/L after 3 days for 5 cm dredging depth and a maximum of about 2.5 mg/L after 12 days for undredged samples. Total phosphorus and total dissolved phosphorus (TDP) were low for the first 3 days of the test and then linearly increased until day 20 (approximately 0.30 mg/L-P). The results also indicate that the release of nutrients and organic matter is significantly higher from riverbed sediment than from bank sediment. Lim et al. (2011) reported similar results for an artificial lake in South Korea unaffected by fire. The phosphate concentrations increased from 0.014 to 0.276 mg/L-P over a period of 20 days and ammonia-

nitrogen peaked on day 11 with a concentration of approximately 5 mg/L-N. Presence of nitrogen and phosphorus in surface waters promote algal blooms that result in release of algal metabolites, and lower dissolved oxygen levels when algae die and decompose (Dodds, 2002). Algal metabolites including impart off-flavors to the surface and drinking waters (Omur-Ozbek and Dietrich, 2005; Ho et al., 2009; Zuo et al., 2009), so are a concern to potable water utilities.

Metals

After fires, total and soluble manganese is increased in soils due to the breakdown of complex organics containing manganese that are present in ash (Smith et al., 2011). Iron, zinc, and copper follow the same trend as manganese (Certini, 2005). The increase in these soluble metals in soils support an increase in streams once precipitation occurs. In Lithuania after drought and fires, copper concentrations increased from a background range 0.6-11.7 µg/L-Cu to 0.6-26.1 µg/L-Cu (Sakalauskiene and Ignatavicius, 2003). Stein et al. (2012) reported a 3 fold increase in the event mean concentrations of copper, lead, nickel and zinc in rivers after southern California fires. After the Cerro Grande fire, the copper concentration ranged from 4-290 µg/L-Cu, which is higher than concentrations prior to the fire (Gallaher and Koch, 2004). Metals, especially iron and manganese, can become soluble and cause aesthetic issues in drinking water including discoloration or metallic taste (MacDonald and Stednick, 2003). Anticipating increases in metals in water after fires permit drinking water utilities to plan for treatment changes.

Taste & Odor Issues

As discussed previously, increased nutrients create the potential for algae to grow and thus for algal metabolites to be introduced into the water body. Geosmin and 2-methylisoborneol (2-MIB) are the most common cyanobacterial compounds detected in source waters around the world, and impart an earthy/musty odor to the drinking water at low ng/L concentrations. (Suffet et al., 1995; Dietrich, 2006; Zaitlin and Watson, 2006). While these compounds are still a concern to water utilities, the potential for smoky flavor in drinking water, as well as in other food and beverage products, is a concern post-fire. Drinking water utilities, breweries, wineries and other members of the food industry are concerned that the smoky flavor might change the flavor of their products and thus cause a negative financial impact.

Parker et al. (2012) studied the effect of smoke on vineyard grapes and the smoky flavors of the resulting wines. It was found that phenols (specifically o-cresol, p-cresol, and m-cresol) and guaiacol were found in smoke treated wines resulting in a smoky/ashy flavor. Another study of wine grapes that were exposed to smoke established that of the many volatile phenols found in the smoke affected wine with guaiacol and 4-methylguaiacol being the most prevalent (Kennison et al., 2008). Members of the wine industry are interested in removing the smoky flavor from the wine, so different removal techniques are considered. In a study conducted by Fudge et al. (2012), activated carbon was effective at removing the smoky and cold ash aromas from Pinot Noir, Cabernet Sauvignon and Merlot wines that were made with smoke-affected grapes. In the Pinot Noir, guaiacol was reduced to 10 µg/L from 24 µg/L and in the cabernet sauvignon and merlot, guaiacol was reduced from 18 and 17 µg/L to 8 and 8 µg/L, respectively (Fudge et al., 2012). Since activated carbon can remove guaiacol from tainted wines, there could be potential applications in drinking water uses. Powdered activated carbon (PAC) has been used in

treatment of taste and odor compounds in drinking water including geosmin and 2-MIB (Koester, 2011), but little is known about its application for removing smoky compounds from drinking water.

2.2 MATERIALS AND METHODS

2.2.1 SENSORY ANALYSIS

2.2.1.1 PILOT PLANT SETUP

A scaled pilot treatment plant at the Fort Collins Water Treatment Facility (FCWTF) was utilized to simulate large-scale water treatment operations. The pilot plant included blending mechanisms, coagulation and flocculation tanks, and a settling basin. Water from the Cache la Poudre River and Horsetooth Reservoir (two primary sources for drinking water for the FCWTF) were blended in varying ratios and fed through the pilot at 8 gal/min. Samples were collected just after the settling basin in 500 mL amber glass bottles with screw caps from Fisher Scientific (Pittsburgh, PA) to eliminate headspace and 1 L Nalgene plastic bottles (Fisher Scientific, Pittsburgh, PA) for metals analysis. Raw water samples of Poudre River and Horsetooth Reservoir were collected in the same manner from separate taps within the facilities. The collected samples were stored in a refrigerator at 3-4 °C until analyzed within several days.

Advanced treatment with powdered activated carbon (PAC) (Hydrodarco-B, Norit Americas Inc., Marshall, Texas) was implemented in the pilot using a Masterflex peristaltic pump to deliver PAC to the blended, untreated water. For PAC runs, the PAC stock solution at 24,000 mg/L concentration was added to the pilot at a rate of 6.5 mL/min, corresponding to 5 mg/L PAC.

Guaiacol was purchased from Sigma Aldrich (St. Louis, MO). Guaiacol stock solution was prepared at 20 g/L methanol in a 40mL amber glass EPA approved volatile organic analysis (VOA) vial (Fisher Scientific, Pittsburgh, PA). This mixture was then placed in a 1L long necked flask containing 980mL deionized water, producing a guaiacol concentration of 400

mg/L. This solution was pumped into the blended water at the start of the pilot with an additional Masterflex pump for specified runs. The guaiacol solution was added at 4 mL/min rate to obtain 50 µg/L in the pilot.

For each ratio of Poudre River to Horsetooth Reservoir, two runs were performed: conventional pilot treatment, and conventional pilot treatment with PAC addition as shown in Table 2.1 below.

Table 2.1- Summary of Pilot Plant Runs

% Poudre	% Horsetooth	PAC dose (mg/L)	Guaiacol dose (µg/L)
10	90	0 and 5	-
25	75	0 and 5	-
50	50	0 and 5	-
75	25	0 and 5	-
100	0	0 and 5	-
50	50	0 and 5	50

2.2.2.2 BENCHSCALE SEDIMENT LEACHING TESTS

Sensory analysis was also performed for benchscale jar testing leachate samples. Water was collected directly from the Poudre River on November 5th, 2012, in 2 gallon amber glass jugs. Sediment from the riverbed (just above the water line and at bankfull) were collected in 70 mL plastic containers (Fisher Scientific, Pittsburgh, PA) and kept in the freezer until analysis. 5 g of sediment was placed in the 2 gallon jug, mixed, and allowed to sit overnight at 4°C. One hour before jar testing, the sample was mixed again and allowed to settle. After 1 hour, 1 L of sample was poured into each of 6 1 L glass beakers (Fisher Scientific, Pittsburgh PA). Two beakers were designated controls and only contained the prepared sample. Two beakers received conventional treatment (alum and coagulant dose- 33 µL alum, 30 µL polymer), and the final two

beakers were given conventional treatment with 5 mg/L PAC. The jar testing conditions were as follows: rapid mix-150 RPM 2 min, flocculation- 25 RPM for 18 min, sedimentation-0 RPM for 40 min. The supernatant from each beaker was collected in 500 mL amber glass bottles to be used for sensory analysis.

Flavor Profile Analysis (FPA), was performed for the water samples following the Standard Method 2170 as briefly explained below (APHA, AWWA, WEF, 2005).

2.2.2.2 GLASSWARE PREPARATION

500 mL wide-mouth glass Erlenmeyer flasks (Fisher Scientific, Pittsburgh, PA) with ground glass stoppers were used for sensory analysis. Without the use of rubber gloves, flasks were washed with Sparkleen® odor-free detergent (Fisher Scientific, Pittsburgh, PA) and rinsed at least three times with tap water and three times with odor-free Nanopure® Diamond ultrapure water (Barnstead/Thermolyne, model D11931, Dubuque, IA) until the flasks were odor-free. The glassware was stored containing 200 mL of purified water. Before use, the flasks were rinsed twice with odor-free water.

2.2.2.2 SAMPLE COLLECTION AND PREPARATION

Samples for FPA analysis were collected in 500 mL amber glass bottles with screw caps (Fisher Scientific, Pittsburgh, PA). The bottles were completely filled to eliminate headspace so volatile compounds were not lost in the headspace. The samples were kept at 3-4 °C until analyzed. For analyses, 200 mL of a sample was gently poured into a prepared 500 mL Erlenmeyer flask.

2.2.2.2 FLAVOR PROFILE ANALYSES

The sensory study was approved by the Institutional Review Board for Research Involving Human Subjects at Colorado State University (Protocol 12-3796H). Two panels were convened at Fort Collins Water Treatment Facility (FCWTF) and New Belgium Brewery (NBB). The FCWTF panel consisted of 5 to 6 people, 3 men and 3 women with ages ranging from early 20s to late 50s. The NBB panel was larger with 9 people, 5 women and 4 men with ages ranging from early 30s to late 40s.

The 500 mL Erlenmeyer flasks were placed in a water bath (Fisher Scientific, model 15-460-S20Q, Pittsburgh, PA) and warmed to 45 °C. Panelists smelled the samples one by one and recorded their individual impressions of odor descriptors and intensities after each sample. Odor-free flasks were provided to restore panelists' sensitivities. After all samples were tested individually by each panelist, the results were discussed as a panel until a consensus was reached.

2.2.2 SEDIMENT LEACHING

2.2.2.1 SAMPLE COLLECTION

Sediment samples were collected from five locations within the Cache la Poudre watershed within the fire burn area: Century Park, Stevens Gulch, Youngs Gulch, Mile Marker 115, Fort Collins Intake, as shown in Figure 2.3.

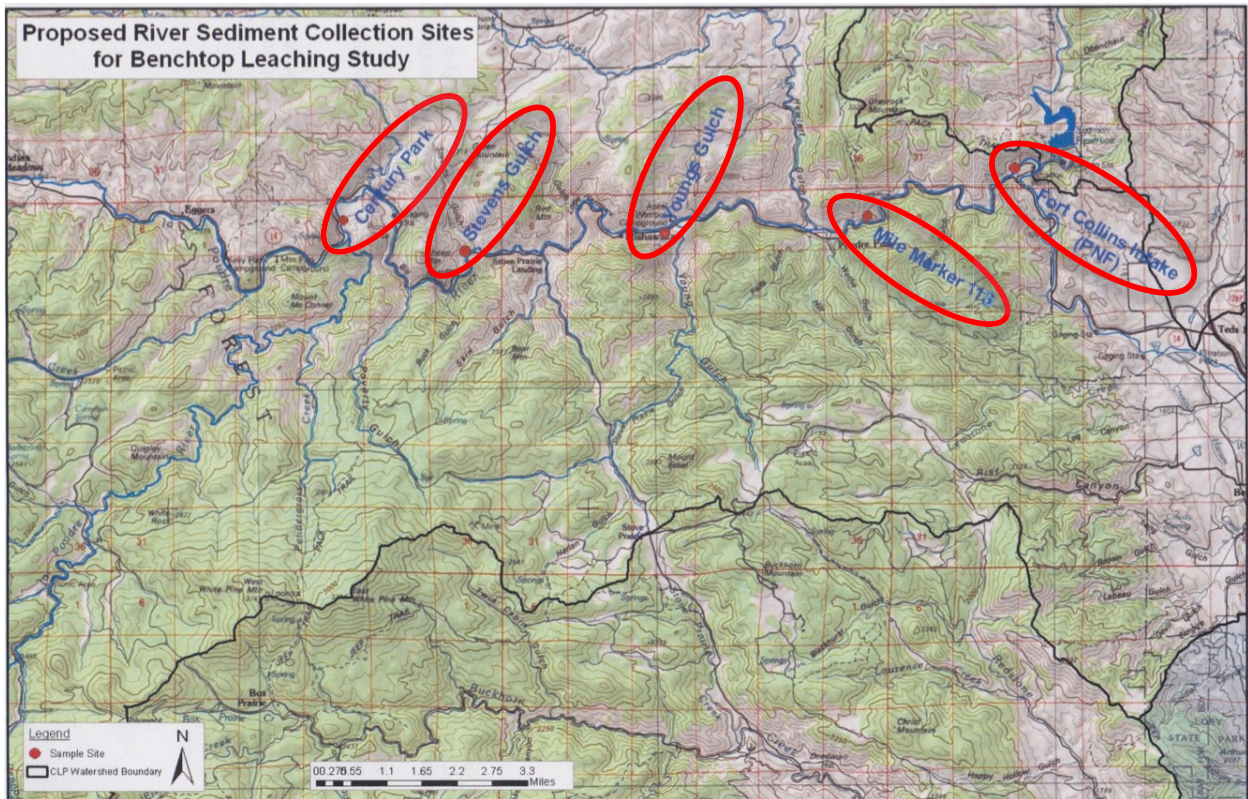


Figure 2.3 - Map of Poudre River with Sediment Sampling Locations

At each location, samples were collected in triplicates from the riverbed (bottom), the riverbank (middle), and just above bankfull (upper). Figure 2.4 shows examples of the riverbed locations. A total of 45 sediment samples were collected from the five sites. Each sample was collected using a shovel and placed in a gallon-sized plastic bag with seal. Samples were dried first at room temperature and then in a muffle furnace at 105 °C. The 105 °C evaporated the water to give dry weight of the soil, but may have also volatilized some compounds, such as

ammonia. The samples were then sifted through a 0.991 mm sieve, which represents fine silt and clays in the American Society for Testing Methods (ASTM) classification (Gee and Or, 2002). The 0.991 mm sieve excluded sands, gravels and cobbles >0.991mm in diameter. The sifted sediment was placed in another gallon-sized plastic bag with seal and placed in the freezer until analysis.



Figure 2.4 - Riverbed Sediment Sample Locations

2.2.2.2 BENCHSCALE LEACHING TESTING PROCEDURES

For leaching test preparations, the samples from one site location, the riverbed sediment (bottom) were mixed, the middle sediments were mixed, and the upper sediments were mixed until homogenous. For each layer of sediment (bottom, middle and upper) 100 g of sediment was placed in a Pyrex glass beaker and received 1 L of Poudre River water,(Fisher Scientific, Pittsburgh, PA) and mixed, in duplicate. This grouping of 6 beakers was designated the “6 hour” leaching samples. Two additional groups of 6 beakers (2 Bottom, 2 Middle, and 2 Upper) were made in the same manner as described above and designated “12 hour” leach and “24 hour”

leach groups. Additionally, 2 1 L glass beakers were each filled with 1 L Poudre River water and served as controls. Figure 2.5 shows the setup of the leaching beakers. The samples were then kept in dark at room temperature.

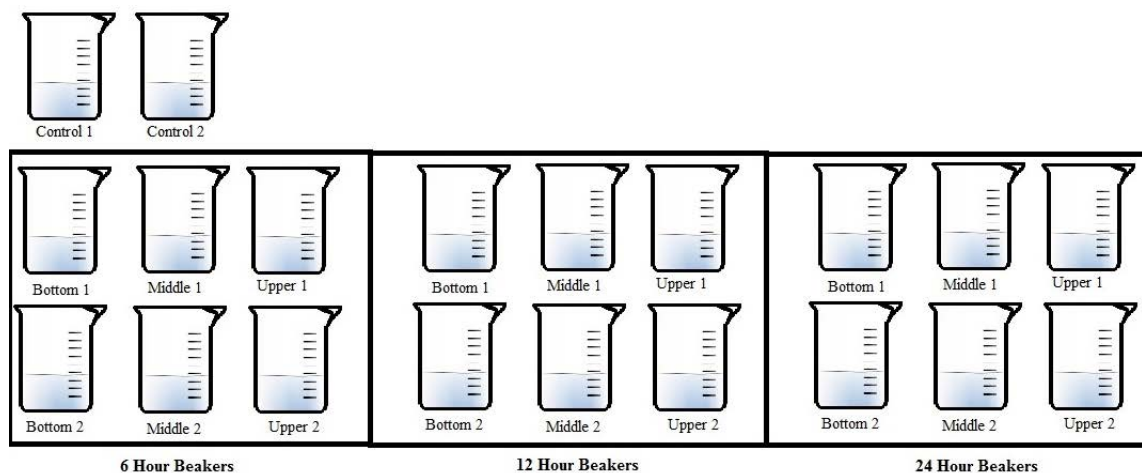


Figure 2.5 - Sediment Leaching Beaker Setup

After 6 hours, 100 mL of the leachate from the “6 hour” beakers were removed, passed through a Whatman 25mm-diameter glass microfiber filter (1.6 μm retention GF/A, Maidstone, UK), and placed in 40 mL amber volatile organic analysis (VOA) vials with Teflon lined caps (Fisher Scientific, Pittsburgh, PA) and kept in the refrigerator at 4 $^{\circ}\text{C}$ until they were analyzed. Metals (iron, copper and manganese) were tested immediately after extraction. Also after 6 hours, 100 mL of each control sample was removed and analyzed. The 1.6 μm filter excluded molecules with diameters larger than 1.6 μm , which limits the number of larger particles, greater than sand/silt size, from influencing results. The leachate and controls were analyzed for Total Organic Carbon (TOC), Total Nitrogen (TN), Total Phosphorus (TP), iron, copper, and manganese, according to Hach Methods (Hach DR 4000, Loveland, CO) 10129, 10071, 8190, 8008, 8506, and 8034, respectively. Water quality parameters including pH (Hach model

51940), Dissolved Oxygen (DO) (Hach DO probe model 51970), temperature, and conductivity (Hach model 51975) were measured using a Hach SensION156 Portable Multiparameter Meter (Loveland, CO). Turbidity was measured using Hach Turbidimeter 2100N (Loveland, CO). After 12 hours of leaching, leachate from the “12 hour” beakers were removed and the same analyses were conducted. The same procedure was repeated after 24 hours.

2.3 RESULTS AND DISCUSSION

2.3.1 SENSORY ANALYSIS

Overall Panel Results

The odor intensity scale for FPA has seven points: the even numbers between 0 (odor-free) to 12 (strong). Threshold value is assigned when the panelists can detect an odor in the sample, yet cannot identify a descriptor due to its low intensity. Comparing the two panels, panel #2 gave consistently higher intensities than panel #1 for the similar odor descriptors. This may be because panel #2 was more sensitive to the odors and were trained to detect various odors in various matrices. Odor free and other spiked samples were also included in the sample pool to verify panelist authenticity. Both panels identified the accuracy check samples correctly.

Raw water

Poudre River water collected directly from the river soon after the fire was contained had a smoky odor that was noticeable at weak to weak-moderate intensities (4 and 7) by both panels. These results confirmed that the fire imparted smoky flavor to the river water. This may have a negative impact on the finished water quality of the drinking water utilities and food industries in Fort Collins due to the smoky off flavor.

Pilot plant- PAC effectiveness

The pilot study runs at the FCWTF consisted of water from the 2 primary drinking water sources (the Poudre River and Horsetooth Reservoir) through a scaled version of conventional treatment and comparing those results to similar runs treated with 5 mg/L PAC. The results from the pilot plant studies are seen in Figure 2.6. The samples containing predominantly Horsetooth Reservoir water were reported to have earthy or musty odors at weak intensities or threshold. These samples were tested in the fall, which is when algae die and release geosmin, and earthy smelling compound, into the water body (Rashash et al., 1995; Watson, 2004). The water level in Horsetooth Reservoir was getting low because it was the only source being used following the onset of the fire, so the concentration of geosmin was higher (less diluted).

Panel #1 consistently reported earthy or musty odors at very-weak (2) intensity for 7 of the 10 samples. This indicates that panelists are sensitive to geosmin. Panel #2 also noticed geosmin at weak (3) intensity for both the samples untreated and treated with PAC. Since the odor intensities did not decrease with the addition of 5 mg/L dose of PAC, this concentration was insufficient to remove geosmin and other taste and odor compounds from the samples. Elevated Total Organic Carbon (TOC) concentrations from the fire may inhibit earthy/musty odor removal with PAC. Higher concentrations and/or longer contact time could remove those odors from the water.

Neither panel recorded smoky as a descriptor for the pilot plant tests. This may be due to the fact the FCWTF utilizes a presedimentation basin prior to the pipeline to the facility, so many of the particles containing smoky compounds were settled before entering the facility. Also, the pilot samples were conducted approximately 4 months after the fire, by which time the flush of

ash could have subsided greatly, so a smoky flavor was not detected in the pilot studies. Regardless of the explanation, a smoky flavor was not detected in the pilot studies.

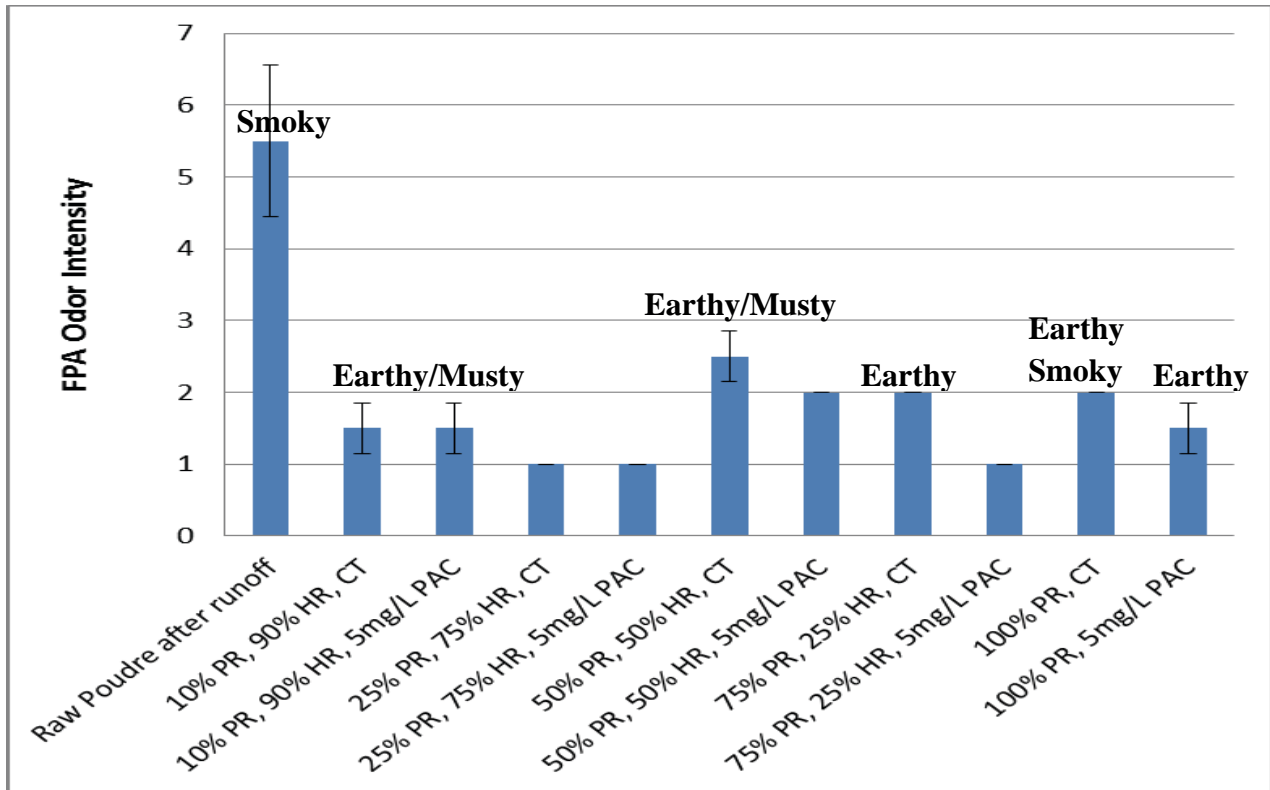


Figure 2.6 – FPA Results for the Pilot Plant Runs

Benchscale & Sensory Tests

The benchscale experiments followed by the sensory tests were conducted to determine whether conventional or PAC treatment was effective in removing possible smoky off flavors in water containing burned river sediment. Figure 2.7 shows the results from FPA studies conducted with the supernatant collected from the benchscale jar testing. Panel 1 described the controls as weak (3) smoky with very-weak (2) musty and earthy odors. Although the alum and polymer treated samples contained an additional fishy descriptor (possibly become noticeable as

the smoky odor masking it was removed), the smoky odor was removed and odor intensity average was decreased to 3.5. The samples with PAC treatment had an average of 2 (very weak) for panel 1, so the results confirm that additional treatment removes taste and odor compounds from the water. Similar results were obtained from panel 2. Panel 2 described most of the samples as earthy smelling. The overall control average odor intensity recorded by panel 2 was 5 (weak-moderate), which is higher than the conventionally treated samples of 4.5 (weak). The samples treated with alum, polymer, and PAC received intensity ratings of 1.5, which is much lower than the conventionally treated samples. The same concentration of PAC (5 mg/L) was used in the benchscale test and in the pilot studies, however this concentration was more effective in the benchscale tests. This may be due to the fact that the batch system of the jar test provided greater contact time and better mixing for the organic compounds to adhere to PAC than in the pilot setup.

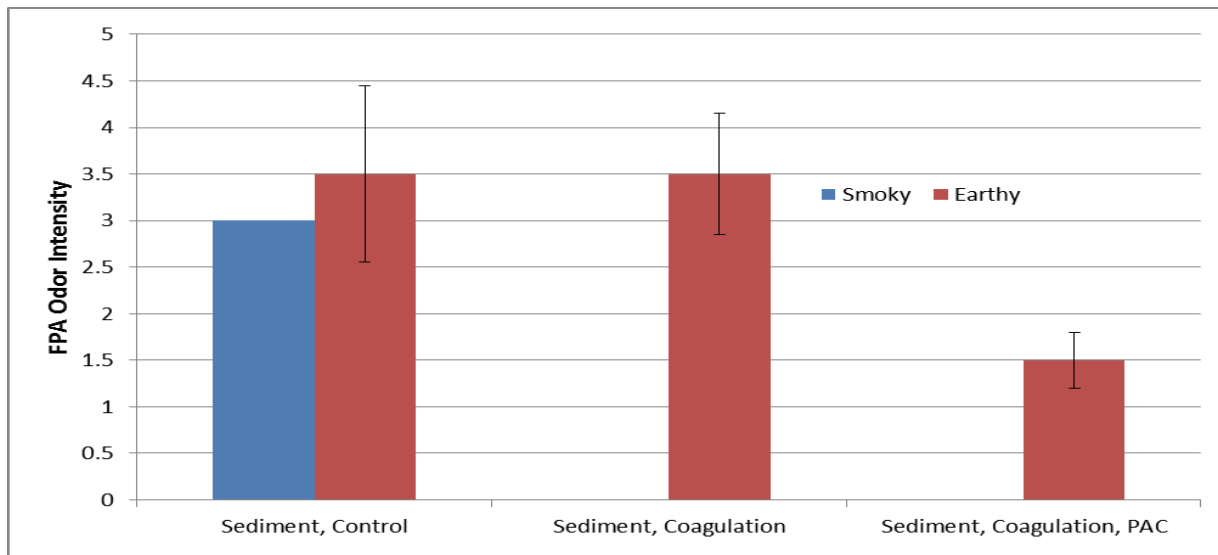


Figure 2.7 - Benchscale FPA Results

Guaiacol tests

Guaiacol, a compound that has a smoky flavor, was selected to test the treatment system's effectiveness in removing smoky compounds and the results are seen in Figure 2.8, below. Other smoky compounds like phenols, carbonyls, syringols and their derivatives have been identified in smoky flavorings from the food industry (Fiddler et al., 1970; Kostyra and Barylko-Pikielna, 2006). Guaiacol was selected due to its low threshold of 27 µg/L (Parker et al., 2012). The 50 µg/L concentration of guaiacol was used to ensure panelists would smell the odor. This compound was also selected because activated carbon has been applications of removing guaiacol from wines made from smoke exposed grapes (Fudge et al., 2012). Both conventional and PAC treatments did not seem to have an impact on the removal of guaiacol. The panels recorded a smoky odor with a weak (4) to weak-moderate (6) intensity that persisted even after the PAC treatment. Since both the conventional treatment and the advanced treatment produced the same odor-intensities for each panel, the dosage of 5 mg/L PAC was determined to be ineffective in removing guaiacol from water. As discussed above, inadequate contact time of the PAC with the contaminated water doesn't allow for the organic compounds to adsorb into the carbon pores. Also the PAC type tested may not be very suitable in removing guaiacol due to its physical (pore size) and chemical properties (Iodine number: 500 mg/g, tannin value: 200 mg/L) (Norit, 2012). The PAC used (Hydrodarco-B, Norit Americas Inc., Marshall, Texas) is designed specifically to remove geosmin and 2-methylisoborneol, not smoky compounds, so another activated carbon type may be considered for removal of smoky compounds. A higher dosage and longer contact time could remove the guaiacol compound and other smoky compounds.

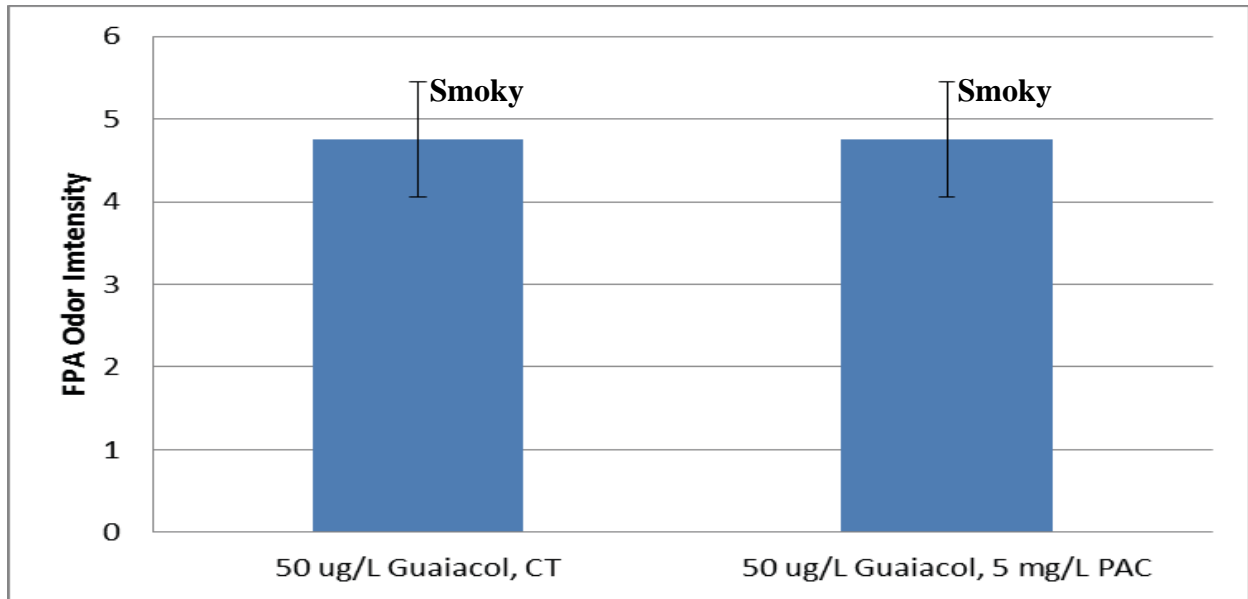


Figure 2.8 - Pilot Studies with 50 µg/L Guaiacol (performed 10/31/12, 11/7/12)

2.3.2 SEDIMENT LEACHING

The results from the leaching studies for organics and nutrients (TOC, TN, and TP) are provided in Table 2.2. Statistical analyses were performed using a 1-way analysis of variation (ANOVA) with $\alpha=0.05$. In comparing the controls concentrations for each parameter, significant differences were not observed except for TOC and TP. The control concentrations at Stevens Gulch for TOC and TP were much lower than controls at other sites, so removing these outliers, there was not a significant difference observed. In comparing the leaching contact times (6, 12, 24 hours) for each riverbed location and site, there is a significant difference in TOC concentrations ($p<0.05$) for 8 of the 15 locations. The highest concentrations occurred after the 24-hour leaching time, which indicates TOC release increased with time. For each site, comparisons of riverbed location concentrations were significant for 3 of the 5 sites ($p<0.027$). The middle riverbed locations had the highest TOC concentrations (averages: Century Park=

41.23 mg/L, Youngs Gulch=20.6 mg/L, FCWTF Intake=42.07 mg/L). In comparing the TOC concentrations from riverbed locations at each site, there was a significant impact of sites on TOC concentrations ($p < 5 \times 10^{-6}$). The FCWTF Intake had the highest concentrations for all riverbed locations (averages: Bottom=36.33 mg/L, Middle=42.07 mg/L, Upper=40.85 mg/L), indicating the accumulation of sediments downstream may affect water quality more. The concentrations were much less than the average post-fire concentration found in the Salt River in Arizona (228 mg/L-C) collected within 2 years after the Rodeo-Chediski fire (Gill, 2004). The increase determined with this study was not as significant because the controls obtained from the Poudre River water were already impacted by the fire. Whereas an increase of 4107% was observed for the Salt River when the before and after fire TOC levels were compared. In contrast, the TOC levels in the leachate (average maximum ranged 19.16 to 44.15 mg/L) from this study contained higher concentrations than the dissolved organic carbon concentrations (<4 mg/L-C) reported after the Mortar Creek fire. This may indicate that even though the samples for this study were filtered with a 1.6 μm glass fiber filter, the particulate organic carbon may contribute a significant amount.

Table 2.2 - Organics and Nutrient Results from Sediment Leaching

	Century Park	Stevens Gulch	Youngs Gulch	Mile Marker 115	FCWTF Intake
TOC	Leaching contact time does not have a significant effect on TOC concentrations. However, time is significant for the middle riverbed location ($p=0.04577$). The highest average from the middle location is at 24 hours (45.3 mg/L). Riverbed location is significant to TOC concentrations ($p=0.0268$). The highest concentration is in the middle location (41.23 mg/L).	Leaching contact time does have a significant effect on TOC concentrations ($p<0.05$). The 12 and 24 hour leaching times have the highest concentrations of 22 mg/L. Riverbed location is not significant to TOC concentrations.	Leaching contact time is not significant for TOC concentrations. Riverbed location is significant to TOC concentrations ($p=6.76 \times 10^{-5}$). The highest average concentration occurred in the middle location (20.6 mg/L).	Leaching contact time does have a significant effect on TOC concentrations ($p<0.05$). Highest average concentrations occur after 24 hours (42 mg/L) for the bottom and middle locations. Riverbed location is not significant to TOC concentrations.	Leaching contact time does have a significant effect on TOC concentrations ($p<0.05$). The highest concentrations occur after 6 hours for the bottom location (39.4 mg/L) & 24 hours for the middle location (45.3 mg/L). Riverbed location is significant to TOC concentrations ($p=0.0058$). The highest average is located in the middle (42.1 mg/L)
TN	Leaching contact time does not have an effect on TN concentrations. Riverbed location does have a significant effect on TN concentrations ($p=9.67 \times 10^{-7}$). The middle location has the highest concentration (6.68 mg/L-N).	Leaching contact time does not have an effect on TN concentrations. The highest average was after 12 hours for the middle location (3.4 mg/L). Riverbed location does have a significant effect on TN conc. ($p=0.0131$). The upper location has the highest concentration (4.04 mg/L).	Leaching contact time does not have an effect on TN concentrations. The highest average was after 6 hours for the middle location (5.45 mg/L). Riverbed location does have a significant effect on TN concentrations ($p=4.59 \times 10^{-8}$). The middle location has the highest concentration (5.02 mg/L).	Leaching contact time does not have an effect on TN concentrations. Riverbed location does have a significant effect on TN concentrations ($p=8.6 \times 10^{-6}$). The upper location has the highest concentration (5.08 mg/L-N).	Leaching contact time does not have an effect on TN concentrations. Riverbed location does not have a significant effect on TN concentrations. However, the highest average concentration occurred the middle location (5.58 mg/L-N).
TP	Leaching contact time does not influence TP concentrations. Riverbed location is significant for TP ($p=1.66 \times 10^{-9}$). The upper location has the highest average concentration of 0.705 mg/L-P.	Leaching contact time does not affect TP concentrations. Riverbed location is significant for TP ($p=0.0089$). The upper location has the highest average concentration of 0.572 mg/L-P.	Leaching contact time does not affect TP concentrations. Riverbed location is significant for TP ($p=2.99 \times 10^{-4}$). The middle location has the highest average 0.747 mg/L-P.	Leaching contact time does not affect TP concentrations. Riverbed location is significant for TP ($p=1.37 \times 10^{-12}$). The upper location has the highest average concentration of 1.32 mg/L-P.	Leaching contact time does not affect TP concentrations. Riverbed location is significant for TP ($p=0.0306$). The middle location has the highest average concentration of 0.705 mg/L-P.

In comparing the TN concentrations from each riverbed location over time, a significant difference was not observed ($p > 0.05$). Time did not significantly affect TN concentrations for 13 of the 15 riverbed locations tested. However, riverbed location had a significant impact on TN concentrations ($p < 0.013$). The middle and upper locations had the highest concentrations at Century Park and Youngs Gulch (averages: 6.68 mg/L-N, 5.02 mg/L-N), and Stevens Gulch and Mile Marker 115 (averages: 4.04 mg/L-N, 5.08 mg/L-N), respectively. Sites also had a significant impact on TN concentrations ($p < 4.8 \times 10^{-5}$) with the highest concentrations for the middle and upper locations from Century Park (averages: 6.68 mg/L-N, 5.17 mg/L-N, respectively). The highest concentration from all the bottom locations came from the FCWTF Intake (average=4.93 mg/L-N).

The maximum average TN concentrations determined by this study ranging from 3.83 to 5.67 mg/L-N are higher than the levels reported for the leaching tests conducted by Wu et al. (2012) in China (1.4 and 2.5 mg/L-N). However the concentrations from Wu et al. (2012) were reported for ammonia-nitrogen so does not account for the other forms of nitrogen that might be present in the leachate, so the TN concentrations would possibly be higher. This study was conducted with 100 g/L of sediment, while the Wu et al. (2012) tested 5 to 10 cm of sediment in 2 L of water (container 10 cm diameter, 50 cm height), which may contribute to the lower nitrogen concentrations. The Wu et al. (2012) study also was not concerned with post fire conditions, but rather a severely contaminated river. In another study by Lim et al. (2011), leaching tests were performed with lake sediments and reported the peak ammonia-nitrogen concentration of 5 mg/L-N. Although this value is similar to the TN values reported in this study, the experiment involved lake sediments that might have higher accumulation of nitrogen

species, providing higher leaching potential of nitrogen. The lake's watershed was also unaffected by fire, so concentrations may not be directly compared.

The TN concentrations found in this study are similar to the concentrations (Stein et al., 2012) found after southern California fires. The mean concentration of nitrate+nitrite after the California fires was 2.45 mg/L-N in river water samples. This study was conducted in a batch system so the higher concentrations found (max average 3.83-5.67 mg/L-N) would be diluted if in the Poudre River, which would correlate better to the concentrations reported in the Stein et al. (2012) study. This study determined much higher levels of nitrogen leaching compared to other studies. For instance, Emelko et al. (2011) reported the dissolved organic nitrogen (DON) at 205.3 µg/L-N after fire in 3 fire affected creeks in 4 years post-fire, which is an order of magnitude smaller than this study's results for TN. It should be noted that the DON concentration does not account for all forms of nitrogen present in the sample. The Red Bench fire in Montana produced a peak of ammonia at 220 µg/L-N and nitrate at 427 µg/L-N (Hauer and Spencer, 1998), also an order of magnitude less than the current study's results, yet ammonia and nitrate were not looked at individually in the current study.

Contact time did not have a significant impact on TP concentrations ($p > 0.05$) for 13 of the 15 riverbed locations. However, riverbed location does significantly affect TP leaching concentrations from all sites ($p < 0.03$). The highest TP concentrations were from the upper location from Century Park, Stevens Gulch, and Mile Marker 115 (averages: 0.71 mg/L-P, 0.572 mg/L-P, 1.32 mg/L-P, respectively). The site the sediment comes from also is significant to TP concentrations ($p < 5 \times 10^{-6}$) with the highest concentrations for the bottom and middle locations from FCWTF Intake (averages: 0.75 mg/L-P, 0.87 mg/L-P, respectively). The site with the highest average TP concentration for the upper location was Mile Marker 115 (1.32 mg/L-P).

The maximum total phosphorus concentrations determined by this study ranged from 0.57 to 1.35 mg/L-P. This is in the same range as in a Lost Creek fire study (maximum average 42.2 $\mu\text{g/L-P}$) (Emelko et al., 2011). This study's range is lower than the Cerro Grande fire concentrations (14.5 mg/L-P) because this value was observed within the first 2 months after fire so a large proportion of nutrient-containing ash and sediment would still be suspended in the river, giving a high concentration (Gallaher and Koch, 2004). Similar phosphorus concentrations (peak of 15 mg/L-P) were seen in the Stein et al. (2012) study mentioned above. The Red Bench fire gave a soluble reactive phosphorus (SRP) peak of 69.2 $\mu\text{g/L-P}$ (Hauer and Spencer, 1998). This concentration is lower than the range in the current study, but SRP does not account for the other forms of phosphorus that may be present in the water.

Results for metals leaching from Poudre River sediment are given in Table 2.3. For the majority of riverbed locations (9 of 15), there was not a significant impact of contact time times on iron concentrations in the leachate. However, the riverbed location had a significant impact on iron concentrations ($p < 0.0008$). The highest concentrations were observed for the upper location (averages: Century Park=1.40 mg/L, Stevens Gulch=2.08 mg/L, Youngs Gulch=1.25 mg/L). The middle and upper riverbed locations significantly impacted the iron concentrations ($p < 1.0 \times 10^{-5}$). The highest iron concentrations for middle location was observed at the FCWTF Intake (average=1.32 mg/L) while the upper location concentration was highest at the Stevens Gulch (average=2.08 mg/L). The maximum averages of iron leached ranged from 0.974 to 2.08 mg/L-Fe. All sites showed an increase in iron concentrations in sediment-spiked samples than the controls. This confirms that the fire increases solubility of iron (Smith et al., 2011).

Table 2.3 - Metal Results from Sediment Leaching

	Century Park	Stevens Gulch	Youngs Gulch	Mile Marker 115	FCWTF Intake
Fe	Leaching contact time does not have an effect on iron concentrations. Except M. Riverbed location is significant for iron concentrations ($p=1.97 \times 10^{-6}$). The highest average concentrations are from the upper location (1.40 mg/L-Fe).	Leaching contact time does not have an effect on iron concentrations. Except U. Riverbed location is significant for iron concentrations ($p=5.44 \times 10^{-6}$). The highest average concentrations are from the upper location (2.08 mg/L-Fe).	Leaching contact time does not have an effect on iron concentrations. Except B. Riverbed location is significant for iron concentrations ($p=3.37 \times 10^{-6}$). The highest average concentrations are from the upper location (1.25 mg/L-Fe).	Leaching contact time does not have an effect on iron concentrations. Except B. Riverbed location is significant for iron concentrations ($p=0.0008$). The highest average concentrations are from the middle location (0.974 mg/L-Fe).	Leaching contact time does have an effect on iron concentrations ($p<0.05$). Except B. Riverbed location is significant for iron concentrations ($p=2.5 \times 10^{-5}$). The highest average concentrations are from the middle location (1.315 mg/L-Fe).
Cu	Leaching contact time does not have an effect on copper concentrations. Except U. Riverbed location is significant for copper concentrations ($p=0.0334$). The highest average concentration is from the bottom location (0.194 mg/L-Cu)	Leaching contact time does not have an effect on copper concentrations. Except B. Riverbed location is not significant for copper concentrations. However, the average concentration based on the bottom and middle locations (to exclude the outlier in the upper region) is 0.361 mg/L-Cu.	Leaching contact time does not have an effect on copper concentrations. Except U. Riverbed location is not significant for copper concentrations. However, the highest concentration of copper is from the middle location (0.315 mg/L)	Leaching contact time does not have an effect on copper concentrations. Riverbed location is not significant for copper concentrations. However, the highest overall average of copper was 0.196 mg/L-Cu at the middle location.	Leaching contact time does not have an effect on copper concentrations. Except M. Riverbed location is not significant for copper concentrations. However, the middle average of all time periods is 0.121 mg/L-Cu
Mn	Leaching contact time is not significant for manganese concentrations. Riverbed location is significant for manganese concentrations ($p=1.3 \times 10^{-4}$). The highest average concentration was from the middle location (0.42 mg/L-Mn)	Leaching contact time is not significant for manganese concentrations. Riverbed location is not significant for manganese concentrations. However, the highest average concentration was 0.32 mg/L-Mn from the bottom location.	Leaching contact time is not significant for manganese concentrations. Riverbed location is not significant for manganese concentrations. However, the highest average concentration was 0.45 mg/L-Mn at the upper location.	Leaching contact time is not significant for manganese concentrations. Riverbed location is significant for manganese concentrations ($p=0.0093$). The highest average concentration was from the middle location (0.30 mg/L-Mn)	Leaching contact time is not significant for manganese concentrations. Riverbed location is not significant for manganese concentrations. However, the highest concentration was from the bottom location (0.35 mg/L-Mn).

Results for copper leaching are provided in Table 2.3. For the majority of riverbed locations (11 of 15), contact-leaching time did not have a significant impact on copper concentrations in the leachate ($p > 0.05$). The riverbed location also did not significantly change the concentrations except at Century Park where the highest concentrations of copper were located in the bottom region (0.19 mg/L, $p = 0.033$). However, the sites have a significant impact on the bottom and middle locations. The highest copper concentrations were observed at Stevens Gulch (averages: Bottom=0.34 mg/L, Middle=0.38 mg/L). The maximum averages of copper concentrations ranged from 0.121 to 0.361 mg/L-Cu. These concentrations are similar to the values of copper after the Cerro Grande fire (4-290 $\mu\text{g/L-Cu}$) (Gallaher and Koch, 2004). The current study's concentrations are slightly higher, probably due to the batch conditions in the beakers. The concentrations would be diluted in the actual Poudre River. The copper results in Table 2.3 are much higher than the concentrations in a study by Sakalauskiene and Ignatavicius, (2003) in Lithuania (maximum post fires: 26.1 $\mu\text{g/L-Cu}$) most likely because of the closer proximity of the High Park fire to the Poudre River than the fires in Lithuania were to their respective rivers. The sediment-leaching test showed higher concentrations of copper than in the controls, so copper is expected to leach into the river, especially with agitation of the riverbed sediment.

Results for manganese in the leachate were also given in Table 2.3. There was not a significant impact of contact times on manganese concentrations in the leachate ($p > 0.05$). Riverbed locations also did not impact manganese concentrations. Sites were only significant for the upper riverbed location ($p = 0.006$) and the highest concentration was observed at Youngs Gulch (0.32 mg/L). The lack of significance for the ANOVA tests may be due to the limited number of data points for manganese. The Hach method 8034 has a detection limit of 0.1 mg/L-

Mn, and many samples had non-detectable manganese levels. The maximum averages of manganese ranged from 0.32 to 0.45 mg/L-Mn. These values were larger than the typical control concentration of 0.2 mg/L-Mn, which indicates that manganese will leach into water from fire affected soils.

The concentrations of organics, nutrients and metals observed in this study were compared to the Environmental Protection Agency (EPA) regulations for drinking water in Table 2.4 (EPA, 2009).

Table 2.4 - Comparison of Sediment Leaching Concentrations to EPA Regulations

Parameter	EPA Regulation	Minimum Concentration	Maximum Concentration
TOC (mg/L)	N/A	8.4	58.2
TN (mg/L-N)	NO ₃ -N: 10 NO ₂ -N: 1	0.6	9.2
TP (mg/L-P)	Limiting algal growth level: 0.005-0.01	0.19	1.5
Iron (mg/L-Fe)	0.3	0.22	2.64
Copper (mg/L-Cu)	Action level: 1.3 Secondary: 1	0	0.56
Manganese (mg/L-Mn)	0.05	0	0.9

There is not a specific EPA regulation for TOC, but the concentrations are higher than the typical TOC concentrations in the Poudre River of 2 to 4 mg/L. The average concentrations discussed above also are larger than the concentration (11 mg/L) found just after the fire on June 30th by the FCWTF. Based on data by Milhous (2009) for the Poudre River during spring flow, the volume to surface area ratio in the river is 0.37 m³/m². The laboratory settings in the beakers give a volume to surface area ratio of 0.11 m³/m², so even with approximately 1:3 dilution, the TOC concentrations in the river would be much higher than concentrations found historically.

The high concentrations may still be problematic for treatment practices, especially the possibility of disinfection-by-product (DBP) formation. TN is not specifically regulated, but nitrate and nitrite have Maximum Contaminant Levels (MCLs) as seen in Table 2.4 due to their harmful health effects including reducing the capability of red blood cells to hold oxygen in young children. The concentrations found in this study do not exceed the regulations, even though the specific forms of nitrogen were not delineated. The EPA does not regulate TP, but literature has shown that the concentration of TP that above which algae flourish ranges from 0.005 to 0.01 mg/L-P (Koester, 2011). The TP concentrations found in this study greatly exceed this level, which may induce algae growth in the river in the future. Even with dilution in the river, these concentrations would be problematic due to the resulting algal metabolites producing taste and odor issues. Several of the iron concentrations in this study exceed the EPA secondary MCL, which may impact the color or corrosion of the water in the distribution system. However, this study focused on the source water, not treated water, so much of the iron would be removed through conventional treatment. Copper concentrations did not exceed either the MCL of 1.3 mg/L-Cu or the secondary, aesthetic MCL of 1 mg/L, so copper would not be problematic for drinking water utilities after the High Park Fire. Manganese concentrations did exceed the secondary MCL of 0.05 mg/L, however the concentrations would not only be more dilute in the river but also would be reduced with conventional treatment.

An approximate dilution factor can be calculated based on data from Milhous (2009) concerning spring conditions in the Poudre River. The flow rate of 2.85 m³/s, average depth of 0.75 m, and average width of 13 m, would give a velocity in the river of approximately 1 m/s. The length of river in the burn area that is of interest is about 24,000 m, which would give the water in the river 6 to 7 hours to travel through the burn area. The contact surface area in the

river ($\sim 314,000 \text{ m}^2$) and the volume of water ($\sim 117,000 \text{ m}^3$) would give a volume to surface area ratio of $0.37 \text{ m}^3/\text{m}^2$. The volume to surface area ratio in the beakers is $0.11 \text{ m}^3/\text{m}^2$. The dilution of the concentrations from the beakers to the river would be approximately 1:3. Even with a 1:3 reduction in concentration, some of the parameters (TOC, TP, iron, and manganese) would not be acceptable for EPA standards. However, with conventional treatment, these parameter could decrease and be acceptable for EPA regulations.

Typical water quality parameters for the sediment leaching tests are found in Table 2.5. Maximum average pH values occurred at the upper riverbed location and ranged from 7.09 to 7.23. Several of the locations (3 of 5) had pH values that decreased from 6 to 12 hours, but then increased after 24 hours. This change in pH may be because of the changing water chemistry over 24 hours. For instance, hydroxide ions could precipitate with iron in the water, thus increasing the pH. Dissolved oxygen (DO) maximum averages ranged from 5.325 to 5.98 mg/L. There was no distinct trend of how location or duration of leaching affected DO values. Temperature was relatively uniform across all samples at 24-25 °C except for Stevens Gulch samples which were kept in the refrigerator just before analysis. Maximum averages of conductivity ranged from 129.2 to 197.98 $\mu\text{S}/\text{cm}$. The upper and middle riverbed locations had the highest conductivity averages, and the values increased with time. This is perhaps due to compounds dissolving into the water over time. The highest averages of turbidity ranged from 80.3 to 209.02 NTU and which typically occurred at the upper riverbed location. There is not a distinct temporal trend for turbidity. The type of particles greatly influences settling rates, so this may be the cause of the variability amongst samples.

Table 2.5 - Water Quality Parameter Results from Sediment Leaching

	Century Park	Stevens Gulch	Youngs Gulch	Mile Marker 115	FCWTF Intake
pH	The upper location had the highest pH (average 7.09). The pH decreased slightly between 6 and 12 hours, but then was at a maximum at 24 hours.	The upper location had the highest pH (average 7.11). The pH decreased from 6 to 12, but increased again at 24 hours.	The largest pH occurred at the upper location (average 7.11). pH increased with time.	The upper location had the highest pH values (average 7.23). The pH decreased from 6 to 12, but increased again at 24 hours.	The highest pH values were at the upper location (average 7.14). pH increased with time.
Dissolved Oxygen	DO was highest at the bottom location (average 5.795 mg/L). The highest DO at each location was at 12 hour (average 12 hour 6.013 mg/L).	The DO concentration decreased at similar rates over time at each location. Overall average: 5.98 mg/L. The middle is slightly higher at an average of 6.002 mg/L	The DO concentration decreased at similar rates over time at each location. Overall average: 5.42 mg/L. The bottom is slightly higher at an average of 5.865 mg/L.	The DO concentration decreased at similar rates over time at each location. Overall average: 5.63 mg/L. The bottom is slightly higher at an average of 5.81 mg/L.	The middle location has the highest DO concentrations (average 5.325 mg/L). The 6-hour time period had the highest DO and the concentration decreased at 12 hour, then increased at 24hr.
Temperature	The temperature was uniform over time and location. The overall average was 25.06 °C.	The highest temperatures were from the 6 hour samples (24.95 °C), and the other samples were ~5°C colder*.	Temperature occurred similarly for each location. The 12-hour samples were higher than the others (24.08 °C average).	Temperature was uniform over time and location. The overall average was 24.39 °C.	Temperature occurred similarly for each location. The 6-hour samples were slightly higher (average 25.47 °C) than the others.
Conductivity	The middle location had the highest conductivity (average 142.03 µS/cm). Conductivity increased with time.	The upper location had the highest conductivity (average 129.2 µS/cm). Conductivity increased with time.	The middle location had the highest conductivity (average 146.17 µS/cm). Conductivity increased with time**.	The upper location had the highest conductivity (average 197.98 µS/cm). Conductivity increased with time.	The upper location had the highest conductivity (average 144.7 µS/cm). Conductivity increased with time.
Turbidity	The upper location had the highest turbidity (average 174.67 NTU). The overall trend is that turbidity decreases over time.	The upper location had the highest turbidity (average 209.02 NTU)***. The 12-hour samples had the highest values for each location.	The upper location had the highest turbidity (average 83.43 NTU). There is not a distinct trend for changes over time.	The upper location had the highest turbidity (average 95.68 NTU). 2 of the 3 locations decrease turbidity with time.	The middle location had the highest turbidity (average 80.3 NTU). 2 of the 3 locations decrease turbidity with time.

* Temperature of 12 and 24-hour samples were delayed, so samples were stored in the refrigerator, which accounts for the difference.

** The bottom, 6-hour sample had higher conductivity than the 12 and 24-hour samples, but since the standard deviation is large, it is neglected.

*** However, the standard deviation is very large for the 6 and 12 hour, upper samples

2.4 CONCLUSIONS

Although the High Park fire was one of Colorado's largest fires, the water quality of the Cache la Poudre was not significantly impacted 5 to 10 months after the fire. There was initially a smoky flavor observed from FPA tests with raw river water collected just after the fire. This is because of the high concentrations of ash in the river, causing the river to run black. However, for the FPA tests of water samples obtained several weeks after the fire, a smoky flavor was not perceived. The 5 mg/L PAC dose was ineffective at removing other earthy/musty odors from the water, so a longer contact time should be considered. Additionally, the FPA panel results indicated that neither the conventional treatment nor the PAC treatment removed the smoky smell from the guaiacol-spiked samples perhaps due to the mismatch between the compound and the PAC type.

The potential for nutrients and metals to enter the water column in the next spring runoff is of great concern. Even with a dilution factor of approximately 1:3 in the river, the concentrations of organics and metals would not satisfy EPA regulations for drinking water. Literature has shown that increases in these aqueous parameters relates to storm events, so it is possible that contaminants from the hillside could be deposited in the river with additional precipitation. The sediment leaching test results show that there is a potential for nutrients and metals to be leached into the water, especially with re-suspension of riverbed sediments. Water utilities that use the Poudre River as a source should monitor water quality parameters and adjust the treatment and add PAC at concentrations of at least 5 mg/L.

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APPENDIX

Table A.1 – Flavor Profile Analysis Results

Sample Contents	Panel #1		Panel #2	
	Descriptor	Intensity	Descriptor	Intensity
Raw Poudre (7/17), after runoff, Panel #1	Smoky	4	Grassy/Smoky	7
Raw Poudre (7/17), after runoff, Panel #2	Smoky	4	N/A	
10% P, 90% HT, No PAC	Musty	2	Threshold	
10% P, 90% HT, 5mg/L PAC	Earthy/musty	2	Threshold	
25% P, 75% HT, No PAC	Threshold		Grassy	6
25% P, 75% HT, 5 mg/L PAC	Threshold		OF	
50% P, 50% HT, No PAC	Earthy/musty	2	Geosmin	3
50% P, 50% HT, 5 mg/L PAC	Earthy/musty	2	Sweet, Geosmin	2, 3
75% P, 25% HT, No PAC	Grassy/Fishy/Earthy	2	Grassy	6
75% P, 25% HT, 5 mg/L PAC	Threshold		Threshold (smoky)	
100% P, 0% HT, No PAC	Musty	2	Phenolic, Earthy	2, 2
100% P, 0% HT, 5 mg/L PAC	Musty/Earthy	2	Threshold	
OF, Panel #1	OF		OF	
OF, Panel #2	OF		N/A	
OF, Panel #3	OF		N/A	
Tap Water	(from FCWTF) Chlorine	4	(from CSU) Threshold	
50 µg/L Guaiacol, 50%P, 50% HT, No PAC- 10/31/12	Smoky	4	Smoky	6
50 µg/L Guaiacol, 50%P, 50% HT, 5 mg/L PAC- 10/31/12	Smoky/grassy*	4	Smoky	6
50 µg/L Guaiacol, 50%P, 50% HT, No PAC- 11/7/12	Smoky/grassy	3	Smoky	6
50 µg/L Guaiacol, 50%P, 50% HT, 5 mg/L PAC- 11/7/12	Grassy/smoky/vegetation	3	Smoky	6
5 g/L sediment, Benchscale control 1	Earthy, smoky	2, 3	Geosmin	6
5 g/L sediment, Benchscale control 2	Musty, smoky	2, 3	Geosmin	4
5 g/L sediment, Benchscale coag/polymer 1	Earthy/musty/fishy	3	Geosmin	5
5 g/L sediment, Benchscale coag/polymer 2	Earthy/musty/fishy	2	Geosmin (dirty/soil)	4
5 g/L sediment, Benchscale coag/polymer, 5 mg/L PAC 1	Earth/musty	2	Threshold	
5 g/L sediment, Benchscale coag/polymer, 5 mg/L PAC 2	Earthy/musty	2	Threshold (geosmin)	

* Grassy most likely from improperly washed glassware

Table A.2 – Total Organic Carbon (mg/L) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	9.9		33.8	38.2	1.5	11.5	8.5	12.3	9.8	16.9	20.6	
	8.4	38.0	38.4	37.8	1.7	8.4	8.4	15.8	8.8	19.3	20.9	19.6
12 hr		34.2	41.7	39.3		10.6	22.0	22.0		15.5	20.5	19.6
		34.5	42.9	41.1		12.5	22.0	22.0		14.7	20.5	19.1
24 hr		33.9	46.2	45.6		22.0	21.4	22.0		16.8	20.5	19.6
		36.6	44.4	40.5		22.0	21.3	22.0		18.2	20.6	20.0

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	9.4	20.1	20.4	20.4	14.4	39.0	38.0	39.4
	7.7	19.9	20.3	20.3	9.2	39.8	39.2	39.2
12 hr		28.8	37.2	37.6		34.2	41.7	39.3
		29.8	36.6	36.8		34.5	42.9	41.1
24 hr		47.1	41.4	38.7		33.9	46.2	45.6
		36.9	42.6	58.2		36.6	44.4	40.5

Table A.3 – Total Nitrogen (mg/L-N) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6hr	0.5	2.5	6.0	4.8	1.2	1.7	1.8	3.6	0.2	1.2	5.4	1.1
	0.2	3.2	17.7*	5.7	0.6	2.1	1.5	16.1*	0	0.8	5.5	1.4
12hr		3.9	7.0	4.8		2.6	3.8	4.9		0.6	5.0	2.0
		3.8	6.7	5.9		3.0	3.0	5.7		1.9	4.7	2.1
24hr		4.1	7.2	5.5		2.7	2.3	2.4		1.3	4.8	2.3
		3.1	6.5	4.3		2.0	1.8	3.6		2.0	4.7	3.4

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6hr	0.3	1.3	3.5	5.6	0.5	5.6	9.4	4.8
	0.3	1.5	17.2*	4.7	0.6	4.5	5.2	17.7*
12hr		2.5	6.4	4.9		4.8	5.4	5.9
		2.3	5.4	5.0		5.3	5.2	5.6
24hr		2.1	3.5	4.9		4.3	4.1	4.6
		1.4	4.2	5.4		5.1	4.2	4.1

* Outlier, not used in calculations

Table A.4 – Total Phosphorus (mg/L-P) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6hr	0.08	0.34	0.42	0.74	0.04	0.34	0.3	0.54	0.07	0.84	0.81	0.24
	0.07	0.39	0.37	0.77	0.05	0.28	0.42	0.52	0.07	0.24	0.8	0.38
12hr		0.38	0.36	0.72		0.31	0.45	0.45		0.19	0.64	0.32
		0.36	0.4	0.68		0.34	0.41	0.98		0.19	0.66	0.35
24hr		0.26	0.44	0.72		0.35	0.35	0.51		0.27	0.78	0.35
		0.34	0.4	0.6		0.34	0.31	0.43		0.26	0.79	0.33

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6hr	0.07	0.43	0.46	1.3	0.06	0.72	0.96	0.84
	0.07	0.32	0.41	1.26	0.05	0.66	0.96	0.82
12hr		0.29	0.6	1.5		0.72	0.84	0.51
		0.3	0.56	1.3		0.87	0.93	0.69
24hr		0.3	0.42	1.26		0.72	0.72	0.75
		0.37	0.4	1.3		0.78	0.81	0.57

Table A.5 – Total Iron (mg/L-Fe) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
		Bottom	Middle	Upper		Bottom	Middle	Upper		Bottom	Middle	Upper
6hr	0.051	0.520	0.678	1.393	0.058	0.530	0.739	2.200	0.094	0.303	0.518	0.840
	0.248	0.670	0.805	1.908	0.07	0.406	0.976	2.065	0.115	0.358	0.539	1.465
12hr		0.580	0.675	1.374		0.777	1.580	2.637		0.375	0.473	0.917
		0.487	0.577	1.147		0.731	1.044	2.489		0.279	0.584	1.549
24hr		0.383	0.403	1.247		0.645	0.679	1.514		0.533	0.629	1.369
		0.470	0.217	1.345		0.332	0.646	1.578		0.637	0.534	1.347

	Mile Marker 115				FCWTF Intake			
		Bottom	Middle	Upper		Bottom	Middle	Upper
6hr	0.088	0.802	1.057	0.870	0.083	0.562	1.708	0.942
	0.051	0.846	0.894	0.644	0.176	0.646	1.588	0.953
12hr		0.563	0.987	0.621		0.524	1.182	0.855
		0.562	1.085	0.473		0.442	1.263	0.829
24hr		0.379	0.910	0.404		0.484	1.005	0.558
		0.371	0.909	0.423		0.455	1.144	0.421

Table A.6 – Copper (mg/L-Cu) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6hr	-0.008*	0.328	0.059	0.044	-0.030*	0.350	0.405	0.556	0.066	0.258	0.369	0.290
	0.062	0.345	0.085	-0.038*	0.157	0.305	0.355	0.465	0.009	0.175	-0.044*	0.301
12hr		0.167	0.088	-0.044*		0.365	0.450	0.172		0.084	0.346	0.039
		0.196	0.153	0.013		0.355	0.300	0.285		0.029	-0.044*	0.029
24hr		0.057	0.154	0.208		0.340	0.266	0.013		0.290	0.166	0.043
		0.069	0.114	0.004		0.341	0.501	0.038		0.161	0.291	0.102

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6hr	0.036	0.189	0.167	0.250	0.121	0.024	0.008	0.030
	0.011	0.209	0.219	-0.003*	-0.044*	0.112	0.045	0.096
12hr		0.111	0.166	-0.044*		0.065	0.146	0.151
		0.214	0.137	-0.044*		-0.044*	0.215	0.095
24hr		0.124	0.199	0.298		0.138	0.140	0.022
		0.144	0.288	0.086		0.031	0.177	0.114

* Below detection limit, zero used for calculations

Table A.7 – Manganese (mg/L-Mn) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6hr	1	0.5	0.4	-0.3*	0.2	0.7	0.8	0	0.2	0.1	0.2	0.8
	0.2	0.2	0.3	-0.3*	0.5	0.3	0.9	0	0.3	0.3	0.4	-0.1*
12hr		0.3	0.5	-0.3*		-0.3*	-0.1*	-0.3*		0.1	0.5	0.3
		0.5	0.5	0.2		-0.2*	0.2	-0.3*		0.3	0.1	0.1
24hr		0.1	0.4	-0.3*		0.2	0	0.1		0	0.2	0.5
		0.2	0.4	0		0.7	0.4	-0.1*		0	0.3	0.2

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6hr	0.2	0.2	0.2	-0.3*	0.1	-0.1*	0.1	-0.3*
	0.2	-0.3*	0.5	-0.3*	0.3	0.5	-0.3*	0
12hr		0.3	0.6	-0.3*		0.1	-0.3*	0.3
		0.2	0.3	-0.3*		0.2	-0.3*	-0.2*
24hr		-0.1*	-0.1*	-0.3*		0.4	0.1	0.1
		0.1	0.2	-0.3*		0.4	0.2	0.2

* Below detection limit, zero used for calculations

Table A.8 – pH Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	6.77	6.82	6.84	7.02	6.58	6.97	7.07	7.18	6.91	6.79	6.74	6.96
	6.88	6.86	6.89	7.12	6.92	6.98	7.14	7.24	6.84	6.94	6.92	7
12 hr		6.86	6.74	6.93		6.75	6.73	6.95		6.96	6.94	7.03
		6.81	6.84	7.02		6.7	6.9	6.93		7.02	6.99	7.1
24 hr		7.07	7.06	7.19		6.97	7.16	7.18		7.24	7.2	7.37
		7.09	7.12	7.24		7.05	7.17	7.19		7.35	7.26	7.38

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	6.74	6.88	7	7.23	6.72	6.76	6.94	7.02
	6.76	6.93	7.13	7.32	6.78	6.88	6.98	7.07
12 hr		6.81	6.97	7.03		6.74	6.97	7.12
		6.87	7.02	7.16		6.89	7.04	7.11
24 hr		7.28	7.26	7.28		7.15	7.21	7.26
		7.33	7.29	7.35		7.17	7.28	7.25

Table A.9 – Dissolved Oxygen (mg/L) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	6.09	5.57	5.59	5.49	7.3	7.08	7.01	6.85	7.41	7.13	6.74	6.71
	5.92	5.64	5.48	5.39	7.25	7.05	6.66	6.56	7.12	7.08	6.94	6.7
12 hr		6.26	6.02	5.84		5.56	5.82	5.96		5.53	4.14	5.45
		6.2	5.94	5.82		5.57	5.86	5.86		5.9	4.46	5.36
24 hr		5.57	5.52	5.52		5.33	5.32	5.25		4.75	3.42	4.59
		5.53	5.46	5.48		5.24	5.34	5.31		4.8	3.3	4.64

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	6.2	6.03	6.37	6.07	6.45	6.07	5.88	5.53
	6.38	6.33	6.21	6.1	6.49	6.01	5.84	5.23
12 hr		5.86	5.66	5.57		3.79	4.77	3.7
		5.9	5.48	5.79		3.55	4.89	4.26
24 hr		5.4	4.49	5.06		3.45	5.33	4.27
		5.34	4.51	5.21		3.58	5.24	3.95

Table A.10 – Temperature (°C) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	25.6	24.9	24.8	24.8	24.7	24.8	24.8	24.9	24.1	23.9	23.4	23.6
	25.2	24.9	24.8	25	24.4	24.8	25.7	24.7	24.4	23.8	23.5	24.1
12 hr		25.1	25.2	25.2		19.9	19.4	20.3		25.3	25	24.8
		25.1	25.2	25.1		19.5	20.1	19.1		25.5	24.7	25.2
24 hr		25.4	25.1	25		19.2	19	18.7		24.7	24.9	24.5
		25.3	25.1	25.1		18.8	18.4	19.3		25.1	24.6	24.4

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	24.8	24.4	24.4	24.4	26.2	25.5	25.4	25.3
	25.2	24.4	24.4	24.5	25.7	25.6	25.5	25.5
12 hr		23.6	24	24.2		24.2	24.1	24.2
		24.1	24	24		24.4	24	24.4
24 hr		24.1	25.1	25.1		24.8	24.4	24.4
		24.3	25	25.1		24.6	24.2	24.2

Table A. 11 – Conductivity ($\mu\text{S}/\text{cm}$) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	67.7	91	131.4	115.9	79.1	112.5	109.5	121.1	81	180.6	132.5	112.8
	67.5	91	134.6	115.3	78.7	110.6	112.3	122.2	82.4	87.7	134	117.6
12 hr		92.7	138.8	124.2		121.8	120.7	126.6		87.9	137.1	117.1
		93.6	140.9	121.6		121.1	118.6	132		87.6	138.7	122.4
24 hr		100.3	151.7	141.1		126.5	128	137.5		93.6	166.3	138.1
		100.5	154.8	138.1		132.8	123.3	135.8		93.5	168.4	143

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	78.7	96.3	129.8	179.4	79.6	134.4	128.7	139
	78.5	95.4	130.3	170.8	77.6	132	130.2	141.7
12 hr		97.3	154	195.8		140.9	130.9	142.8
		96.6	136.6	191.9		145.8	133.8	140.4
24 hr		102.6	150.2	228		142.9	137.6	152.5
		102.8	150.7	222		148	141.9	151.8

Table A. 12 – Turbidity (NTU) Results from Sediment Leaching

	Century Park				Stevens Gulch				Youngs Gulch			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	2.12	50.7	53.3	296	3.2	40.8	45.3	123	1.93	32.2	59.5	46.9
	2.79	63.1	51.1	260	7.24	36.4	67.6	298	1.36	27.4	54.3	83.9
12 hr		26.1	41	130		49.6	52.5	244		23.8	47.7	78.4
		36.3	36.6	127		48.3	68.1	363		26	42.5	105
24 hr		35.9	47.4	126		16.4	31.7	97.1		35.5	43	88.9
		31	29.8	109		25.3	29.8	129		34.7	65.8	97.5

	Mile Marker 115				FCWTF Intake			
	Control	Bottom	Middle	Upper	Control	Bottom	Middle	Upper
6 hr	1.63	53.4	53.4	140	3.07	86.1	89.4	89.5
	2.38	43.8	40.9	97.8	5.73	87.7	78.4	88
12 hr		43.3	72.5	99.8		62.9	92.6	75.1
		43.6	50.7	96.6		75.5	94.5	74.2
24 hr		36	33.6	55.3		43.7	68.9	44.2
		28.4	28.9	84.6		56.2	58	35.8

Figure A.1: Total Organic Carbon Results from Sediment Leaching

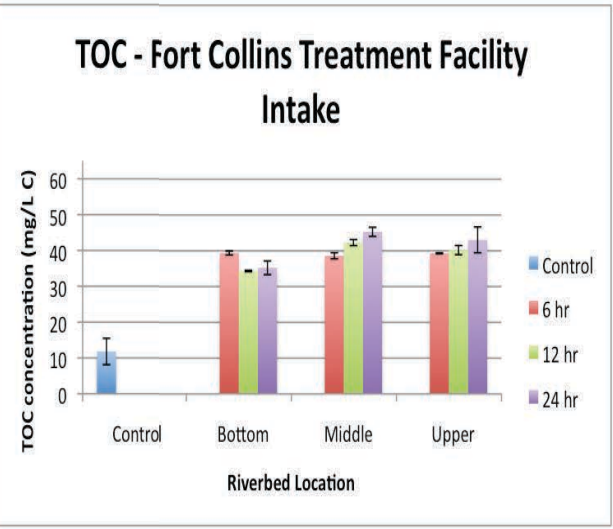
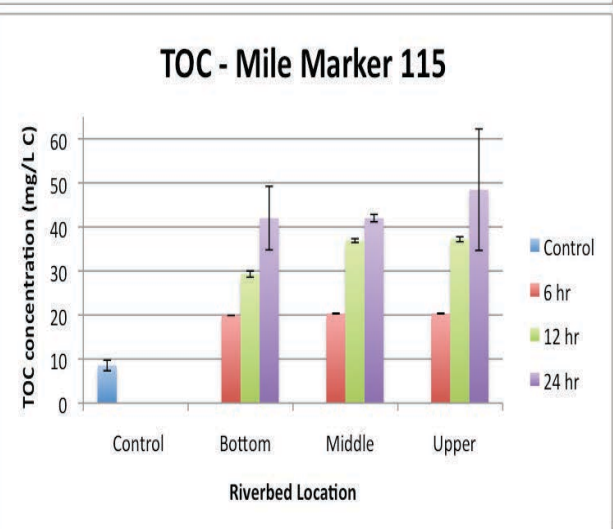
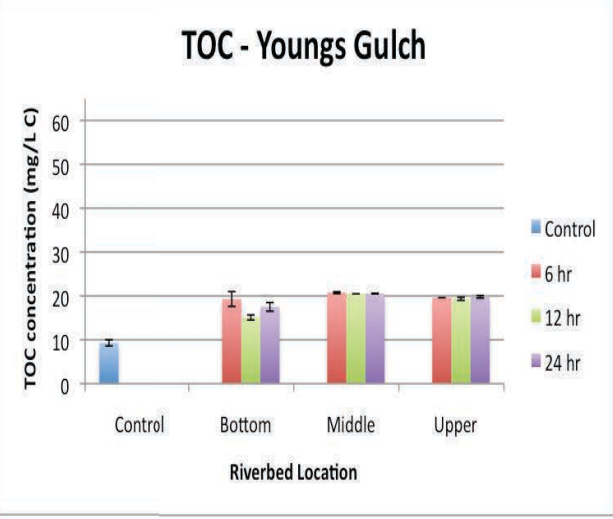
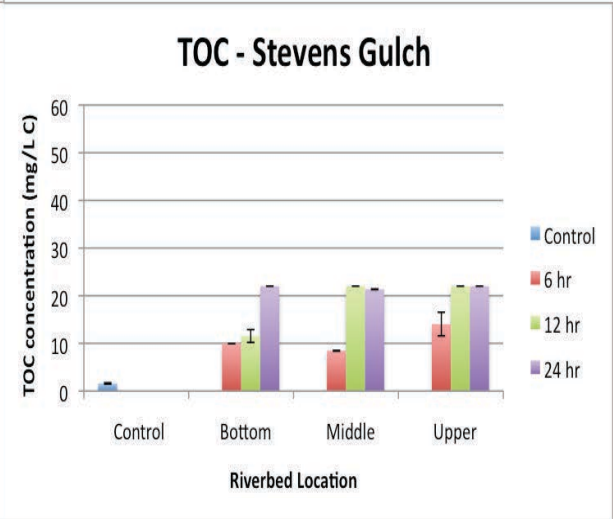
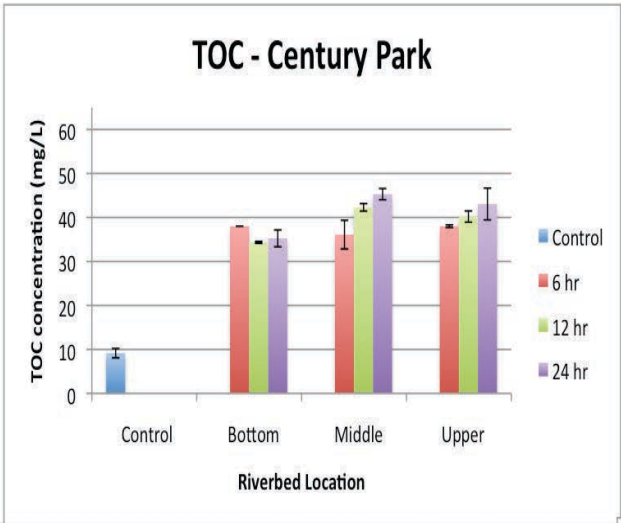


Figure A.2: Total Nitrogen Results from Sediment Leaching

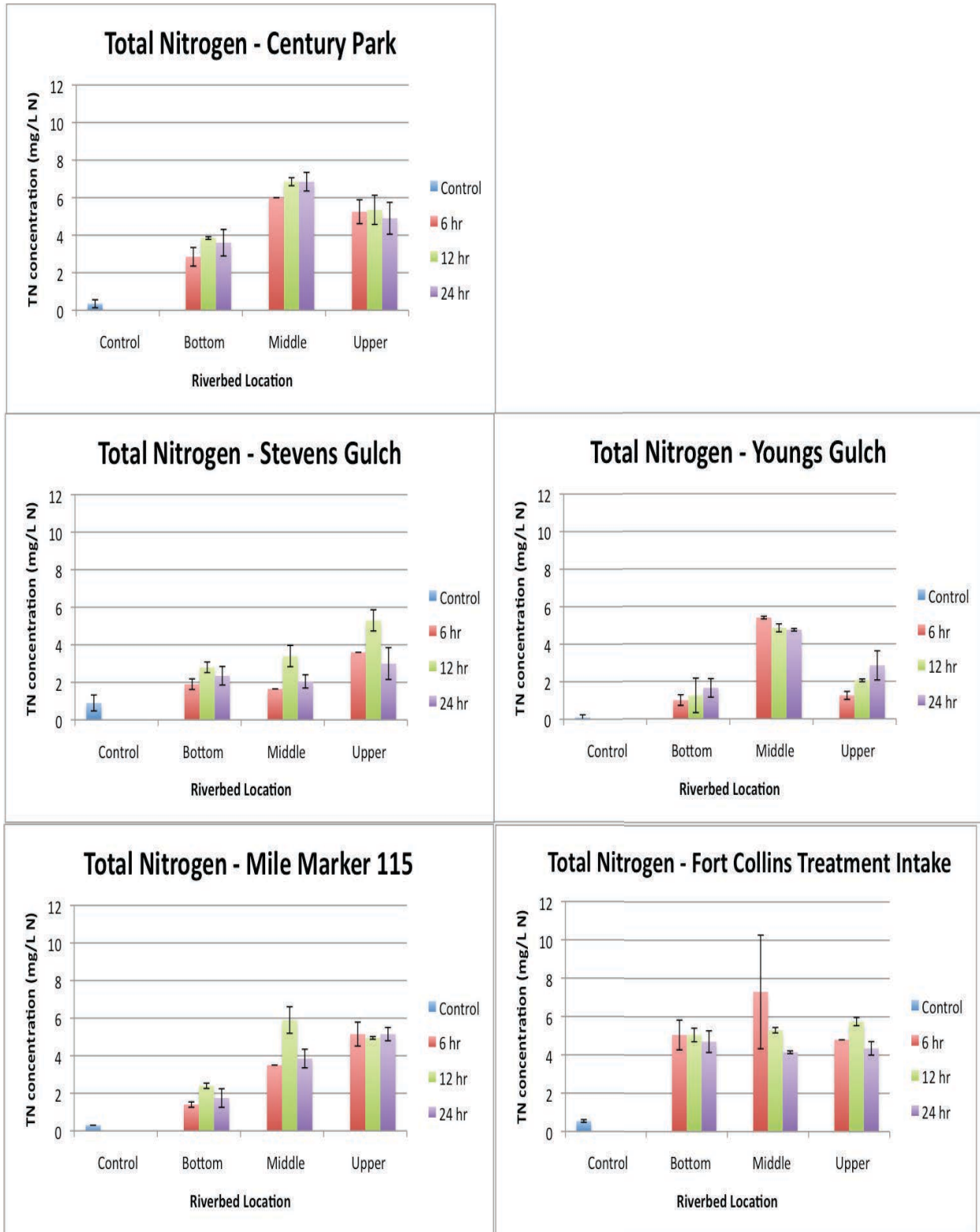


Figure A.3: Total Phosphorus Results from Sediment Leaching

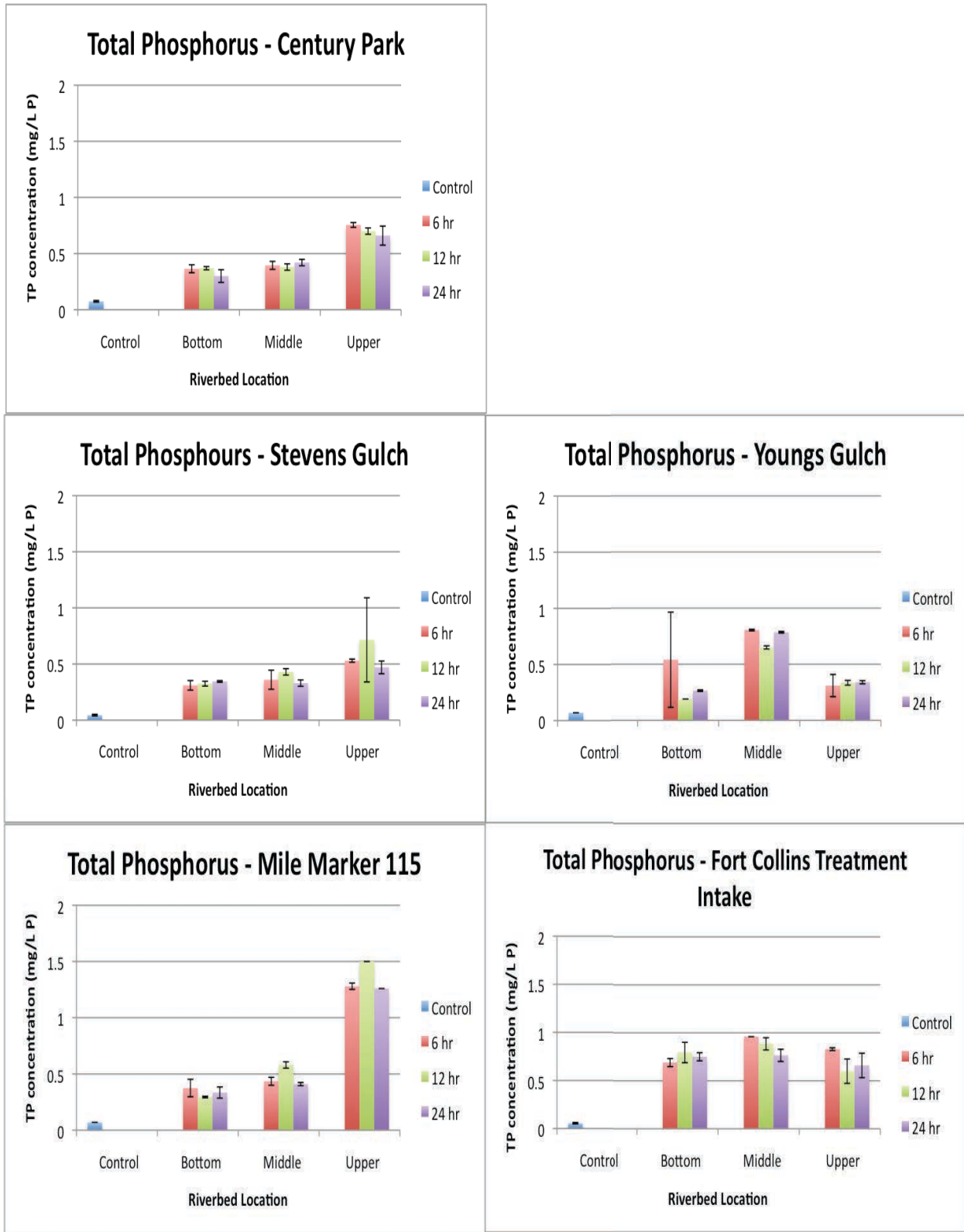


Figure A.4: Iron Results from Sediment Leaching

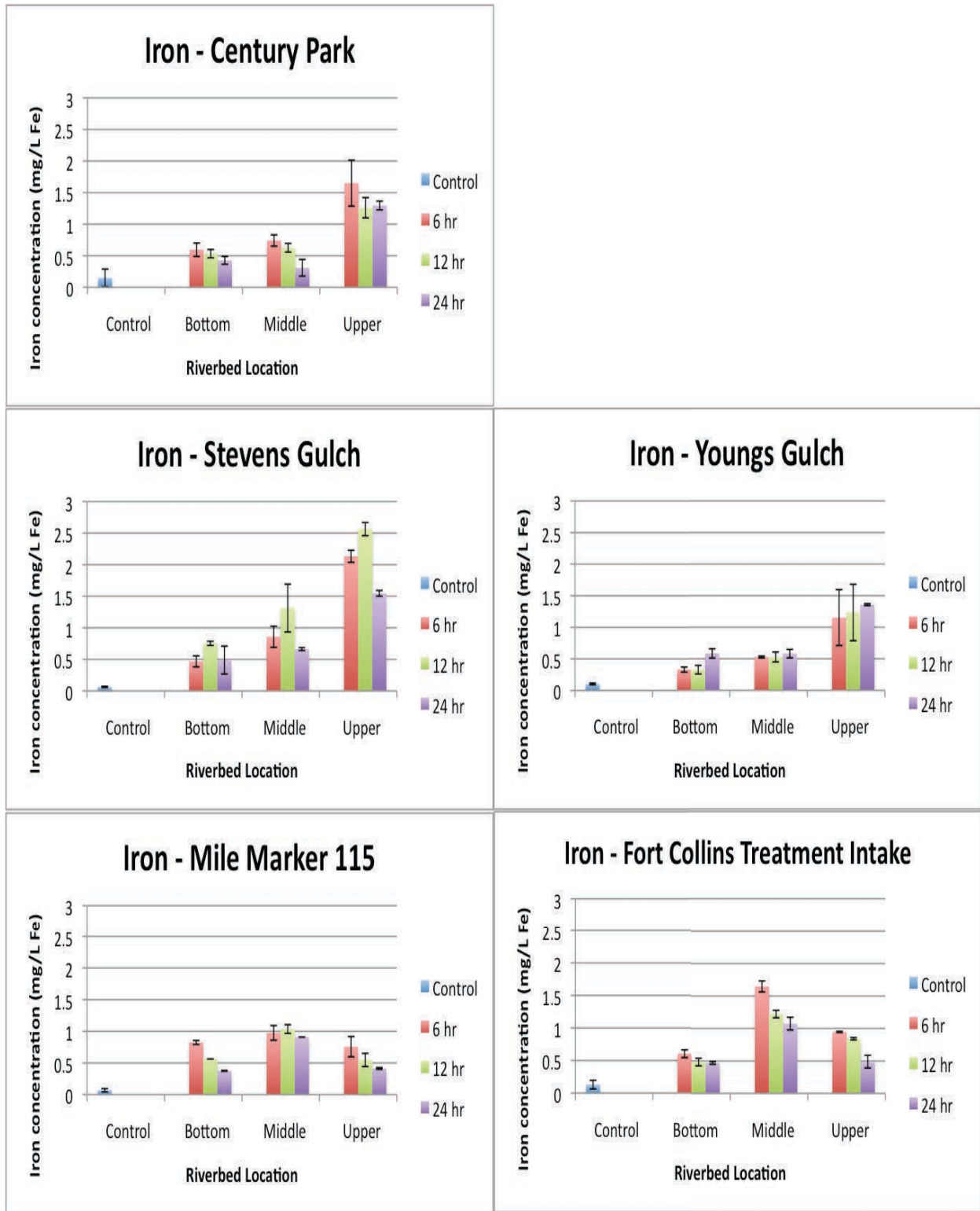


Figure A.5: Copper Results from Sediment Leaching

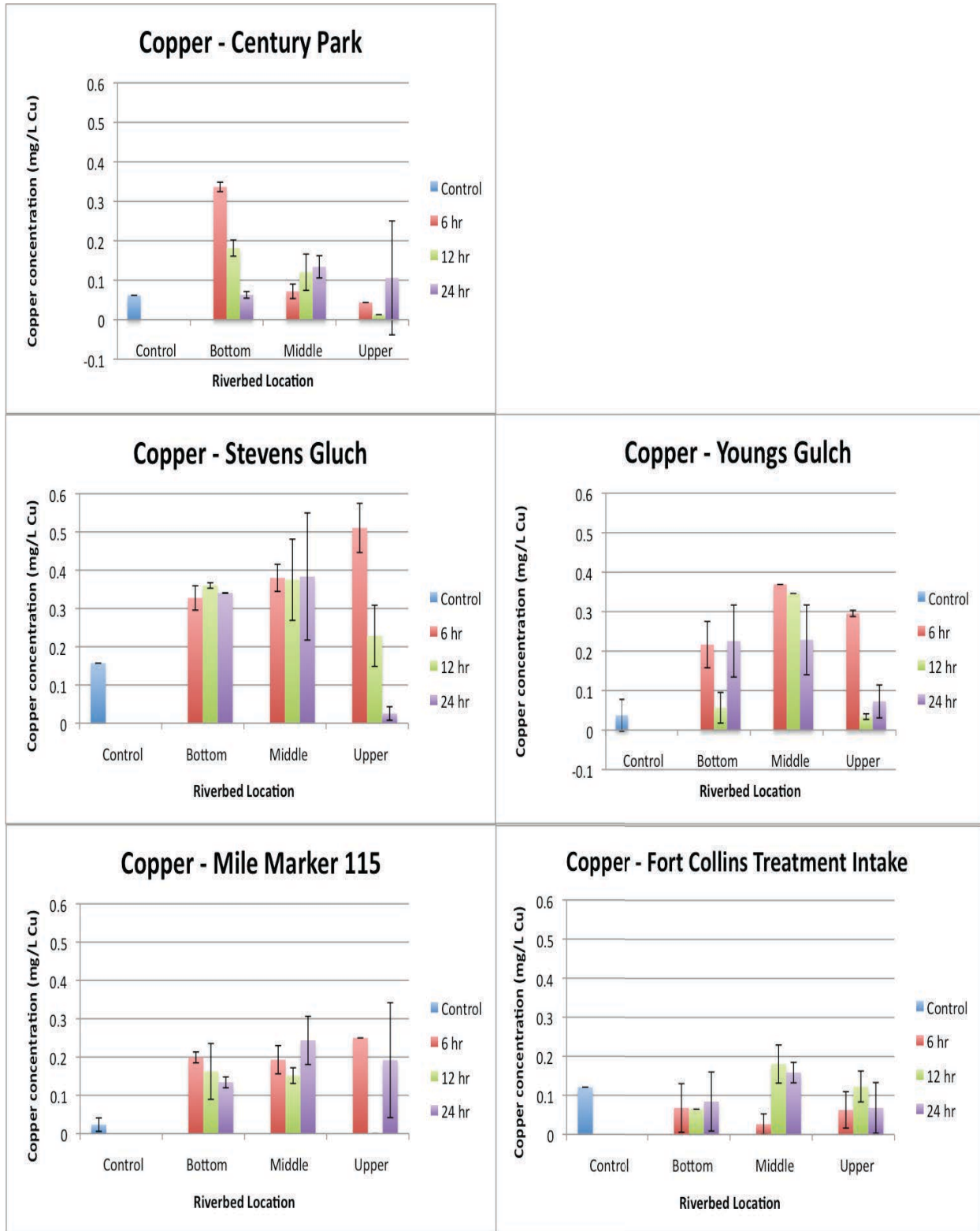


Figure A.6: Manganese Results from Sediment Leaching

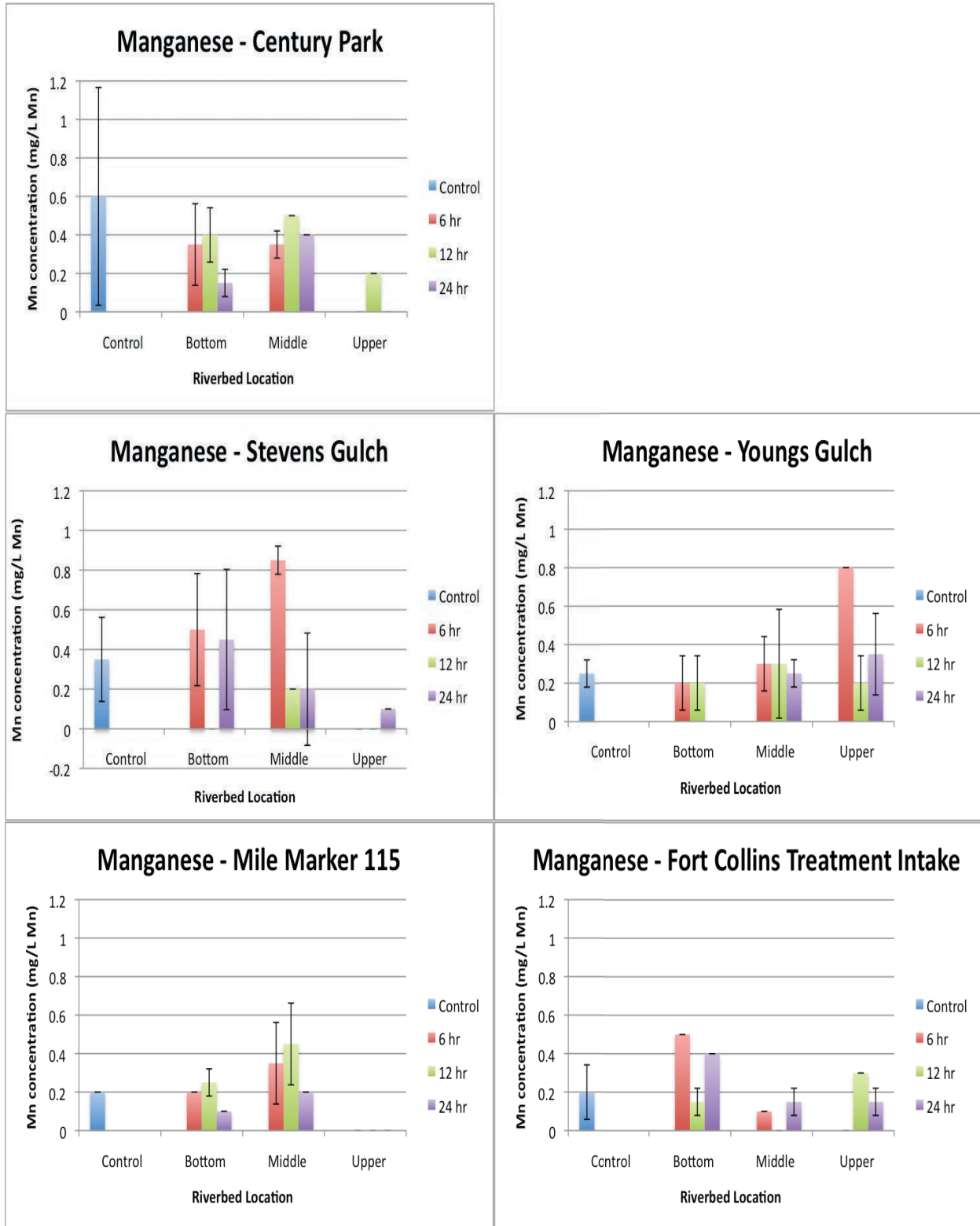


Figure A.7: pH Results from Sediment Leaching

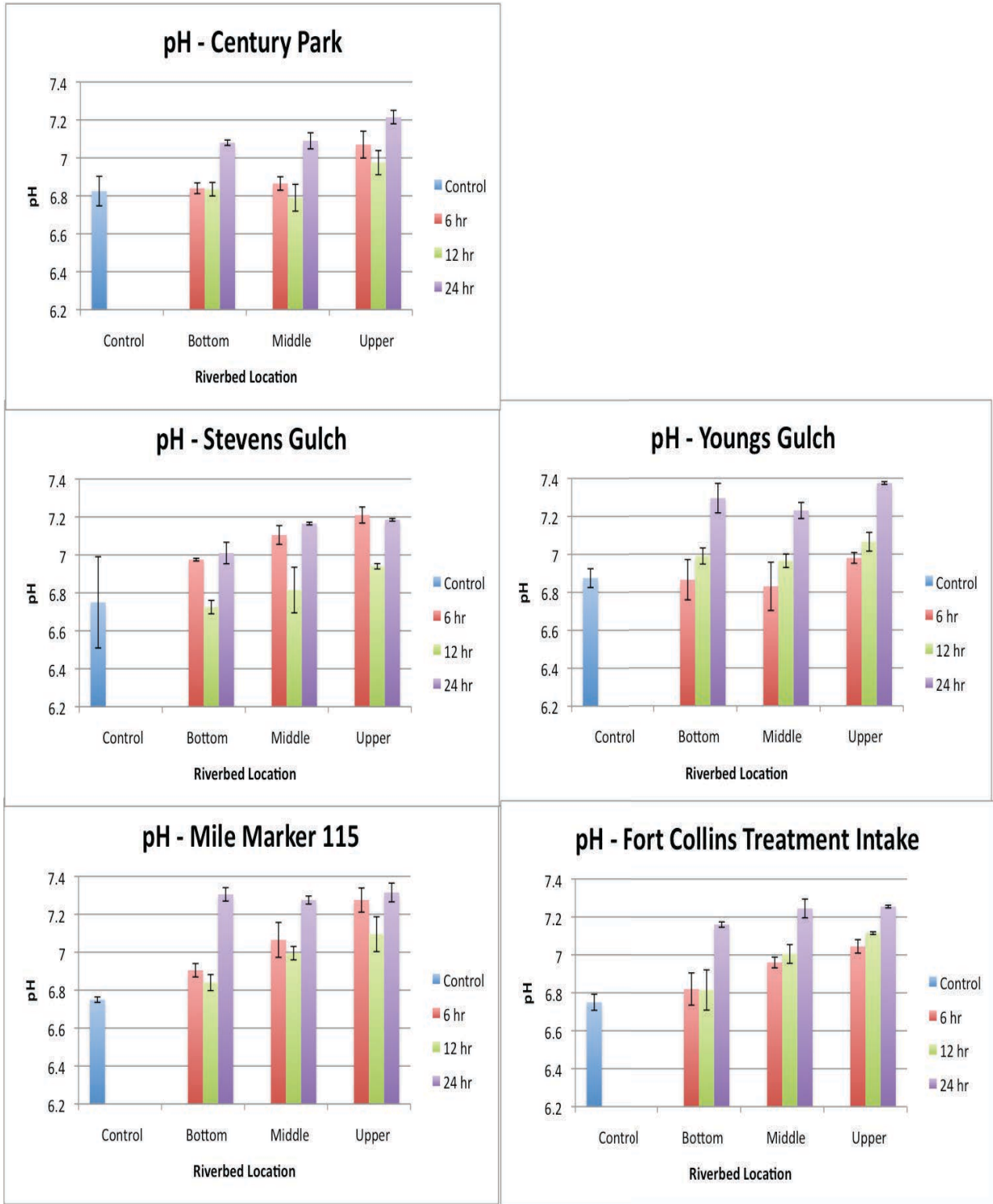


Figure A.8: Total Organic Carbon Results from Sediment Leaching

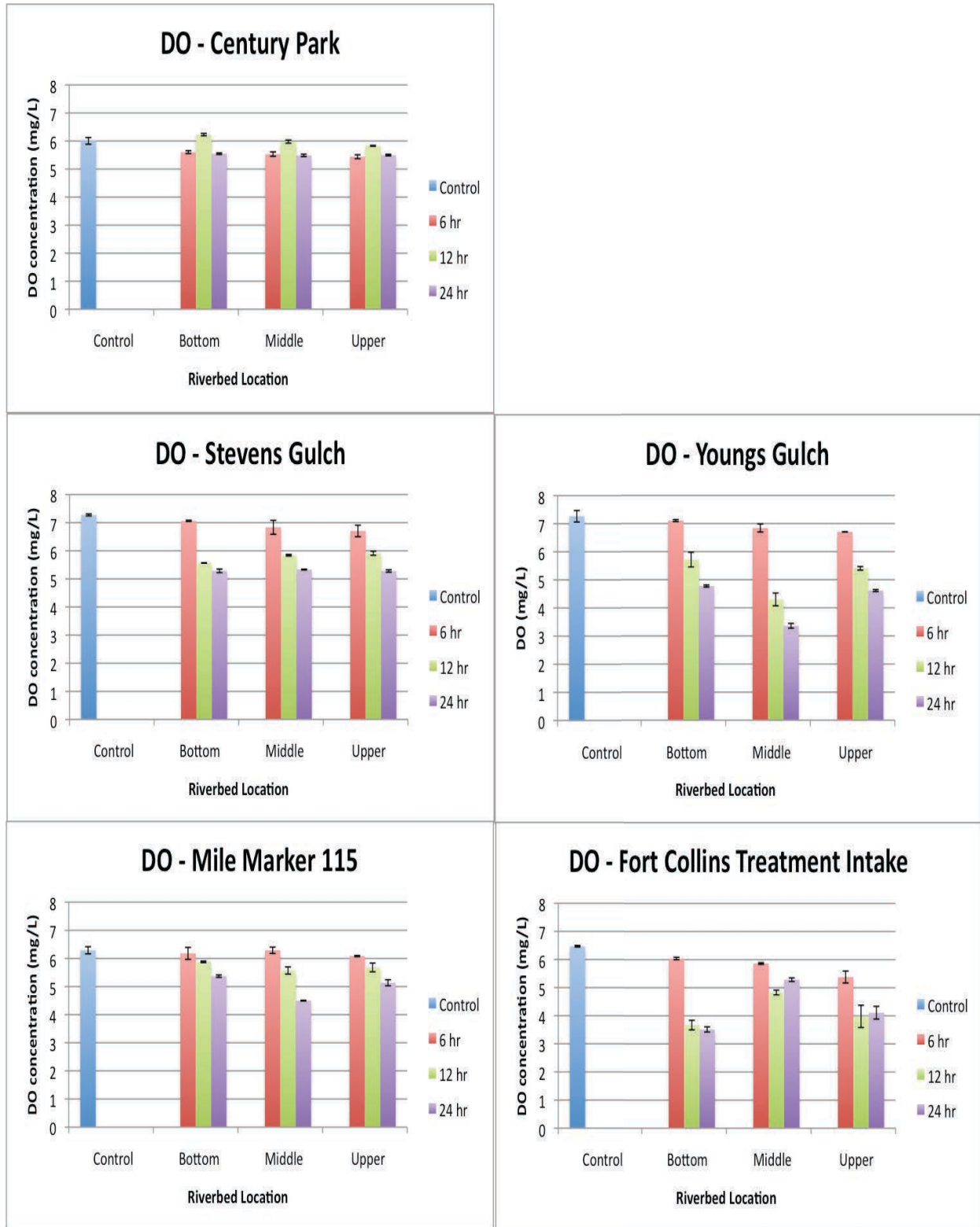


Figure A.9: Temperature Results from Sediment Leaching

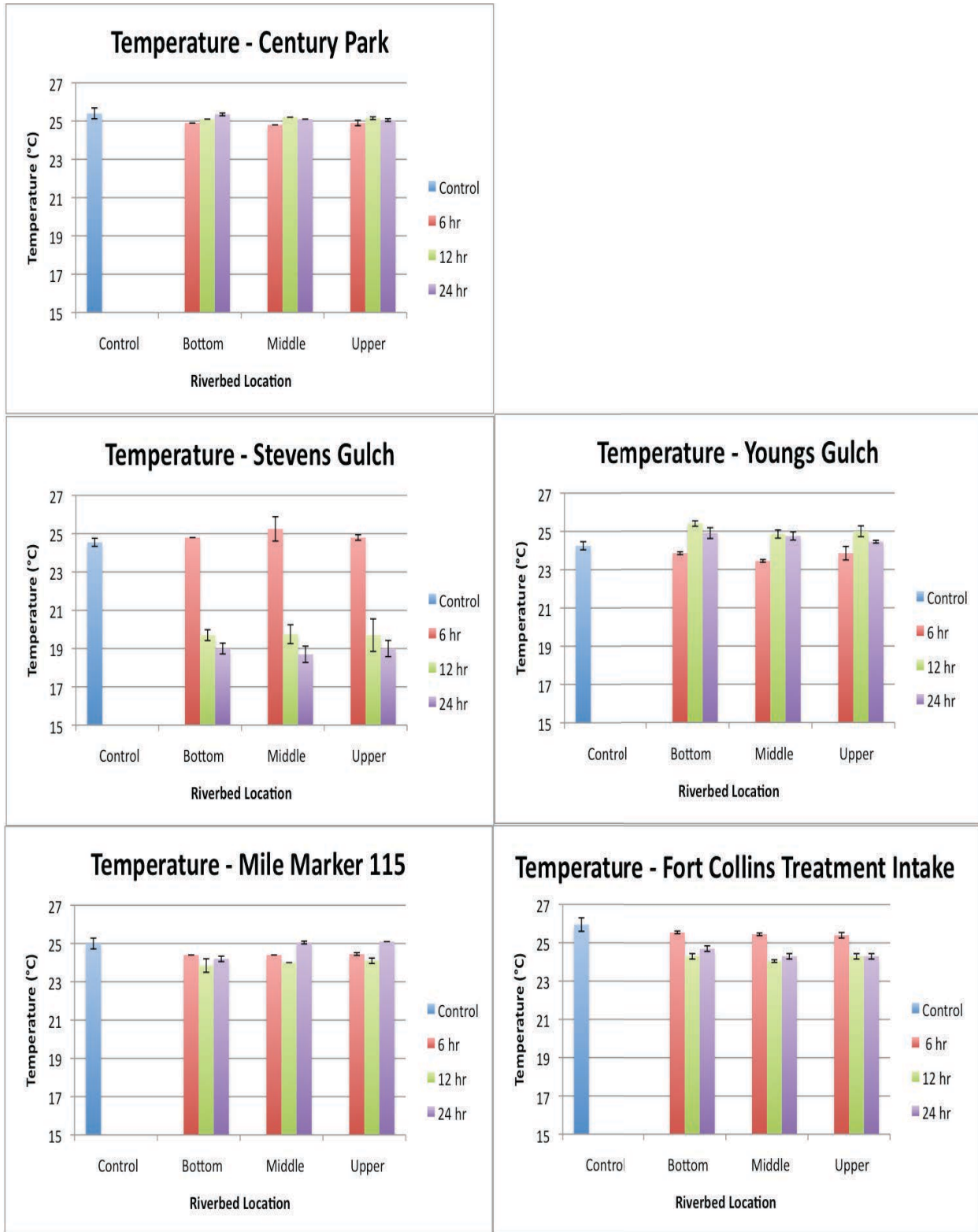


Figure A.10: Conductivity Results from Sediment Leaching

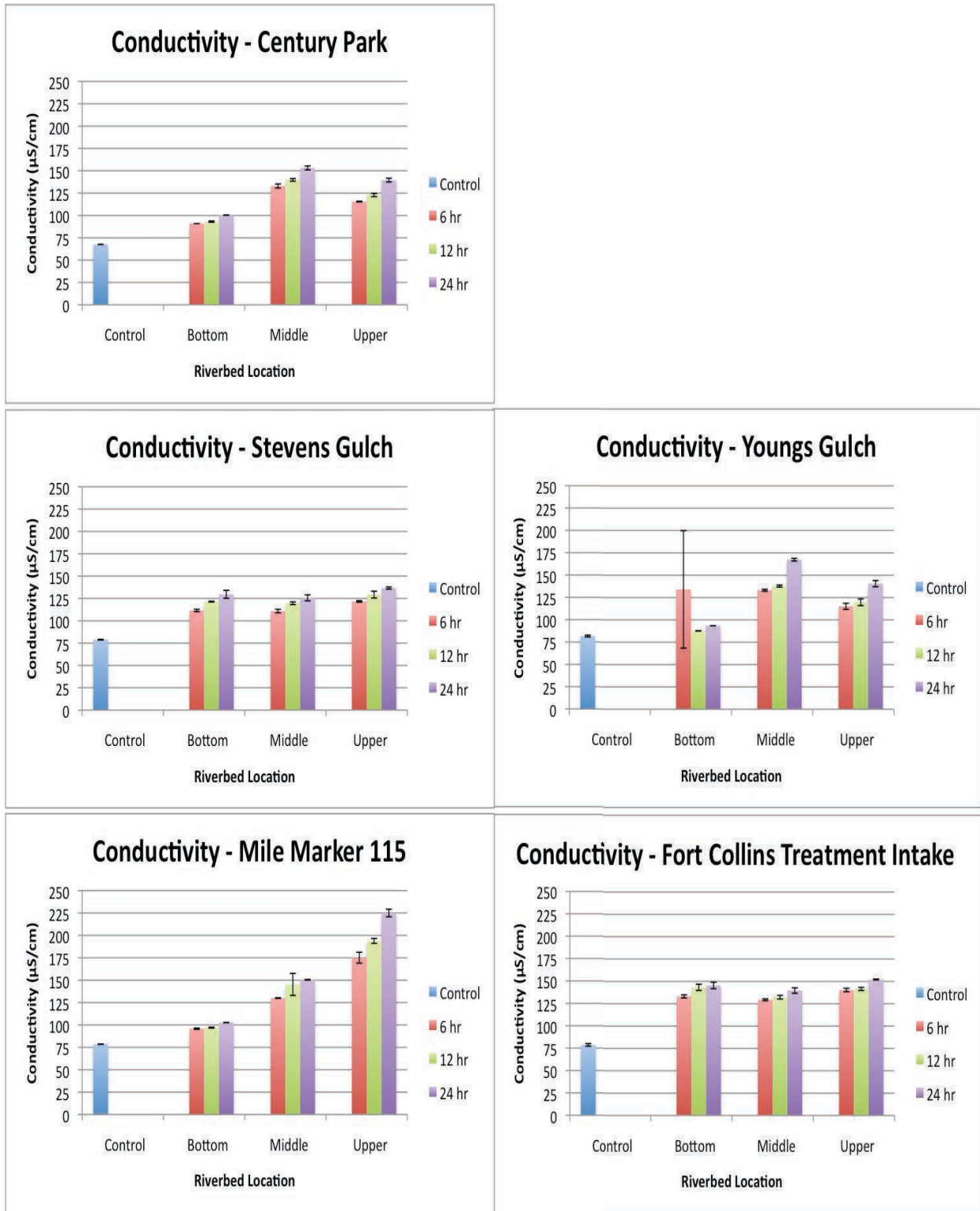


Figure A.11: Turbidity Results from Sediment Leaching

