THE IMPACTS OF FOREST FIRES ON DRINKING WATER QUALITY

by

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A Thesis Presented in Partial Fulfillment
of the Requirement for the Degree
Master of Science

ARIZONA STATE UNIVERISTY

December 2004

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has been approved

November 2004

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ABSTRACT

The objective of this research was to determine the long-term effects of catastrophic forest fires on drinking water quality. The watersheds burned by the Rodeo-Chediski fire in Arizona and the Hayman fire in Colorado were the focus of this research. Both fires occurred in watersheds that serve as a drinking water source for a major metropolitan area.

Concentration data and mass loading analyses were conducted utilizing pre- and post-fire water quality data. The results of the Rodeo-Chediski fire and Hayman fire analysis were compared. In addition, an unburned watershed in Arizona was analyzed and served as the reference watershed.

The short- and long-term impacts on stream water that serves as a drinking water source were determined. Short-term effects consisted of elevated nutrient and particulate concentrations in the burned watersheds during subsequent storm events. The post-fire storm events also elevated the metal concentrations in runoff water. However, the maximum contaminant level set by the United States Environmental Protection Agency (EPA) for the metals analyzed was only exceeded by selenium.

Long-term effects will originate from the mobilized particulate fraction settling to the stream or lake bottom. The organic material and metals adsorbed to the particulates will be slowly dissolved, diffusing into the water. The dissolved organics will serve as an energy source for microorganisms that have the ability to cause taste and odor problems in drinking water. While dissolved metals such

as arsenic and selenium are regulated by the EPA due to negative human health effects.

ACKNOWLEDGEMENTS

I would like to thank Dr. Paul Westerhoff for supervising the research, as well as Dr. Jordan Peccia and Dr. Morteza Abbaszadegan for being on my committee. The Cities of Phoenix, Tempe, and Peoria, the Salt River Project, Central Arizona Project, and the United States Geological Survey provided funding and/or assistance in conducting this research.

I want to express my gratitude to my lab mates for all the effort they put forth to assist me in completing my research: Mario Esparza, Wontae Lee, Mohammad Badruzzman, Youngil Kim, Baiyang Chen, Aaron Dotson, Andy Baumgardner, Jorge Navarro, Steve Acquafredda, Joby Carlson, Renaud Bordier, and Bo Song. Thank you to Erin Lyons, Erika Wilder, Brad Brown, Jeff Brahs, Frank Wayne LaFaveIII, and Kevin Robertson for going to collect water samples with me. Also, I appreciate the support I received from Dr. Dillner, Charity Coury, Jennifer Triplett, and my family. Thank you!

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CHAPTER 1 : INTRODUCTION

During the summer of 2002 two catastrophic forest fires burned in Arizona and Colorado. The Rodeo-Chediski fire burned the Salt River watershed, while the Hayman fire burned the South Platte River watershed. Runoff from portions of these watersheds is a source of drinking water in the Phoenix, Arizona and Denver, Colorado metropolitan areas. The purpose of this chapter is to provide background information for each fire and the existing drought conditions for the burned and unburned watersheds, as well as to present the objectives of this research.

RODEO-CHEDISKI FIRE

The summer of 2002 marked the occurrence of the largest and most severe forest fire in recorded Arizona history, the Rodeo-Chediski fire. The Rodeo-Chediski fire began as two individual fires that merged to form one wildfire in eastern Arizona along the Mogollon Rim as shown in Figure 1.1. The Rodeo fire began on June 18 on the Fort Apache Reservation northeast of Cibeque, Arizona. By June 20, the fire had burned nearly 121km^2 (Bureau of Land Management, 2002). Meanwhile on June 20, the Chediski fire began northwest of Cibeque near Chediski Peak. The Rodeo and Chediski fires began approximately 15 miles apart. The Chediski fire had consumed over 40km^2 within one day of being reported (Wilderness Society, 2004). The long-term regional drought, crosswinds, and fire suppression caused the fires to burn out of control. The fires merged to form the Rodeo-Chediski fire on June 23. At this point, the fire had consumed more than 951km^2, an area equivalent to the combined areas of Tempe, Scottsdale, Mesa, and Chandler. The fire continued to burn for the next two weeks, until the blaze was contained on July 7. The fire devastated privately owned land as well as land within the Fort Apache Reservation, Apache-Sitgreaves National Forest, and Tonto National Forest. The fire burned approximately 1871km^2 of land, 1,235km^2 were within the Salt River watershed. This represents 11% of the Salt River watershed, as shown in Figure 1.1. Water from the burned portion of the watershed flowed into the drinking water reservoirs for the Phoenix metropolitan area, as shown in Figure 1.2.

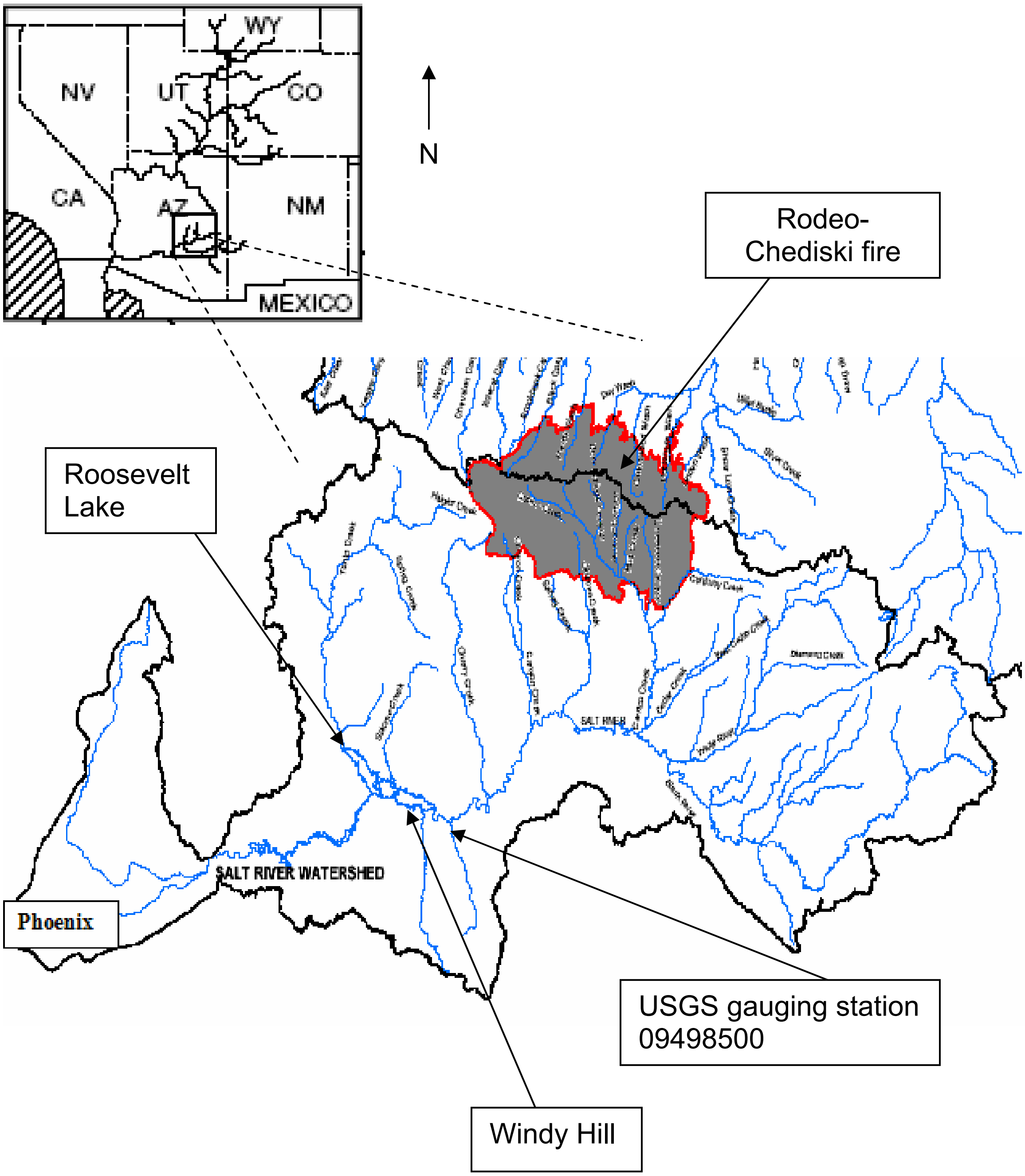

Figure 1.1 Salt River watershed in Arizona and USGS gauging station 09498500
Sources: State map, Earl and Blinn (2003); Salt River watershed map, Conway, Arizona Department of Water Resources (2004)

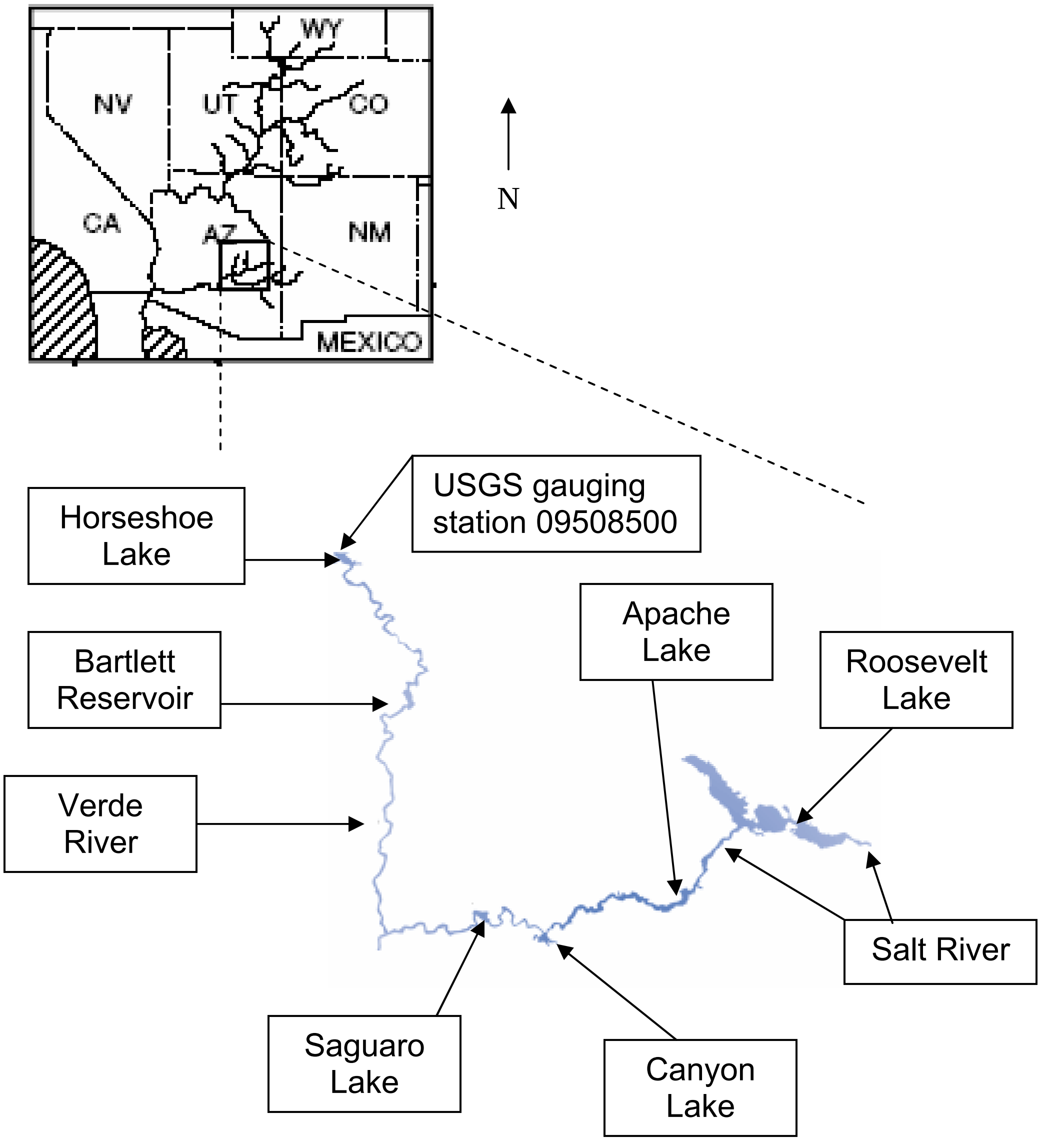

Figure 1.2 Map of the Verde River and Salt River Reservoir Systems and USGS gauging station 09508500
Source: State map, Earl and Blinn (2003)

HAYMAN FIRE

During the summer of 2002 Colorado also experienced the largest wildfire in the history of the state, the Hayman fire. The fire began southwest of Denver near Cheesman Lake on June 8, as shown in Figure 1.3. Within one day, the fire had burned 243km^2 on the Colorado Front Range of the Rocky Mountains between Denver and Colorado Springs. Due to similar weather conditions and fire suppression that fueled the Rodeo-Chediski fire, the Hayman fire burned out of control for several weeks. On July 2 the fire was contained and declared controlled on July 18. The fire burned nearly 560km^2 of land. Most of the burned land was within the boundaries of Pike National Forest, which includes Cheesman Lake, a drinking water source for Denver.

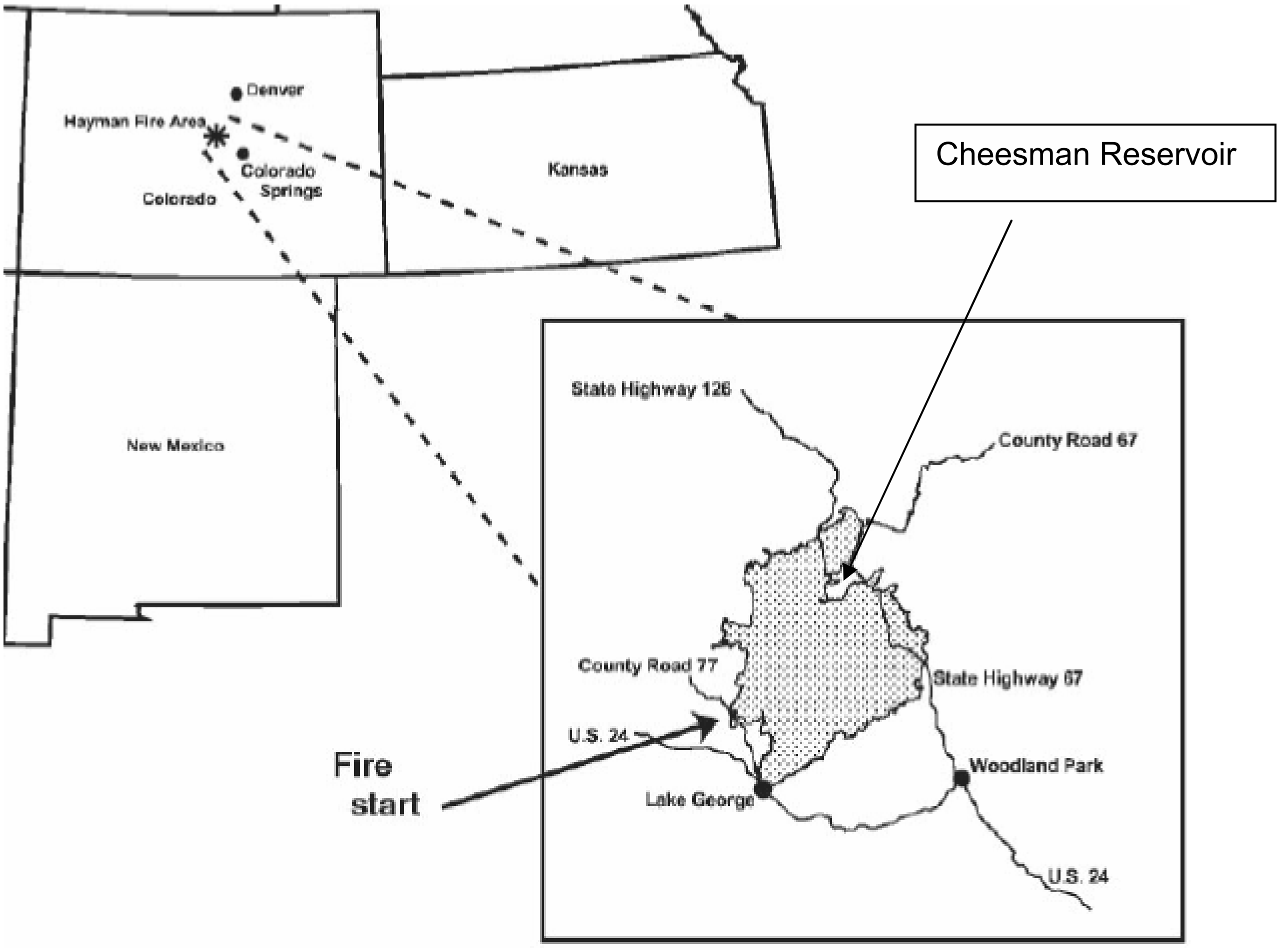

Figure 1.3 Hayman fire location map
Source: Unknown

DROUGHT

As a result of inadequate rainfall, the Salt River and South Platte River watersheds are currently in a drought. The Palmer Drought Severity Index (PDSI) is a method used to measure the severity of a drought. It utilizes rainfall and temperature to predict long-term dryness within a region.

A major cause of rainfall fluctuations is El Nino and La Nina. El Nino and La Nina influence climate on a global scale. El Nino refers to the warming of ocean currents in the eastern and central portions of the Pacific Ocean, while La Nina refers to the cooling of these ocean currents. The interactions between the water and air cause a shift in atmospheric pressure gradients. Therefore, changing the amount of moisture in the air and shifting the trade winds that carry the air into the Southwestern United States. Typically, El Nino causes flooding and La Nina causes drought. However, the effect of El Nino and La Nina at a specific location changes with each cycle.

According to the PDSI, the Salt River and South Platte River watersheds are part of a regional drought within their respective state. The PDSI is shown in Figure 1.4 and 1.5. The scale use zero to represent normal conditions. Wet years are represented by positive numbers and dry years are represented by negative numbers. Over the past eight years, drought conditions have dominated the Salt River watershed. The South Platte River watershed has experienced drought conditions since 2000.

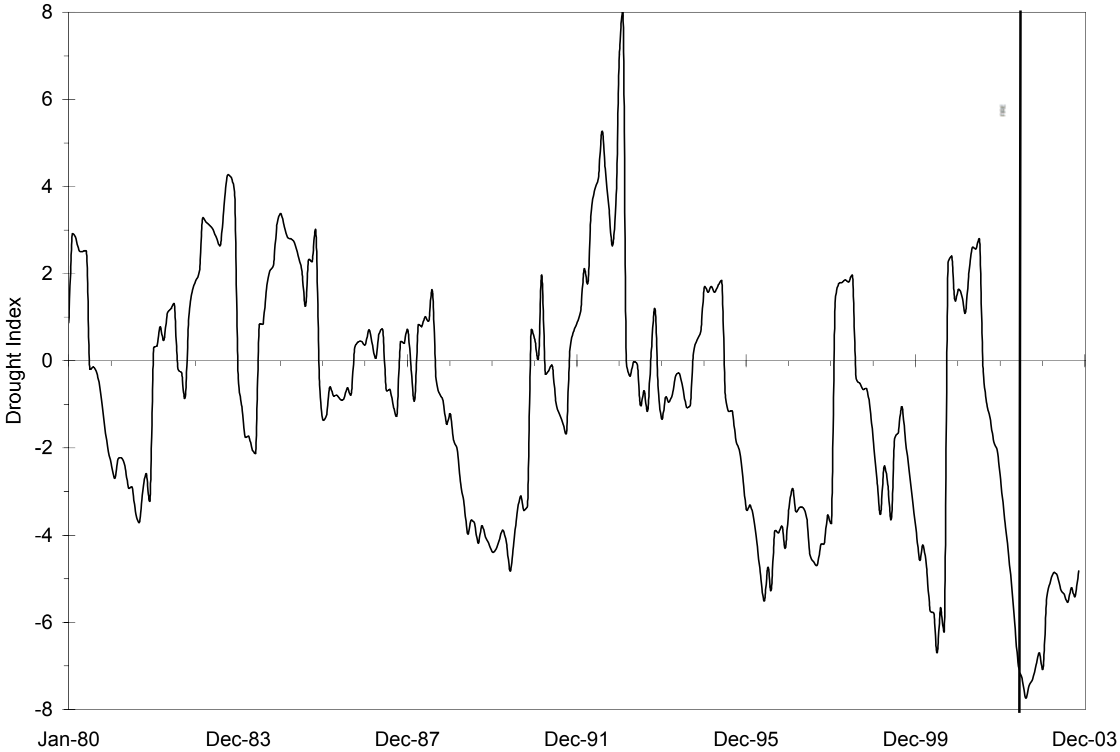

Figure 1.4 Palmer Drought Severity Index for the Salt River watershed
Source: National Oceanic and Atmospheric Administration (2004)

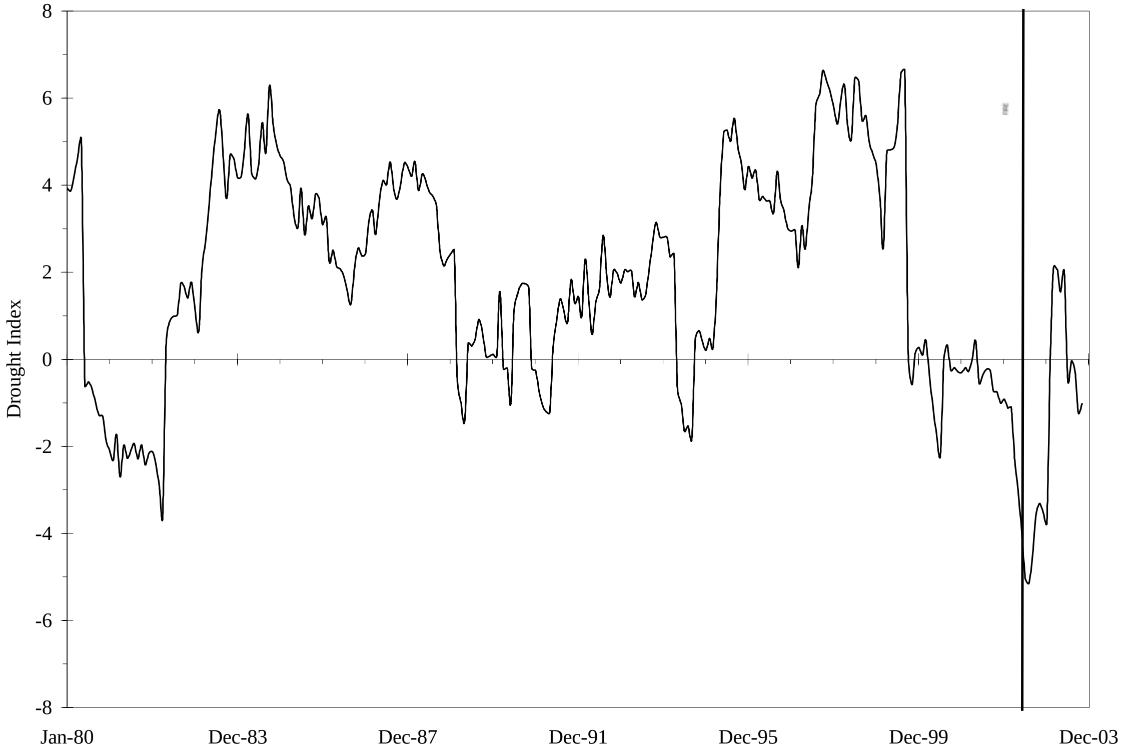

Figure 1.5 Palmer Drought Severity Index for the South Platte River watershed
Source: National Oceanic and Atmospheric Administration (2004)

OBJECTIVE

The long-term effects of forest fires on drinking water quality are becoming increasingly important. Rodeo-Chediski and Hayman fires are sources for this concern. Forest fires stimulate the release of nutrients, metals, and ions from vegetation, litter, and soil. Wind and runoff then mobilize these constituents, which are eventually transported into a stream or lake. The fate of the nutrients, metals, and ions as they travel through the water system will ultimately determine the effects of forest fires on drinking water quality.

An increase in nutrients in the Salt River and Cheesman Lake has the potential to degrade water quality. Nutrients such as organic carbon, phosphorus, and nitrogen stimulate the growth of algae and bacteria. The elevated levels of bacteria and algae may adversely impact drinking water due to their abilities to produce metabolites. Geosmin and methylisoborneol are two metabolites linked to taste and odor problems in drinking water.

The release and fate of metals is a concern since trace metals can negatively impact human health. Trace metals are regulated by the Environmental Protection Agency (EPA) due to adverse health effects. For example, arsenic is known to affect the circulatory system and is a potential carcinogen; lead is linked with delayed physical and mental development in children and high blood pressure in adults; and mercury and copper exposure can lead to kidney damage. Since human health is of utmost importance, the fate of metals released due to forest fire needs to be determined.

Water quality data collected within the watersheds burned by the Rodeo-Chediski fire in Arizona and the Hayman fire in Colorado are the focus of this research. A pre- and post-fire concentration comparison of water quality data will be completed for both watersheds. A mass loading analysis will be conducted for the Salt River, since the Rodeo-Chediski fire burned a portion of the Salt River watershed. In addition, the nutrient concentration data and mass loadings for the Salt River will be compared with the Verde River; the Verde River represents the unburned watershed in Arizona. The results will be useful since catastrophic forest fires are becoming more common in the United States; therefore, more people and their water supply will be affected. The objectives of this research are listed below.

- Determine the effects of the Rodeo-Chediski fire on the water quality of the Salt River.
- Determine the impacts of the Hayman fire on the water quality of the South Platte River, Goose Creek, and Cheesman Reservoir.
- Determine long-term impacts of forest fires on drinking water quality.

CHAPTER 2 : LITERATURE REVIEW

This literature review focuses on the short- and long-term effects of wildfires on stream water chemistry and drinking water quality. Cummings (2002) also completed a literature review pertaining to the effects of wildfires on water quality, it is located in Appendix A. Constituents released and created by forest fires will be discussed.

Forest fires cause the release and production of various constituents. Examples of released constituents are nitrogen, phosphorus, and suspended sediment. These are also commonly measured water quality constituents. In addition, some metals and ions are released and measured. Polycyclic aromatic hydrocarbons (PAHs) and dioxins are created by forest fires. Although PAHs and dioxins are regulated by the Environmental Protection Agency (EPA), they are not consistently measured in stream water samples. Therefore, only the sources and effects of these two constituents will be discussed.

As a result of forest fires and storm events, all of these constituents will potentially be mobilized and transported into a stream or lake. The literature reviewed focuses mainly on the impacts of fire on stream water quality and the environment. Studies evaluating the impacts of fire on drinking water quality are limited.

Wildfires play an increasing role in the quality of stream water and drinking water. The short-term impacts of fire on stream water quality are intensely studied relative to the long-term effects. Several studies have focused on the long term effects of prescribed burns (Davis 1989, Debano and Conrad 1978, Monleon and Cromack 1995). Although wildfires tend to burn at higher temperatures than prescribed burns, these results can be used as an indicator to determine the long term impacts on water quality. However, studies pertaining to drinking water impacts are limited.

This literature review will emphasize the long-term impacts of forest fires on water quality and drinking water quality when possible. The short- and long-term effects of fire on stream flow, erosion, nutrients, metals, PAHs, and dioxins will be discussed. In addition, EPA drinking water regulations of constituents addressed in the literature review will be included where applicable.

STREAM FLOW

Stream flow is affected by forest fires. Gottfried et al. (2003) reported historically high peak stream flows during the initial storm events following two different wildfires. After the Rodeo-Chediski fire burned in the Stermer Ridge watershed, a peak flow of 232ft^3/s was estimated, which is 2,350 times higher than the historic peak flow. The Coon Creek fire was the other fire included in the study. It burned within the Sierra Ancha Mountains in Arizona and caused a post-fire peak flow of approximately 2,000ft^3/s. This was more than seven times greater than the historic peak flow.

A study conducted by Belillas and Roda (1993) in the heathlands of northeastern Spain found that the mean annual stream flow increased by 36% during the two-year post-fire period. Elliott and Parker (2001) reported flooding following a severe wildfire in central Colorado after a post-fire storm event that caused only minor runoff in surrounding, unburned areas. Similar results were found for the first year after a wildfire in central Idaho by Minshall et al. (2001).

In contrast, Kutiel and Inbar (1993) reported a decrease in runoff after a moderate wildfire burnt a pine forest plantation in Israel. The decrease is attributed to increased infiltration capacity. The low-intensity burn only burned the understory of the forest; the canopy intercepted the rainfall and reduced the intensity. In addition, infiltration rates remained constant since the low-intensity burn did not alter the soil structure by making it hydrophobic. Hydrophobic soil occurs as a result of heat-induced changes to soil structure (DeBano 1999) and increases runoff (Mataix-Solera and Doerr 2003).

The short-term effects of fire on stream flow have been thoroughly studied. Typically, the stream flow will peak during the initial rain events subsequent to the fire (Gottfried et al. 2003, Elliott and Parker 2001). The long-term effects range from one to two years post-fire. Belillas and Roda (1993) and Minshall et al. (2001) reported elevated stream flows during this post-fire time period.

EROSION AND SUSPENDED SEDIMENT

Erosion is an immediate response to forest fires that will have a long-term effect on drinking water quality. Erosion is affected by three major factors: fire intensity, steepness of slopes, and post-fire re-growth of vegetation (Tiedemann et al. 1979; Wondzell and King 2003). Nutrient loss due to water erosion directly correlates too fire intensity (Gimeno-Garcia et al. 2000). The lack of vegetation can also make a burned area more susceptible to wind erosion, which is another method of nutrient transport (Lewis, W.M. Jr. 1974, Wondzell and King 2003). In addition, the soil structure can be altered by fire, becoming hydrophobic. This causes an increase in soil erodibility due to the decreased infiltration capacity of hydrophobic soil (Gottfried et al. 2003, Neary 2004). The increase in erosion causes more nutrients to be transported and deposited into streams and rivers.

Gimeno-Garcia et al. (2000) reported increases in erosion four months after experimental fires in Mediterranean shrubland. The plots exposed to high-intensity burns lost 4,077kg/ha of soil. The moderate-intensity burns caused 3,280kg/ha of soil to be transported. The unburned plots in this study lost 72.8kg/ha of soil.

In central Idaho, Minshall et al. (2001) reported high suspended sediment and particulate organic matter concentrations in streams one year after the Mortar Creek wildfire. These concentrations were especially high in runoff from subsequent snowmelt and storm events. In contrast, Kutiel and Inbar (1993) reported a decrease in erosion rates after a moderate wildfire burned a pine forest plantation in Israel. The burned, mulch-like material on the forest floor created during the fire prevented erosion.

Fire-induced erosion can impact drinking water quality (Dissmeyer 2000). Fire influences short- and long-term sediment transport trends (Swanson 1981).

Increased turbidity of drinking water sources is a short-term impact. Long-term impacts result when sediment settles to the bottom of a stream or lake. Dissmeyer (2000) noted long-term water quality impacts from stored sediment occur if metals or other chemicals are contained within the sediment.

Fire can cause erosion to increase in a burned watershed. In general, the studies reviewed concluded increases in erosion will occur after a fire. Increased turbidity in water sources is a short-term impact of the elevated suspended sediment loads. Long-term effects were studied by Minshall et al. (2001). The authors reported elevated suspended sediment concentrations one year after a wildfire. In addition, sediment settling to the bottom of a stream or lake can affect long-term water quality (Dissmeyer 2000); metals and chemicals contained within the sediment will diffuse into the drinking water source.

NUTRIENTS

Organic nutrients such as nitrogen and phosphorus do not pose a threat to human health. However, nitrate and nitrite which are byproducts of nitrogen decomposition do pose a health risk. The EPA has set a maximum contaminant level (MCL) of 10mg-N/L and 1mg-N/L for nitrate and nitrite, respectively. At concentrations above the MCL, infants less than six months of age may experience shortness of breath and blue-baby syndrome. If the infant does not receive treatment, death may result.

Nitrogen and phosphorus are nutrients commonly measured in stream water. Studies were conducted to determine the fate of these nutrients after a fire. During a wildfire in northwest Montana, Spencer and Hauer (1991) found a five- to 60-fold increase in phosphorus and nitrogen relative to background concentrations. Within several weeks the dissolved nutrient concentrations returned to background levels. The leaching of ash deposits was the phosphorus source, while atmospheric deposition was the dominant nitrogen source. Five years after this wildfire in northwest Montana, Spencer, Gabel, and Hauer (2003) noted periodic increases in nitrogen and phosphorus concentration during spring runoff.

Similar trends for nitrogen and phosphorus concentrations were reported in a study conducted by Earl and Blinn (2003). The results are summarized in Table 2.1. The authors found that within four months after two wildfires in southwestern New Mexico, nitrate, ammonia, and soluble reactive phosphate had increased concentrations in stream water. After the Bonner fire the change in nitrate and ammonia concentration was 250% and 400%, respectively. After the Langstroth fire soluble reactive phosphate concentration change was 567% and nitrate concentration change was 250%. The elevated concentrations returned to background levels within four months. The increased concentrations were attributed to a monsoon rain transporting ash into the West Fork of the Gila River after the Langstroth fire and atmospheric fallout after the Bonner fire. In addition, the authors conducted a five year ash study following a wildfire. The effects of the ash were evident, but short-lived.

Minshall et al. (2001) found elevated nitrate concentrations during spring runoff in the first year after a wildfire in central Idaho. However, Gerla and Galloway (1997) reported nitrate concentrations in stream water following a wildfire in Yellowstone Park, Wyoming were not elevated until two years post-fire. The nitrate transported from the watershed reached over 1.5mmoles/ha-s; however, five years after the fire, a maximum of 0.2mmoles/ha-s of nitrate was transported. The delay was potentially due to the natural decomposition processes followed by leaching of nitrate produced by these processes. In addition, transport of total phosphorus concentration from the burned basin reached 0.011mmoles/ha-s the year following the fire. However, during the next three years the phosphorus transport decreased to 0.003mmoles/ha-s. The decreased transport rate is attributed to the re-growth of vegetation utilizing the phosphorus.

Tiedemann et al. (1978) also reported elevated total phosphorus concentrations in streams after a severe wildfire located in eastern Washington. An increase of 1.5 to 3 times relative to an unburned watershed was reported. The elevated concentrations occurred three to five years post-fire. However, the water quality was only minimally impacted, therefore, still available for municipal use.

As a result of a prescribed burn in a South Carolina pine forest, the first rainfall had higher nitrate and phosphate concentrations; 14.000mg-N/L and 1.033mg-P/L, respectively. These concentrations are high relative to the reference watershed concentrations of 5.300mg-N/L for nitrate and 0.077mg-P/L for phosphate. However, this trend did not exist in other storm events (Lewis 1974).

A long-term drought caused above average nitrate, total nitrogen, and ammonia concentrations in stream flow after a wildfire in Malaysian Borneo. The results are summarized in Table 2.1. The concentrations were high relative to stream flow in a controlled burn in the same area that was not in a drought. However, the total phosphorus concentrations remained at background levels (Malmer 2004). Two months post-fire the concentration change of nitrate, total nitrogen, and ammonia was 2,970%, 217%, and 670%, respectively. However, nitrate and total nitrogen concentrations returned to background levels one year post-fire. Ammonia concentration declined one year post-fire, but remained at a level twice the pre-fire concentration.

Studies conducted by Tiedemann et al. (1978) and Davis (1989) reported results that contrast the other short-term trends for nitrate as previously discussed. Tiedemann et al. (1978) determined that no significant impact on the nitrate concentration occurred as a result of the wildfire in northeastern Washington. Davis (1989) reached the same conclusion for a prescribed burn in Arizona chaparral. The difference in the results could be due to factors such as fire intensity, climate, steepness of slopes, and soil type. These results are summarized in Table 2.1.

As mentioned before, erosion causes nutrient transport. Debano and Conrad (1978) found significant losses of nitrogen due to erosion after a prescribed fire in southern California. The net loss was 15kg/ha of nitrogen. Gimeno-Garcia, Andreu, and Rubio (2000) reported a loss of 8.1mg-N/m^2 ammonia and 7.5mg-N/m^2 nitrate due to water erosion after moderate-intensity experimental burns in Mediterranean shrub land.

Nitrogen and phosphorus concentrations can also be influenced by litter decomposition. Five years after a low-intensity prescribed burn in central Oregon, Monleon and Cromack (1995) found significantly elevated concentrations of nitrogen and phosphorus being released from litter. The delay in the release is attributed to the slow microbial decomposition, which is common in burnt areas.

Short-term impacts of fire on nutrient transport varied throughout the literature, while long-term impacts were more consistent; Table 2.1 summarizes these impacts. The short-term results for post-fire nutrient trends conflicted. The difference in the results could be due to factors such as fire intensity, climate, steepness of slopes, and soil type. The long-term results were consistent. Nitrogen and phosphorus concentrations were consistently higher 1-5years post-fire. However, the water quality was only minimally impacted, therefore, still available for municipal use.

MAJOR IONS

Cations

The EPA does not regulate the cations discussed in this literature review since they do not pose a human health hazard. Calcium, magnesium, potassium, and sodium are commonly measured cations in stream water. Cations can be mobilized and deposited into stream water by runoff or wind (Lewis 1974). Tiedemann et al. (1978) found that the average combined concentration of calcium, magnesium, potassium, and sodium increased within the first year after the wildfire. However, concentrations of the four cations declined two years after the fire. The decrease was attributed to dilution due to increased discharge. These results are summarized in Table 2.1.

Earl and Blinn (2003) determined that elevated potassium concentrations were a result of ash flow from the Langstroth wildfire in southwestern New Mexico. The change in the post-fire average potassium concentration was 189%. However, within four months the background concentrations were re-established. The Bonner fire was also included in this study. The post-fire concentration was the same as the pre-fire average.

In Malaysian Borneo, Malmer (2004) found a direct relationship between fuel supply and the base flow potassium concentration of streams draining the burned watershed. For at least two years, the potassium concentration remained significantly higher than the pre-fire concentration of 1mg/L in base flow. Malmer (2004) also correlated stream flow with calcium concentration for a controlled burn and a wildfire. The post-fire stream flow had a concentration of potassium increased by 413% within two months. A concentration nearly twice the background level was still present one year after the fire. The higher concentration of calcium after the wildfire is attributed to a long-term drought.

Davis (1989) and Minshall et al. (2001) reported results that contrasted other studies. Davis (1989) found that a prescribed burn in Arizona chaparral had no effect on the stream water concentration of these cations. Minshall et al. (2001) came to the same conclusion after studying a wildfire in central Idaho.

The impacts of fire on the release and mobilization of cations are inconsistent. The effects ranged from no impact (Davis 1989, Minshall et al. 2001) to elevated concentrations two years post-fire (Malmer 2004). The varied results are potentially due to climatic conditions, such as a drought (Malmer 2004).

Anions

Chloride and bicarbonate are commonly measured anions in stream water. The EPA has set a secondary MCL of 250mg/L for chloride, but does not regulate bicarbonate concentration. Davis (1989) concluded that a prescribed burn in Arizona chaparral had no significant effect on the bicarbonate and chloride concentration in the stream water. Belillas and Roda (1993) reported that a prescribed burn in northeastern Spain had minimal effects on the chemistry of stream water within the watershed; the average bicarbonate concentration and chloride concentration did not significantly change after the fire. The results of these studies are summarized in Table 2.1.

Table 2.1 Comparison of pre- and post-fire changes in ions and nutrients

Location	Type of Fire	Post-fire Sample Collected (years post-fire)	Constituents Analyzed	Results (% change)†	Author
Central Arizona	Presribed	1	Bicarbonate	-6%	Davis (1989)
Northeastern Spain	Presribed	2*	Bicarbonate	2%	Belillas and Roda (1993)
Malaysia	Wildfire	0.17	Calcium	413%	Malmer (2004)
Central Arizona	Presribed	1	Calcium	-29%	Davis (1989)
Eastern Washington	Wildfire	0	Calcium	1%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	1	Calcium	-27%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	2	Calcium	-54%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	3	Calcium	-66%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	4	Calcium	-63%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	5	Calcium	-52%	Tiedemann, et al (1978)
Northeatern Spain	Presribed	2*	Calcium	8%	Belillas and Roda (1993)
Central Arizona	Presribed	1	Chloride	12%	Davis (1989)
Northeastern Spain	Presribed	2*	Chloride	-4%	Belillas and Roda (1993)
Central Arizona	Presribed	1	Magnesium	0%	Davis (1989)
Eastern Washington	Wildfire	0	Magnesium	30%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	1	Magnesium	0%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	2	Magnesium	-40%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	3	Magnesium	-20%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	4	Magnesium	-10%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	5	Magnesium	10%	Tiedemann, et al (1978)
Northeatern Spain	Presribed	2*	Magnesium	-2%	Belillas and Roda (1993)
Malaysia	Wildfire	0.17	Ammonia	670%	Malmer (2004)
Southwestern New Mexico	Wildfire	<0.25	Ammonia	400%	Earl and Blinn (2003)
Malaysia	Wildfire	0.17	Total Nitrogen	217%	Malmer (2004)

(continued)

Table 2.1 (continued)

Location	Type of Fire	Post-fire Sample Collected (years post-fire)	Constituents Analyzed	Results (% change)†	Author
Malaysia	Wildfire	0.17	Nitrate	2970%	Malmer (2004)
Southwestern New Mexico	Wildfire	<0.25	Nitrate	250%	Earl and Blinn (2003)
Central Arizona	Presribed	1	Nitrate	71%	Davis (1989)
Northeastern Spain	Presribed	2*	Nitrate	469%	Belillas and Roda (1993)
Eastern Washington	Wildfire	1	Nitrate	25%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	2	Nitrate	-90%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	3	Nitrate	-97%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	4	Nitrate	-97%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	5	Nitrate	-99%	Tiedemann, et al (1978)
Southwestern New Mexico	Wildfire	<0.25	Soluble Reactive Phosphorus	567%	Earl and Blinn (2003)
Southwestern New Mexico	Wildfire	<0.25	Potassium	189%	Earl and Blinn (2003)
Central Arizona	Presribed	1	Potassium	16%	Davis (1989)
Eastern Washington	Wildfire	0	Potassium	70%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	1	Potassium	10%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	2	Potassium	-10%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	3	Potassium	-20%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	4	Potassium	10%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	5	Potassium	10%	Tiedemann, et al (1978)
Northeastern Spain	Presribed	2*	Potassium	130%	Belillas and Roda (1993)
Central Arizona	Presribed	1	Sodium	23%	Davis (1989)
Eastern Washington	Wildfire	0	Sodium	22%	Tiedemann, et al 1978
Eastern Washington	Wildfire	1	Sodium	0%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	2	Sodium	-13%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	3	Sodium	4%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	4	Sodium	17%	Tiedemann, et al (1978)
Eastern Washington	Wildfire	5	Sodium	0%	Tiedemann, et al (1978)
Northeastern Spain	Presribed	2*	Sodium	11%	Belillas and Roda (1993)
Central Arizona	Presribed	1	Sulfate	-48%	Davis (1989)
Northeatern Spain	Presribed	2*	Sulfate	18%	Belillas and Roda (1993)

*Average of all samples collected 0-2yrs post-fire

†%Change = (Final concentration - Initial concentration)/(Initial Concentration)

METALS

Metals released due to forest fires are mobilized by runoff during storm events. The metals are then transported into streams, rivers, and lakes. The newly deposited metals are potentially toxic to humans. The EPA has set drinking water MCLs to protect human health. Table 2.2 is a summary of the EPA MCLs and the potential health effects from drinking water contaminated by metals.

The mobilization of metals can be influenced by fire and fire-induced pH changes. Ballard (2000) reported mobilization of manganese, zinc, and aluminum under these conditions. Sakalauskiene and Ignatavicius (2003) reported similar results following a fire in Lithuania during a drought. Copper, lead, and zinc concentrations were 21%-74% above background levels in Lithuanian rivers within a few months of the fire. In addition, metals can be lost due to volatilizations. Gao et al. (2003) reported only minor iron concentrations in smoke from a savanna fire in southern Africa. The elevated concentrations of metals are potentially harmful to human health and the environement (Sakalauskiene and Ignatavicius 2003).

In addition to human health effects, metals may impact algae, diatoms, and periphyton living in contaminated water. Genter and Lehman (2000) reported a depleted community of diatoms and algae as a result of arsenic and copper contamination from mine sites near a stream in Idaho. In South Korea, Shin et al. (2002) noted a reduction in the growth rate of microalgal communities as a result of cadmium exposure; the additional cadmium within the system was attributed to forest fires. Medley and Clements (1998) studied periphyton communities in the Cache la Poudre River in Colorado. The periphyton displayed little response to zinc concentrations below 50ug/L, but at concentrations exceeding 200µg/L the composition of the periphyton communities was altered. Also, as a part of this study, diatoms were exposed to cadmium, copper, and zinc. Species that were initially present were more tolerant than the successional species.

Table 2.2 Maximum contaminant level and health effects of various metals

Metal	MCL (mg/L)	Health Effects
Arsenic	0.05	Skin damage; circulatory system problems; increased risk of cancer
Barium	2	Increase in blood pressure
Copper	1.3†	Short term exposure: Gastrointestinal distress
		Long term exposure: Liver or kidney damage
Iron	0.3*	N/A
Lead	0.015†	Infants and children: Delays in physical or mental development
		Adults: Kidney problems and high blood pressure
Manganese	0.05*	N/A
Mercury	0.002	Kidney damage
Selenium	0.05	Circulatory system problems; loss of hair and fingernails
Zinc	5*	N/A

Source: EPA 2004
*Secondary MCL
†Action Level: the level which, if exceeded, triggers treatment or other requirements that a water system must follow (EPA 2003)

POLYCYCLIC AROMATIC HYDROCARBONS

Polycyclic aromatic hydrocarbons can be generated during incomplete combustion or organic compounds (ATSDR 1995, Muri et al. 2003). Forest fires are one source of PAHs. Retene, phenanthrene, and pyrene are examples of PAHs produced from the burning of biomass (Muri et al. 2003). Typically, PAHs are released into the air or sorbed onto sediment. PAHs have a high affinity for particulates rather than to volatilize into the air or dissolve into water. As established by the Environmental Protection Agency (EPA), the maximum contaminant level (MCL) of benzo(a)pyrene and similar PAHs is 0.2ppb.

PAHs have the potential to cause detrimental health effects. In humans, PAHs are typically retained in the kidneys, liver, spleen, and fat and are excreted within a few days, causing little harm (ATSDR 1995). However, during prolonged exposure to PAHs can be harmful. An increase in the development of tumors and cancer has been found in animals and humans as a result of prolonged exposure to PAHs (ATSDR 1995). In other animal studies, tumor formation has occurred at very low doses of a common PAH, benzo(a)pyren (Heil 1998). Reproduction difficulties, birth defects and a weakened immune system are also a result of animal studies (ATSDR 1995, Nielsen et al. 1995). Although the Department of Health and Humans Services, the International Agency for Research on Cancer, and the EPA have determined that phenanthrene and pyrene cannot be classified as carcinogens to humans, these compounds exist in the environment as complex mixtures. Within these mixtures, other mutagenic and carcinogenic compounds can exist causing concerns for human health (Nielsen et al. 1996).

Since wood is considered to have a low energy density, more PAHs will be emitted than by the combustion of high-energy density material. PAHs are created during forest fires and are a concern due to the MCL. Bundt et al. (2001) found that wood ash has high concentrations of PAHs; therefore, PAHs can be leached and transported into streams.

Due to the hydrophobic nature of PAHs, as well as resistance to biodegradation and chemical stability, persistence of these compounds in the environment is expected (Muri et al. 2003). On land, PAHs can be broken down by photodegradation or microbial degradation, the latter taking much longer. When particulates are transported by water, the PAHs adsorbed onto the particulates are mobilized as well. The PAHs are likely to stay attached to the particulate, which may then settle to the bottom of the river or lake and persist even longer. Once in the water, photo and chemical oxidation processes also aid in breaking down the compounds (ATSDR 1995). However, bioaccumulation within plants and animals may occur before degradation takes place.

DIOXINS

Dioxins are a byproduct of combustion processes such as volcanic eruptions and forest fires. Dioxin is actually a chemical known as 2, 3, 7, 8-tetrachlorodibenzo-para-dioxin (TCDD). In addition to TCDD, polychlorinated dibenzo-para-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs) and specific polychlorinated biphenyls (PCBs) make up a family of structurally and chemically similar compounds are considered dioxins; TCDD is the most toxic. Dioxins are very chemically stable, persist in the environment, and exist in air, soil, water, and food; the EPA has established a MCL of .00003µg/L.

Disposing of materials contaminated with dioxins is difficult. Ironically, the method used by humans to destroy dioxins is the same method nature uses to create them: fire. However, incinerating dioxin-contaminated materials requires extreme temperatures, 850°C-1000°C (WHO 1999).

Due to their hydrophobic nature, dioxins are likely to attach to soil particles, although some dioxins do evaporate. Prior to erosion, photodegradation is one pathway in which dioxins are broken down (ATSDR 1998). However, when these soil particles are eroded and deposited into a stream, bioaccumulation is likely to occur. The most common pathway of exposure for humans to dioxins is through the consumption of contaminated food (WHO 1999, ATSDR 1998).

Dioxins have a negative effect on human health. The health effects are progressively more extreme and detrimental as exposure level and time increase. At a high exposure level and a low exposure time, skin lesions and altered liver function are common responses (WHO 1999). At a high exposure level and a long exposure time, the human immune, nervous, endocrine, and reproductive systems exhibit impaired functions. At the chronic exposure level, only animals have been tested (WHO 1999). The result was the classification of TCDD as a known human carcinogen.

Dioxins can be removed during the water treatment process. Their hydrophobic nature actually makes the removal processes of coagulation, sedimentation, and sand filtration more effective (Kim et al. 2002). In Japan, Kim et al. (2002) reported 87% removal efficiency during water treatment for most dioxin congeners. In Russia, Smirnov et al. (1996) reported the highest removal efficiency of 90%-95% was obtained by filtering the contaminated water through granulated sorbents. This included the removal of the most toxic dioxin, TCDD. Although dioxins are very toxic, they can be effectively and efficiently removed through general drinking water treatment processes.

SUMMARY

Fire has the potential to impact water quality. Treating the runoff water from a burned watershed will vary based on the intensity of the storm event and proximity to a water treatment plant. Short-term effects consist of pulses of soluble material, particulates, and metals in stream water. The mobilization typically occurs during the initial storm events following a fire. A flux of nutrients can stimulate eutrophication and bacteria growth causing negative aesthetic effects on drinking water quality (Tiedemann et al. 1979).

The long-term effects consisted of an overall increase in concentration of a constituent relative to background concentrations. Elevated stream flow and increased nutrient and potassium concentrations were reported from 2-5 years post-fire. As previously mentioned, elevated nutrient concentrations can stimulate growth of algae and bacteria within a stream.

The EPA has set MCLs for metals, PAHs, and dioxins, making their post-fire fate important. Metals are commonly measured in water samples. However, PAHs and dioxins are typically measured not due to the cost of the analysis. Only the short-term trends of metals were reported; post-fire concentrations increased in air and water but returned to background levels within four months. Due to the health risks posed by PAHs and dioxins, more studies need to be conducted to determine their post-fire fate and transport.

CHAPTER 3 : METHODS AND MATERIALS

This chapter presents information pertaining to the collection and analysis of water samples from watersheds in Arizona and Colorado. Characteristics of the Salt River and Verde River watersheds in Arizona and the South Platte River watershed in Colorado are presented, as well as the sampling sites within these watersheds. Methods used to collect and analyze the samples are also discussed.

SAMPLING SITES

Rodeo-Chediski Fire

The Salt River watershed spans across 11,000km^2 of eastern Arizona, as shown in Figure 1.1. It extends north to the Mogollon Rim, east to the Black River and White River, west until the confluence with the Gila River, and south to the Salt River. The Salt River flows into a series of reservoirs (Roosevelt Lake, Apache Lake, Canyon Lake, and Saguaro Lake) before it is diverted into canals for drinking water and irrigation within the Phoenix metropolitan area, as shown in Figure 1.2.

The upper Salt River (the portion above Roosevelt Lake) maintains a perennial flow. This is due to the many tributaries that exist within the Salt River watershed that carry runoff and snowmelt. Storm flow is typically due to monsoon rains that cause a temporary increase in stream flow, while snowmelt provides a consistent base flow during the winter months.

The Verde River watershed is adjacent to the Salt River watershed, as shown in Figure 1.2. The Verde River watershed drains approximately 14,000km^2 in north-central Arizona. The Verde River is about 225km in length, beginning at Sullivan Lake Dam, near Paulden, Arizona, passing through Horseshoe Reservoir and Bartlett Reservoir, and ending at the confluence with the Salt River.

Perennial flow in the Verde River is maintained beginning three miles below the Sullivan Lake Dam. Tributaries and groundwater discharge continually flow into the Verde River. Similar to the Salt River, storm flows are a result of monsoon rains causing very brief increases in stream flow, while snowmelt provides a steady base flow during the winter months.

Two sites were selected for Salt River sampling and one site for Verde River sampling. The sampling conducted by Arizona State University (ASU) began at the two Salt River sites in October, 2003 and ended in November, 2004. The Salt River sampling site was selected since it is the location of the United States Geological Survey (USGS) gauging station 09498500; continuous flow rates, as well as historical nutrient, trace metal, and ionic data exist for this location. The Windy Hill sampling site on Roosevelt Lake was selected based on accessibility. The location of the USGS gauging station 09498500 and Windy Hill are shown in Figure 1.1. USGS gauging station 09508500 is located at the Verde River site. ASU and the USGS have conducted the sampling. Continuous flow rate data is recorded and nutrient analysis is conducted on water samples at this site. The Verde River sampling site is shown in Figure 1.2.

Samples were collected on a monthly basis for both sites. Grab samples were taken from mid-depth. At the Salt River site, samples were taken at least five feet from the south side of the bank. During increased stream flow periods within the Salt River samples were collected more frequently. The dates and locations of all sampling trips are listed in Table 3.1.

When collecting samples, three 1L amber glass bottles with lined screw caps were used at each of the locations. The caps and bottles were both soaked in a 10% hydrochloric acid bath for a minimum of 24 hours. The bottles were then ashed at 550°C for six hours. All bottles were stored covered with aluminum foil to prevent contamination. This same procedure was followed for the 40mL amber bottles used during lab analysis.

Table 3.1 Sample collection dates

Date of Sampling Trip	Salt River	Windy Hill
10/24/2003	X	X
11/21/2003	X	X
12/5/2003	X	
1/23/2004	X	
2/6/2004	X	X
2/20/2004	X	X
3/5/2004	X	X
3/26/2004	X	X
4/3/2004	X	
4/9/2004	X	
4/23/2004	X	X
5/11/2004	X	X
6/16/2004	X	X
7/14/2004	X	X
8/10/2004	X	X
8/21/2004	X	

Hayman Fire

The South Platte River watershed drains an area of approximately 72,700km^2. The river travels 725km originating in the Rocky Mountains at the Continental Divide and ending at the confluence with the North Platte River in North Platte, Nebraska. The watershed spans three states, Colorado, Nebraska, and Wyoming. Nearly 80% of the drainage area is located within Colorado. The Denver metropolitan area is located entirely within this watershed. The perennial flow within the South Platte River is maintained by many tributaries, snowmelt, and rainfall.

The City of Denver (Steve Lohman) provided pre- and post-fire water quality data. The sampling sites utilized in this research are listed in Table 3.2 and displayed in Figure 3.1. Metal concentrations and flow rate data were analyzed at each of the four sites.

Table 3.2 City of Denver sampling sites

Sampling Site	Abbreviation
Goose Creek inlet at Chessman Reservoir	GC
South Platte River inlet at Cheesman Reservoir	CLI
South Platte River outlet at Chessman Reservoir	CLO
South Platte River below the confluence of of the North	NFSP

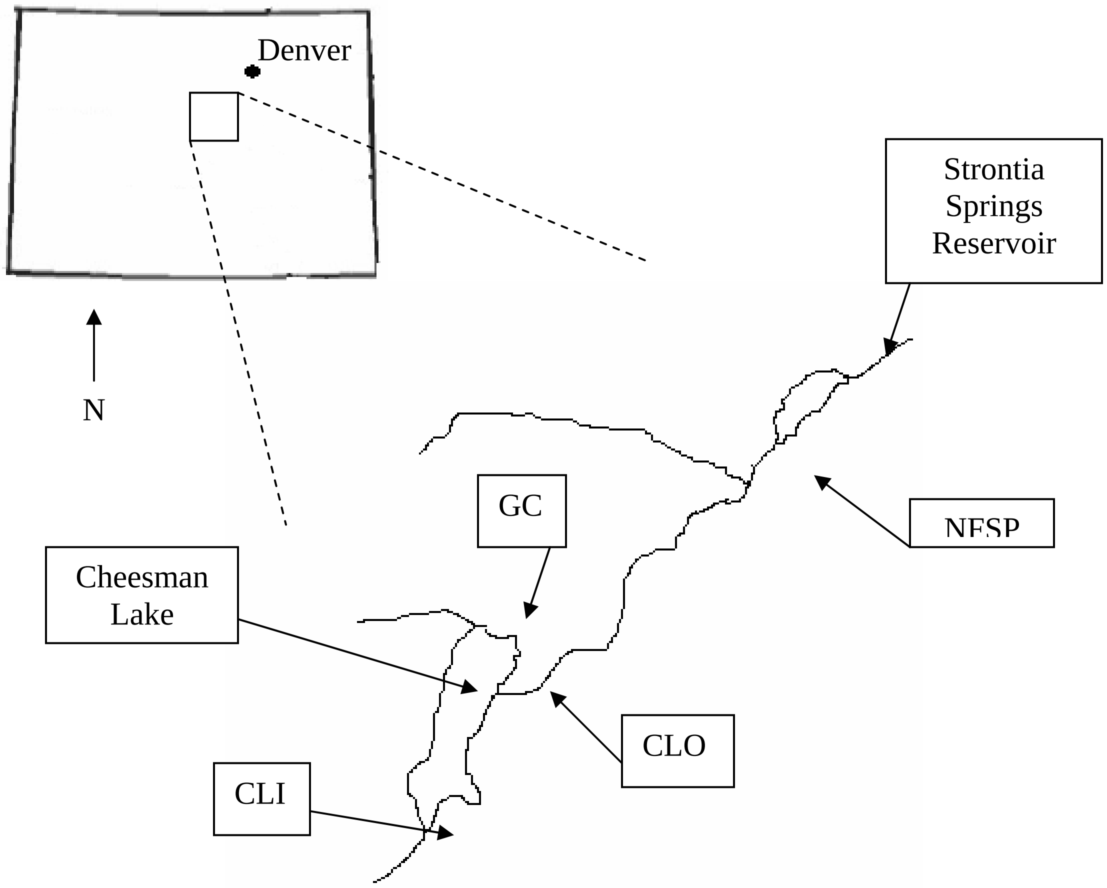

Figure 3.1 Map of Hayman fire sampling sites

ANALYTICAL METHODS

During every sampling trip two field measurements were always recorded, pH and temperature. A YSI Model 60 Handheld pH and Temperature System was used to take these measurements. Samples were transported to the Arizona State University (ASU) lab in a cooler filled with blue ice and stored at 4°C until analysis. All filtration and preservation was completed within 24 hours of sample collection. All samples were filtered through GF/C glass fiber filters. The filters had been ashed at 550°C for a minimum of 20 minutes and rinsed with a minimum of 200mL of nanopure water prior to use.

All samples were analyzed for the same constituents and parameters. Filtered samples acidified with hydrochloric acid to a pH less than two were used for ultraviolet absorption, total dissolved nitrogen, and dissolved organic carbon analysis. Ultraviolet absorption was measured at a wavelength of 254nm using the Schimadzu MultiSpec-1501. Total dissolved nitrogen and dissolved organic carbon were measured using the Schimadzu TOC Analyzer. Non-filtered samples were analyzed for conductivity using the VWR EC meter, Model 2052. These samples were stored in glass, amber bottles prepared as described above.

Dissolved arsenic, dissolved phosphorus, and total phosphorus analysis was conducted as well. For dissolved arsenic preservation, samples were filtered using ashed and rinsed GF/C filters then preserved with nitric acid at a pH less than two and stored in 60mL HDPE nitric acid washed bottles. A Varian

SpectrAA 400Zeeman Graphite Furnace Atomic Absorption Spectrometer was used for dissolved arsenic analysis. Non-filtered samples for total and dissolved phosphorus analysis were preserved with hydrochloric acid to a pH less than two and stored glass, amber bottles prepared as described above. Prior to analysis, the portion of the sample to be used for DP was filtered using a GF/C glass fiber filter that was rinsed with 50mL of 0.1N hydrochloric acid followed by 50mL of deionized water. Digestion for the TP and DP samples was completed by using potassium persulfate and the autoclave. A Bran-Luebbe TrAAcs 800 Autoanalyzer was used for the phosphorus analysis.

Samples were also analyzed for suspended solids (Method 2540G; APHA 1998). Samples were filtered through ashed and rinsed GF/C glass fiber filters. For each filtered sample, at least, one filter was exposed to only 105°C. These filters were preserved in glass, amber bottles for carbon, hydrogen, nitrogen (CHN) analysis; the bottles were prepared as described above.

CHAPTER 4 : WATER QUALITY ON THE SALT RIVER POST RODEO-CHEDISKI FOREST FIRE

From June 18, 2002 to July 7, 2002 the Rodeo-Chediski fire burned approximately 1,870km^2, burning 11% of the Salt River watershed. Forest fires cause the release of materials from soil and vegetative matter. The burned material does not retain water or prevent erosion, therefore surface runoff and erosion increase and infiltration rates decrease. The fire-induced release of nutrients, trace metals, and ions once retained in vegetation and litter are mobilized by post-fire runoff. The Salt River concentration data analysis was completed for these constituents. For further comparison, the Verde River suspended sediment and nutrient concentration data is analyzed. The Verde River watershed is adjacent to the Salt River watershed and will represent the reference watershed.

The continuous stream flow data for the Salt and Verde Rivers was analyzed to generate a hydrograph. The first major storm event after the fire was emphasized. The objective of this chapter is to establish the effect of the Rodeo-Chediski fire on the water quality of the Salt River.

SALT RIVER FLOW RATE DATA

The Salt River watershed drains approximately 11,000km^2; this is the area above the United States Geological Survey (USGS) gauging station 09498500 on the Salt River above Roosevelt Lake. Due to a prolonged regional drought, the runoff and precipitation contributing to the stream flow in the Salt River has been low for the past eight years when compared to historical data. Salt River stream flow is shown in Figure 4.1, the hydrograph begins in 1919 and ends in 2004. The maximum flow rate achieved by the Salt River was 2,500m^3/s and the minimum flow rate sustained was 1.5m^3/s. Within the past eight years, the maximum flow rate achieved was only 250m^3/s, with an average stream flow of 13.6m^3/s. This eight year drought increases the possibility of fires and allows the accumulation of soluble materials near the sediment surface.

An additional hydrograph was created to emphasize the first major rain event following the Rodeo-Chediski fire, termed the first flush. The hydrograph begins in 2000, as shown in Figure 4.2. The first flush occurred in early August 2002. The stream flow during the first flush ranged from 30m^3/s to 100m^3/s. The minimum stream flow, 2m^3/s, in the Salt River since the Rodeo-Chediski fire occurred shortly after the fire was declared contained in July 2002.

The eight year drought has caused the historically low stream flows in the Salt River. Therefore, the effects of dilution are minimal, making the river susceptible to changes in concentrations of constituents transported by runoff. The first flush deposited an initial high flux of nutrients, sediment, trace metals, and ions into the Salt River. This time period is critical since it marks the beginning of the long term impact of the Rodeo-Chediski fire on Salt River water quality.

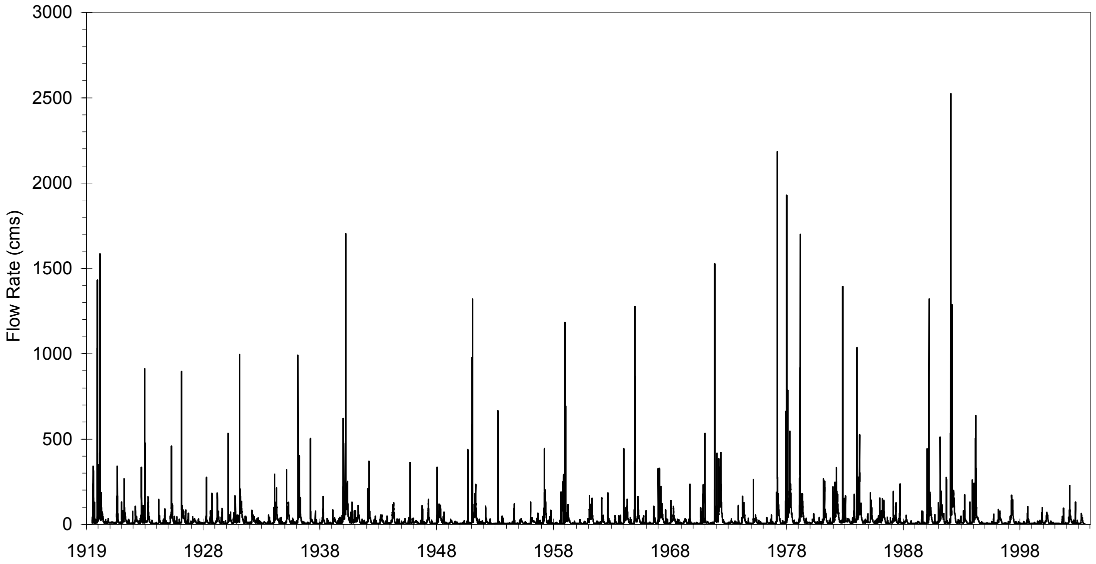

Figure 4.1 Salt River hydrograph 1914-2004 above Roosevelt Lake at USGS gauging station 09498500
Source: USGS

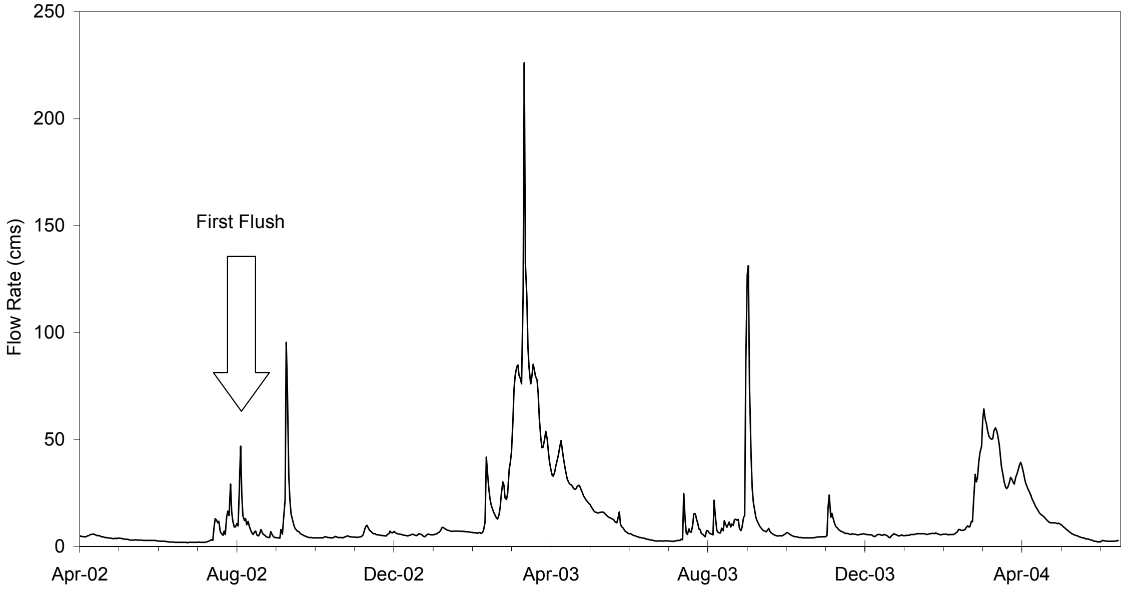

Figure 4.2 Salt River hydrograph 2000-2004 above Roosevelt Lake at USGS gauging station 09498500
Source: USGS

SALT RIVER CONCENTRATION DATA ANALYSIS

The data analysis in this chapter consists of comparing pre and post-fire concentration data for various constituents. A list of the constituents analyzed is in Table 4.1. The results of this analysis established the response of the Salt River due to the Rodeo-Chediski forest fire. Comparing historical data prior to the fire with post-fire data was the method used for this analysis. This comparison established the presence of short-term effects. Chapter 5 used the mass loading analysis to compare the data.

Table 4.1 List of constituents analyzed for the Salt River

Constituents	Pre 2002	2002-2003	2003-2004	Data collected by ASU
Nutrients				
Ammonia	X	X	X	X
Dissolved Organic Carbon	X	X	X	X
Dissolved Phosphorus	X	X	X	X
Nitrate-Nitrite	X	X		X
Total Nitrogen	X	X		
Total Organic Carbon	X	X		
Total Phosphorus	X	X	X	X
Trace Metals				
Total Arsenic	X	X		
Dissolved Arsenic	X	X	X	X
Total Barium	X	X		
Dissolved Barium	X	X		
Total Copper	X	X		
Dissolved Copper	X	X		
Total Iron	X	X		
Dissolved Iron	X	X		
Total Lead	X	X		
Dissolved Lead	X	X		
Total Manganese	X	X		
Dissolved Manganese	X	X		
Total Mercury	X	X		
Dissolved Mercury	X	X		
Dissolved Selenium	X	X		
Total Selenium	X	X		
Total Zinc	X	X		
Dissolved Zinc	X	X		
Ions				
Potassium	X	X		
Manganese	X	X		
Calcium	X	X		
Sodium	X	X		
Chloride	X	X		
Bicarbonate	X	X		
Other				
Suspended Sediment	X	X	X	X
Stream Discharge	Continuous data			

Suspended Sediment

Runoff from the area burned by the Rodeo-Chediski forest fire caused an increase in suspended sediment concentration in the Salt River, as shown in Figure 4.3. Before the Rodeo-Chediski fire, the maximum suspended sediment concentration was 12,000mg/L. As a result of the first flush, the suspended sediment concentration in the Salt River reached 22,900mg/L, exceeding the historical maximum concentration. The maximum concentration one year post-fire increased to 42,500mg/L. A high intensity storm event probably occurred in a severely burned area that had not re-established sufficient vegetation to prevent erosion. However, the suspended sediment concentration remained at background levels two years post-fire

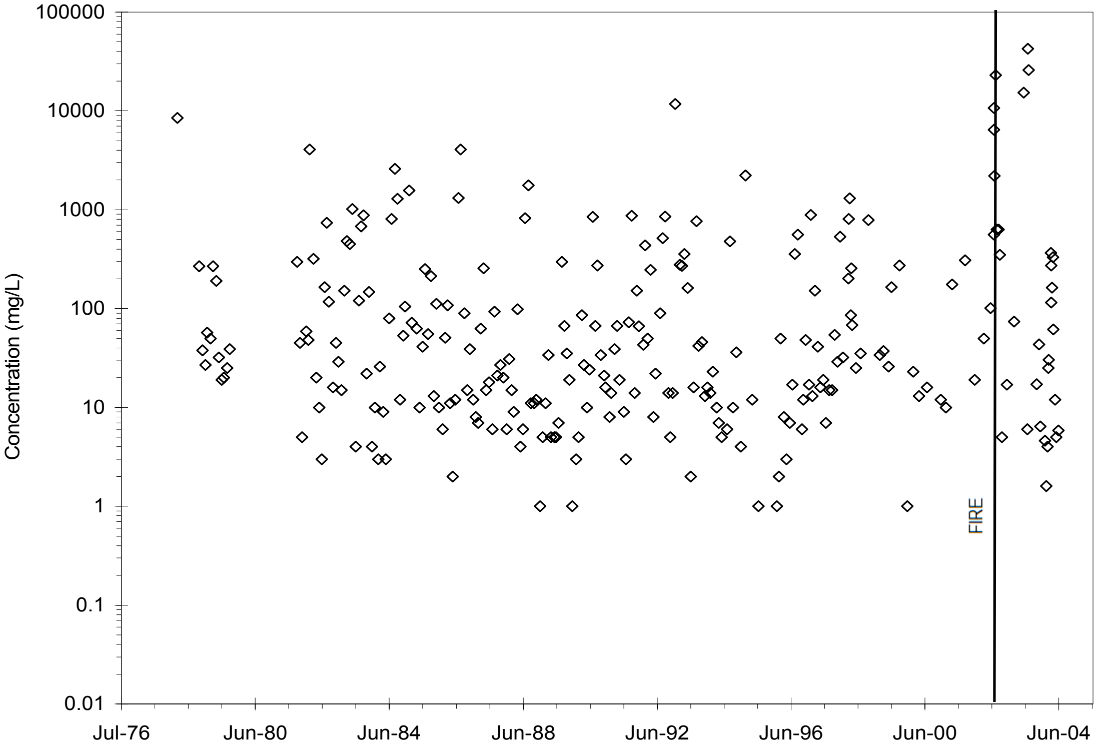

Figure 4.3 Suspended sediment concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500
Source: USGS

Nutrients

As a result of the Rodeo-Chediski fire, dissolved and particulate fractions of nutrients were released. The first flush mobilized the nutrients, increasing nutrient concentrations in the Salt River. The pre- and post-fire average concentration and concentration range are listed in Table 4.2.

The first flush caused historically high total and dissolved nitrogen concentrations. Prior to the Rodeo-Chediski fire the ammonia concentration range was below the MRL-0.06mg-N/L, as shown in Figure 4.4. During the first flush, the ammonia concentration increased by two orders of magnitude. The first flush also impacted the nitrate plus nitrite concentration, as shown in Figure 4.5. The post-fire maximum nitrate plus nitrite concentration exceeded the historical maximum by one order of magnitude. Similarly, the total nitrogen concentration surpassed the historical maximum of 3.8mg/L by two orders of magnitude, as shown in Figure 4.6. Greater than 95% of the total nitrogen concentration is probably associated with particulates.

In July 2003, one year post-fire, the total nitrogen, ammonia, and nitrate plus nitrite concentrations all exceeded their pre-fire range again. Total nitrogen, ammonia, and nitrate plus nitrite surpassed the pre-fire maximum by one order of magnitude. This corresponds with a storm event that was minor relative to the first flush. Within the first year after the fire, other storm events of greater magnitude than this event occurred. However, this storm event probably flushed out a different portion of the burnt watershed than the minor storm event that mobilized the ammonia and total nitrogen. Two years after the fire the ammonia and nitrate plus nitrite concentrations returned to pre-fire range; no data is available for total nitrogen.

Dissolved and total organic carbon both exceeded maximum historical values post-fire. These trends are displayed in Figure 4.7 and Figure 4.8. Dissolved organic carbon concentration in the Salt River increased prior to the first flush. This was due to a minor storm event that mobilized dissolved organic carbon and caused the stream flow within the Salt River to temporarily increase during mid-July, 2002. When compared to the historical pre-fire maximum concentration, the post-fire dissolved organic carbon concentration was one order of magnitude higher. Unlike dissolved organic carbon, total organic carbon was affected by the first flush. A new maximum of 1140mg/L was reached, which exceeded the pre-fire maximum concentration 33mg/L. The first flush had a greater impact on the particulate fraction than the dissolved fraction. Also, dissolved organic carbon concentrations have returned to typical pre-fire levels two years post-fire; total organic carbon data does not exist for this time period.

Dissolved phosphorus had a unique post-fire trend. Prior to the Rodeo-Chediski fire, the dissolved phosphorus range was below the MRL-0.1mg/L, as shown in Figure 4.9. After the fire, the dissolved phosphorus concentration was 0.57mg/L. The concentration spike occurred during the minor storm event in mid-July 2002. This increase is attributed to the fire. In March 2004, a storm event comparable to the first flush took place. The dissolved phosphorus concentration of 0.17mg/L surpassed the pre-fire maximum again. At this time, the stream flow had increased to 30m^3/s from a base flow of 10m^3/s. This storm event probably flushed out a different portion of the burned watershed than the other previous storm events. After this storm event, the dissolved phosphorus concentration returned to pre-fire levels.

Total phosphorus concentration displayed a trend similar to total organic carbon one year post-fire. The total phosphorus maximum concentration jumped from 4mg/L to 39mg/L, as shown in Figure 4.10. It took a more significant rain event, the first flush, to cause an increase in total phosphorus concentration than dissolved phosphorus. This is because the total phosphorus concentration consists of the particulate and dissolved fractions and is governed by the particulate fraction. Two years after the fire, the total phosphorus concentrations decreased to pre-fire levels.

Table 4.2 Summary of nutrients and suspended sediment analyzed for the Salt River

Constituent	Time Period	Number of Samples	Minimum Concentration (mg/L)	Maximum Concentration (mg/L)	Average Concentration (mg/L)
Ammonia	Pre-fire	81	0.01 (MRL)	0.06	0.020
	Post-fire	21	0.0040	1.09	0.18
Total Nitrogen	Pre-fire	93	0.020	3.8	0.74
	Post-fire	12	0.84	220	52.8
Nitrate+Nitrite	Pre-fire	143	0.01 (MRL)	0.62	0.081
	Post-fire	20	0.013	2	0.42
Dissolved Organic Carbon	Pre-fire	34	0.90	4.9	2.16
	Post-fire	23	0.703	56.6	7.66
Total Organic Carbon	Pre-fire	102	0.9	33	5.42
	Post-fire	9	4.8	1140	228
Dissolved Phosphorus	Pre-fire	81	0.01 (MRL)	0.08	0.019
	Post-fire	20	0.0035	0.57	0.12
Total Phosphorus	Pre-fire	276	0.01 (MRL)	4	0.12
	Post-fire	23	0.006	39	3.89
Suspended Sediment	Pre-fire	224	1	11700	293
	Post-fire	32	1.6	42500	4050

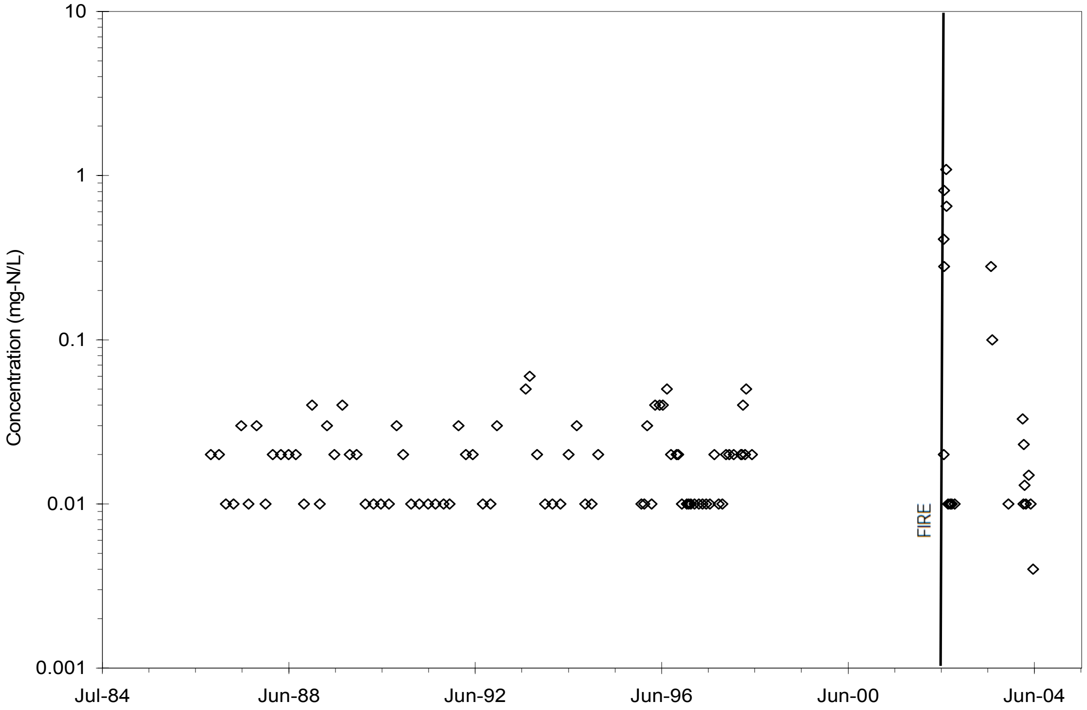

Figure 4.4 Ammonia concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

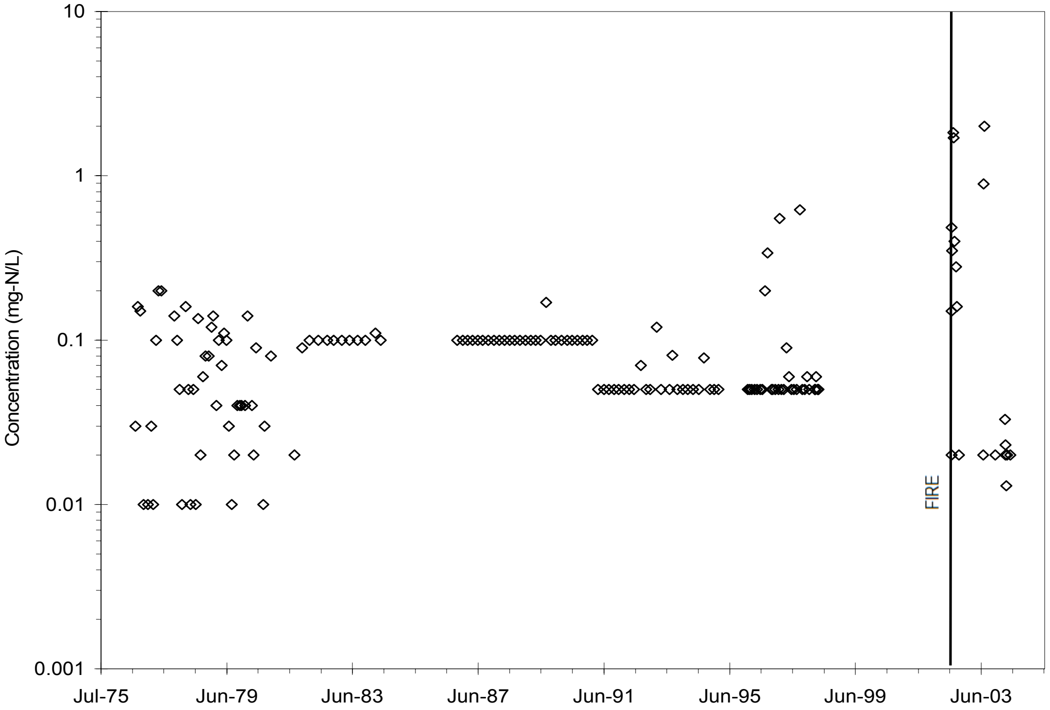

Figure 4.5 Nitrate plus nitrite concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

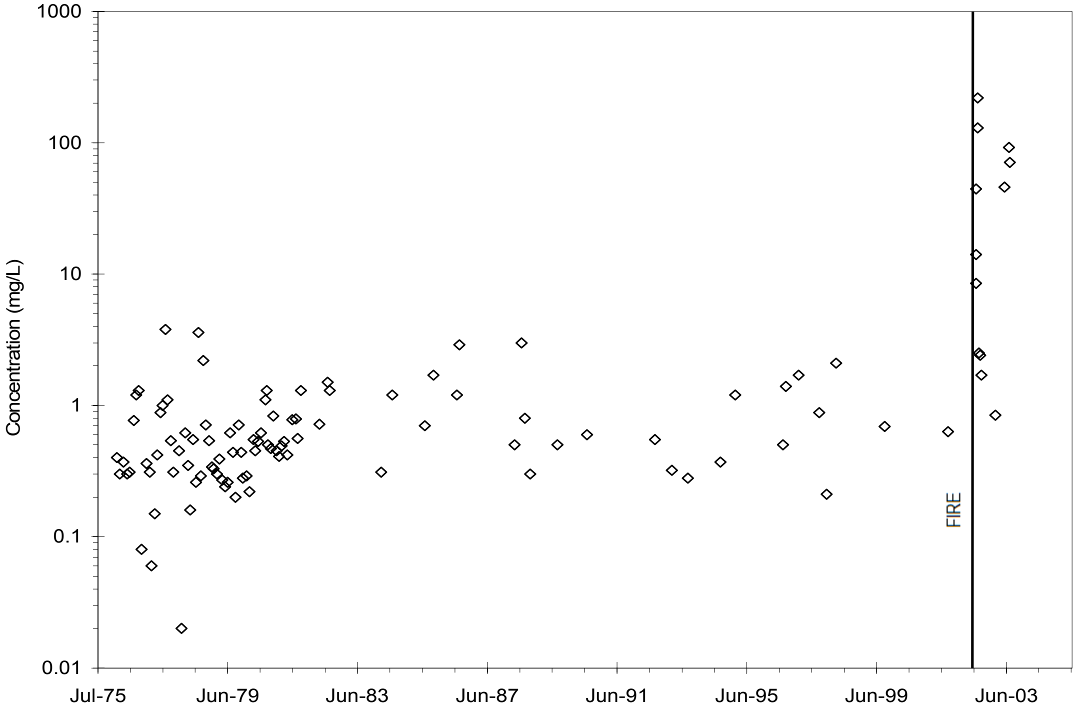

Figure 4.6 Total nitrogen concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

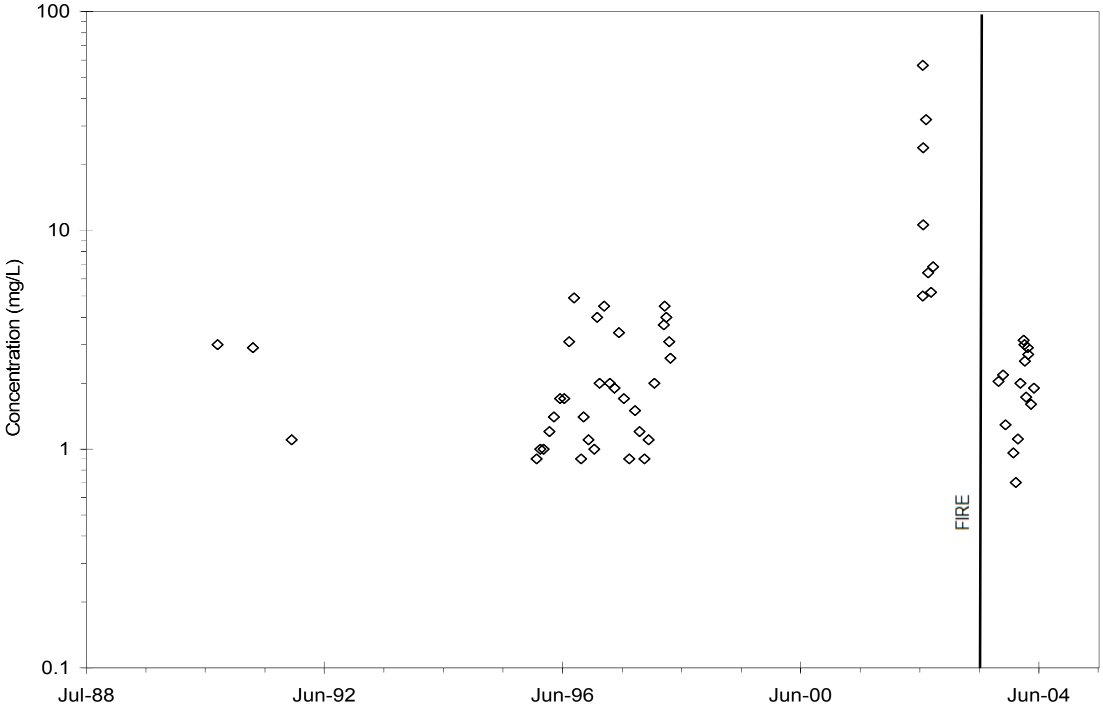

Figure 4.7 Dissolved organic carbon concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

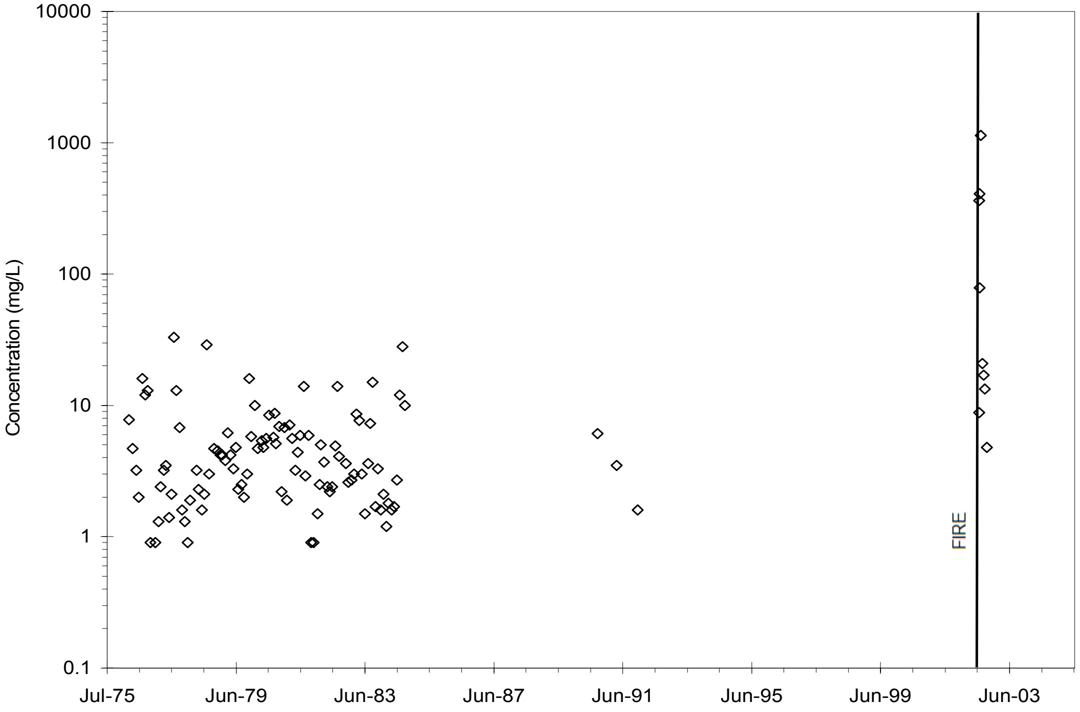

Figure 4.8 Total organic carbon in the Salt River above Roosevelt Lake at USGS gauging station 09498500

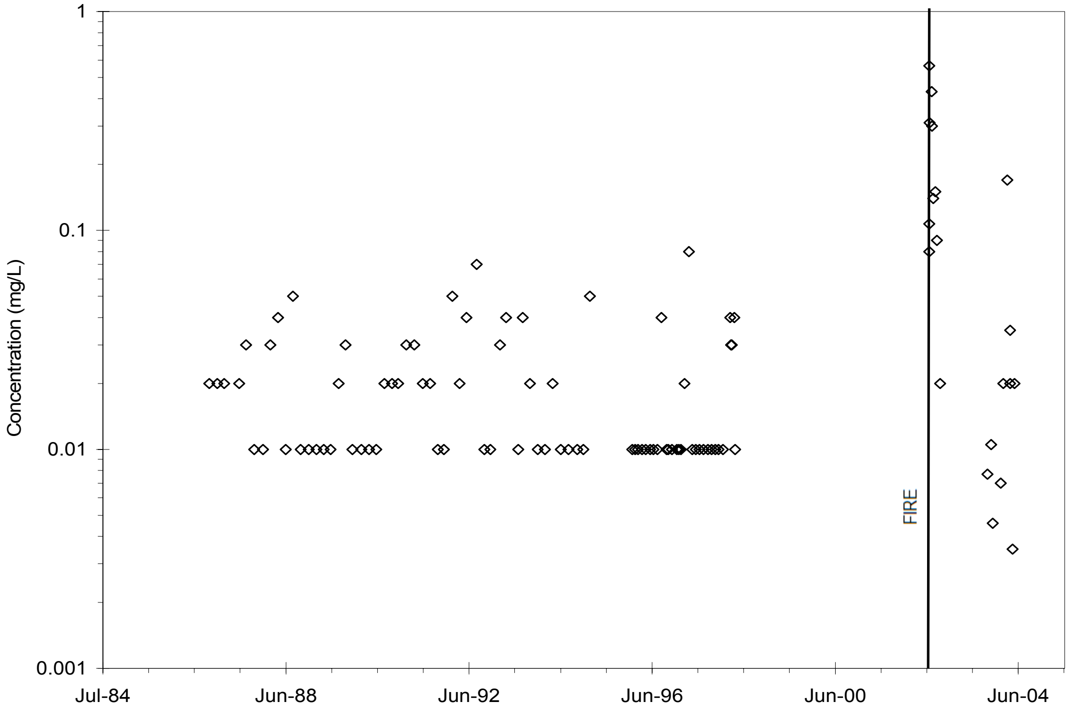

Figure 4.9 Dissolved phosphorus concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

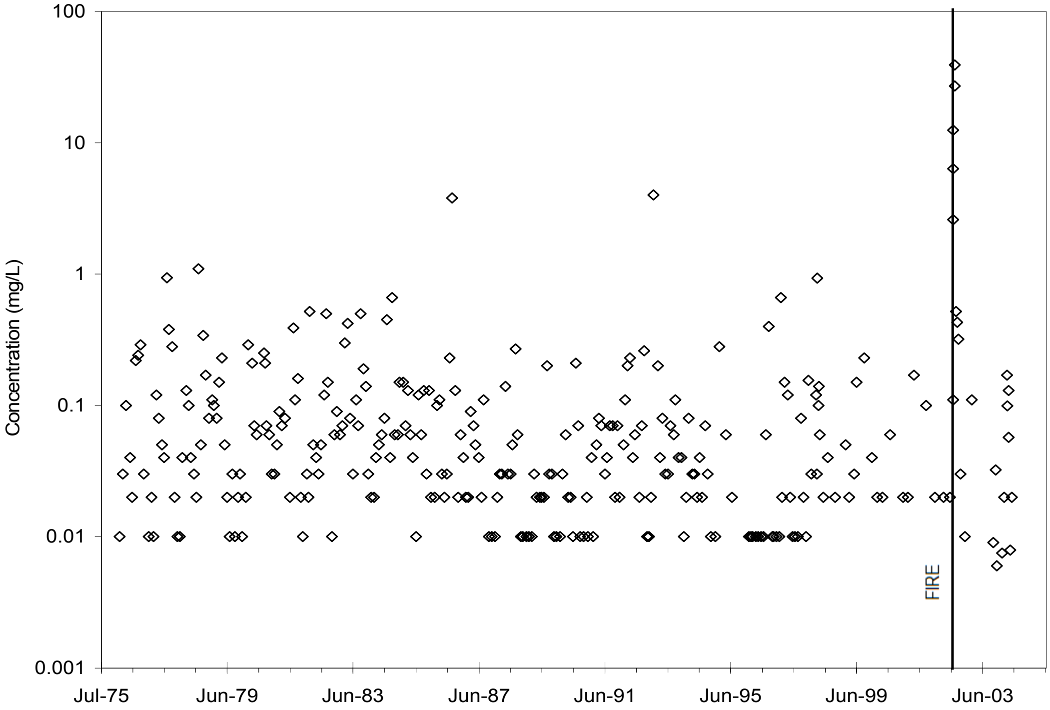

Figure 4.10 Total phosphorus concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

Metals

The Rodeo-Chediski fire impacted the particulate fraction of trace metals differently than the dissolved trace metal fraction. The pre- and post-fire dissolved fraction concentration stayed comparable, whereas the particulate fraction increased post-fire for all of the trace metals considered. Table 4.3 summarizes the concentration range and MCL for each metal analyzed, pre and post-fire.

The dissolved and particulate arsenic fractions were affected differently by the first flush. The dissolved arsenic concentration, shown in Figure 4.11, ranged from below the MRL-0.012mg/L. The post-fire dissolved arsenic concentration reached a maximum of 0.017mg/L during the first flush. During the first flush, runoff caused an increase in the particulate fraction of arsenic, as shown in Figure 4.12. Historically, the total arsenic concentration ranged from below the MRL-0.04mg/L. As a result of the first flush the concentration tripled to 0.13mg/L. The first flush transported more particulate arsenic into the Salt River than dissolved arsenic; therefore, the total arsenic concentration is dependent on the particulate fraction.

Like arsenic, the combination of the fire and the first flush caused an increase in the particulate fraction of barium and had minimal effect on the dissolved fraction. The dissolved barium concentration exceeded the historical range of 0.013mg/L-0.2mg/L during the first flush, as displayed in Figure 4.13. The new maximum of 0.23mg/L slightly surpassed the historical maximum concentration. The increase in the total barium concentration was much more noticeable. A post-fire concentration of 5.55mg/L was eight times greater than the maximum pre-fire concentration of 0.7mg/L. This data is shown in Figure 4.14. The trend of the first flush causing a significant increase in the concentration of the particulate fraction and having minimal impact on the concentration of the dissolved fraction was consistent for barium.

Copper concentration was minimally impacted by the Rodeo-Chediski fire and the first flush. There was no significant mobilization of the dissolved fraction, this is clearly shown in Figure 4.15. When compared to pre-fire data, dissolved copper concentrations after the fire remained within the historical range, below the MRL-0.03mg/L. Unlike the dissolved copper fraction, the particulate fraction showed a decreasing trend since the early 1990's. Relative to the historic data, the total copper concentration remained below the maximum concentration of 0.7mg/L, as shown in Figure 4.16. However, when compared to the decreasing copper concentration since the early 1990's, the first flush caused a noticeable increase in the particulate fraction of copper within the Salt River. Overall, the copper concentration within the Salt River was not significantly altered by the Rodeo-Chediski fire.

The post-fire dissolved and total iron concentrations were comparable to pre-fire levels. The dissolved iron fraction in the Salt River, shown in Figure 4.17, remained within the historical range of below the MRL-0.5mg/L. The concentration of the particulate fraction remained within the background range of 0.02mg/L-110mg/L. However, one year after the fire, the total iron concentration was elevated, the source is unknown. This data is shown in Figure 4.18.

While the post-fire dissolved lead concentration remained below detection limits, the total lead concentration increased, shown in Figure 4.19 and 4.20. The pre-fire dissolved lead concentration range was below the MRL-0.1mg/L. Any post-fire fluctuations in concentration were not noticeable since the concentration was less than the MRL. The historical range for total lead concentration is from below the MRL-0.11mg/L. The total lead concentration was nearly 400% higher than the historical maximum concentration during the first flush. The increase in the total concentration of lead is consistent with the mobilization of particulates after the fire. The particulate fraction has a greater influence than the dissolved fraction on the total lead concentration.

The dissolved fraction of manganese achieved a maximum post-fire concentration during the first flush. Dissolved manganese concentration has been on a rapidly decreasing trend since 1993, as shown in Figure 4.21. The initial increase in dissolved manganese concentration was the result of a leaking acid pond used in mining operations. A pump and treat method was used to remediate the water, therefore, dissolved manganese concentrations decreased. Prior to this decrease, the historical maximum concentration was 4.9mg/L. During the first flush, the highest dissolved manganese concentration measured was 0.126mg/L. Although the full extent of the effect of the fire and the first flush on the dissolved fraction cannot be determined due to the contamination, when compared to the decreasing dissolved concentration since 1993, the first flush did mobilize dissolved manganese.

The concentration of total manganese exceeded the historical record high concentration, as shown in Figure 4.22. In the past, the highest total manganese concentration was 5.2mg/L. During the first flush, the total manganese concentration in the Salt River was elevated by one order of magnitude. Unlike the dissolved manganese, the effect of the forest fire and the first flush on the on the particulate fraction is evident. In addition, the post-fire total manganese concentration was significantly influenced by the particulate fraction, a trend that is consistent with the elevated post-fire suspended sediment concentration.

Mercury mobilization was minimal after the fire. Historically, dissolved mercury concentration in the Salt River remained below the MRL. The post-fire dissolved mercury concentration remained below detection limits, shown in Figure 4.23. The total mercury concentration showed a response to the fire which is displayed in Figure 4.24. This response is unique when compared to the other trace metals. It occurred in mid-July, not in early August during the first flush; in mid-July, the stream flow increased from $2m^3/s$ to $12\ m^3/s$. The post-fire mercury concentration reached 0.3μg/L, when compared to the historical maximum concentration of 5.6μg/L achieved at a stream flow of $7m^3/s$, the response was minute. Although the slight increase in the particulate fraction of the mercury concentration was not due to the first flush, the fire did cause the release of mercury from vegetation and plant litter.

The particulate and dissolved fractions of selenium were mobilized during the first flush, as shown in Figure 4.25 and 4.26. The dissolved concentration of 2mg/L exceeded the MDL of 1mg/L. Dissolved selenium is the only metal to exceed the MCL of 0.05mg/L established by the EPA. Total selenium concentration was elevated post-fire; however it did not exceed the historical maximum concentration. After the first flush, the selenium concentrations were at or below the MDL.

Post-fire, the total zinc and dissolved zinc concentrations displayed different trends. Dissolved zinc was not mobilized as a result of the fire. The dissolved fraction of zinc remained within the historical range of 1μg/L-60μg/L;

this data is shown in Figure 4.27. However, the maximum post-fire dissolved zinc concentration of 20µg/L was achieved during the first flush. The total zinc concentration of 1,170µg/L exceeded the historical value of 450µg/L during the first flush, as shown in Figure 4.28. Before the Rodeo-Chediski fire the particulate fraction dominated the total zinc concentration. After the fire, the same trend was followed, however, to a greater extent.

Table 4.3 Summary of trace metals analyzed for the Salt River

Constituent	MCL (mg/L)	Time Period	Number of Samples	Minimum	Maximum	Average
Dissolved Arsenic	0.05	Pre-fire	147	0.001 (MRL)	0.012	0.0044
		Post-fire	24	0.0020	0.017	0.01
Total Arsenic		Pre-fire	200	0.001 (MRL)	0.038	0.0053
		Post-fire	8	0.015	0.13	0.05
Dissolved Barium	2	Pre-fire	146	0.013	0.2	0.07
		Post-fire	9	0.087	0.23	0.2
Total Barium		Pre-fire	76	0.027	0.7	0.15
		Post-fire	9	0.15	5.55	1.8
Dissolved Copper	1.3[†]	Pre-fire	147	0.001 (MRL)	0.03	0.004
		Post-fire	9	0.002 (MRL)	0.006	0.003
Total Copper		Pre-fire	224	0.001 (MRL)	0.70	0.026
		Post-fire	9	0.004 (MRL)	0.33	0.10
Dissolved Iron	0.3[*]	Pre-fire	244	0.001 (MRL)	0.5	0.03
		Post-fire	9	0.002	0.021	0.01
Total Iron		Pre-fire	192	0.020	110	3.25
		Post-fire	9	2.28	69.1	23.7
Dissolved Lead	.015[†]	Pre-fire	159	0.001 (MRL)	0.1	0.01
		Post-fire	9	0.002 (MRL)	0.002	0.002
Total Lead		Pre-fire	155	0.001 (MRL)	0.20	0.03
		Post-fire	9	0.004	0.56	0.17

(continued)

Table 4.3 (continued)

Constituent	MCL (mg/L)	Time Period	Number of Samples	Minimum	Maximum	Average
Dissolved Manganese	.05*	Pre-fire	242	0.003	4.9	0.38
		Post-fire	9	0.0010	0.13	0.020
Total Manganese		Pre-fire	197	0.01 (MRL)	5.2	0.49
		Post-fire	9	0.1930	37.8	9.04
Dissolved Mercury	0.002	Pre-fire	59	0.0001 (MRL)	0.0013	0.0001
		Post-fire	9	0.0001 (MRL)	0.0001	0.0001
Total Mercury		Pre-fire	209	0.001 (MRL)	0.0056	0.0002
		Post-fire	9	0.0001 (MRL)	0.0003	0.0002
Dissolved Selenium	0.05	Pre-fire	149	1 (MRL)	1 (MRL)	1.0
		Post-fire	13	1 (MRL)	2	1.1
Total Selenium		Pre-fire	227	1 (MRL)	17.8	1.1
		Post-fire	15	1 (MRL)	3	1.5
Dissolved Zinc	5*	Pre-fire	185	0.001 (MRL)	0.06	0.01
		Post-fire	9	0.002 (MRL)	0.02	0.01
Total Zinc		Pre-fire	214	0.001 (MRL)	0.45	0.03
		Post-fire	9	0.017	1.17	0.36

*Secondary MCL

†Action Level: the level which, if exceeded, triggers treatment or other requirements that a water system must follow (EPA 2003)

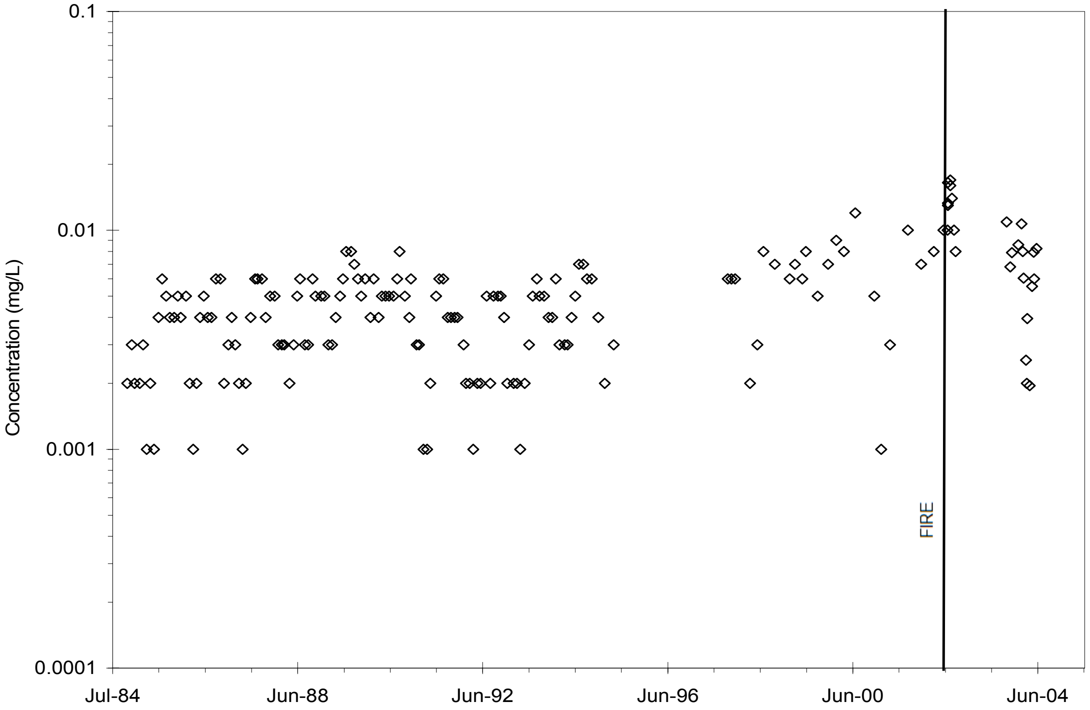

Figure 4.11 Dissolved arsenic concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

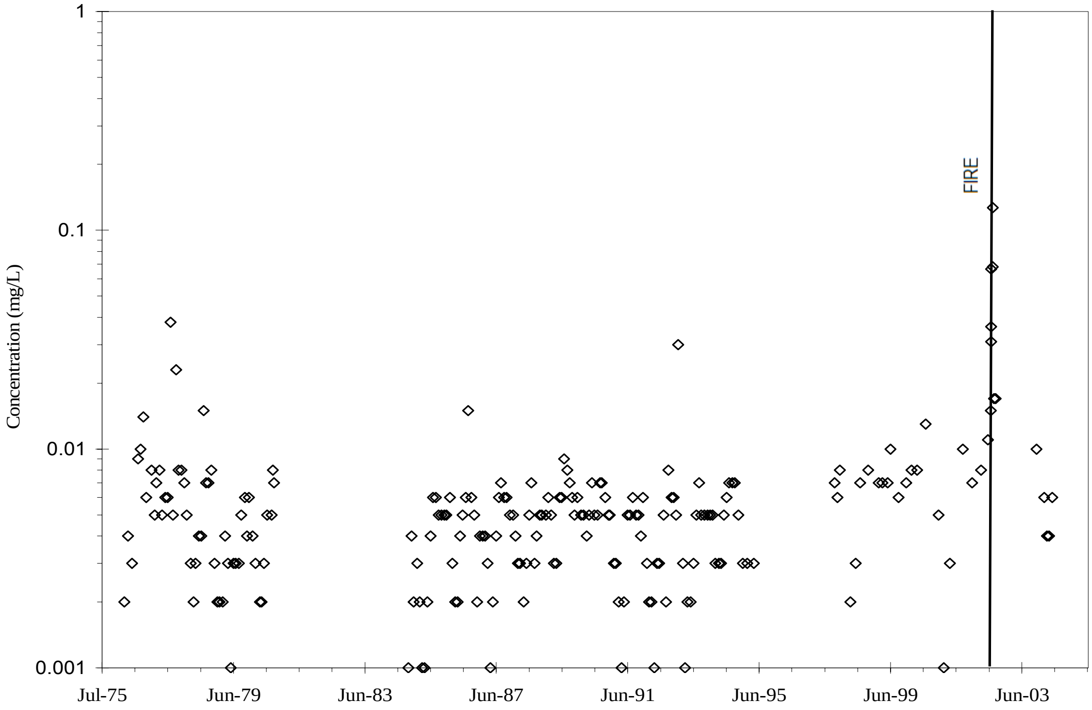

Figure 4.12 Total arsenic concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

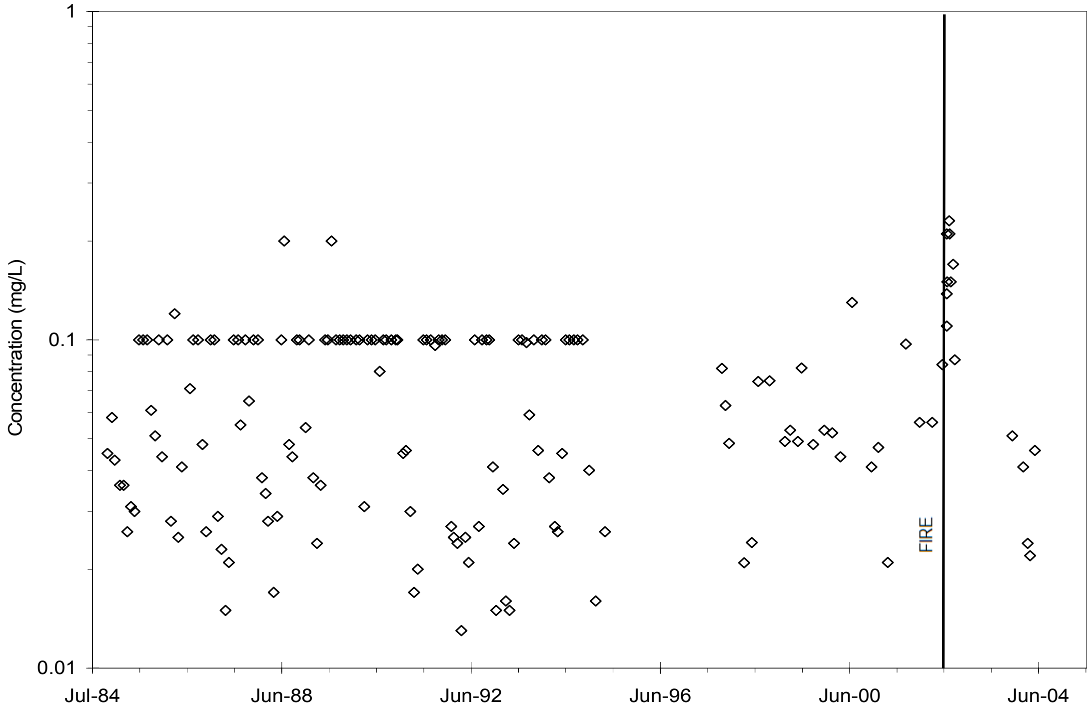

Figure 4.13 Dissolved barium concentration above Roosevelt Lake at USGS gauging station 09498500

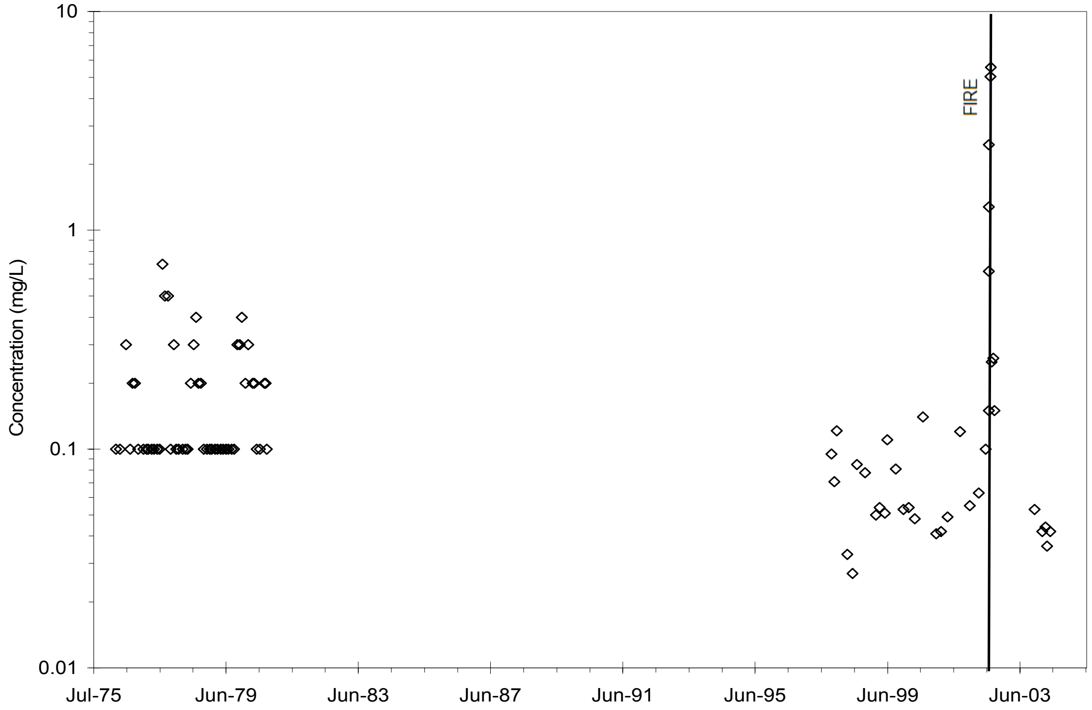

Figure 4.14 Total barium concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

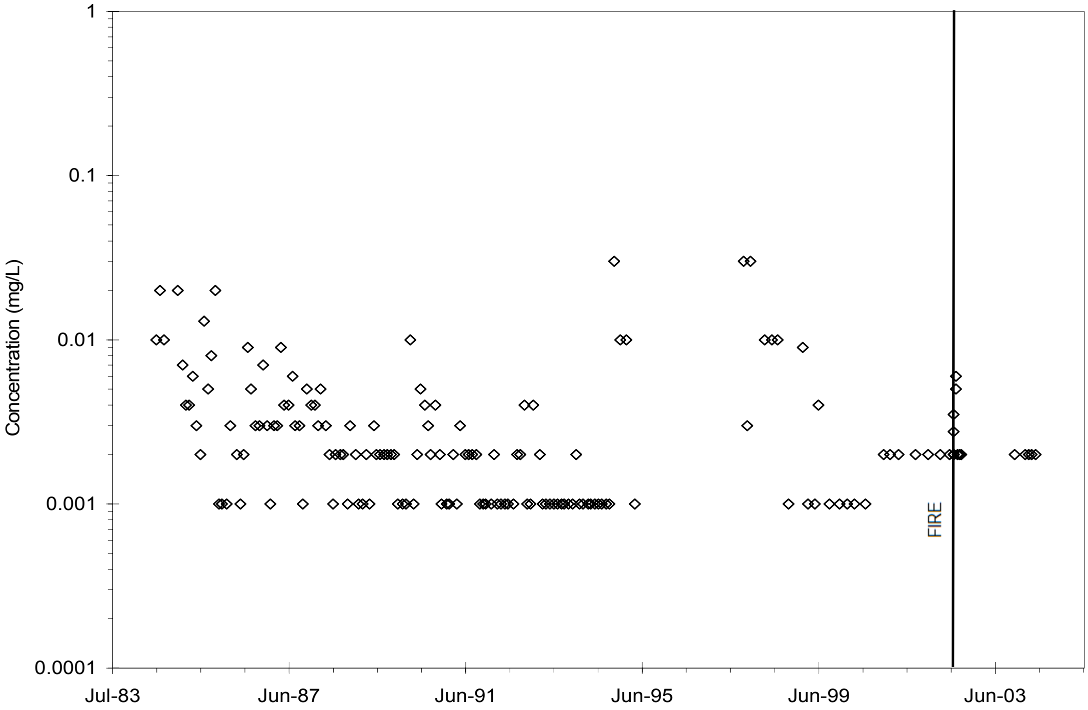

Figure 4.15 Dissolved copper concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

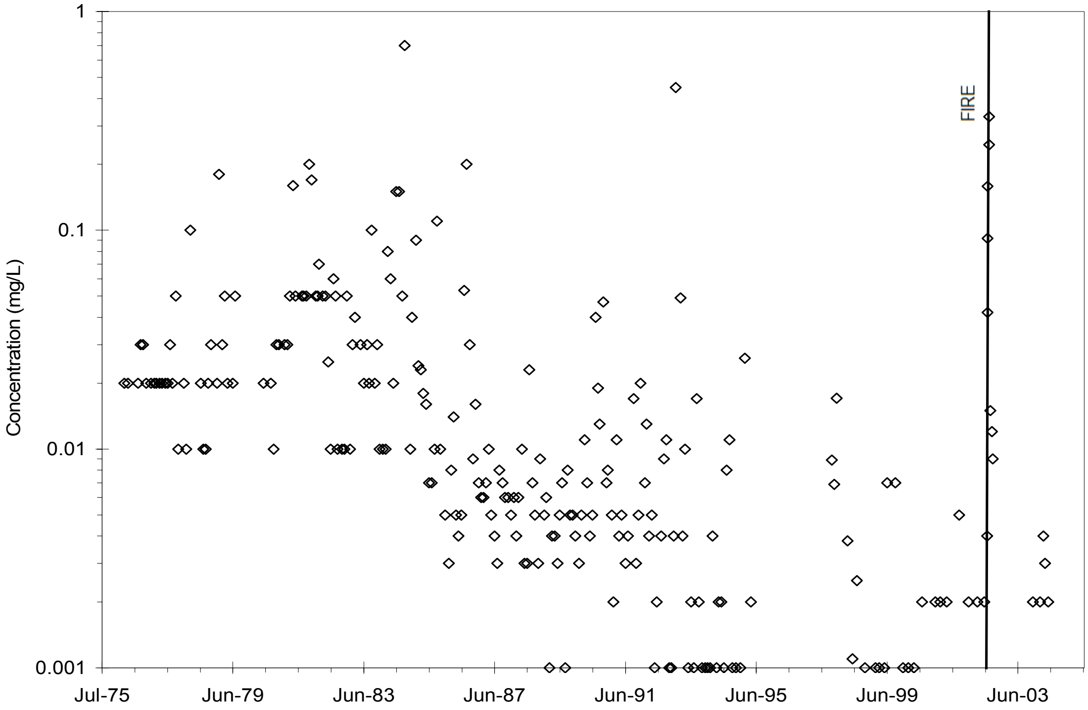

Figure 4.16 Total copper concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

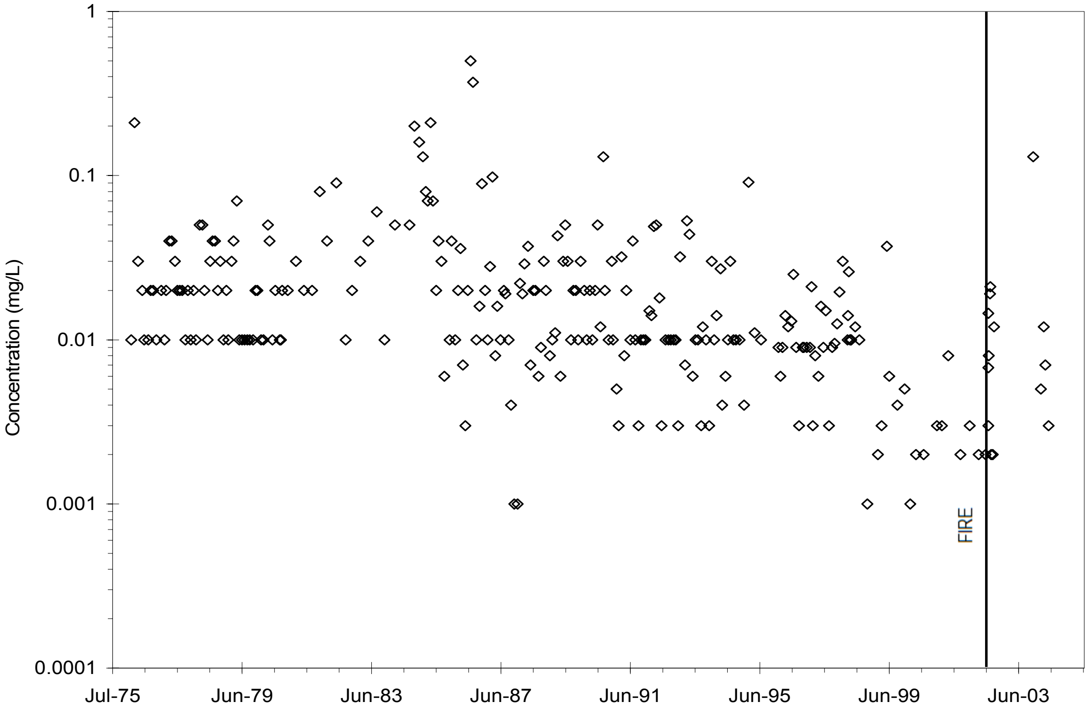

Figure 4.17 Dissolved iron concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

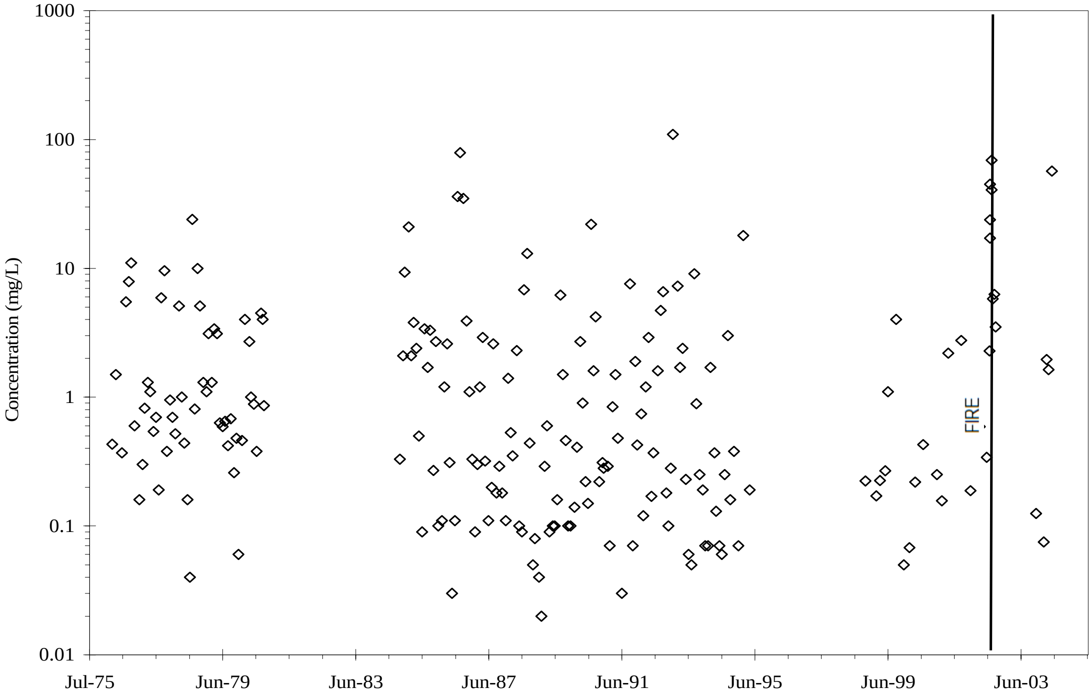

Figure 4.18 Total iron concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

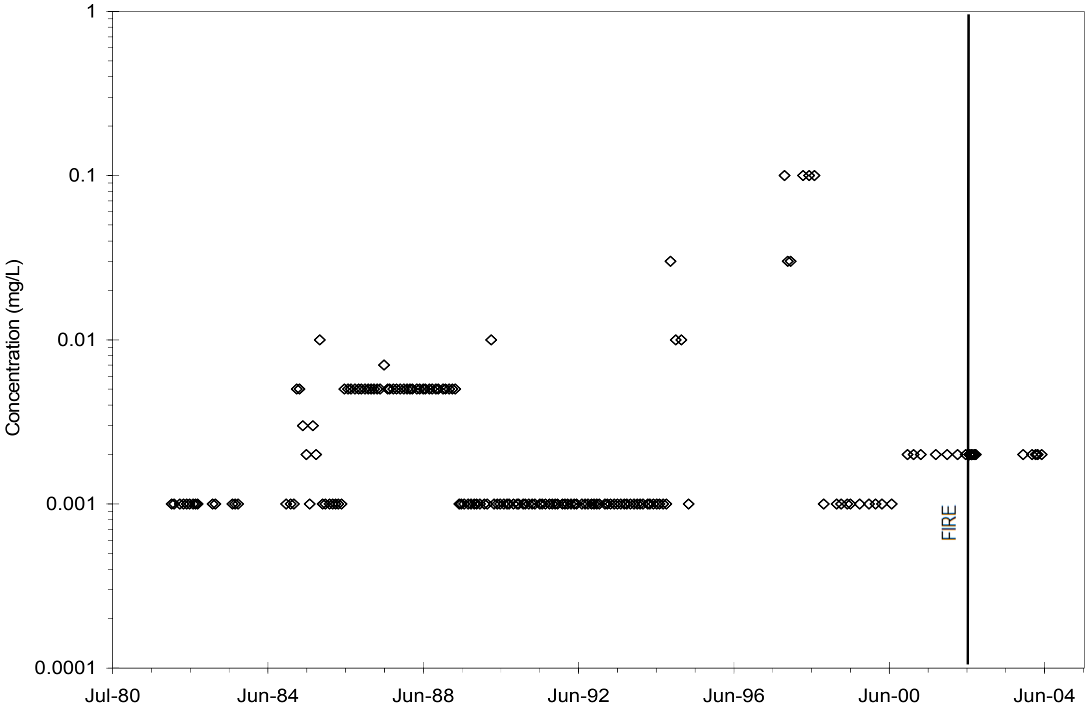

Figure 4.19 Dissolved lead concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

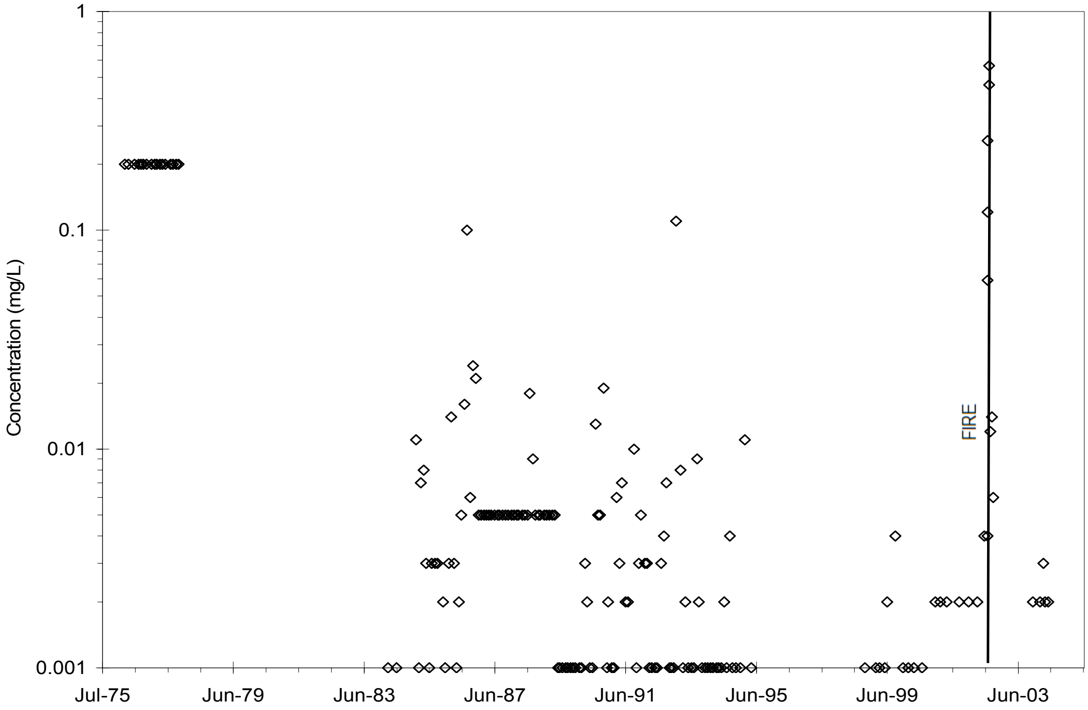

Figure 4.20 Total lead concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

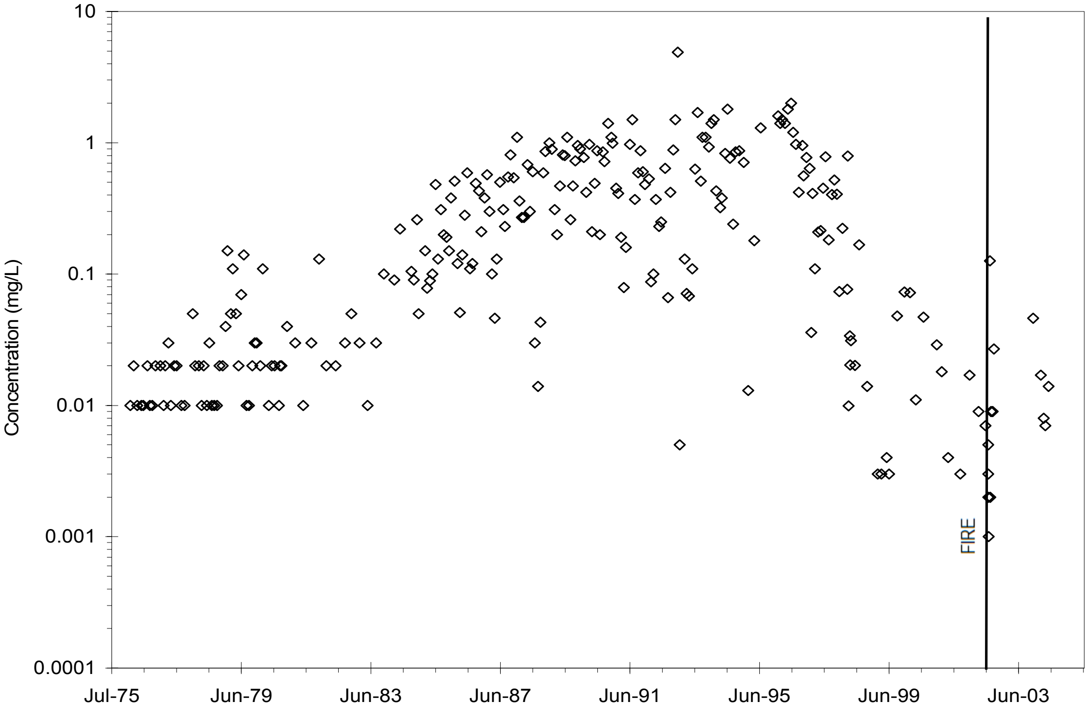

Figure 4.21 Dissolved manganese concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

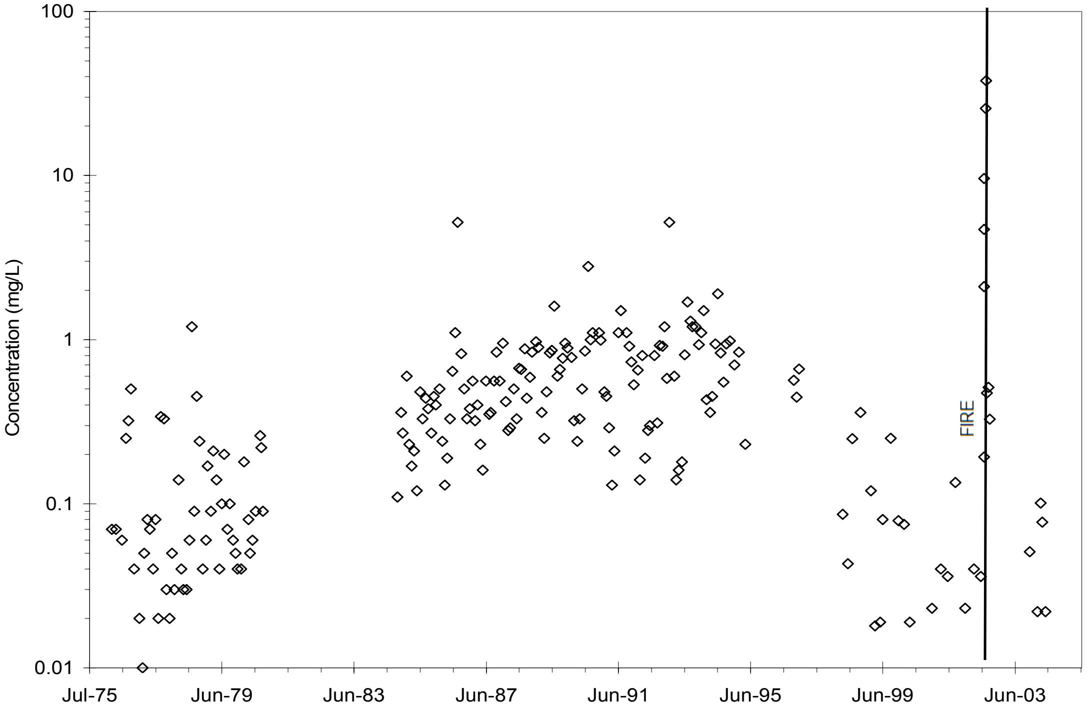

Figure 4.22 Total manganese concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

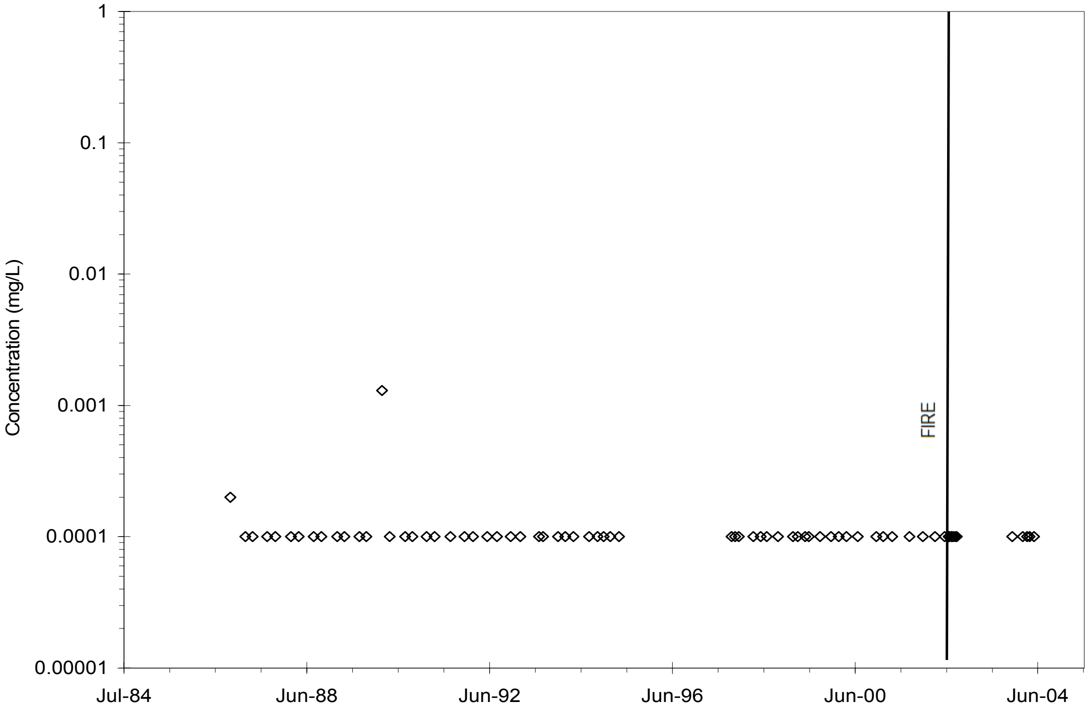

Figure 4.23 Dissolved mercury concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

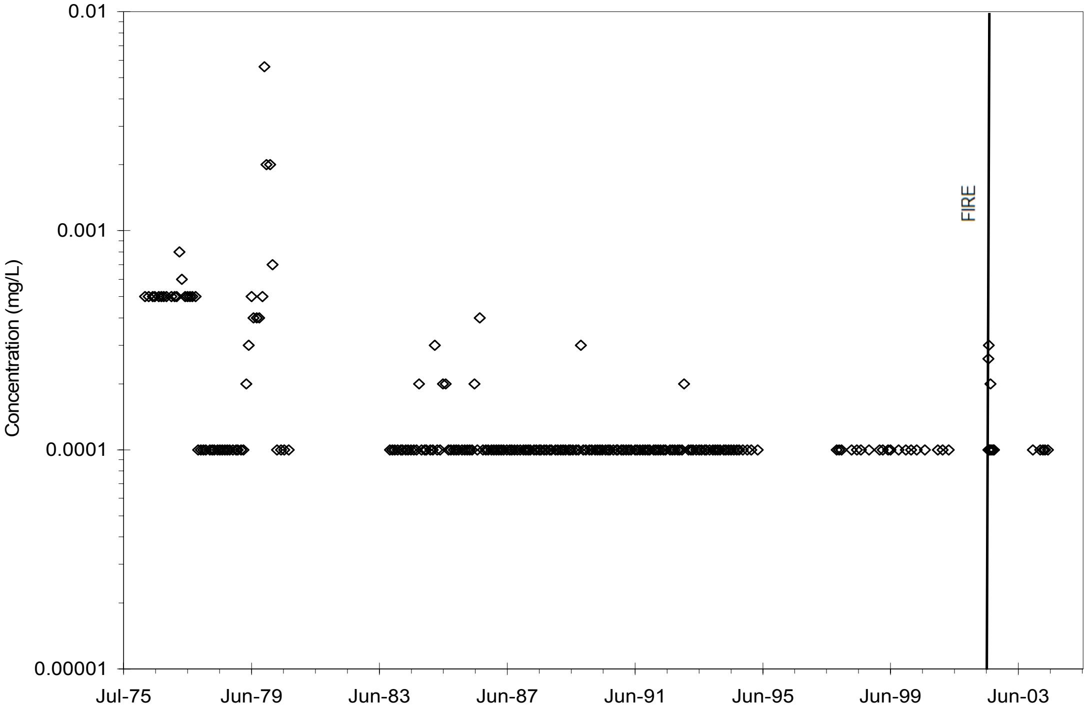

Figure 4.24 Total mercury concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

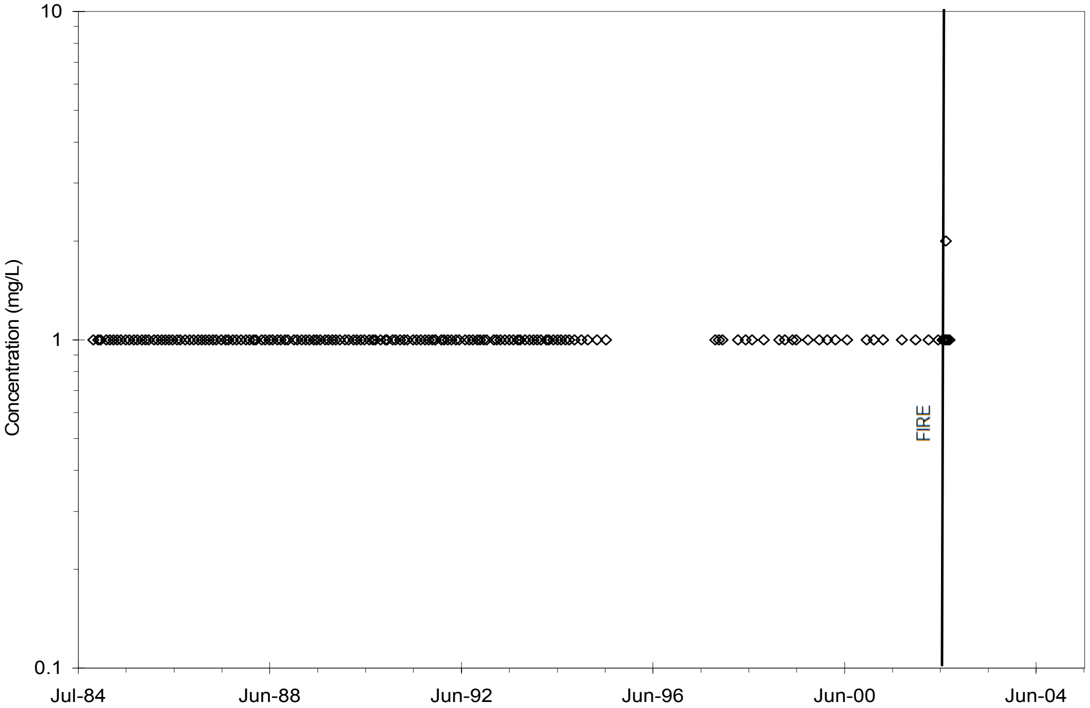

Figure 4.25 Dissolved selenium concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

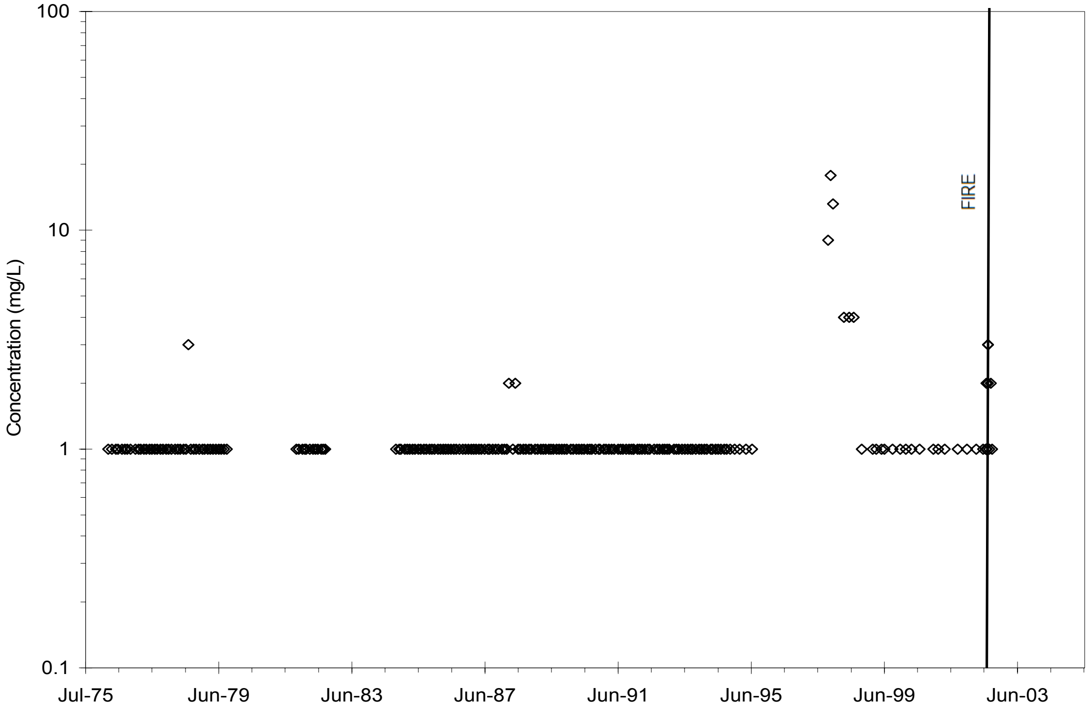

Figure 4.26 Total selenium concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

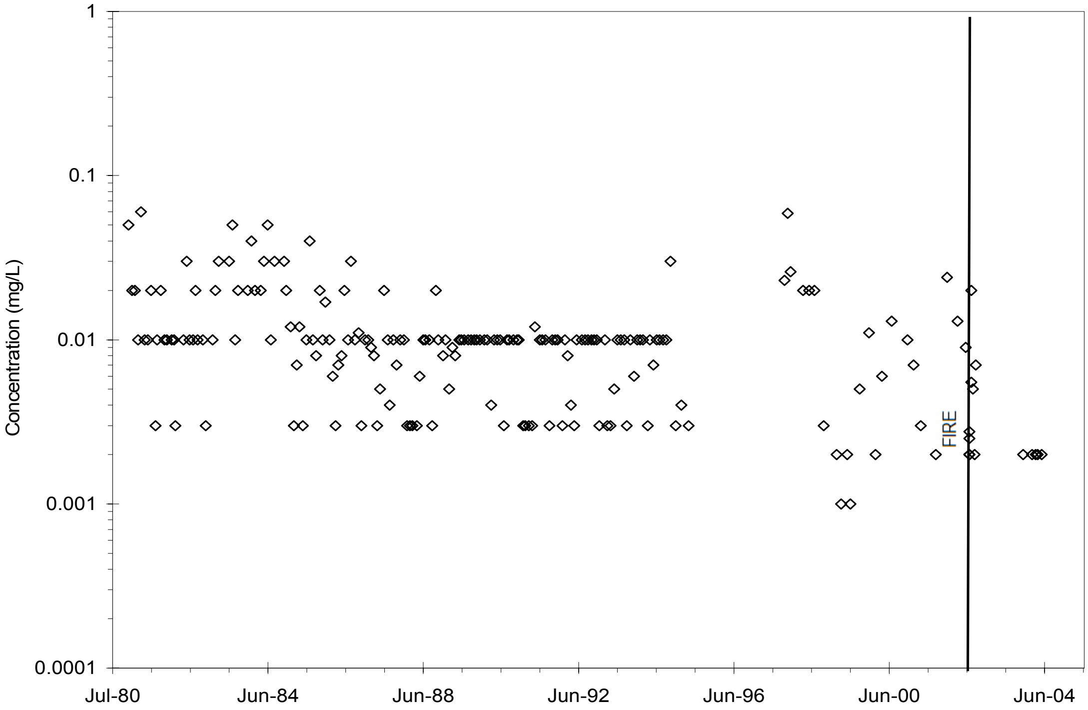

Figure 4.27 Dissolved zinc concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

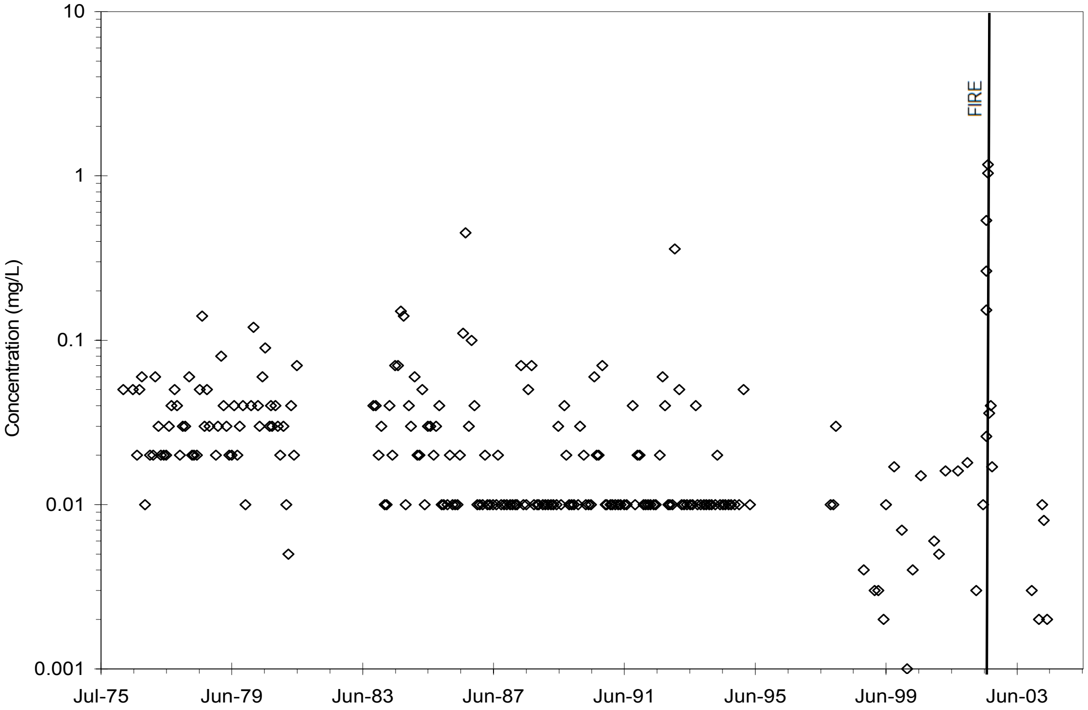

Figure 4.28 Total zinc concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

Major Ions

Cations

Major cations were not significantly impacted by fire or the first flush. Post-fire concentrations of potassium and magnesium were consistent with pre-fire trends as shown in Figures 4.29 and 4.30. Calcium and sodium displayed a minimal response to the fire as shown in Figures 4.31 and 4.32.

Potassium and magnesium post-fire concentrations remained within the limits of background levels. The highest post-fire concentrations achieved by potassium and magnesium were 26mg/L and 45mg/L, respectively. These concentrations were reached when the stream flow in the Salt River increased from 2m^3/s to 12 m^3/s during mid-July, not during the first flush. Overall, the effect of the fire and the first flush were negligible for the potassium and magnesium concentration.

Calcium and sodium concentrations were elevated after the Rodeo-Chediski fire. Like potassium and magnesium, sodium reached a post-fire maximum concentration of 1270mg/L in mid-July. This concentration surpasses the historical maximum of 1150mg/L. Unlike the other major cations analyzed, calcium achieved a maximum concentration during the first flush. The 130mg/L post-fire calcium concentration exceeded the historical maximum of 110mg/L. Although sodium and calcium showed post-fire responses at different times, both were impacted by the fire. Two years post-fire the calcium and sodium concentrations have returned to pre-fire levels.

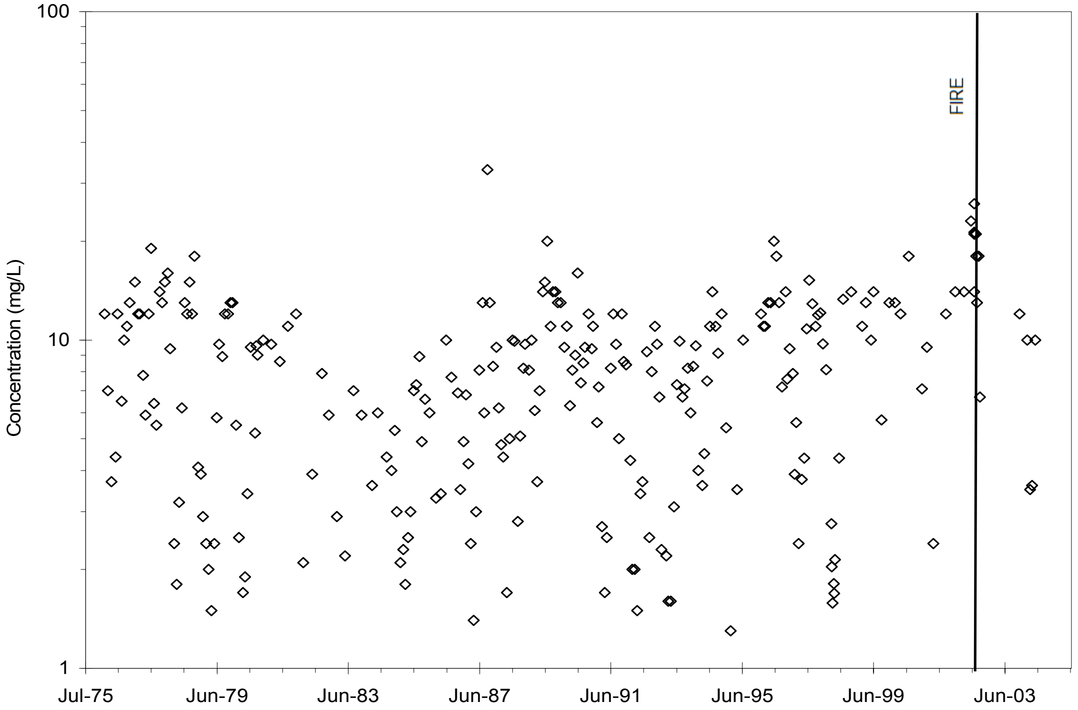

Figure 4.29 Potassium concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

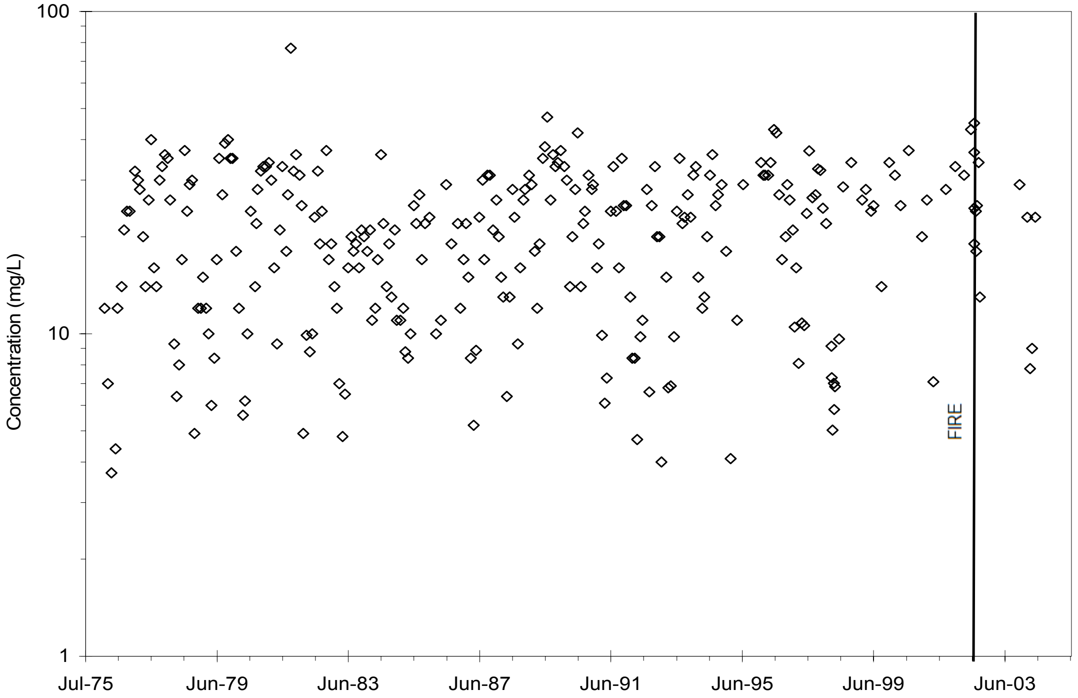

Figure 4.30 Magnesium concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

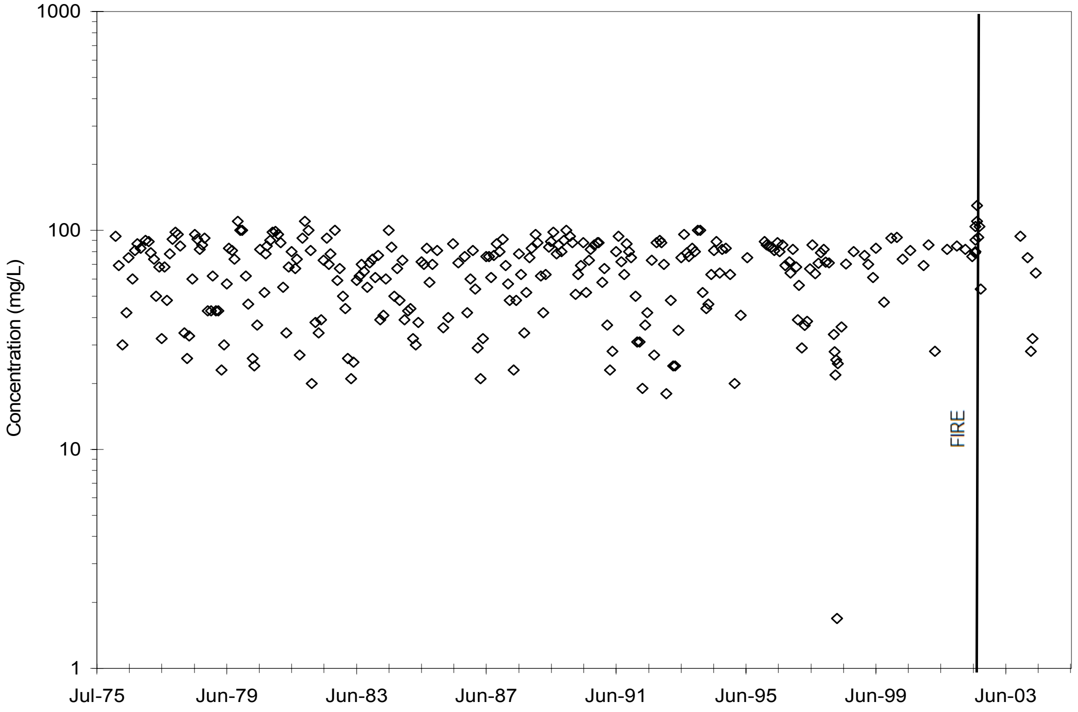

Figure 4.31 Calcium concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

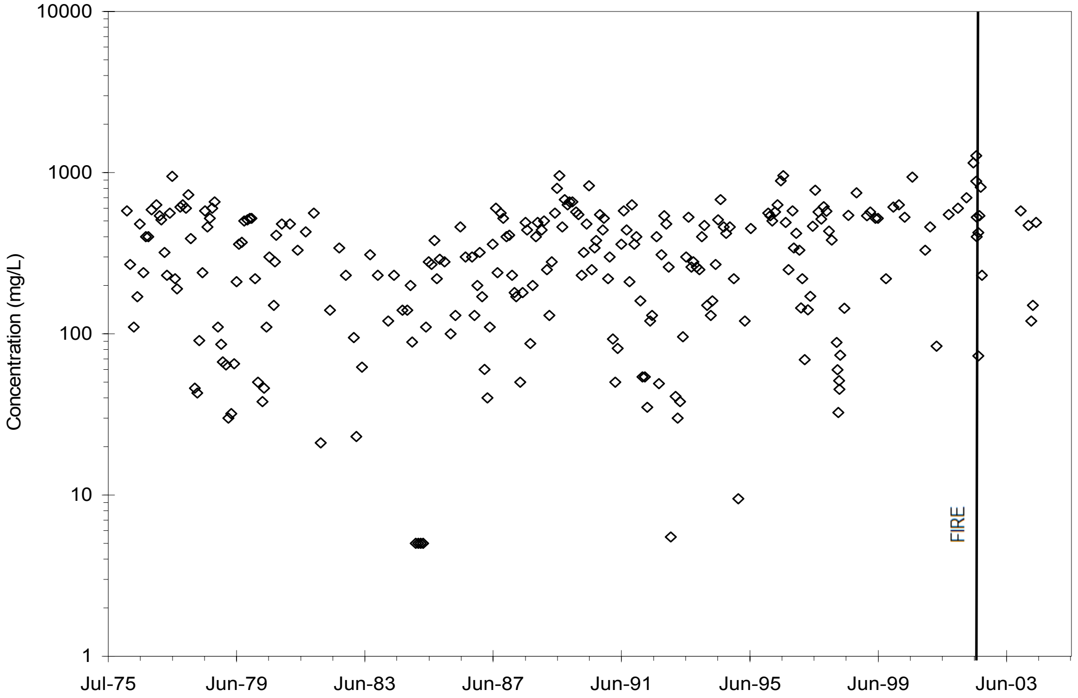

Figure 4.32 Sodium concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

Anions

Post-fire bicarbonate and chloride concentrations were minimally impacted by the fire, as shown in Figure 4.33 and 4.34. The first flush increased the bicarbonate concentration in the Salt River to 286mg/L, which surpassed the historical maximum of 254mg/L. The increase in concentration of chloride occurred during mid-July when stream flow rose from 2m^3/s to 12 m^3/s. Post-fire chloride concentration was 2110mg/L, which exceeded the pre-fire maximum concentration of 1900mg/L. Although both anions did exceed their pre-fire maximum concentration, the impact of the fire was minimal.

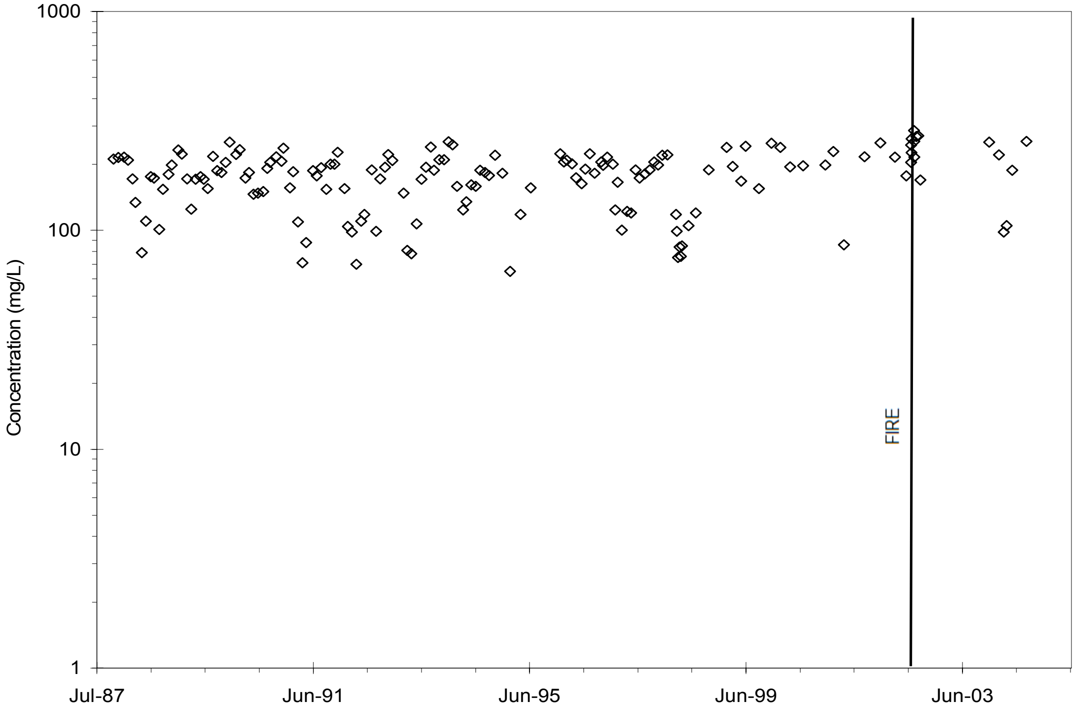

Figure 4.33 Bicarbonate concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

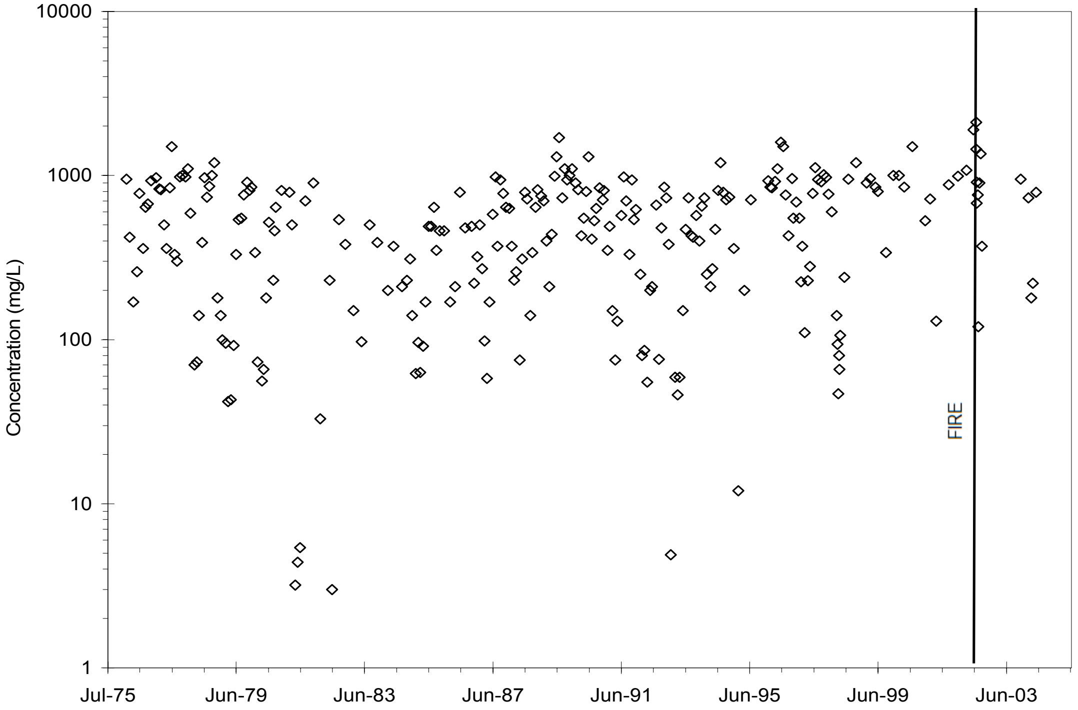

Figure 4.34 Chloride concentration in the Salt River above Roosevelt Lake at USGS gauging station 09498500

VERDE RIVER FLOW RATE DATA

The Verde River watershed is adjacent to the Salt River watershed and is comparable in size; the Verde River watershed drains approximately 14,000km^2. The stream flow trends in the Verde River are shown in the hydrograph in Figure 4.35. The hydrograph begins in 1945 and ends in 2004. The maximum and minimum stream flows recorded for the Verde River are 3,113m^3/s and 1.6m^3/s, respectively. During the eight-year, the maximum stream flow was 450m^3/s and the average stream flow was 9.6m^3/s. Prior to 1996, the average stream flow was 17m^3/s. The drought allows for the accumulation of soluble materials near the sediment surface; therefore, when a storm event occurs, these materials are readily mobilized.

Unlike the Salt River watershed, the Verde River watershed has not endured any catastrophic forest fires. However, the watershed is in a drought. The Verde River watershed will be used for comparison and represent the unburned watershed with drought conditions.

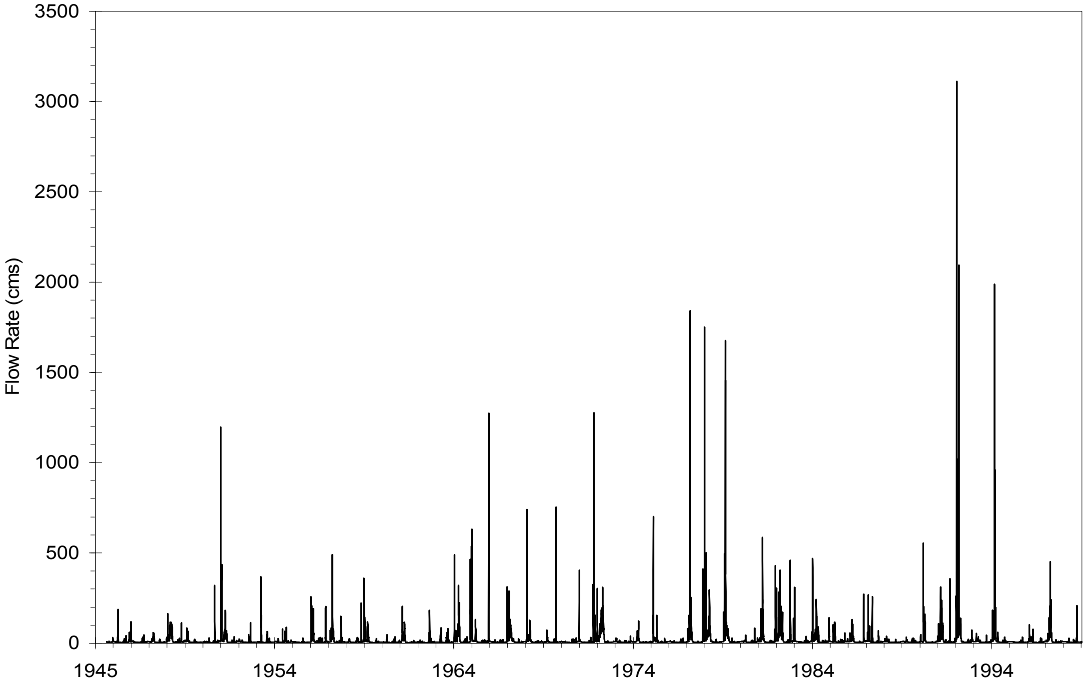

Figure 4.35 Verde River hydrograph

VERDE RIVER CONCENTRATION DATA ANALYSIS

Concentration data for the Verde River were analyzed to determine the effects of the drought water quality. The constituents analyzed for the Verde River are listed in Table 4.4. The pre- and post-fire concentration trends were compared to the Salt River. Therefore, the effects of the fire on water quality were distinguishable from drought effects.

Table 4.4 List of constituents analyzed for the Verde River

Constituents	Pre 2002	2002-2003	2003-2004	Data collected by ASU
Nutrients				
Ammonia	X	X	X	X
Dissolved Organic Carbon	X	X	X	X
Total Nitrogen	X	X	X	X
Total Phosphorus	X	X	X	X
Other				
Suspended Sediment	X	X	X	
Stream Discharge	Continuous data			

Suspended Sediment

Since the summer of 2002, suspended sediment concentrations in the Verde River have remained steady, as shown in Figure 4.36. Historically, the concentration varied three orders of magnitude. Since the summer of 2002, the concentration range has remained narrow, varying from 14mg/L-62mg/L. This narrow range is uncharacteristic for the Verde River. In a typical year, the suspended sediment concentration will range from 1mg/L-1,000mg/L. The lower, steadier concentrations are probably due to the drought since less runoff is available to mobilize sediment.

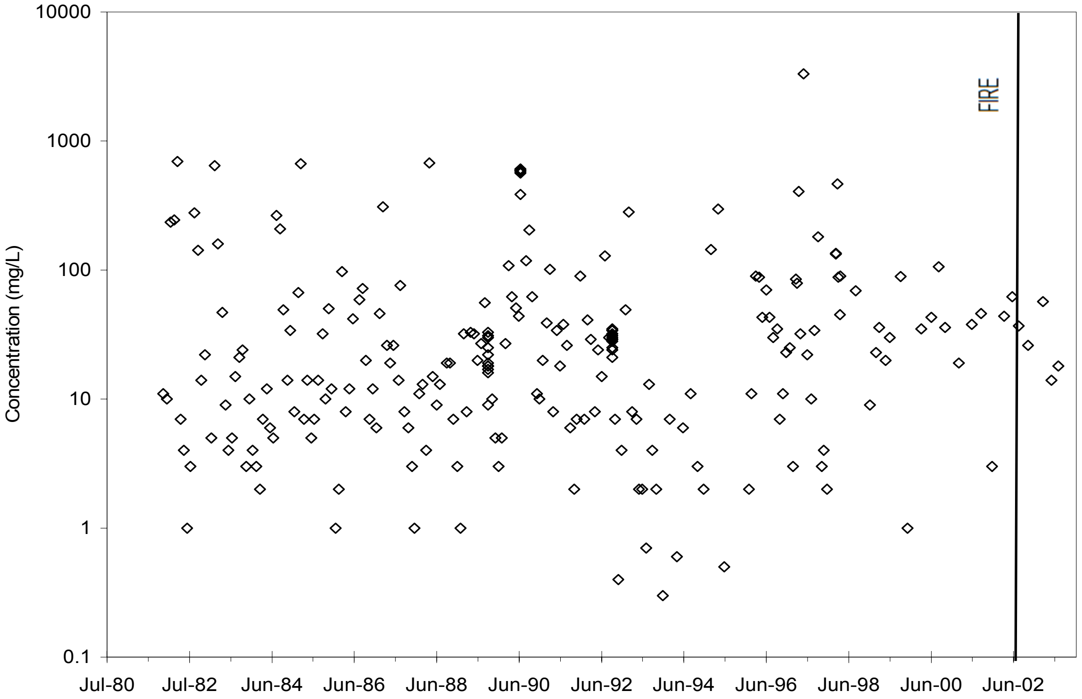

Figure 4.36 Suspended sediment concentration in the Verde River above Horseshoe Lake at USGS gauging station 09508500

Nutrients

The ammonia concentration within the Verde River has consistently maintained a range of 0.001mg-N/L-0.1mg-N/L, as shown in Figure 4.37. This trend was sustained after the Rodeo-Chediski fire. However, in December 2002, a pulse of ammonia entered the Verde River. The concentration reached 3.9mg-N/L; this surpassed the historical value by more than one order of magnitude. Within two weeks the concentration returned to background levels. The drought is likely the reason for the build up of ammonia within the watershed. Although base flow conditions were maintained within the Verde River in December 2002, a small rain event probably mobilized the ammonia.

The total nitrogen concentration data is not continuous, as shown in Figure 4.38. Initially, data was collected from 1980-83; the data collection then resumed in 1999. During the initial sampling period, the total nitrogen concentration range was 0.3mg/L-3.3mg/L. Currently, the concentration range has shifted to 0.04mg/L-0.23mg/L. However, a common trend exists; the highest concentrations of ammonia are reached during August and September. Since 1999, the total nitrogen concentration has surpassed the upper limits of the concentration range during these months. Since these elevated concentrations occur on a yearly basis and started prior to any drought conditions, this is simply a characteristic of the Verde River.

The post-fire dissolved organic carbon concentration has remained at background levels, as shown in Figure 4.39. However, one year before the Rodeo-Chediski fire the dissolved organic carbon concentration exceeded historical values by a factor of three. This is due to a minor rain event that mobilized the dissolved organic carbon build-up caused by the drought. Typically, a direct correlation exists between flow rate and dissolved organic carbon concentration in the Verde River.

Total organic carbon data is limited for the Verde River. The data collection began in 1981 and ceased in 1984, as shown in Figure 4.40. A decreasing trend in the total organic carbon concentration occurred during the sample collection time period. However, due to the inadequate data the impacts of the drought cannot be determined.

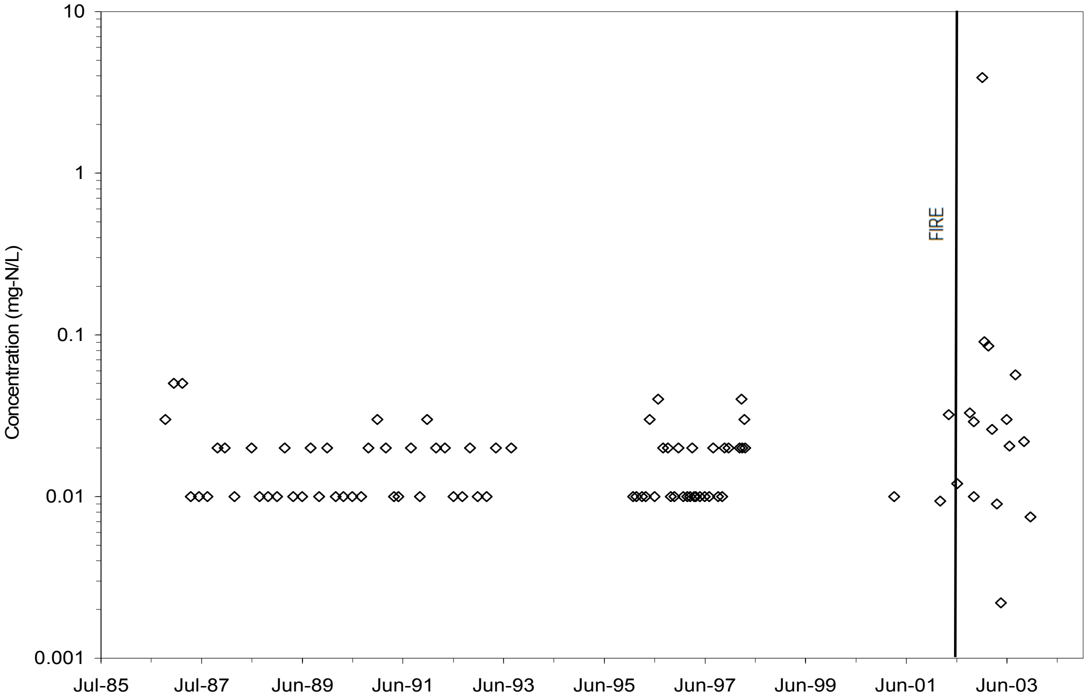

Figure 4.37 Ammonia concentration in the Verde River above Horseshoe Lake at USGS gauging station 09508500

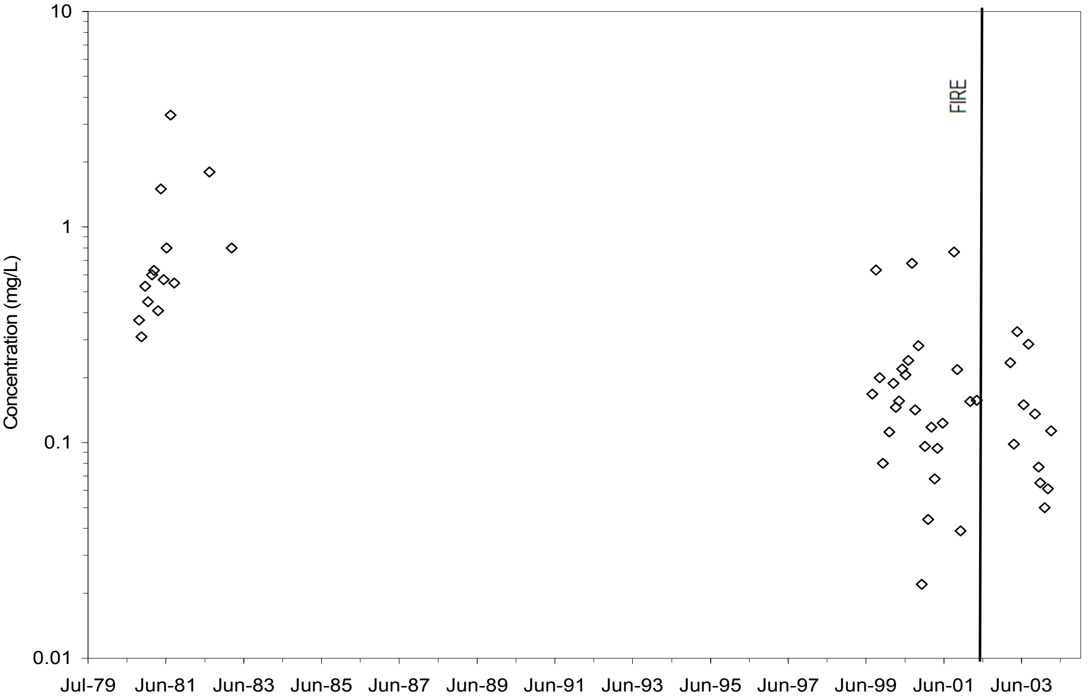

Figure 4.38 Total nitrogen concentration in the Verde River above Horseshoe Lake at USGS gauging station 09508500

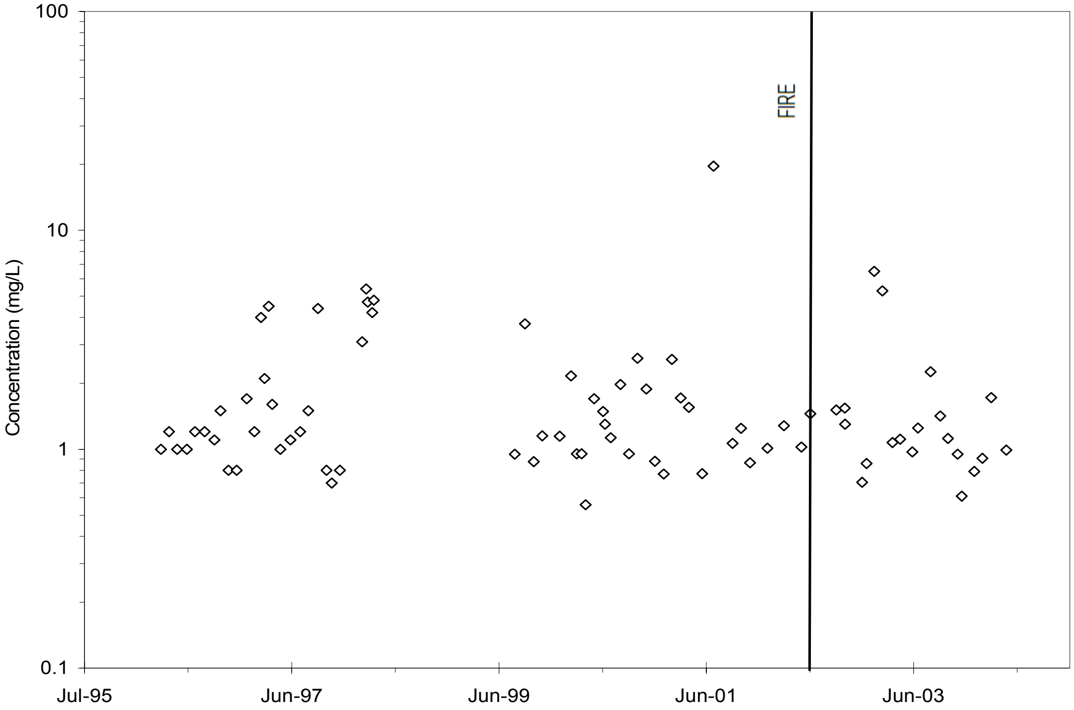

Figure 4.39 Dissolved organic carbon concentrations in the Verde River above Horseshoe Lake at USGS gauging station 09508500

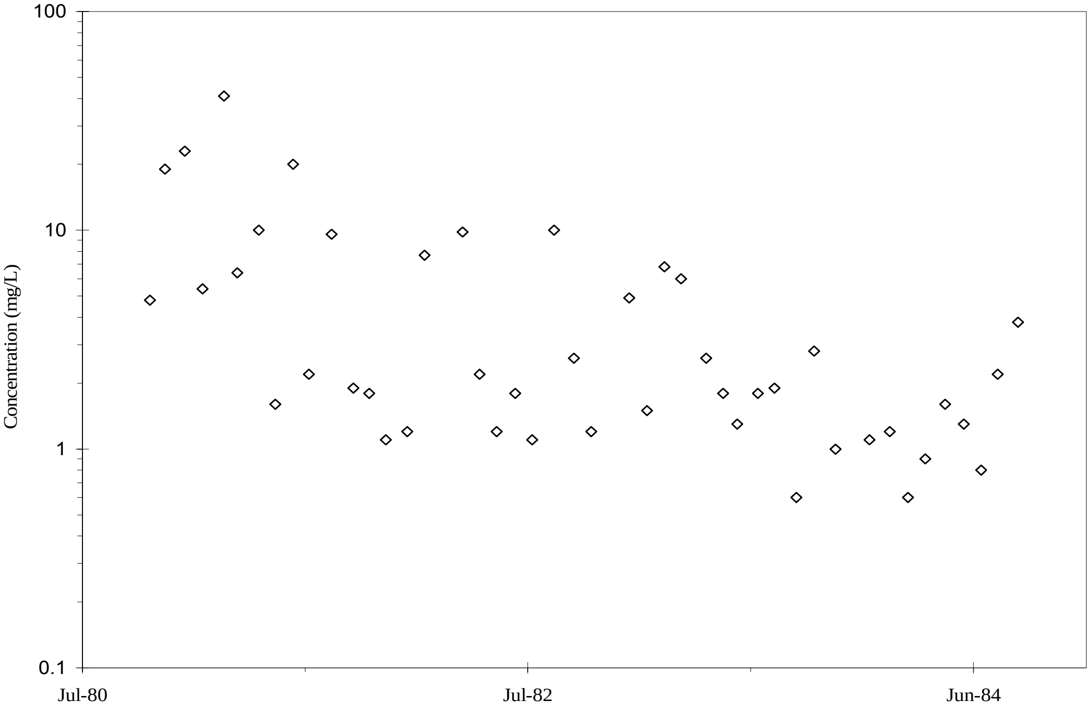

Figure 4.40 Total organic carbon concentrations in the Verde River above Horseshoe Lake at USGS gauging station 09508500

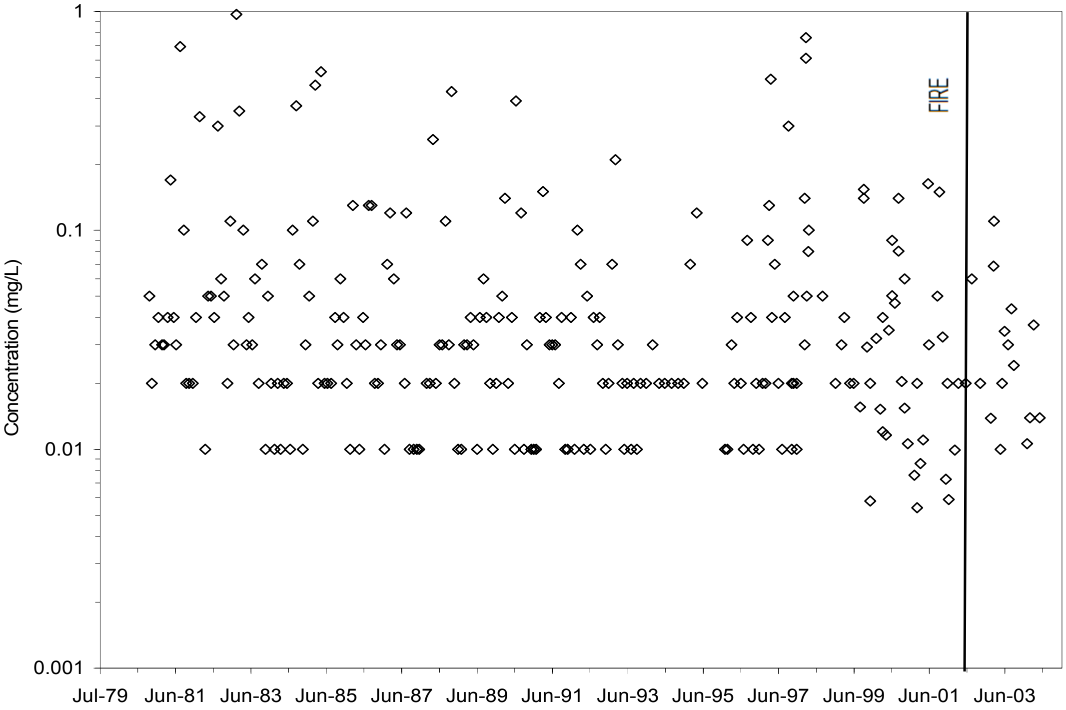

Figure 4.41 Total phosphorus concentrations in the Verde River above Horseshoe Lake at USGS gauging station 09508500

DISCUSSION

Stream Flow

The Rodeo-Chediski fire impacted the Salt River stream flow. The percentile ranking of stream flow for the Salt River and Verde River is displayed in Figure 4.42. During the first flush the Salt River stream flow peaked at $47m^3/s$ which is 88% higher than all other recorded flow rates. At this time, the Verde River stream flow of $2.4m^3/s$ was 2% higher than other recorded flow rates. The low flow rate in the Verde River is indicative of the regional drought, which includes the Salt River watershed. The elevated stream flow in the Salt River is abnormally high for drought conditions; therefore, the Rodeo-Chediski forest fire influenced the Salt River stream flow.

Elevated stream flows commonly occur after a forest fire. Gottfried et al. (2003) studied the Stermer Ridge watershed which was also burnt by the Rodeo-Chediski fire. The stream flow was 2,350 times higher than the historic flow. Although the historic maximum stream flow was not exceeded in the Salt River watershed, a high post-fire peak flow did occur. In addition, a 36% increase in average stream flow two years after a fire in Spain was reported by Belillas and Roda (1993). The two year post-fire average stream flow for the Salt River is 40% below the long-term average; however, this is due to the drought.

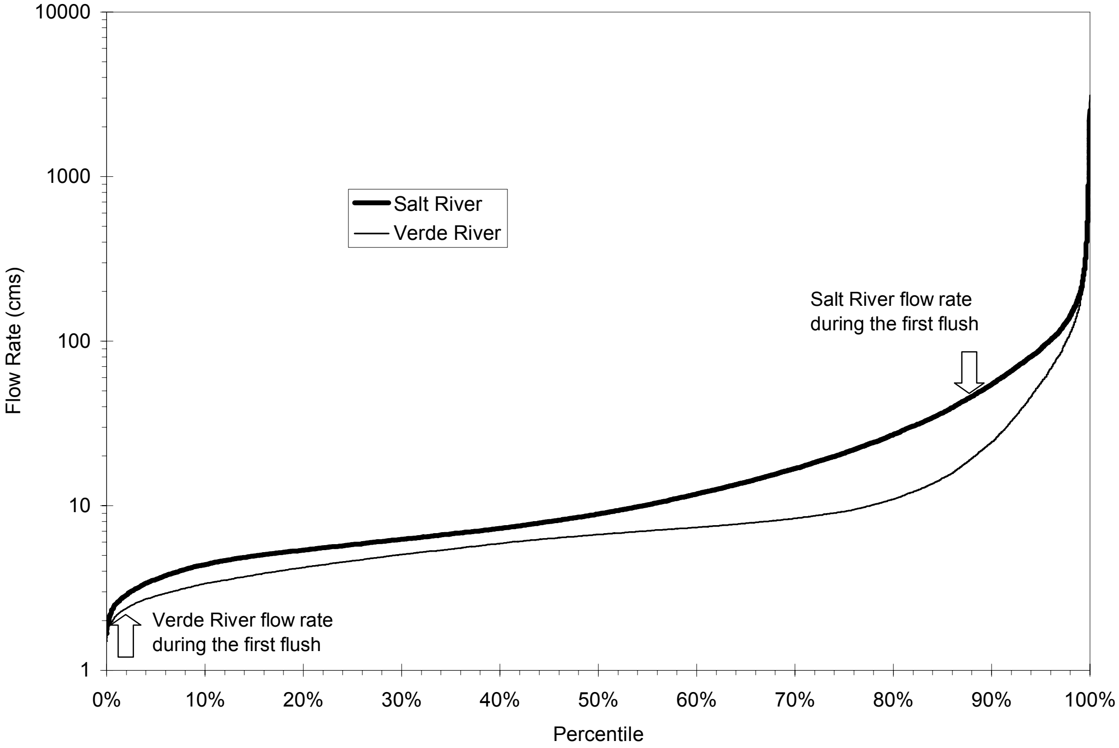

Figure 4.42 Percentile ranking of stream flow in the Salt River and Verde River

Suspended Sediment

The Rodeo-Chediski fire caused an increase in erosion; especially during the first flush. The influence of the fire was evident based on the suspended sediment concentration graph, Figure 4.3. A historically high concentration of 23,000mg/L was recorded during the first flush; this post-fire concentration exceeded the previous maximum by 96%. Gottfried et al. (2003) studied the area burned by the Rodeo-Chediski fire within the Stermer Ridge watershed. An increase of more than 3,200% in sediment transport was reported. Neary (2004) also observed an increase in erosion after a fire. In both studies, the increase in erosion was attributed to hydrophobic soil; similar soil conditions probably existed after the Rodeo-Chediski fire. In addition, the Verde River did not experience increased suspended sediment concentrations; this is expected since the Verde River watershed was not burned by the fire.

The fire also influenced the organic content of the particulates. Assuming 50% of the particulate organic carbon is carbon, the mobilized particulates had an average organic content of 2% prior to the fire; post-fire, the average organic content was 11%. Elevated suspended sediment concentrations and organic content of the particulates after a wildfire are consistent with the study conducted by Minshall et al (2001).

Nutrients

The Salt River and Verde River watersheds are both experiencing an eight year drought. The accumulation of soluble materials occurs on the sediment surface during a drought; the excess materials are then mobilized by major and minor storm events. The nutrient concentrations in the Verde River have remained at background levels since the Rodeo-Chediski fire. However, a temporary increase in ammonia and dissolved organic carbon concentration did occur within the past five years; the elevated concentrations are attributed to minor storm events that occurred within the watershed and possibly due to new development near Sedona, Arizona.

The Rodeo-Chediski fire contributed to the accumulation of soluble materials. The Salt River experienced high nutrient concentrations during the first flush, the first major storm event after the Rodeo-Chediski fire. The dissolved and particulate fractions of all the nutrients analyzed exceeded background concentration levels. During this time period, the Verde River nutrient concentrations remained at background levels. Therefore, the effects of the drought within the Salt River watershed were minor relative to the Rodeo-Chediski fire.

The historically high nitrate plus nitrite and total nitrogen concentrations within the Salt River are attributed to the fire. The nitrate plus nitrite and total nitrogen concentration increased by 195% and 5,690%, respectively. Similarly, Malmer (2004) reported a 2,970% increase in nitrate concentration and 217% increase in the total nitrogen concentration in stream flow following a fire in a drought-stricken watershed. Minshall et al. (2001) and Earl and Blinn (2003) also reported elevated post-fire nitrate concentrations. In all of these studies, the concentrations returned to pre-fire levels within one year.

High phosphorus concentrations in stream flow after a fire have been reported by Spencer and Hauer (1991), Tiedemann et al. (1978), and Gerla and Galloway (1997). Spencer and Hauer (1991) reported a significant phosphorus concentration increase of five to 60 times higher than background levels within a few weeks of a fire. Background phosphorus concentration levels were elevated 1.5 to 3 times post-fire, as reported by Tiedemann et al. (1978). Gerla and Galloway (1997) reported one order of magnitude increase in post-fire phosphorus concentration one year after a wildfire. Similarly, the post-fire total phosphorus and dissolved phosphorus concentrations were increased by one order of magnitude in the Salt River watershed; the elevated concentrations occurred within a month of containment of the Rodeo-Chediski fire.

Although seasonal increases in nutrient concentration have not been observed, the nitrate plus nitrite and total nitrogen concentrations were elevated one year post-fire in the Salt River. The dissolved phosphorus concentration was elevated eight months after the fire. Similar seasonal trends were reported by Spencer, Gabel, and Hauer (2003) five years after a wildfire.

The mobilization of burned organic matter and newly released nutrients altered the organic carbon-nitrogen-phosphorus (C:N:P) ratio in the Salt River. The C:N:P ratio is commonly used to determine the limiting nutrient within a system. Table 4.5 displays the pre- and post-fire C:N:P ratios for the total and dissolved fraction. The change in the pre- and post-fire total C:N:P ratio indicates more nitrogen was mobilized into the Salt River than organic carbon. While the difference in the pre- and post-fire ratio of dissolved organic carbon to dissolved nitrogen was less than 2%. Therefore, the dissolved C:N ratio remained constant while the dissolved phosphorus concentration increased. However, the Salt River system remained phosphorus-limited pre- and post-fire.

Table 4.5 Total and dissolved C:N:P ratio for the Salt River

	Total C:N:P	Dissolved C:N:P
Pre-fire	6:46:1	116:111:1
Post-fire	4:59:1	66:62:1

Metals

Metals were mobilized after the Rodeo-Chediski fire. The concentrations of the particulate fraction of the metals analyzed for the Salt River were elevated after the fire. Dissolved arsenic, dissolved barium, dissolved manganese, and dissolved selenium concentrations were above background levels after the fire, as well. However, after the first flush the concentrations of these metals returned to background levels. The background levels have been maintained, except for the particulate iron concentration. It cannot be determined if the elevated particulate iron concentration was a pulse or a trend due to the two year time period between sample collections. The elevated concentrations of the particulate fraction of the metals correlate with the increased suspended sediment concentrations.

Ballard (2000) reported mobilization of manganese, zinc, and aluminum after a fire-influenced pH change. Although the change in soil pH is not known following the Rodeo-Chediski fire, it is likely that it did change since similar trends were observed. In addition, Sakalauskiene and Ignatavicius (2003) reported similar results following a fire in Lithuania during a drought. Copper, lead, and zinc concentrations were 21%-74% above background levels in Lithuanian rivers within a few months of the fire.

The Rodeo-Chediski fire did not mobilize a significant portion of regulated dissolved metals. The EPA has set MCLs for the dissolved fraction of arsenic, barium, and mercury; secondary MCLs are established for the dissolved fraction of iron, zinc, and manganese. The MCLs are listed in Table 4.3. Manganese was the only metal to exceed the secondary MCL; the other regulated dissolved metal concentrations stayed below the MCLs.

SUMMARY

The results of the concentration data analysis indicate that the Rodeo-Chediski fire did impact the Salt River. The concentrations of all the constituents analyzed for the Salt River were compared pre- and post-fire to determine the impacts of the fire on each constituent. Nutrient and suspended sediment concentrations were compared with the Verde River as well. The Verde River watershed is also in a drought and has not endured a catastrophic wildfire. The conclusions reached as a result of the concentration data analysis are listed below.

- The first flush caused a 96% increase in the total suspended sediment concentration within the Salt River.
- The organic content of the particulates increased from 2% to 11% after the fire.
- The increased nutrient concentrations in the Salt River are a result of the fire.
- Total nitrogen, ammonia, and nitrate plus nitrite concentrations exceeded historical limits during the first flush and two years post-fire:

Constituent	% Change*
Ammonia	1717%
Total Nitrogen	5689%
Nitrate+Nitrite	223%

*%Change = (Final concentration - Initial concentration)/(Initial Concentration)

- The Salt River system remained phosphorus-limited post-fire.
- The dissolved C:N:P ratio indicated the mobilization of phosphorus exceeded the mobilization of organic carbon and nitrogen.
- The total C:N:P ratio indicated the mobilization of total nitrogen exceeded the mobilization of organic carbon and phosphorus.
- The Rodeo-Chediski fire and subsequent storm events caused high concentrations for all trace metals and ions to increase except for; total

iron, dissolved manganese dissolved zinc, potassium, magnesium, and bicarbonate.

- Selenium was the only EPA regulated metal to exceed the MCL after the fire.

CHAPTER 5 : LOADING ANALYSIS IN ARIZONA

The purpose of this chapter is to perform a mass loading analysis to determine the long term water quality changes as a result of the fire in the Salt and Verde Rivers. The constituents analyzed for the Salt River are listed in Table 4.1. The Verde River mass loading analysis was completed for dissolved organic carbon, ammonia, total phosphorus, suspended sediment, and stream discharge. The concentration data and continuous stream flow measurements presented in Chapter 4 were used for the mass loading analysis.

METHODS TO ESTIMATE CONSTITUENT MASS LOADING

Two methods were used to estimate the mass loading of the constituents listed in Table 4.1 and 4.4. These methods were the Simple Method and Beale's ratio estimator, Equation 5.1, and Equation 5.2, respectively (Dolan et al. 1981). To estimate the mass loading for samples with concentrations below the MRL, a concentration one-half of the MRL was assumed.

$$L = \frac{2.45 * \sum_{j=1}^{365} A_j * C_j * Q_j}{\sum_{j=1}^{365} A_j} \qquad (5.1)$$

L = mean load estimate for the time interval (kg/d)
Q_j = daily flow rate (cfs)
C_j = concentration on day j (mg/L)
A_j = 1 if the concentration data is available or 0 if it is not available
2.45 is a conversion factor for converting mg/L and cfs to kg/day

$$\hat{\mu}_y = \mu_X \frac{m_y}{m_x}\left(\frac{1+\frac{1}{n}\frac{S_{xy}}{m_x m_y}}{1+\frac{1}{n}\frac{S_x^2}{m_x^2}}\right) \qquad (5.2)$$

$\hat{\mu}_y$ = estimated load (kg/day)
μ_x = mean daily flow for the year (m^3/day)
m_y = mean daily loading for the days on which concentrations were determined (kg/day)
m_x = mean daily flow for the days on which daily concentrations were determined (m^3/day)
n = number of days on which concentrations were determined (day)

$$S_{xy} = \frac{1}{\left(n-1\right)}\sum_{i=1}^{n} x_i y_i - n m_x m_y \qquad (5.3)$$

$$S_x^2 = \frac{1}{\left(n-1\right)}\sum_{i=1}^{n} x_i^2 - n m_x^2 \qquad (5.4)$$

x_i = individual measured flows (m^3/day)
y_i = daily loading for each day on which concentration was determined (kg/d)

Both methods have advantages and disadvantages. The Simple method has the advantage of being a quick and simplistic technique to estimate the average annual mass loading. However, it does not account for flow data obtained on days when no samples were collected. Beale's ratio estimator takes advantage of continuous flow data. This was accomplished by incorporating the mean of the flow data for all days; it is then multiplied by the ratio of mean daily loading to mean daily flow for days on which concentrations were determined.

Another advantage of Beale’s ratio estimator is that the flow rate, high or low, at the time of sample collection has less of an overall impact on the estimated load than when using the simple method (Dolan et al. 1981). Also, the bracketed term in Equation 5.2 accounts for bias. As the number of samples collected increases, the bias decreases, therefore, a more accurate load estimate is obtained. Overall, Beale’s ratio estimator is better suited for this analysis than the Simple method. The Simple method was used to check all mass loading calculations made using Beale’s ratio estimator; ideally, the results should be the same order of magnitude.

An example of how to use both methods is described by using data for dissolved arsenic for the year 2003-2004. All flow rate data was obtained from Figure 4.1.

Simple Method

The required data for the simple method is listed in Table 5.1.

Table 5.1 Data for the simple method

Dissolved Arsenic Data			
Date of Sample Collection	C_j (mg/L)	Q_j (cfs)	A_j^*
10/24/2003	0.0109	143	1
11/21/2003	0.0068	286	1
12/5/2003	0.0079	201	1
1/23/2004	0.0086	213	1
2/20/2004	0.0107	201	1
3/5/2004	0.00605	414	1
3/26/2004	0.00255	1840	1
4/9/2004	0.00395	983	1
4/23/2004	0.00195	739	1
5/11/2004	0.00555	386	1
5/25/2004	0.00795	197	1
6/16/2004	0.00825	96	1

*A_j is 0 for all other dates since there were only 12 dates in which concentration data was collected.

Step 1: Write Equation 5.1

$$L = \frac{2.45 * \sum_{j=1}^{366} A_j * C_j * Q_j}{\sum_{j=1}^{366} A_j}$$

Since 2004 was a leap year, the upper limit is 366 rather than 365.

Step 2: Calculate $A_j*C_j*Q_j$ for each day concentration data was collected using data in Table 5.1

Date of Sample Collection	$A_j*C_j*Q_j$
10/24/2003	1.56
11/21/2003	1.94
12/5/2003	1.59
1/23/2004	1.83
2/20/2004	2.15
3/5/2004	2.50
3/26/2004	4.69
4/9/2004	3.88
4/23/2004	1.44
5/11/2004	2.14
5/25/2004	1.57
6/16/2004	0.79

Step 3: Calculate L using Equation 5.1

$$\sum_{j=1}^{366} A_j * C_j * Q_j = 26.09$$

$$\sum_{j=1}^{366} A_j = 12$$

$$L = \frac{2.45 * 26.09}{12}$$

$$\mathbf{L = 5.331\ kg/d}$$

Step 4: Calculate the Average Annual Mass Loading (kg/yr)

Average Annual Mass Loading = 366*L

Average Annual Mass Loading = $\underline{1.95\text{x}10^3\ \text{kg/yr}}$

Beale's ratio estimator

The required data for Beale's ratio estimator is listed in Table 5.2; a summary of the parameter values are listed in Table 5.3.

Table 5.2 Data for Beale's ratio estimator

Date of Sample Collection	x_i	x_i^2	y_i	x_iy_i	n
10/24/2003	$3.5\text{x}10^5$	$1.23\text{x}10^{11}$	3.81	$1.33\text{x}10^6$	1
11/21/2003	$7\text{x}10^5$	$4.90\text{x}10^{11}$	4.76	$3.33\text{x}10^6$	1
12/5/2003	$4.92\text{x}10^5$	$2.71\text{x}10^{11}$	3.88	$1.91\text{x}10^6$	1
1/23/2004	$5.21\text{x}10^5$	$2.42\text{x}10^{11}$	4.48	$2.33\text{x}10^6$	1
2/20/2004	$4.92\text{x}10^5$	$1.02\text{x}10^{12}$	5.26	$2.59\text{x}10^6$	1
3/5/2004	$1.01\text{x}10^6$	$2.03\text{x}10^{13}$	6.13	$6.19\text{x}10^6$	1
3/26/2004	$4.5\text{x}10^6$	$5.81\text{x}10^{12}$	11.5	$5.18\text{x}10^7$	1
4/9/2004	$2.41\text{x}10^6$	$3.28\text{x}10^{12}$	9.5	$2.29\text{x}10^7$	1
4/23/2004	$1.81\text{x}10^6$	$8.91\text{x}10^{11}$	3.53	$6.39\text{x}10^6$	1
5/11/2004	$9.44\text{x}10^5$	$2.32\text{x}10^{11}$	5.24	$4.95\text{x}10^6$	1
5/25/2004	$4.82\text{x}10^5$	$5.52\text{x}10^{10}$	3.83	$1.85\text{x}10^6$	1
6/16/2004	$2.35\text{x}10^5$	$1.35\text{x}10^{12}$	1.94	$4.56\text{x}10^5$	1
Total	$\mathbf{1.39\text{x}10^7}$	$\mathbf{3.29\text{x}10^{13}}$	**63.8**	$\mathbf{1.06\text{x}10^8}$	**12**
Average	$\mathbf{1.16\text{x}10^6}$		**5.32**		

Table 5.3 Summary of parameter values for Beale's ratio estimator

Parameter	Value
μ_x*	$1.06\text{x}10^6$
m_x	$1.16\text{x}10^6$
m_x^2	$3.29\text{x}10^{13}$
m_y	5.32
x_i	$1.39\text{x}10^7$
x_i^2	$3.29\text{x}10^{13}$
y_i	63.8
$\sum x_i y_i$	$1.06\text{x}10^8$
n	12

*μ_x calculated using data displayed in Figure 4.1

Step 1: Calculate S_{xy} using Equation 5.3, Table 5.2, and Table 5.3

$$S_{xy} = \frac{1}{(12-1)}\sum_{i=1}^{12} 1.06*10^8 - 12(1.16*10^6)5.32$$

$$\mathbf{S_{xy} = \boldsymbol{2.9x10^6}}$$

Step 2: Calculate S_x^2 using Equation 5.4, Table 5.2, and Table 5.3

$$S_x^2 = \frac{1}{(12-1)}\sum_{i=1}^{12} 3.29*10^{13} - 12(1.16*10^6)^2$$

$$\mathbf{S_x^2 = 1.52x10^{12}}$$

Step 3: Calculate the estimated load, $\hat{\mu}_y$, using Equation 5.2

$$\hat{\mu}_y = 1.06*10^6 \frac{5.32}{1.16*10^6}\left(\frac{1+\frac{1}{12}\frac{2.9*10^6}{(1.16*10^6)5.32}}{1+\frac{1}{12}\frac{1.52*10^{12}}{(1.16*10^6)^2}}\right)$$

$$\hat{\mu}_y \mathbf{= 4.61\ kg/day}$$

Step 4: Calculate the average annual mass loading (kg/yr)

Average Annual Mass Loading = 366 (d/yr)* $\hat{\mu}_y$ (kg/day)

366d/yr was used since 2004 was a leap year

Average Annual Mass Loading = <u>1.69x10^3 kg/yr</u>

Comparison of methods to estimate mass loading

The estimated mass loading was different for both methods used in the previous example. However, the result was within the same order of magnitude for this ideal situation. The behavior of the constituent concentration during high flow and low flow conditions will affect the overall bias of the estimate. Loads will be underestimated if concentrations increase during runoff periods and overestimated if concentrations decrease during runoff periods (Richards and Holloway 1987). Although Beale’s ratio estimator corrects for the bias, the final result may be skewed depending on the characteristics of the specific constituent analyzed.

A comparison of the two methods to estimate average annual mass loading for dissolved zinc is displayed in Figure 5.1. Dissolved arsenic is a constituent that historically decreased in concentration during runoff periods. The outliers occurred during periods of increased runoff, which can skew the estimate. The first post-fire year estimate was very skewed, as indicated in Figure 5.1. The concentration was historically high during the runoff period; therefore, the estimate was biased high.

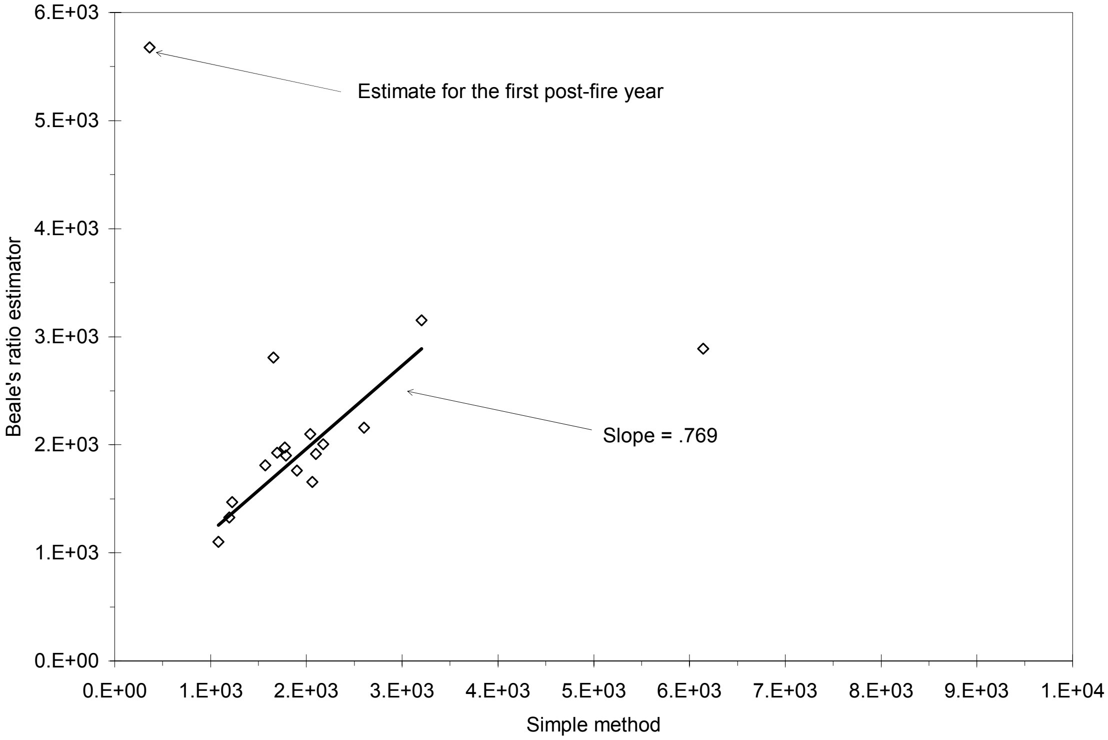

Figure 5.1 Beale's ratio estimator and Simple method comparison

SALT RIVER LOADING ANALYSIS

The objective of this analysis was to compare pre-fire and post-fire mass loadings of the constituents listed in Table 4.1. Table 5.4 displays the years that data were collected and analyzed for each constituent. Table 5.5 lists the three different time periods used to estimate the mass loading. Since the Rodeo-Chediski fire was contained within the first week of July the beginning of each year was July 1.

Table 5.4 The years of data collection on the Salt River above Roosevelt Lake at USGS gauging station 09498500

Constituents	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
Nutrients																													
Total Organic Carbon	X	X	X	X	X	X	X	X	X							X												X	
Dissolved Organic Carbon																X					X	X	X					X	X
Total Phosphorus	X	X	X	X	X	X	X	X			X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Phosphorus		X										X	X	X	X	X	X	X	X	X	X	X	X					X	X
Total Nitrogen	X	X	X	X	X	X	X	X			X			X				X		X		X	X					X	X
Nitrate-Nitrite												X	X	X	X	X	X	X	X	X	X	X	X					X	
Total Organic Nitrogen																													
Ammonia		X	X	X	X	X	X	X	X			X	X	X	X	X	X	X	X	X	X	X	X					X	X
Major Cations																													
Potassium	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Magnesium	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Calcium	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Sodium	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Major Anions																													
Bicarbonate													X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Chloride	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Other																													
Sediment				X	X		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Stream Flow*	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X

(Continued)

Table 5.4 (continued)

Constituents	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
Trace Metals																													
Total Manganese	X	X	X	X	X	X				X	X	X	X	X	X	X	X	X	X	X				X	X	X	X	X	
Dissolved Manganese	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Total Iron	X	X	X	X	X	X				X	X	X	X	X	X	X	X	X	X	X				X	X	X	X	X	
Dissolved Iron	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Total Copper	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Dissolved Copper										X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Total Zinc	X	X	X	X	X				X	X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Dissolved Zinc						X	X	X	X	X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Total Barium	X	X	X	X	X	X																	X	X	X	X	X	X	
Dissolved Barium										X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Total Mercury	X	X	X	X	X	X				X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Dissolved Mercury												X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Total Lead	X	X	X						X	X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Dissolved Lead							X	X	X	X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Total Selenium																													
Dissolved Selenium										X	X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	
Total Arsenic	X	X	X	X	X	X				X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	
Dissolved Arsenic										X	X	X	X	X	X	X	X	X	X	X			X	X	X	X	X	X	X

*Data collection began in 1914

Table 5.5 Time Periods for Mass Loading Calculations

Time Period	Description of data included in analysis
Pre-2002	All data for each constituent before July 1, 2002*
2002-03	All data for each constituent from July 1, 2002 to June 30, 2003*
2003-04	All data for each constituent from July 1, 2003 to June 30, 2004†

*Data obtained from USGS
†Data collected by ASU sampling

The average annual mass loading results produced from Beale's ratio estimator were selected for use in the mass loading figures. Beale's ratio estimator is the more appropriate and accurate method for the purpose of this data analysis. It also utilizes the continuous flow rate data and is useful when concentration data is limited. The results of Beale's ratio estimator were checked by the Simple Method; the mass loading estimates should be within one order of magnitude.

Stream Flow

The mass of water entering the Salt River has continued to decrease since the fire, as shown in Figure 5.2. A 34% decrease was observed the first year after the fire. Another 18% decrease occurred during the second year after the fire. The continual decline in the mass loading of water is due to the eight year drought the watershed is enduring.

Suspended Sediment

The suspended sediment load entering the Salt River system increased 135% the first year after the fire, as shown in Figure 5.2. Two years post-fire the mass loading decreased by 42%, however, it still exceeded the long-term average annual loading by 35%. Erosion was enhanced by the lack of ground cover and vegetation. Runoff easily mobilized the particulate fraction of nutrients and metals during storm events.

Nutrients

Nutrient mass loadings increased the first year after the fire, as shown in Figure 5.2. Table 5.6 lists the percent change in pre- and post-fire nutrient mass loadings. The elevated loadings one year post-fire occurred even though the mass loading of water in the Salt River was lower. Two years post-fire the mass loading of nutrients, for which data was available, decreased.

The dissolved organic carbon, total and dissolved phosphorus, and total nitrogen loadings were less than the long-term average annual loading two years post-fire. Dissolved organic carbon and dissolved phosphorus loadings were 56% and 79% lower than the average annual load, respectively. The total phosphorus and total nitrogen loads were 100% lower.

The estimated loading for ammonia and nitrate plus nitrite increased two year post-fire. However, the ammonia and nitrate plus nitrite loadings were not accurately estimated. The sum of the dissolved fraction of nitrogen exceeded the total fraction of nitrogen, this is impossible. The skewed estimate is likely due the MRL approximation (as previously discussed) or error resulting from the water sample analysis.

Table 5.6 Percent change in pre- and post-fire nutrient, suspended sediment, and stream discharge average annual mass loadings for the Salt River at USGS gauging station 09498500

Constituent	**%Change 1 year post-fire**	**%Change 2 years post-fire**
Nutrients		
Total Organic Carbon	4334%	-
Carbon	470%	-92%
Total Phosphorus	1241%	-100%
Dissolved Phosphorus	417%	-96%
Total Nitrogen	4935%	-100%
Ammonia	2059%	-93%
Nitrate-Nitrite	755%	-84%
Other		
Suspended Sediment	135%	-42%
Stream Discharge	-34%	-18%

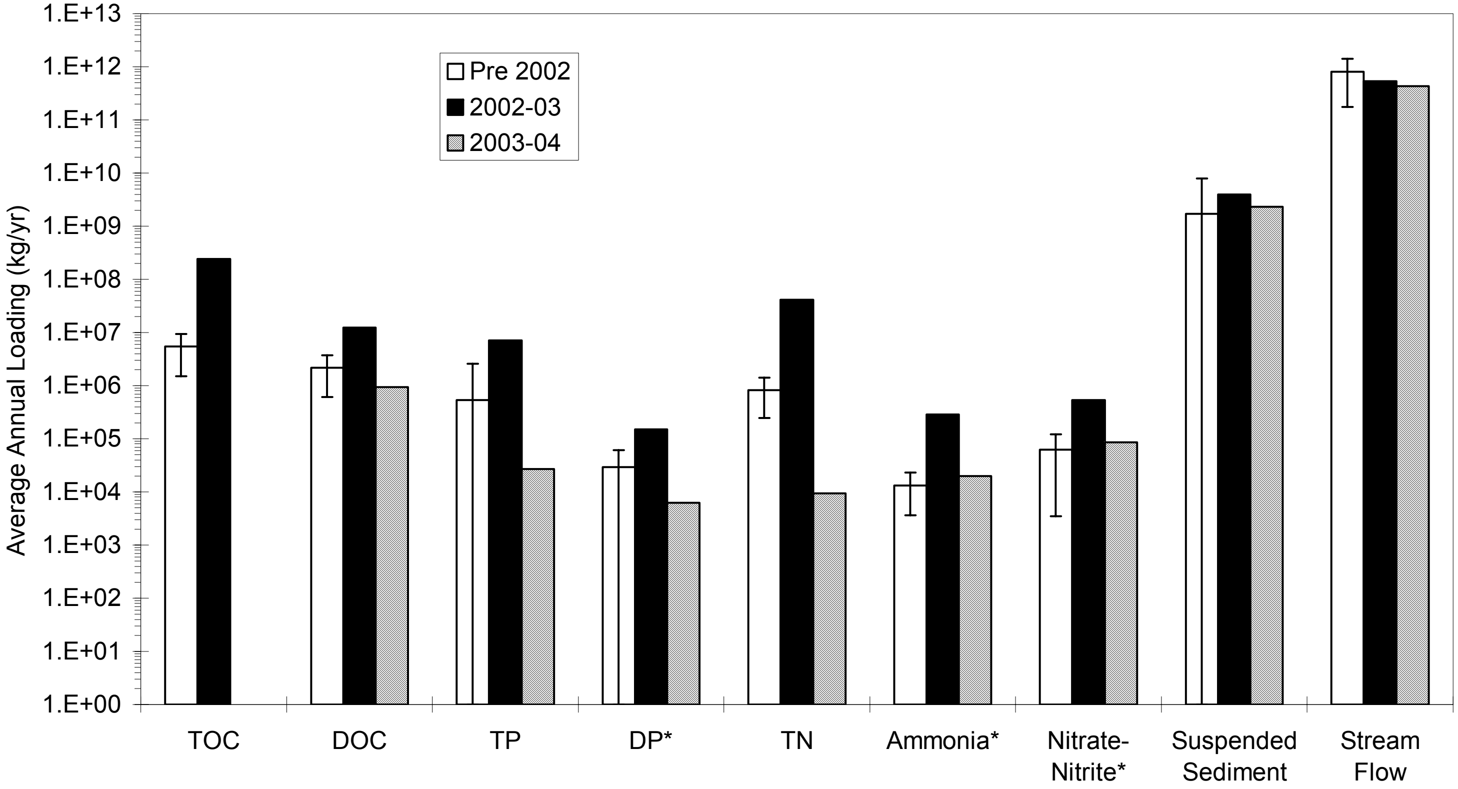

Figure 5.2 Pre- and post-fire loading of nutrients and suspended sediment for the Salt River at USGS gauging station 09498500

*More than half of the concentrations for this constituent are below the MRL

Metals

The response of the metals to the Rodeo-Chediski fire varied. Iron and copper displayed similar post-fire behavior; while the responses of the manganese, zinc, and lead were comparable. Arsenic, barium, and mercury exhibited unique post-fire behavior when compared to the other trace metals. Figure 5.3 and 5.4 show the responses of the total and dissolved mass loading for these trace metals. Table 5.7 displays the percent change in the post-fire mass loading after the fire.

Iron, copper, and selenium responded similarly to the fire. One year post-fire, the dissolved mass loading of iron and copper decreased. The 39% decrease in dissolved copper mass loading and 80% decline in dissolved iron mass loading correlates with the 34% decrease in the mass loading of water. The dissolved selenium mass loading was increased by 9%. However, the total mass loading of iron, copper, and selenium were elevated post-fire, 21%, 12%, and 35%, respectively. The increase in total mass loading is due to the elevated post-fire mass loading of particulates.

Manganese, zinc, and lead displayed similar trends after the fire. Like iron and copper, the dissolved mass loadings decreased the first year after the fire due to a decrease in water within the Salt River. Although an 85%, 63%, and 84% decrease occurred the first year post-fire for the mass loading of dissolved manganese, dissolved zinc, and dissolved lead, respectively, the total trace metals loading increased. The mass loadings of total manganese, total zinc, and total lead were elevated by 960%, 402%, and 414%, respectively. The increase in the total manganese, total zinc, and total lead mass loading is consistent with the elevated post-fire mass loading of total suspended sediment.

The post-fire mass loadings trend of arsenic and barium were different from the other trace metals. After the fire, the total and dissolved mass loadings increased for both constituents. Dissolved arsenic mass loading one year post-fire exceeded the long-term average annual loading by 152%, while dissolved barium was increased by 71%. The increase in the total mass loading is attributed to an increase in, both, the dissolved and particulate fractions. Arsenic and barium have been responsive to minor storm and runoff events since the drought began eight years ago. Therefore, the increase in mass loading for the dissolved fraction is not an unusual trend during a runoff event. However, historically, arsenic and barium have not been have not been available for mobilization to the extent observed post-fire.

Dissolved arsenic is the only trace metal for which data has been collected and analyzed for two years post-fire. The dissolved arsenic mass loading decreased 71% during the second year after the fire. Therefore, the increased mobilization of dissolved arsenic was temporary.

Mercury was the only trace metal to display a decrease in mass loading after the fire. A 42% and 37% reduction in the mass loadings of dissolved and total mercury occurred the first year post-fire, respectively. The total mercury mass loading decreased less than the dissolved mercury mass loading due to the elevated particulate mass loading.

Table 5.7 Percent change in pre- and post-fire average annual mass loading of metals Salt River at USGS gauging station 09498500

Constituent	% Change
Dissolved Arsenic	152%
Total Arsenic	484%
Dissolved Barium	71%
Total Barium	1991%
Dissolved Copper	-39%
Total Copper	21%
Dissolved Iron	-80%
Total Iron	12%
Dissolved Lead	-84%
Total Lead	414%
Dissolved Manganese	-85%
Total Manganese	960%
Dissolved Mercury	-42%
Total Mercury	-37%
Dissolved Selenium	9%
Total Selenium	35%
Dissolved Zinc	-63%
Total Zinc	402%

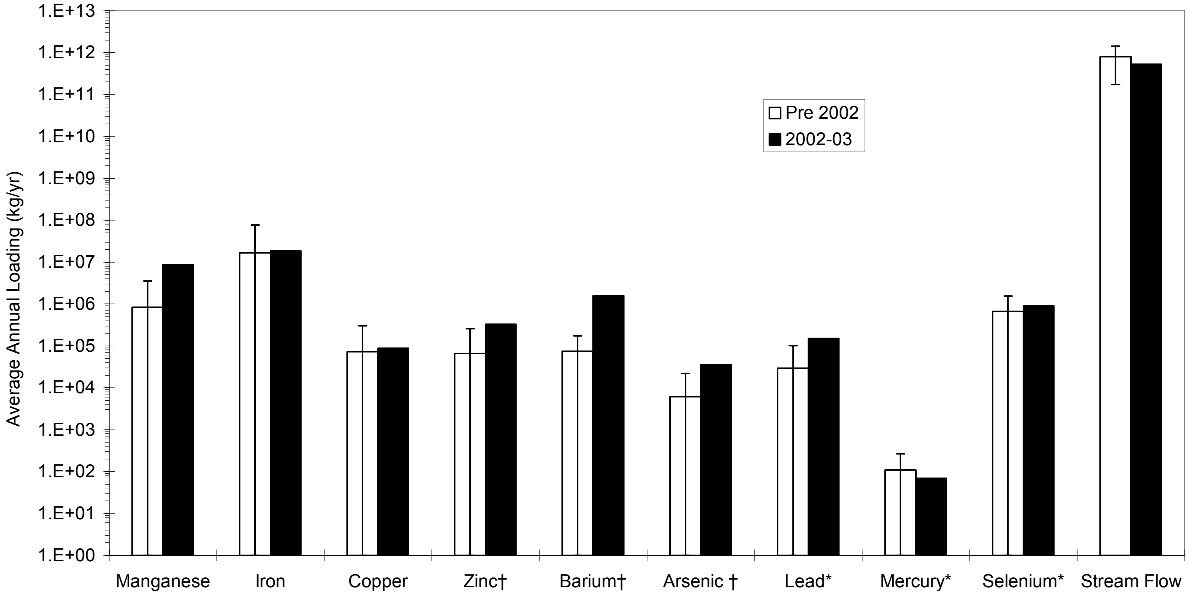

Figure 5.3 Pre- and post-fire loading of total trace metals for the Salt River at USGS gauging station 09498500

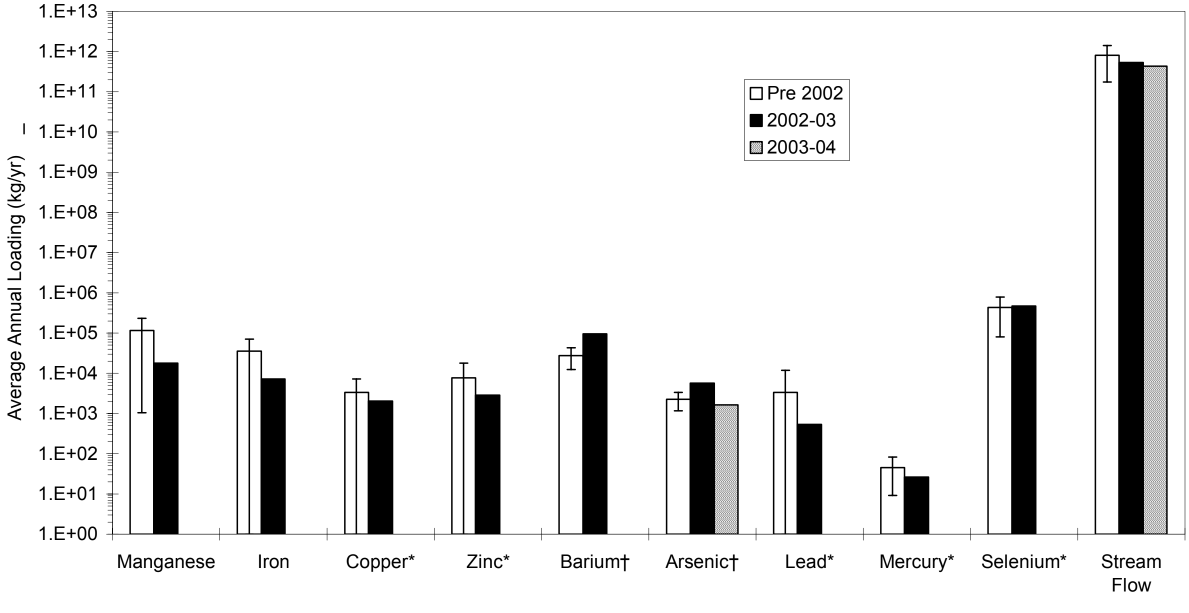

Figure 5.4 Pre- and post-fire loading of dissolved trace metals for the Salt River at USGS gauging station 09498500

*More than half of the concentrations for this constituent are below the MRL

†More than half of the concentrations for this constituent are below the MRL for pre-fire data only

Major Ions

Cations

The four cations analyzed were potassium, magnesium, calcium, and sodium. The mass loadings for all the cations did increase one year post fire, as displayed in Table 5.8 and Figure 5.5. The potassium and sodium post-fire mass loading increased by 192% and 132%, respectively. The fire had a more significant impact on potassium and sodium than on magnesium and calcium. The post-fire magnesium and calcium mass loading was elevated by 30% and 53%, respectively. In comparison to other constituents, such as the nutrients, the increase in mass loading one year post-fire of the cations is minimal.

Table 5.8 Percent change in pre- and post-fire average annual mass loading of cations for the Salt River at USGS gauging station 09498500

Constituent	% Change
Potassium	192%
Magnesium	30%
Calcium	53%
Sodium	132%

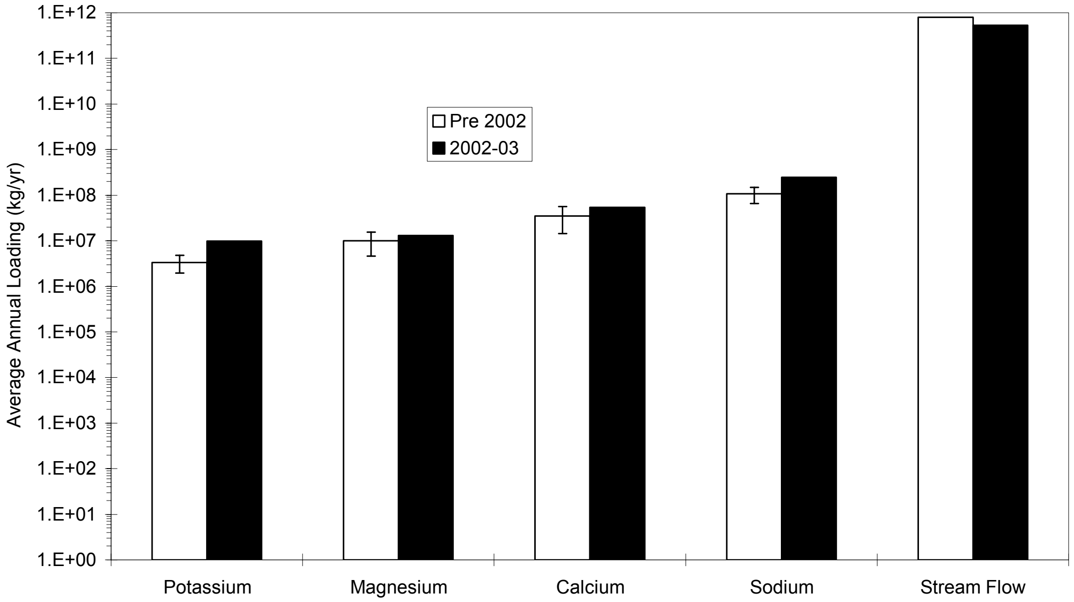

Figure 5.5 Pre- and post-fire loading of major cations for the Salt River at USGS gauging station 09498500

Anions

Two anions were analyzed, chloride and bicarbonate. The post-fire trend of the anions was similar to that of the cations. The mass loading of the anions increased post-fire, as displayed in Table 5.9 and Figure 5.6. A 139% and 49% increase in the mass loading of chloride and bicarbonate occurred one year post-fire, respectively. Once again, the impact of the fire on the mass loadings was minimal.

Table 5.9 Percent change in pre- and post-fire average annual mass loading of anions for the Salt River at USGS gauging station 09498500

Constituent	% Change
Chloride	139%
Bicarbonate	49%

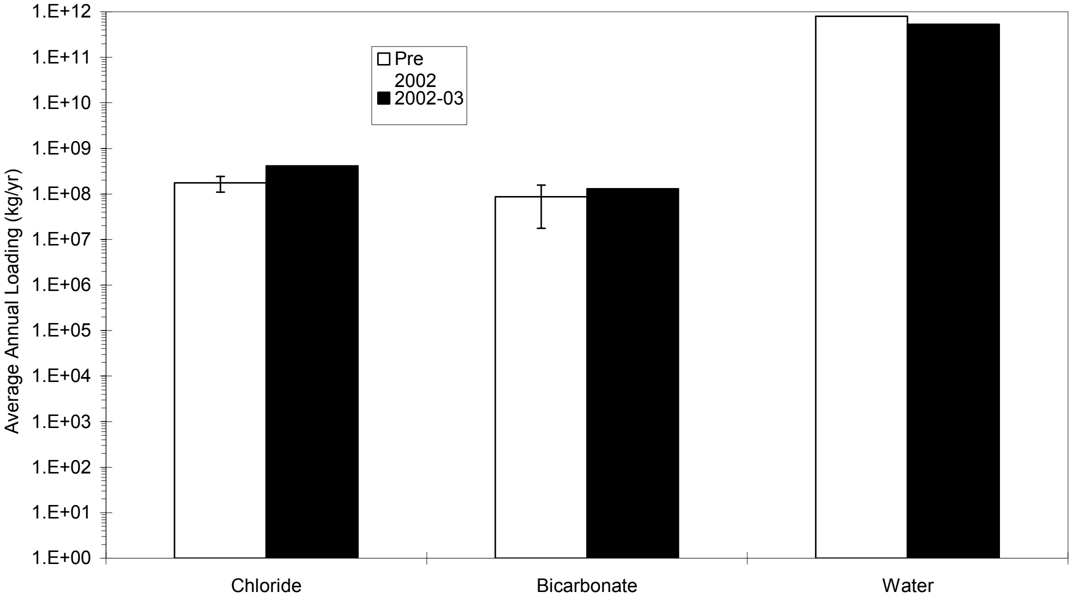

Figure 5.6 Pre- and post-fire loading of anions for the Salt River at USGS gauging station 09498500

VERDE RIVER LOADING ANALYSIS

The objective of this analysis was to compare pre-fire and post-fire mass loading of the constituents included in Table 5.10. In addition, Table 5.10 displays the years that data was collected and analyzed for each constituent. Since the unburned watershed is represented by the Verde River watershed, the time periods used to estimate the mass loading were consistent with the Salt River calculations. Table 5.5 lists the three different time periods used to estimate the mass loading. Ultimately, the results of the Verde River mass loading analysis were compared with the Salt River mass loading analysis.

Table 5.10 The years of data collection on the Verde River above Horseshoe Lake at USGS gauging station 09508500

Constituents	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
Nutrients																								
Dissolved Organic Carbon																X	X	X		X	X	X	X	X
Total Phosphorus	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Dissolved Phosphorus													X			X	X	X		X	X	X	X	X
Total Nitrogen	X	X	X																	X	X	X	X	X
Ammonia								X	X	X	X	X					X	X				X	X	X
Other																								
Suspended Sediment		X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Stream Flow*	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X

Stream Flow

The mass of water entering the Verde River has continued to decrease since the fire, as shown in Figure 5.7. A 34% decrease was observed the first year after the fire. Another 49% decrease occurred during the second year after the fire. The continual decline in the mass loading of water is due to the eight-year drought the watershed is experiencing.

Suspended Sediment

The suspended sediment load that entered the Verde River system decreased the first year after the Rodeo-Chediski fire burned the Salt River watershed, as shown in Table 5.11 and Figure 5.7. The decrease of 81% in the sediment load correlates with the decrease of 34% in the water load within the Verde River; therefore, less runoff was available to transport sediment.

Nutrients

Nutrient mass loadings displayed varied post-fire trends, as shown in Table 5.11 and Figure 5.7. The dissolved organic carbon and ammonia mass loadings were elevated the first year following the Rodeo-Chediski fire. The mass loadings of total phosphorus and total nitrogen decreased during this time period. The mass loadings of all the nutrients declined two years post-fire.

The first year after the fire, the mass loadings of dissolved organic carbon and ammonia showed a 39% and 22% increase, respectively. The elevated mass loading is probably because of the build up of soluble material on the sediment surface as a result of the drought. Two years after the fire, the mass loadings for the dissolved organic carbon and ammonia decreased by 85% and 39%, respectively. These changes in mass loading are consistent with the water loading being 49% less than the first year after the fire. Therefore, during 2002-2003 a source of dissolved nutrients existed that was not available during 2003-0204.

Total phosphorus and total nitrogen loadings have continually decreased since 2002. During the first year post-fire, the total phosphorus load was reduced by 72% and the total nitrogen load showed an 86% reduction. The decrease correlates with the reduced water load and sediment load. This decrease occurred although the loading of the dissolved fraction of both nutrients increased minimally during the first year post-fire. Therefore, the particulate fraction had a greater impact on the overall loading of phosphorus and nitrogen than the dissolved fraction. Two years after the Rodeo-Chediski fire, both total phosphorus and total nitrogen loads were reduced again by 82% and 72%, respectively.

Table 5.11 Percent change in pre- and post-fire nutrient, suspended sediment, and stream discharge mass loadings for the Verde River at USGS gauging station 09508500

Constituent	post-fire	post-fire
Nutrients		
Dissolved Organic Carbon	39%	-85%
Total Phosphorus	-72%	-82%
Total Nitrogen	-86%	-72%
Ammonia*	22%	-39%
Other		
Suspended Sediment	-81%	-
Stream Discharge	-34%	-49%

*More than half of the concentrations for this constituent are below the MRL for pre-fire data only

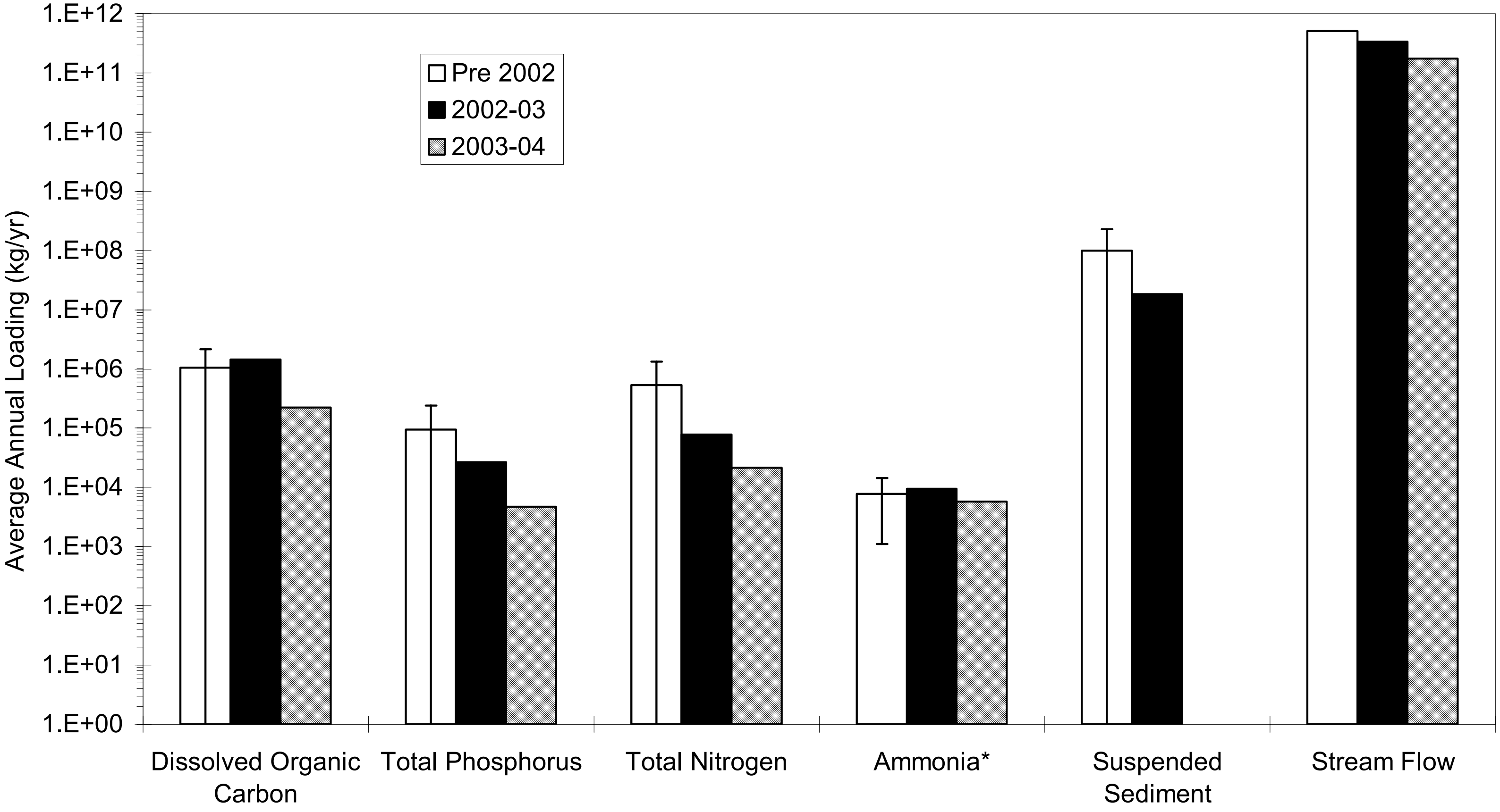

Figure 5.7 Pre- and post-fire loading of nutrients and suspended sediment for the Verde River at USGS gauging station 09508500

*More than half of the concentrations for this constituent are below the MRL for pre-fire data only

DISCUSSION

The mass loading analysis was completed for the Verde River and Salt River. Both watersheds are in an eight-year drought. However, only the Salt River watershed was burned. Therefore, the unburned Verde River watershed represents a baseline for the effects of drought conditions on a watershed. In addition, the annual mass loading and flow rate for each constituent is displayed in Appendix B.

Stream Flow

Since the Rodeo-Chediski fire, stream flow has continually decreased in the Salt River and Verde River. Both rivers experienced a 34% reduction in the mass loading of water the year following the fire. Two years post-fire, the mass loading of water decreased another 18% within the Salt River. An additional 49% in reduction of the mass loading of water was estimated for the Verde River. The decline in the mass loading of water observed in the Verde River and Salt River is attributed to the drought. Although forest fires can cause elevated stream flows, the drought is the controlling factor.

Suspended Sediment

The fire caused an increase in erosion within the Salt River watershed. The first year post-fire, the suspended sediment load entering the Salt River increased by 135% relative to the long-term average annual loading. The Verde River suspended sediment load was reduced by 81%. The decreased load in the Verde River was due to the drought since less runoff was available to transport sediment. Two years after the fire the suspended sediment load within the Salt River decreased 42%. This indicates the fire-induced erosion was depleted due to the re-growth of vegetation.

Nutrients

The mass loading analysis of nutrients was completed for the Salt River and Verde River. The Salt River mass loadings of nutrients all increased the first year post-fire. The elevated mass loading of total nutrients within the Salt River is consistent with the high suspended sediment mass load. This increase is attributed to the fire; the decreased mass loading of water within the Verde River caused a decrease in the total nutrient mass loading. Two years post-fire, the total nutrient loading decreased within the Salt River; this is consistent with the Verde River and the decreased suspended sediment load in the Salt River. The lower nutrient total loads indicate the impacts of the forest fire lasted less than one year.

The elevated dissolved nutrient mass loading is also due to the fire. The Verde River mass loading of ammonia and dissolved organic carbon increased the first year after the fire, this is attributed to the drought. During a drought materials accumulate on the sediment surface until a runoff event occurs to mobilize them; therefore, accumulated ammonia and dissolved organic carbon were finally mobilized. The dissolved nutrient mass loading in the Salt River and Verde River decreased the second year after the fire. The lower mass loading in the Salt River is a result of the watershed recovering from the fire. Nutrients were absorbed by vegetative re-growth, preventing further mobilization. In addition, the decreased mass loading of water in both rivers indicates less runoff was available to transport nutrients.

Metals

A metals mass loading analysis was completed for the Salt River. The impacts of the forest fire were evident. Total mercury was the only total metal immobilized post-fire, while dissolved barium and dissolved arsenic were the only dissolved metals mobilized. The total mass loading is governed by the particulates since the post-fire dissolved mass loading decreased for all metals, except barium and arsenic. This is consistent with the elevated post-fire suspended sediment load. The decreased dissolved loadings correlate with the decreased mass loading of water for the Salt River.

Mass loading ratio for nutrients and metals

The mass loading analysis for the Salt River showed a change in loading for trace metals and nutrients. A mass loading ratio was calculated to illustrate the influence of the particulate and dissolved fraction on the constituents analyzed. A ratio decrease indicates the constituent is influenced by the particulate fraction; a ratio increase indicates a dissolved fraction influence. The results are displayed in Table 5.12.

The ratio decreased the first year after the fire for the 11 nutrients and metals considered. This is consistent with the elevated suspended sediment loading within the Salt River during this time frame. The ratio indicates the particulate fractions zinc and manganese were readily mobilized relative to the other metals; particulate organic carbon was readily mobilized relative to phosphorus and nitrogen. The particulate fraction of mercury, copper, arsenic, selenium, and phosphorus was minimally mobilized, this is evident due to the minimal change in the ratio.

Total and dissolved data were available for phosphorus and nitrogen two years post-fire. The phosphorus ratio is still high relative to the pre-fire ratio. However, the total nitrogen ratio of 1120% is unrealistic. The estimated mass loading for the dissolved fraction of nitrogen exceeds the estimated mass loading for the total nitrogen. The estimated load is biased high since during periods of runoff the ammonia and nitrate plus nitrite concentration decreased. However, the concentration went below the minimum detection limit (MDL); therefore, the actual concentration is unknown. The approximation of one-half the MDL was used to estimate the concentration. This was not a good approximation for the dissolved fractions of nitrogen.

Table 5.12 Dissolved to total mass loading ratio for metals and nutrients

Constituent	Pre 2002	2002-03	2003-04
Nutrients			
Organic Carbon	40%	5.1%	-
Phosphorus	5%	2.1%	23%
Nitrogen	9.2%	2.0%	1120%
Trace Metals			
Manganese	14%	0.20%	-
Iron	0.21%	0.039%	-
Copper	4.6%	2.3%	-
Zinc	12.0%	0.88%	-
Barium	37%	6.1%	-
Arsenic	37%	16%	-
Lead	11%	0.35%	-
Mercury	42%	38%	-
Selenium	65%	52%	

Major Ions

The fire did not impact the average annual loading of cations and anions, on the Salt River, when comparing the year after the fire to the historical average as shown in Figures 5.5 and 5.6. However, the average annual loading in the first year post-fire fire did not follow the direct correlation with flow rate as in previous years. The average annual loading for calcium, magnesium, potassium, and bicarbonate each individual year prior to the fire increased when the flow rate surpassed 1000m^3/s, as shown in Figures B.27-B.30 in Appendix B. The average annual loading in the year following the fire increased to a level comparable to the pre 2002 years with an increase in flow rate; however, the flow rate did not surpass 1000m^3/s. The average annual loading for sodium and chloride did not display a correlation with flow rate as shown in Figure B.31 and B.32 in Appendix B. Therefore, the fire did affect the mass loading of ions when flow rate was considered, except sodium and chloride.

SUMMARY

During the first year after the forest fire, the composition of the Salt River did change based on the mass loading analysis completed. Nutrients, trace metals, ions, and suspended sediment experienced a shift in mass loading, typically an upward shift. However, two years after the fire, the loading of the constituents were near or below pre-fire levels. A summary of the results is below.

- All Salt River nutrient and suspended sediment mass loadings increased one year post-fire.
    - Total and dissolved organic carbon
    - Total and dissolved phosphorus
    - Total nitrogen, nitrate plus nitrite, and ammonia
    - Suspended sediment.
- Sodium and chloride loadings were not significantly impacted by the forest fire; magnesium, potassium, and bicarbonate loadings increased the year after the fire.

- All Salt River total trace metals displayed an increase in mass loading during the first year after the fire, except for mercury. The increase correlates with the elevated suspended sediment loading.
    - Manganese
    - Zinc
    - Barium
    - Arsenic
    - Lead
    - Copper
- Total mercury mass loading in the Salt River decreased the first year after the fire. The mercury concentration was typically below detection limits; during the first flush the concentration was detectable. The estimated mass loading was biased low due to the increased concentration during runoff.
- All dissolved trace metal mass loadings decreased the year following the fire, except for barium and arsenic.
    - Manganese
    - Zinc
    - Lead
    - Copper
    - Mercury

- Two years after the fire, arsenic loading in the Salt River was consistent with pre-fire levels.
- Two years post-fire, the nutrient and arsenic loading were at or below pre-fire levels.
    - Dissolved organic carbon
    - Dissolved and total phosphorus
    - Total nitrogen
- Two years post-fire, the estimated loading for ammonia and nitrate plus nitrite was unrealistic. The sum of the loading for the dissolved fractions of nitrogen exceeded the total nitrogen loading.

To determine if the change in mass loading within the Salt River was due to an overall environmental change or the Rodeo-Chediski fire, Verde River nutrient and suspended sediment loadings were analyzed. The loading for the Verde River showed different trends the first year after the fire than the Salt River loadings for the same time period. However, both rivers have experienced a continual decrease in the loading of water. The comparison of these two river systems allowed the following conclusions to be drawn.

- The first year after the fire, total phosphorus, total nitrogen, and suspended sediment loading decreased within the Verde River. The ammonia and dissolved organic carbon loadings increased slightly.
- The second year after the fire, dissolved organic carbon, total phosphorus, and total nitrogen loading decreased within the Verde River.

- The Rodeo-Chediski fire did impact the Salt River water quality. The mass loadings of total phosphorus, total nitrogen, and suspended sediment increased the first year after the fire in the Salt River; the loading for these same constituents within the Verde River decreased.
- The fire did affect the amount of dissolved organic carbon and ammonia within the Salt River. An increase in the loading of dissolved organic carbon and ammonia occurred within both river systems the first year after the fire. However, the increase within the Verde River was minimal when compared to the Salt River.

Due to the Rodeo-Chediski forest fire, the Salt River water quality was affected by both dissolved and particulate fractions of the constituents analyzed. The dissolved fraction will have more of an immediate and noticeable effect since they are a readily available food source for bacteria and algae. However, the particulate fraction will settle to the bottom of the Salt River or any of the reservoirs, and be released slowly over time.

The slow release of particulate fraction of nutrients will provide a steady source of energy to promote bacteria and algae growth. The particulate fraction of trace metals can be broken down by bacteria, therefore creating prolonged health concerns. Although two years post-fire the mass loadings are at or below pre-fire levels the, effects of the particulate phase that has settled to the bottom of the river and reservoirs may take longer to be fully realized.

CHAPTER 6 : CONCENTRATION DATA ANALYSIS IN COLORADO

Forest fires cause the release of materials from soil and vegetation. The soluble material and particulate matter is mobilized during storm events. To assess the mobilization of metals and suspended sediment, concentration data collected before and after the Hayman fire were analyzed. The Hayman fire began June 8, 2002 and was declared controlled on July 18, 2002. The fire consumed nearly 560km^2, as shown in Figure 1.3. The objective of this chapter is to establish the effect of the Hayman fire on water quality in the South Platte River, Goose Creek, and Cheesman Lake areas.

CONCENTRATION DATA ANALYSIS

The impact of the fire-induced release of various constituents was evaluated through analysis of concentration data. The constituents analyzed and sampling sites Table 6.1. The data analysis consisted of comparing pre- and post-fire data for each constituent at each sampling site. In addition, the average concentration for each constituent was compared for the four sites: South Platte River above Cheesman Lake (CLI), Goose Creek above Cheesman Lake (GC), Cheesman Lake outlet (CLO), and the South Platte River below the confluence with the North Fork River (NFSP). A map of these site locations is in Figure 3.1. However, due to the limited pre-fire data, the extent of the analysis was limited. The available data were utilized to establish the effects potentially attributed to the Hayman fire.

Table 6.1 Constituents analyzed for each site in Colorado

Constituent	CLI	GC	CLO	NFSP
Total Aluminum			X	X
Dissolved Aluminum	X	X	X	X
Total Arsenic				X
Dissolved Arsenic	X	X		X
Total Barium				X
Dissolved Barium				X
Dissolved Cadmium	X	X	X	X
Dissolved Copper	X	X	X	X
Total Iron	X	X	X	X
Dissolved Iron	X	X	X	X
Total Lead				X
Dissolved Lead		X	X	X
Total Manganese	X	X	X	X
Dissolved Manganese	X	X	X	X
Dissolved Mercury	X	X	X	X
Dissolved Selenium	X	X	X	X
Dissolved Silver	X	X		X
Total Zinc				X
Dissolved Zinc		X		X
Suspended Sediment	X	X	X	X
Stream Flow	X	X	X	X

Not all trace metals analyzed showed a response to the fire. Table 6.2 metals that did not appear to change as a result of the fire. The concentrations of these metals were consistently at or below the minimum reporting limit. Therefore, the potential impacts of the fire cannot be determined.

Table 6.2 Metals with concentrations at or below the MRL

Constituent	CLO	GC	CLI	NFSP
Dissolved Arsenic	X	X		X
Dissolved Cadmium	X		X	X
Dissolved Copper	X	X	X	X
Dissolved Lead		X	X	X
Dissolved Mercury	X	X	X	X
Dissolved Selenium	X	X	X	X
Dissolved Silver	X	X		X
Dissolved Zinc	X		X	

South Platte River Above Cheesman Lake

Stream Flow

The flow rate data for Cheesman Lake Inlet (CLI) is limited. Figure 6.1 displays the South Platte River stream flow data collected at CLI. Before the fire, the flow rate fluctuated from 2m^3/s-15 m^3/s. Since the fire, a stream flow of 2m^3/s-6m^3/s has been maintained. The slower flow rate may delay the effects of the fire since particulates are likely to settle out of suspension before reaching Cheesman Lake.

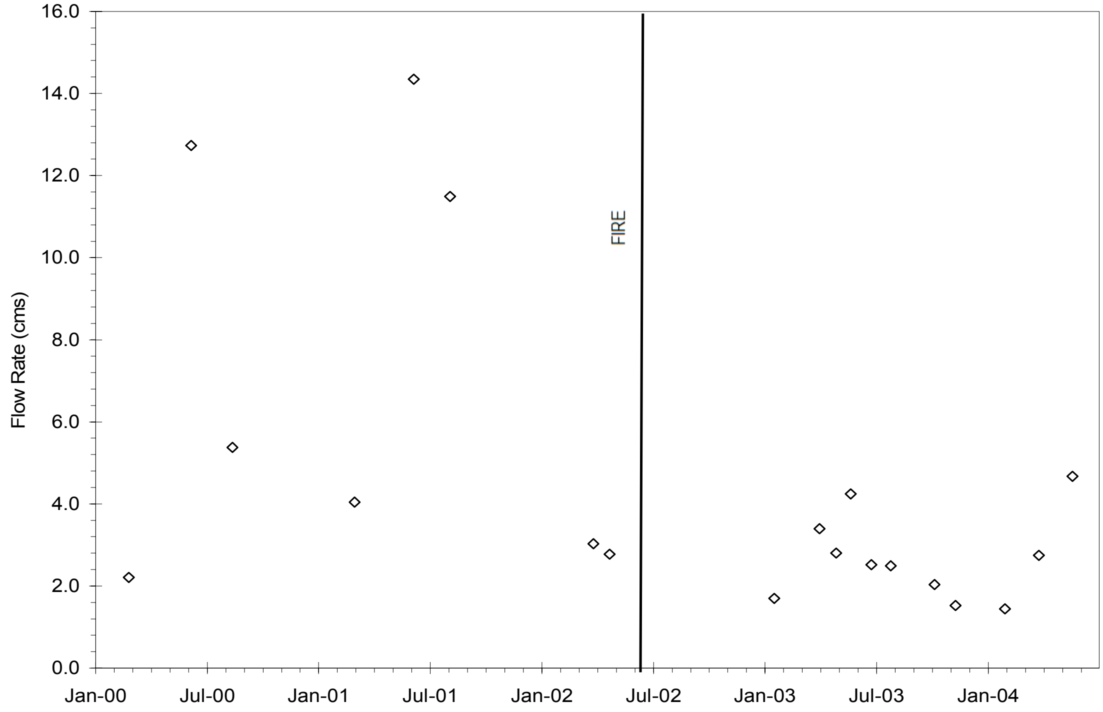

Figure 6.1 Flow rate at Cheesman Lake inlet

Suspended Sediment

During and after the Hayman fire, the total suspended sediment concentrations increased in the South Platte River above Cheesman Lake, as shown in Figure 6.2. The high suspended sediment concentrations contrast the low stream flow post-fire. The elevated particulate concentrations are a result of post-fire erosion. The lack of vegetative ground cover and presence of hydrophobic soil structure increase soil erosion (Lewis, W.M. Jr. 1974, Wondzell and King 2003, Gottfried et al. 2003, Neary 2004).

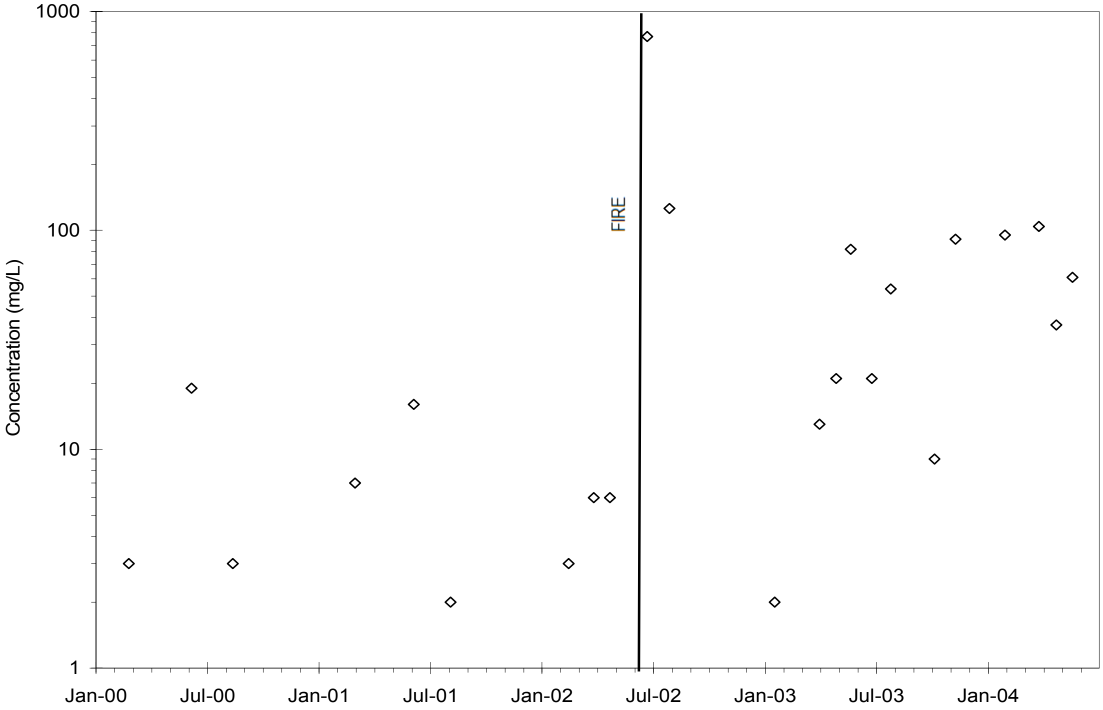

Figure 6.2 Total suspended sediment concentration at CLI

Metals

The impact of the Hayman fire on trace metals was variable. The particulate fraction for trace metals analyzed increased during or after the fire. The response of the dissolved fraction of the trace metals oscillated between no measurable change and increased concentration. However, determining the extent of increase is not possible for most trace metals due to insufficient historical data.

Dissolved and total manganese concentrations exceeded historical values during the fire, as shown in Figures 6.3 and 6.4. The total manganese concentration increased more than one order of magnitude when compared to other pre- and post-fire values. The dissolved manganese concentration was elevated to a level more than one order of magnitude higher than pre-fire values. However, an increasing trend in dissolved and total manganese concentration has occurred since 2002. This trend is consistent with the slower stream flow potentially delaying the effects of the fire. Overall, the total and dissolved manganese concentrations were elevated during the fire and have since decreased. The post-fire manganese concentration remains higher than pre-fire levels.

Dissolved and total iron concentrations are shown in Figures 6.5 and 6.6. The dissolved iron concentration did not exceed the 0.14mg/L maximum that was achieved prior to the fire and one year post-fire. The total iron concentration increased nearly two orders of magnitude when compared to pre-fire values. Since the fire, elevated levels of dissolved and total iron concentrations have been maintained. Similar to manganese, this trend is consistent with a decrease in flow rate.

Dissolved aluminum and arsenic data were collected starting in February 2002. Approximately, four months before the fire. The samples collected and analyzed for dissolved aluminum concentration prior to the fire were higher than the post-fire samples, as shown in Figure 6.7. Two years post-fire, the dissolved aluminum concentrations remain within the pre-fire range of below the minimum reporting limit (MRL) of 0.02mg/L-0.14mg/L. However, the dissolved arsenic concentration exceeded the MRL of 0.001mg/L, as shown in Figure 6.8. Since the fire, dissolved arsenic concentration has returned to the MRL. Although the dissolved arsenic concentration increased while the dissolved aluminum concentration appeared to remain unaffected by the fire, a lack of historical data prevents a complete assessment of the impacts due to the fire.

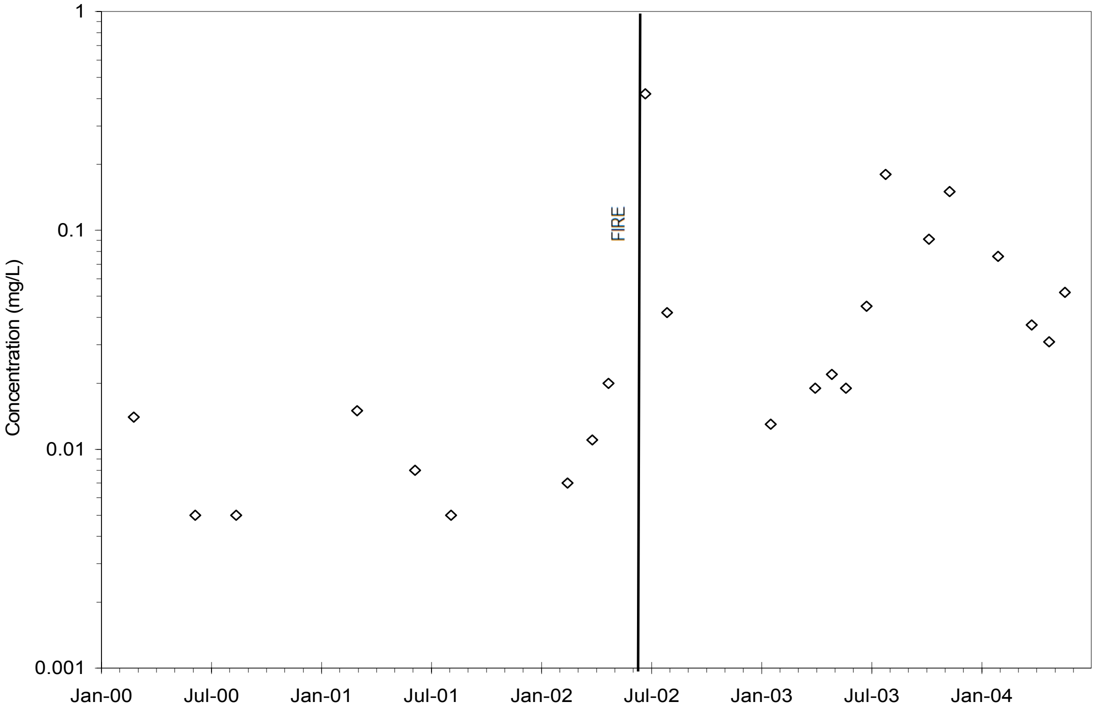

Figure 6.3 Dissolved manganese concentration at CLI

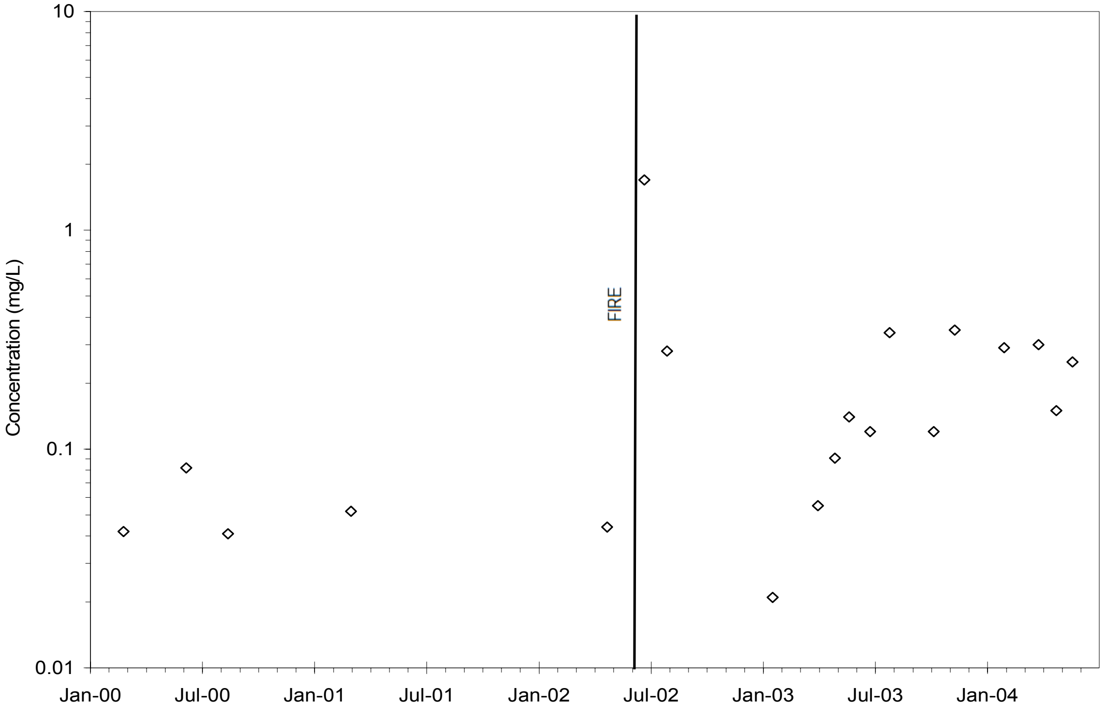

Figure 6.4 Total manganese concentration at CLI

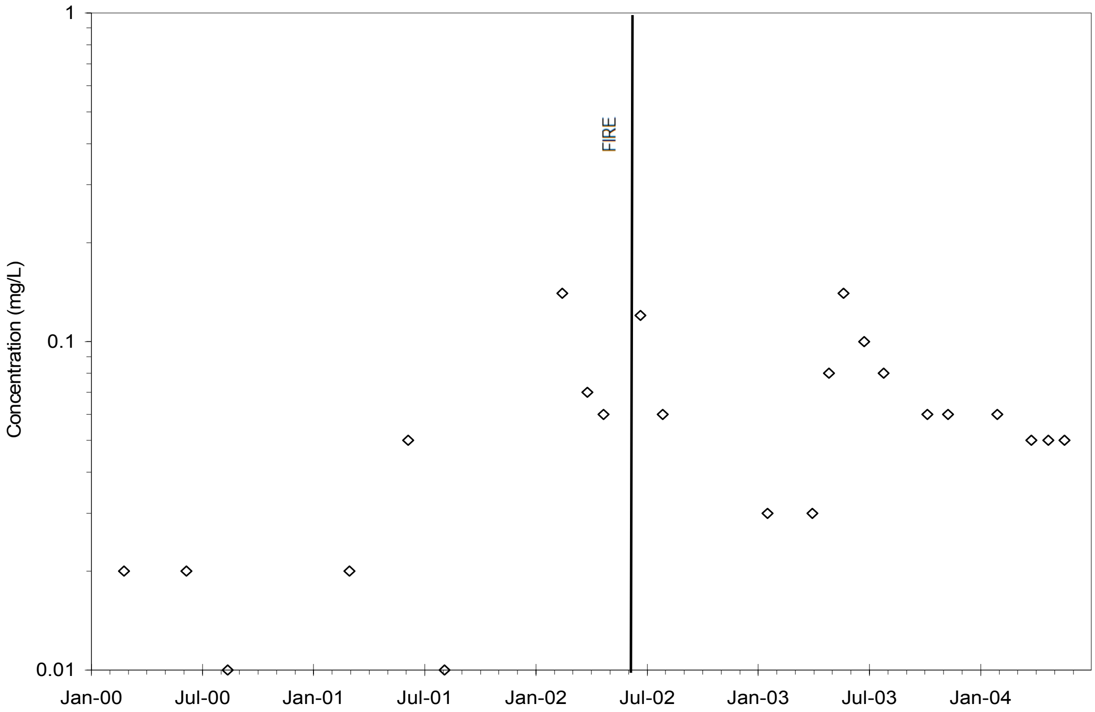

Figure 6.5 Dissolved iron concentration at CLI

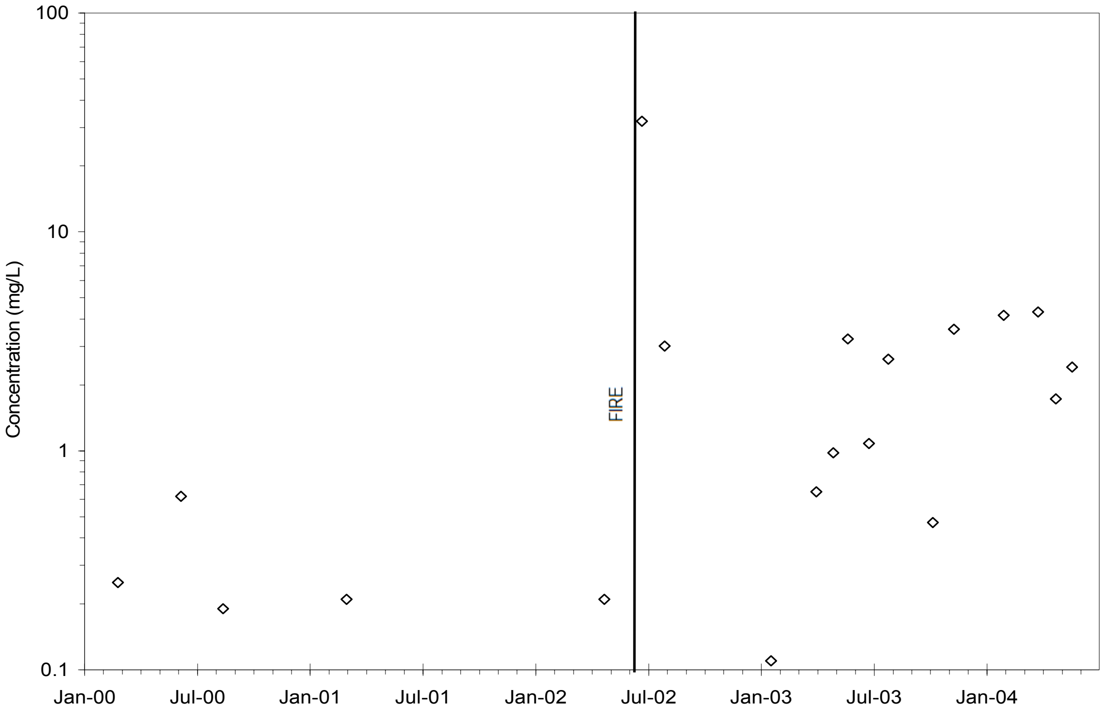

Figure 6.6 Total iron concentration at CLI

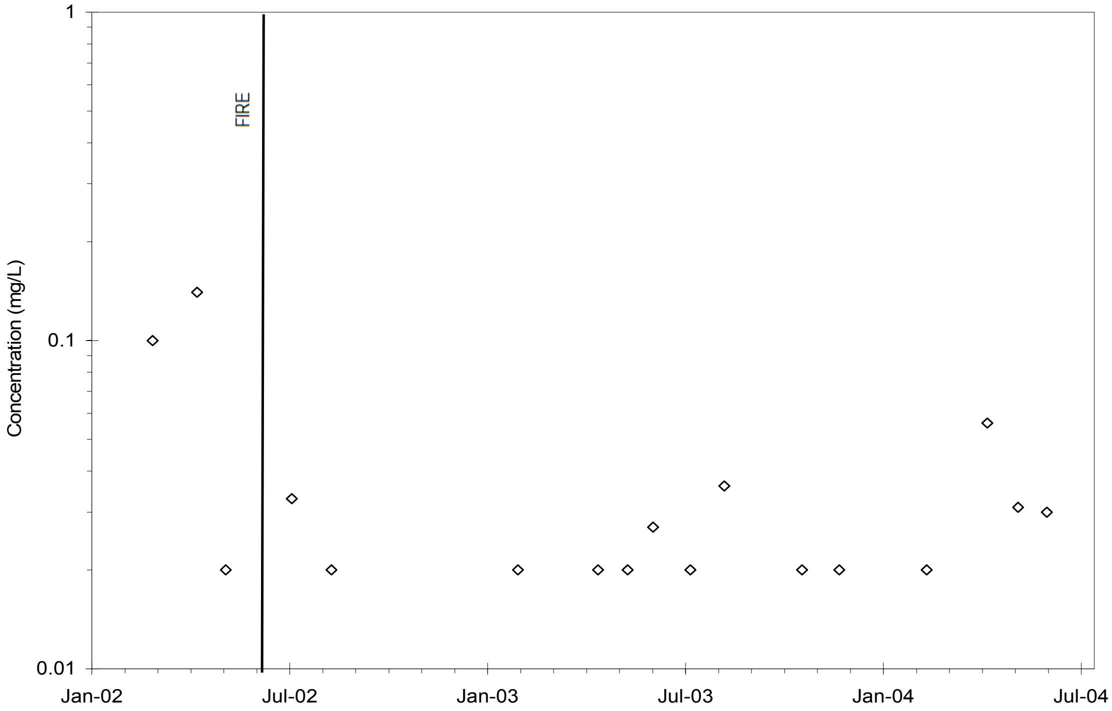

Figure 6.7 Dissolved aluminum concentration at CLI

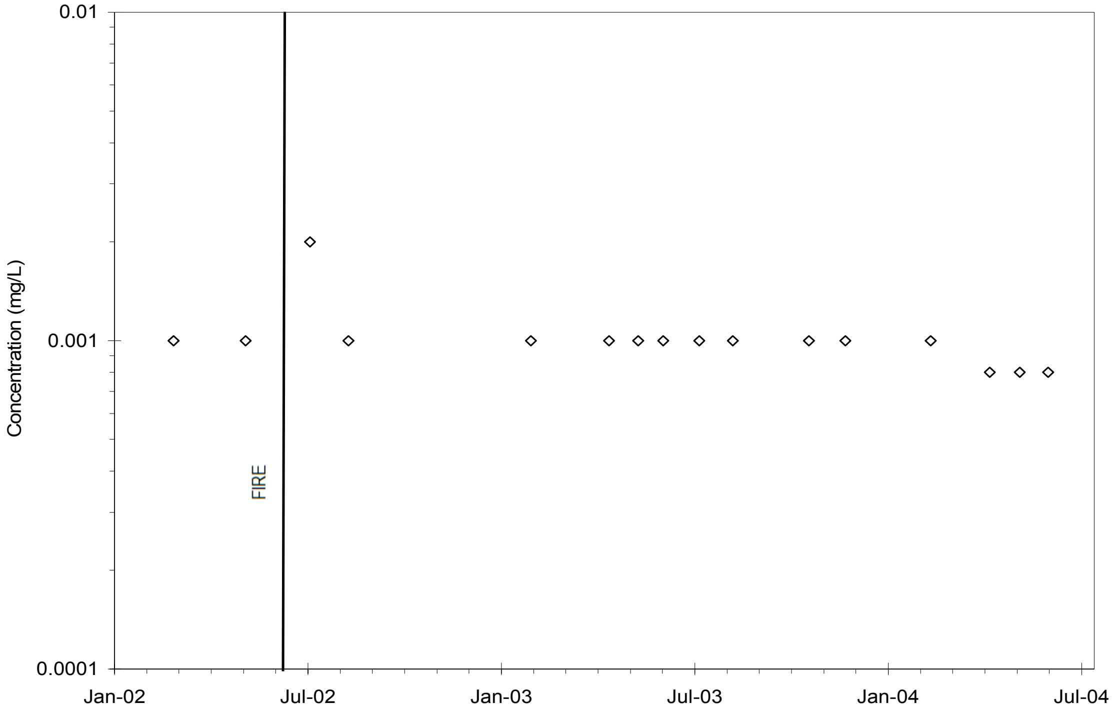

Figure 6.8 Dissolved arsenic concentration at CLI

Goose Creek

Stream Flow

Relative to the South Platte River, Goose Creek (GC) is a minor tributary to Cheesman Lake. The flow rate for GC is typically less than 1m^3/s, as displayed in Figure 6.9. This is less than the minimum South Platte River stream flow of 2m^3/s at the Cheesman Lake inlet. However, similar to the South Platte River, slower stream flow may cause a delay in the impacts of the fire. The stream flow reached a maximum of 2m^3/s approximately 11 months after the fire. A storm event is presumably the cause for the elevated flow rate.

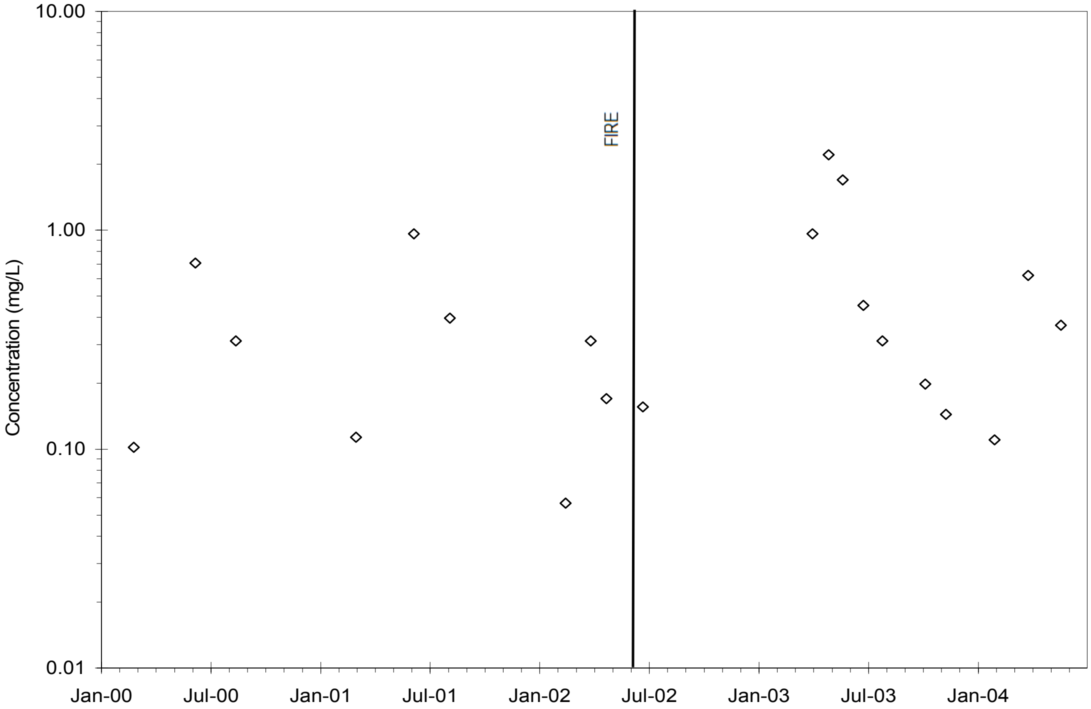

Figure 6.9 Flow rate at Goose Creek

Suspended Sediment

The total suspended sediment concentration surpassed pre-fire values by more than one order of magnitude, as shown in Figure 6.10. From January through July of 2002, the flow rate in Goose Creek was less 0.5m^3/s. Since the flow rate did not increase, but the concentration of particulates within Goose Creek did rise, the elevated concentration can be attributed to erosion caused by hydrophobic soil and a lack of vegetation and litter to hold soil in place. Two years post-fire elevated particulate concentrations are still present even though the stream flow is comparable to pre-fire rates.

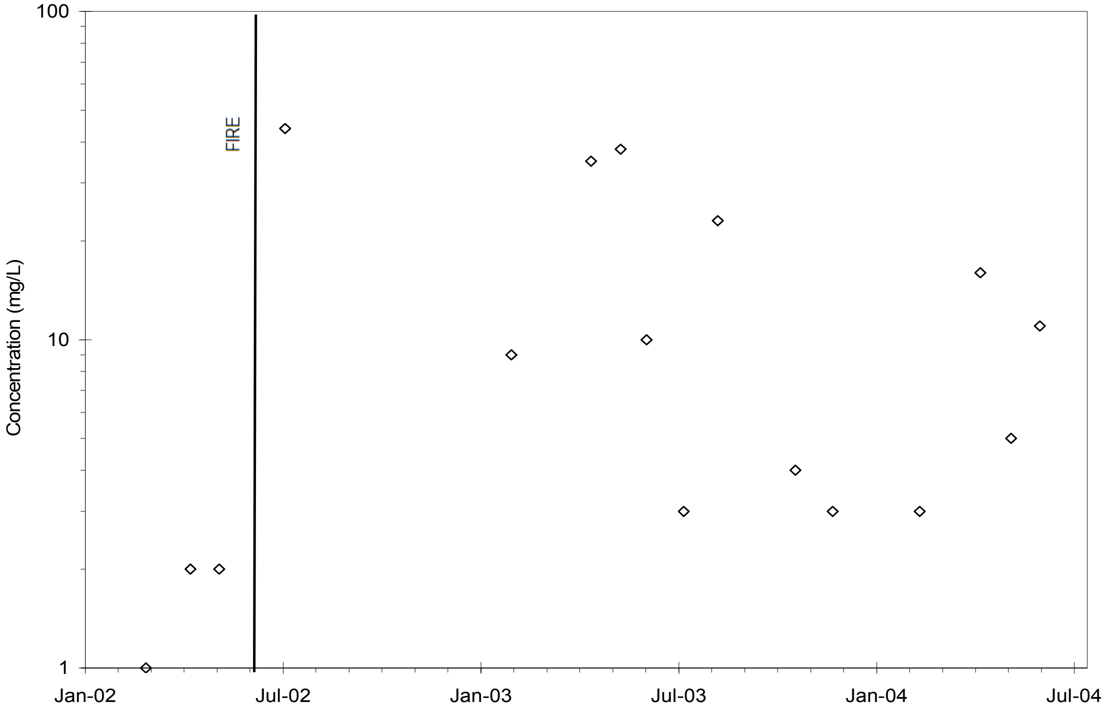

Figure 6.10 Total suspended sediment concentration at GC

Metals

The Hayman fire did not impact Goose Creek water quality to the extent observed at CLI. The dissolved fraction for most trace metals analyzed remained within pre-fire limits. Total iron was the only metal analyzed that included the particulate fraction, which was potentially impacted by the fire.

The dissolved manganese concentration displayed a unique post-fire behavior, as shown in Figure 6.11. The concentration surpassed pre-fire levels by nearly two orders of magnitude and the levels have remained high relative to pre-fire levels. The total manganese concentration has the same trend as the dissolved fraction, as shown in Figure 6.12. In addition, the post-fire total manganese concentration has remained high when compared to pre-fire levels. The dissolved fraction comprises a larger portion of the total manganese concentration than the particulate fraction.

Total iron concentration exceeded pre-fire values, while the dissolved fraction responded in a similar manner as dissolved aluminum. The dissolved iron concentration during the fire was not elevated beyond pre-fire levels, as displayed in Figure 6.13. However, a gradually increasing trend in dissolved iron concentration began after the fire. Total iron concentration was elevated during the fire, as shown in Figure 6.14. This was primarily attributed to an increase in the particulate fraction, which is consistent with higher suspended sediment concentrations. Similar to the post-fire concentration of the dissolved fraction, the total iron concentration remains elevated relative to the pre-fire concentration data.

The dissolved concentrations of aluminum, cadmium, and zinc showed no response to the fire. The dissolved aluminum concentration varied from below the MRL of 0.02mg/L-1mg/L prior to the fire, as show in Figure 6.15. The post-fire concentration has remained within these pre-fire concentration boundaries. The pre-fire dissolved cadmium concentration limits were below the MRLof 0.0001mg/L-0.005mg/L, as shown in Figure 6.16. The concentration has remained within this range since the fire. The pre-fire limits for dissolved zinc ranged between 0.005mg/L-0.01mg/L, as displayed in Figure 6.17. The post-fire concentrations have stayed within these pre-fire limits. Overall, the fire did not impact the dissolved fraction of aluminum, cadmium, and zinc at the GC sampling site.

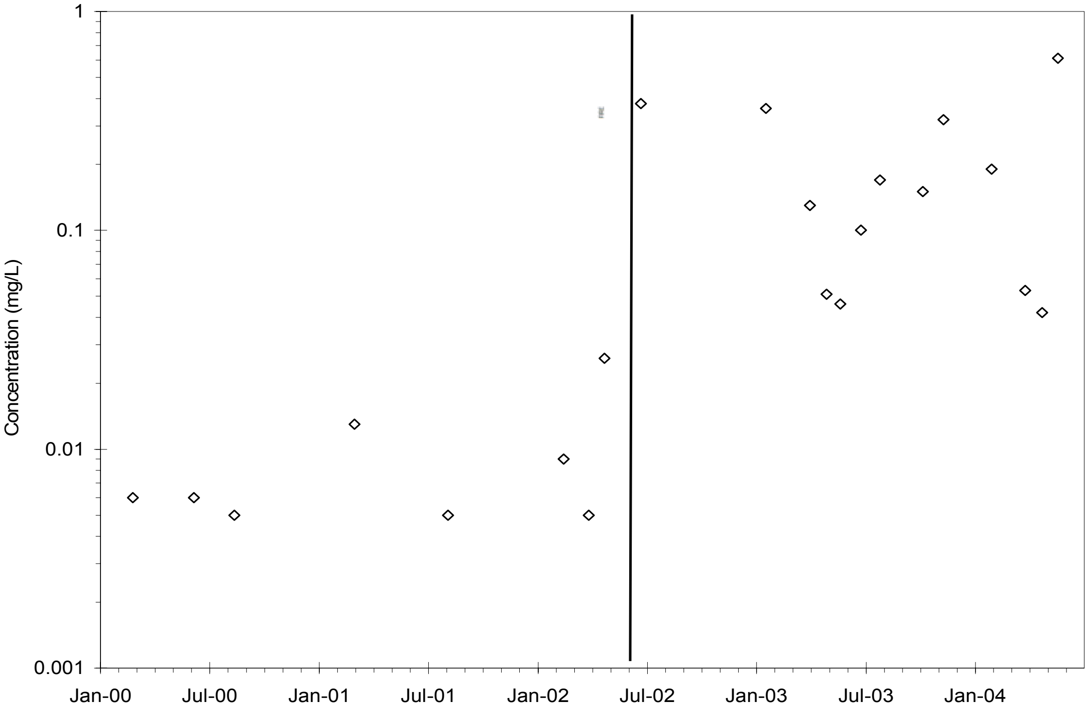

Figure 6.11 Dissolved manganese concentration at GC

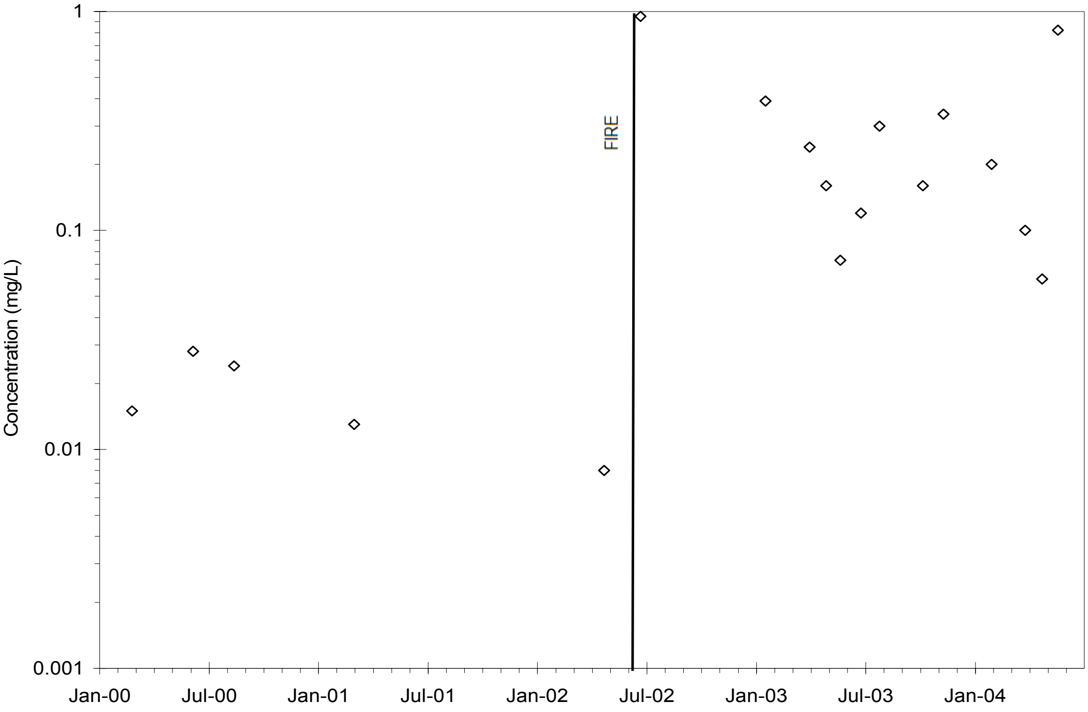

Figure 6.12 Total Manganese concentration at GC

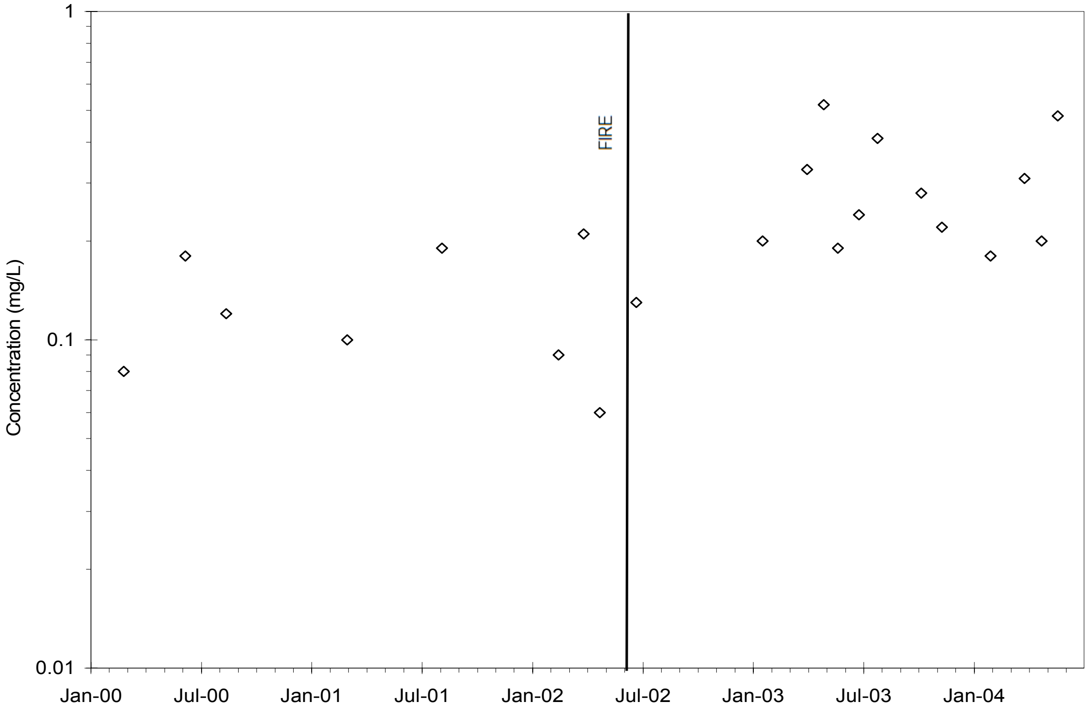

Figure 6.13 Dissolved iron concentration at GC

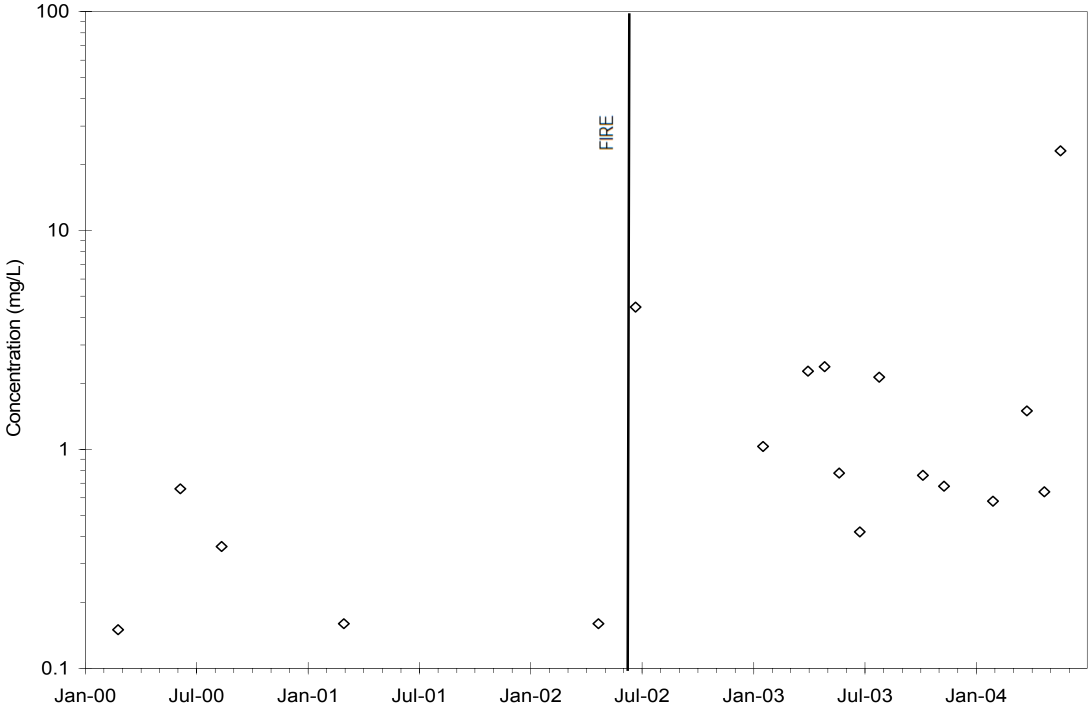

Figure 6.14 Total iron concentration at GC

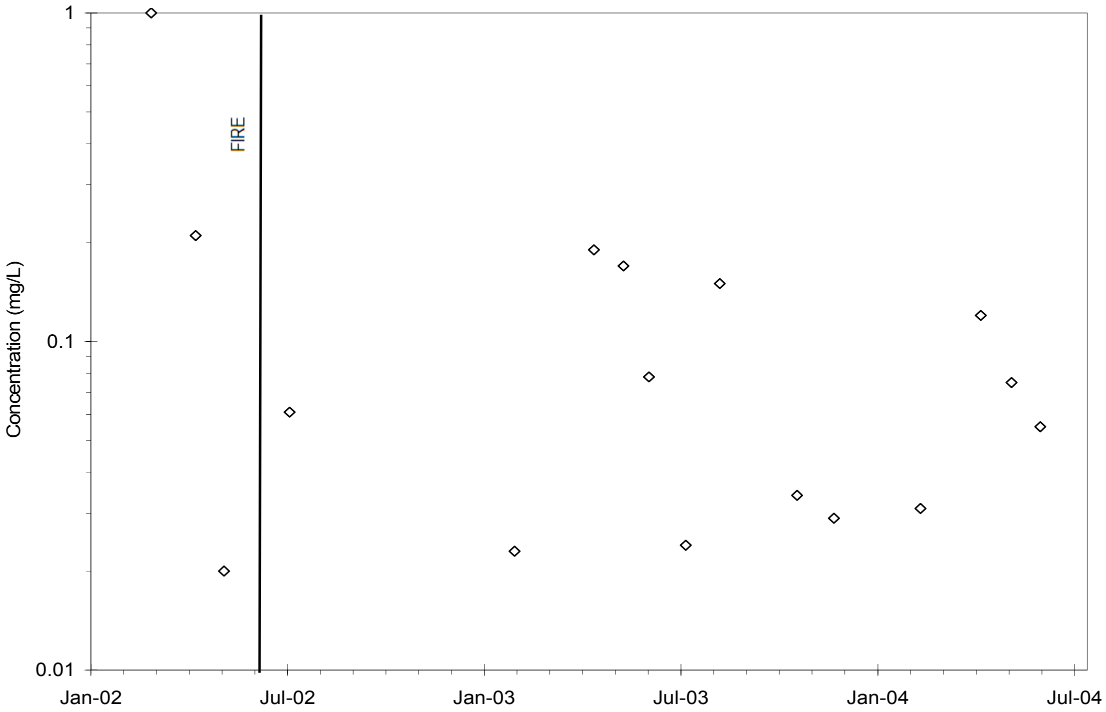

Figure 6.15 Dissolved aluminum concentration at GC

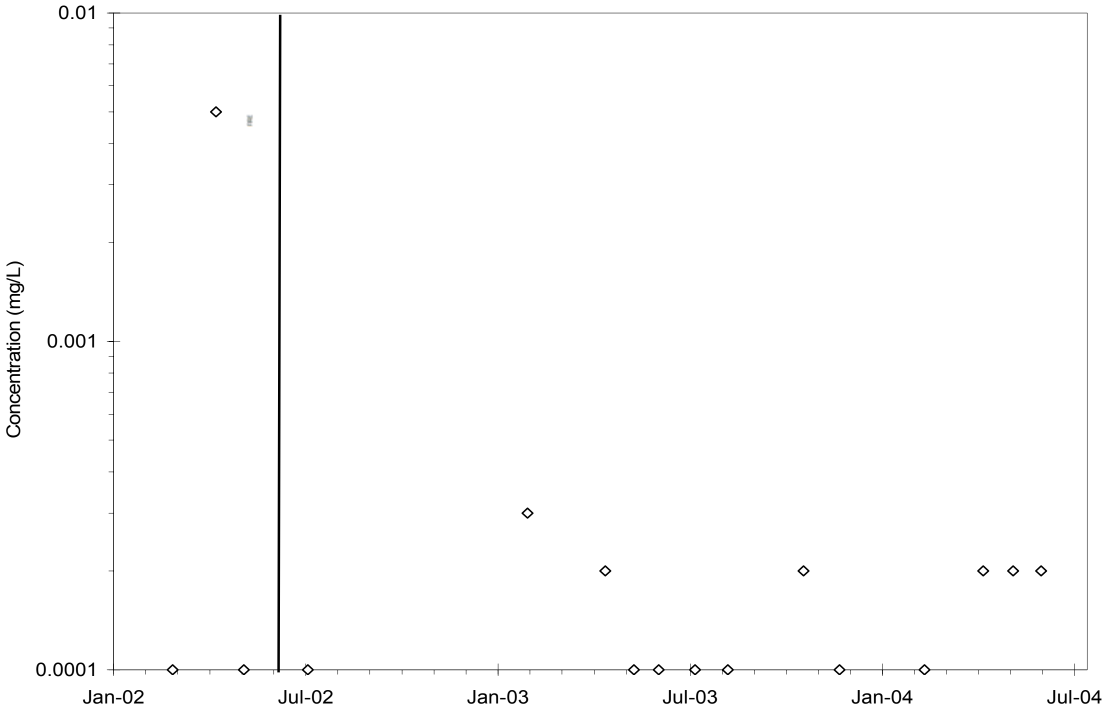

Figure 6.16 Dissolved cadmium concentration at GC

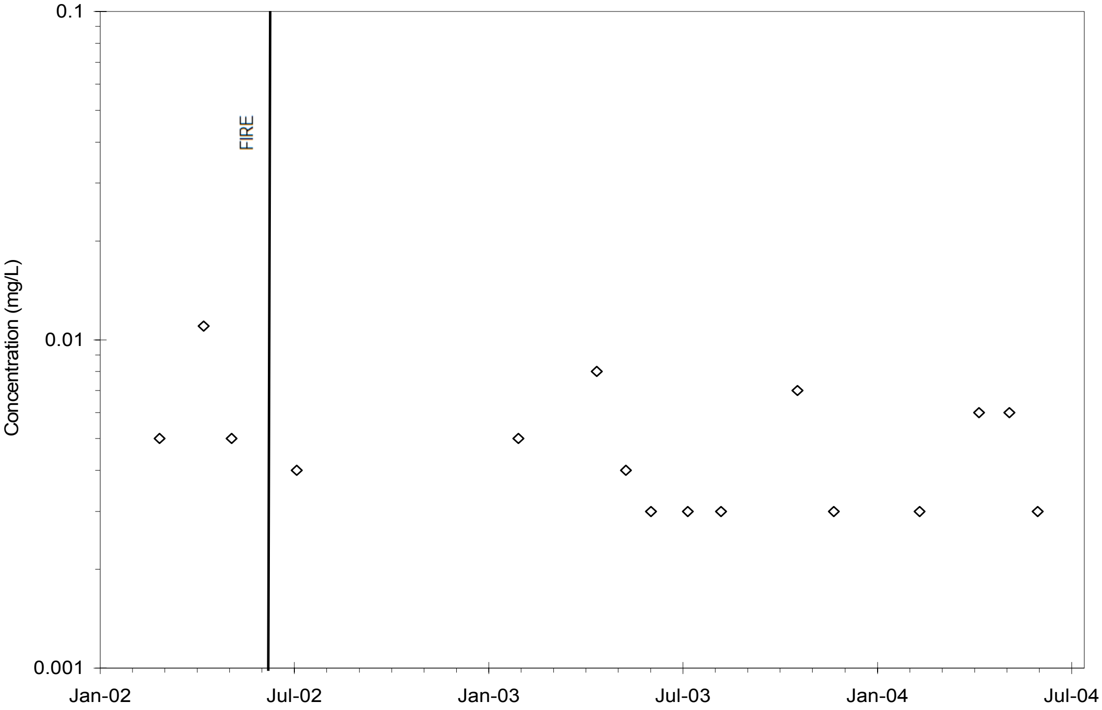

Figure 6.17 Dissolved zinc concentration at GC

Cheesman Lake Outlet

Stream Flow

Utilizing the flow rate data for the South Platte River at Cheesman Lake outlet, the hydrograph in Figure 6.18 was created. Before the fire, the flow rate fluctuated from 0.05m^3/s-130 m^3/s. During the fire approximately 10m^3/s was being released from Cheesman Lake. The post-fire stream flow range is 1m^3/s-16 m^3/s. The stream flow at this point on the South Platte River is controlled by the amount of water being released from the dam.

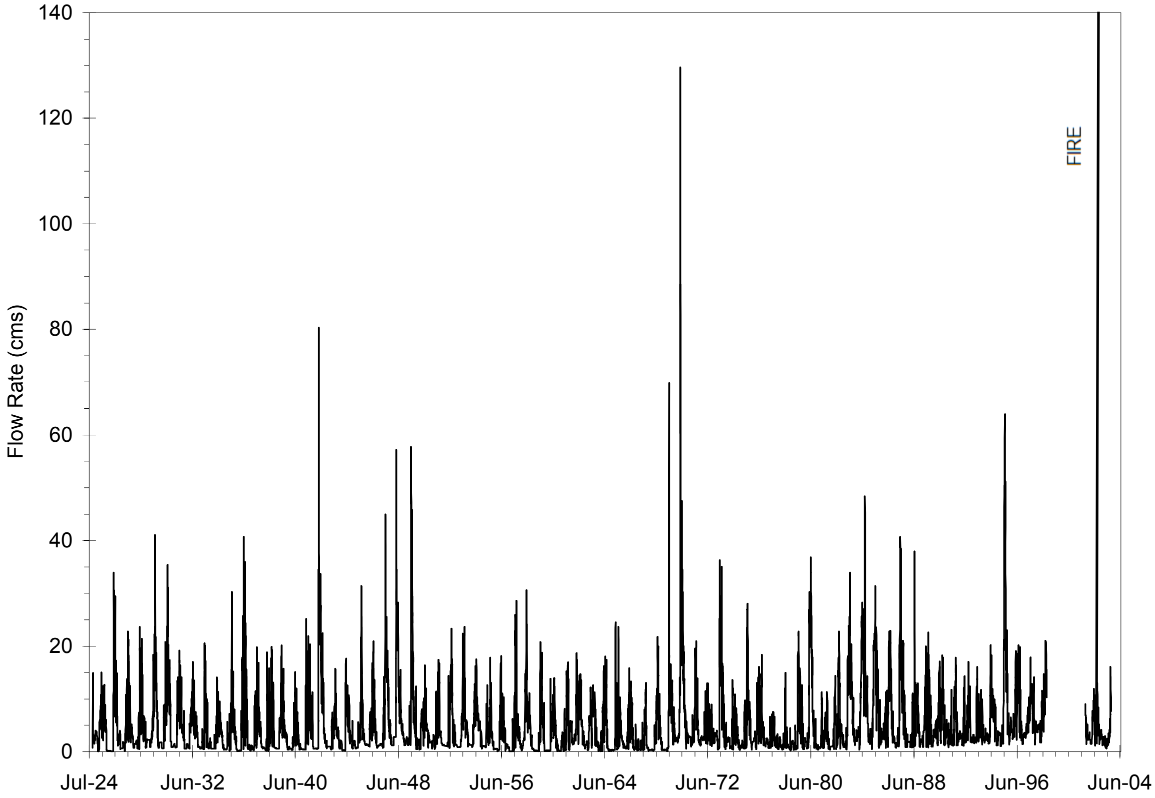

Figure 6.18 Flow rate at CLO
(*Source:* USGS)

Suspended Sediment

Since this sampling site is below Cheesman Lake, the suspended sediment concentration is expected to be low due to particulates settling to the bottom of the lake. However, during the Hayman fire the total suspended sediment concentrations increased more than one order of magnitude in the South Platte River below Cheesman Lake, as shown in Figure 6.19. The post-fire suspended sediment concentration returned to pre-fire levels soon after the fire. The increase in suspended sediment concentration during the fire is the result of an excess amount of particulates being deposited into Cheesman Lake and its tributaries. A portion of these particulates were not retained in the lake for a sufficient amount of time to settle to the bottom, therefore increasing the concentration at CLO.

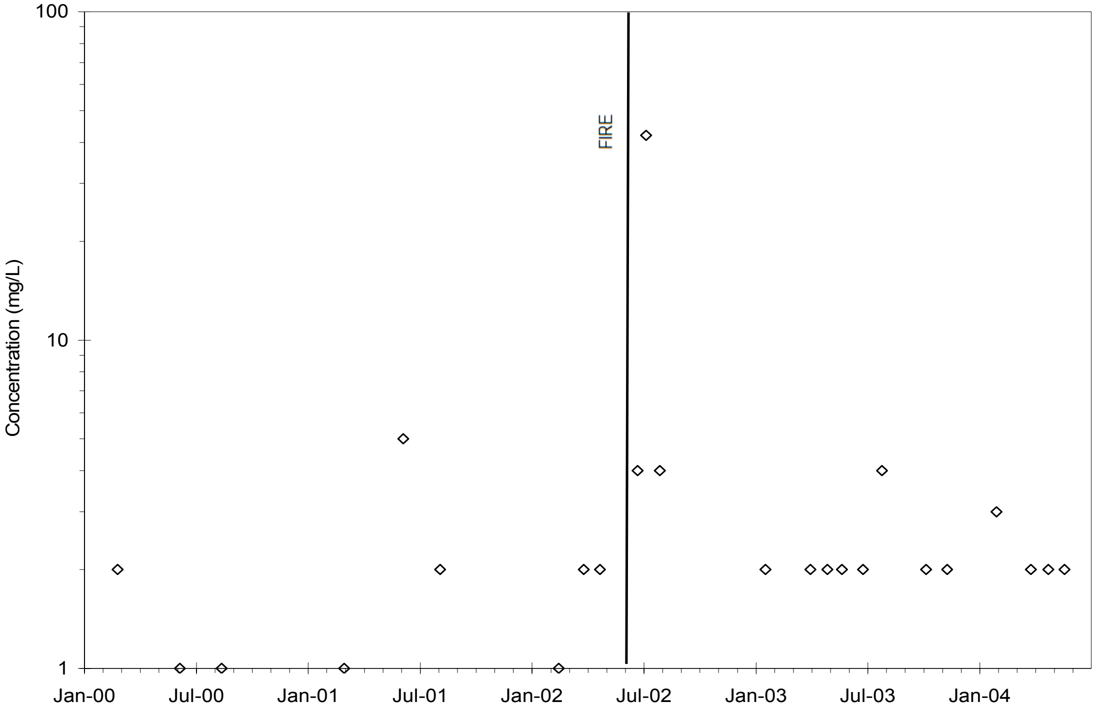

Figure 6.19 Total suspended sediment concentration at CLO

Trace Metals

Trace metal concentrations at CLO showed varying trends during and after the Hayman fire. The outlet concentrations are dependent upon the Goose Creek and South Platte River inlet concentrations. The dissolved and particulate fraction of trace metals analyzed at CLO displayed concentration increases. However, the particulate fraction for the trace metals was more responsive to changes in concentration.

The total and dissolved iron concentrations responded differently to the fire. The dissolved iron concentration was 33% higher than the MRL of 0.06mg/L, as shown in Figure 6.20. Since the fire, the dissolved iron concentration has remained at or below the MRL. The total iron concentration exceeded pre-fire values by nearly two orders of magnitude, as displayed in Figure 6.21. This increase was due to mobilization of the particulate fraction which correlates with the higher total suspended sediment concentration. The South Platte River and Goose Creek both contributed to the elevated concentration. Post-fire total iron concentrations remain above pre-fire levels at CLO. This result is unexpected since the suspended sediment and dissolved iron concentrations have remained at pre-fire levels.

In contrast to total iron, total manganese was impacted by the dissolved fraction. The dissolved manganese concentration ranged from 0.005mg/L-0.25mg/L prior to the fire, as shown in Figure 6.22. During the fire, dissolved manganese exceeded the maximum pre-fire value by 100%. However, two years post-fire the concentration was 152% higher than the pre-fire maximum concentration. Total manganese surpassed the pre-fire high by 177%, as shown in Figure 6.23. Total manganese during the fire consisted of 80% dissolved manganese. Two years post-fire, the dissolved fraction is approximately 90% of the total concentration. The increasing trend in the dissolved fraction is consistent with the elevated concentrations recorded at CLI and GC.

The dissolved aluminum concentration did increase after the Hayman fire, as shown in Figure 6.20. During the fire, the concentration measured was above the MRL of 0.01mg/L at 0.078mg/L, but it did not surpass the pre-fire maximum of 0.12mg/L. Prior to the fire and the year following the fire the dissolved aluminum concentration was at or below the MRL. The elevated concentration during the fire is attributed to the dissolved aluminum entering Cheesman Lake from Goose Creek.

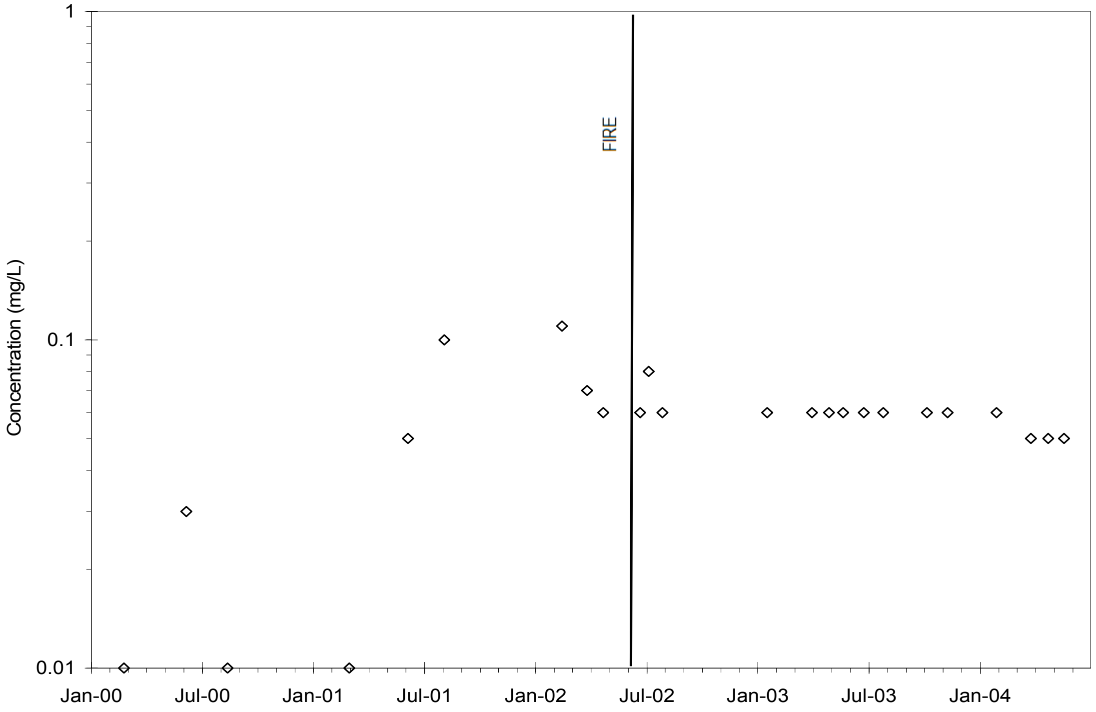

Figure 6.20 Dissolved iron concentration at CLO

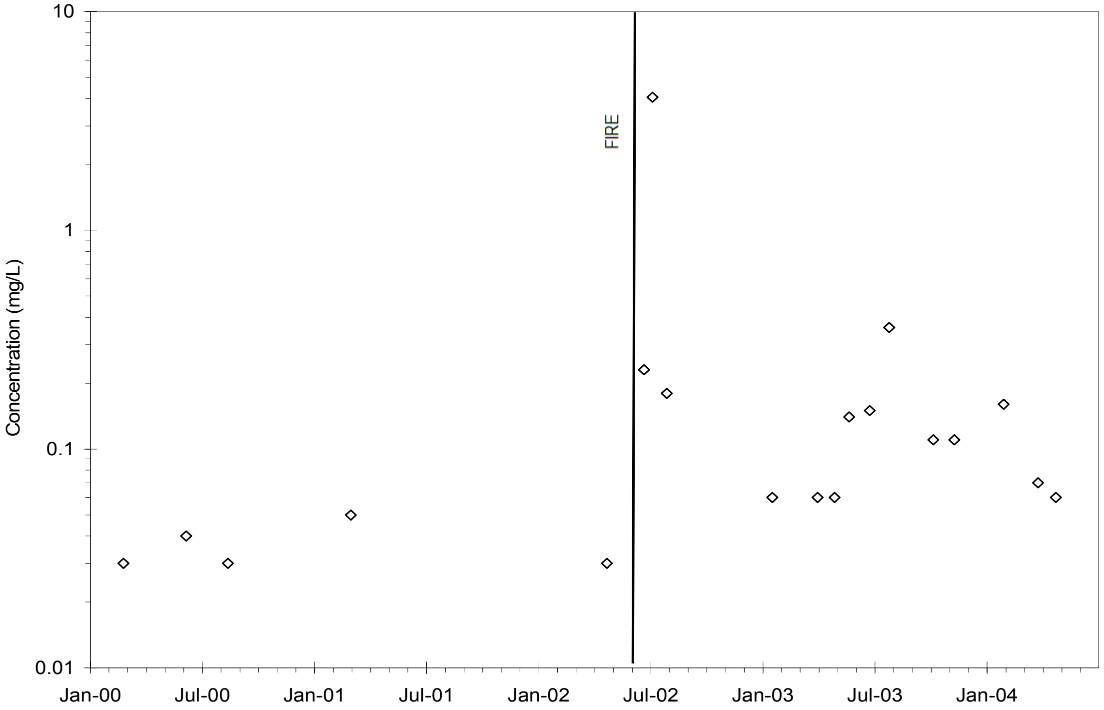

Figure 6.21 Total iron concentration at CLO

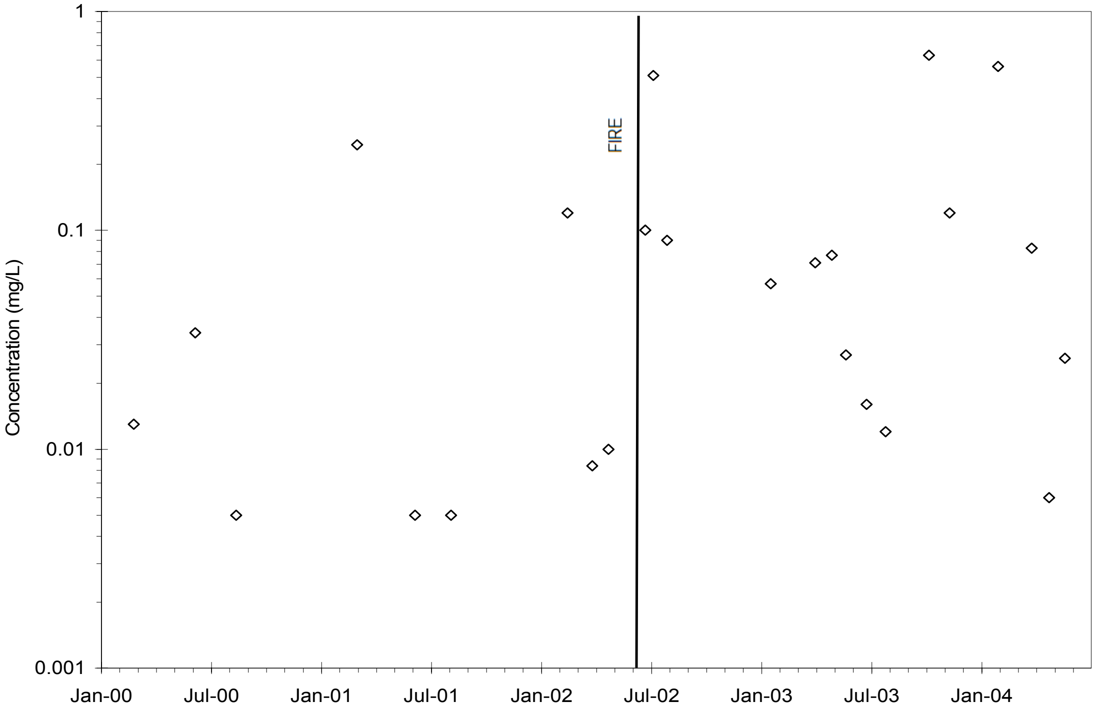

Figure 6.22 Dissolved manganese concentration at CLO

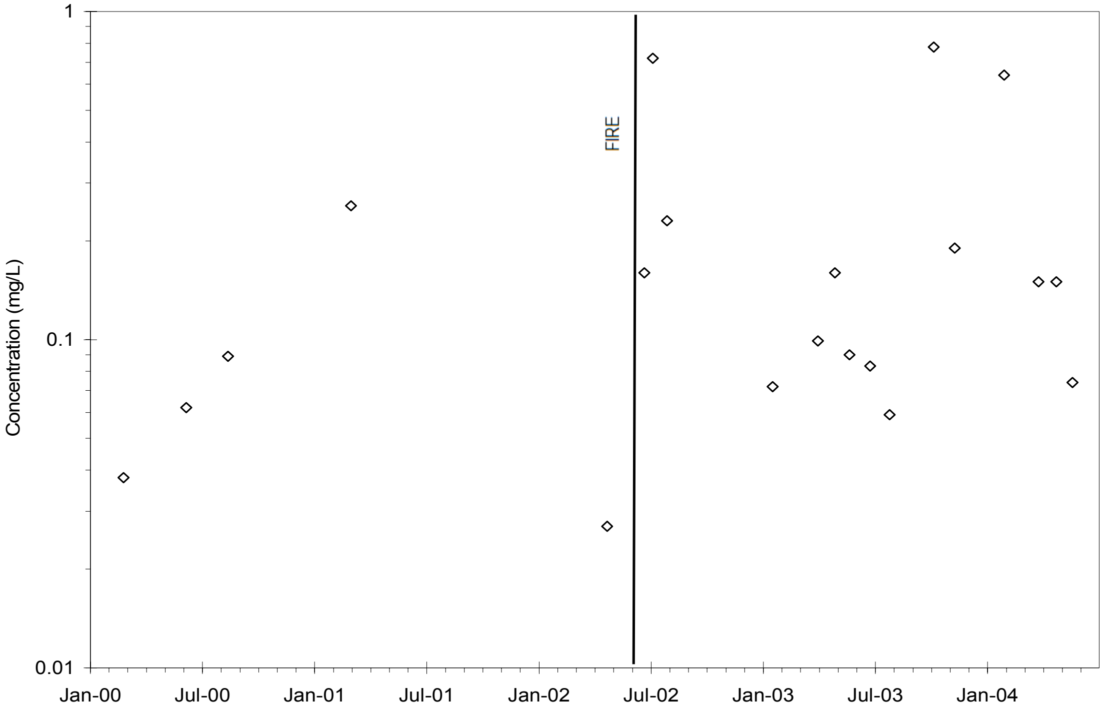

Figure 6.23 Total manganese concentration at CLO

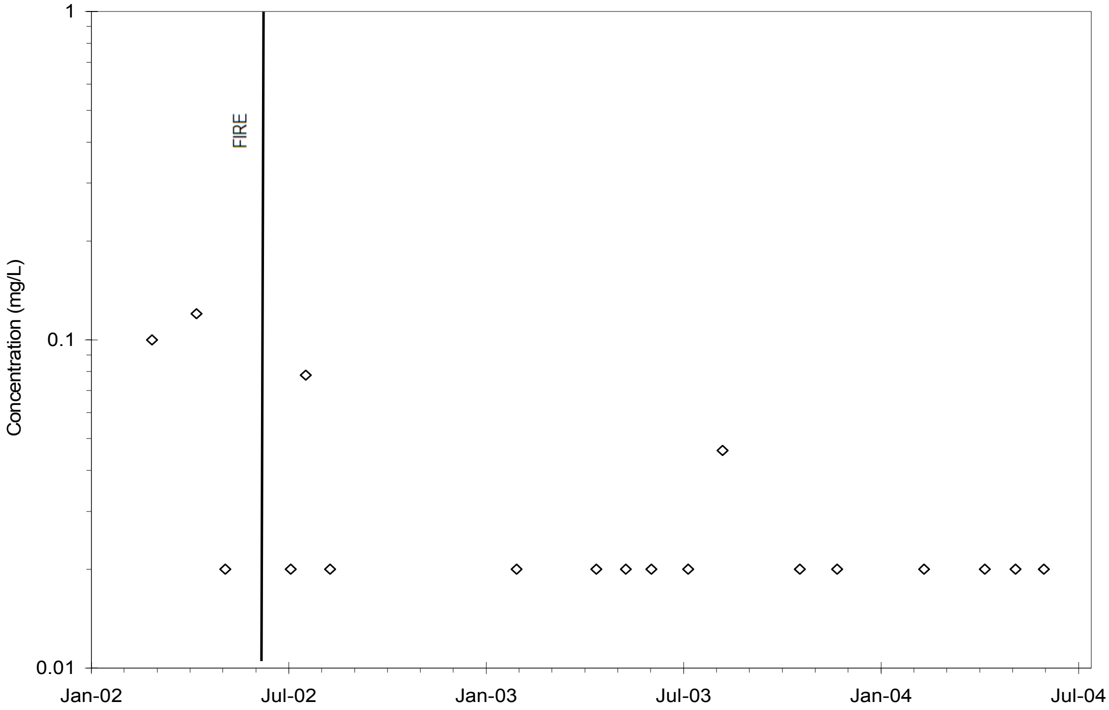

Figure 6.24 Dissolved aluminum concentration at CLO

Below the Confluence of the North Fork River and the South Platte River

The flow rate at a location approximately one mile downstream from the confluence of the North Fork and South Platte River (NFSP) is displayed in Figure 6.25. The peak stream flow of $18m^3/s$-$25m^3/s$ is reached during the months of June, July, and August. The lowest flow rate range is $3m^3/s$-$10m^3/s$ and occurs between December and February. The stream flow ranged between $15m^3/s$-$20m^3/s$ during the Hayman fire in June and July.

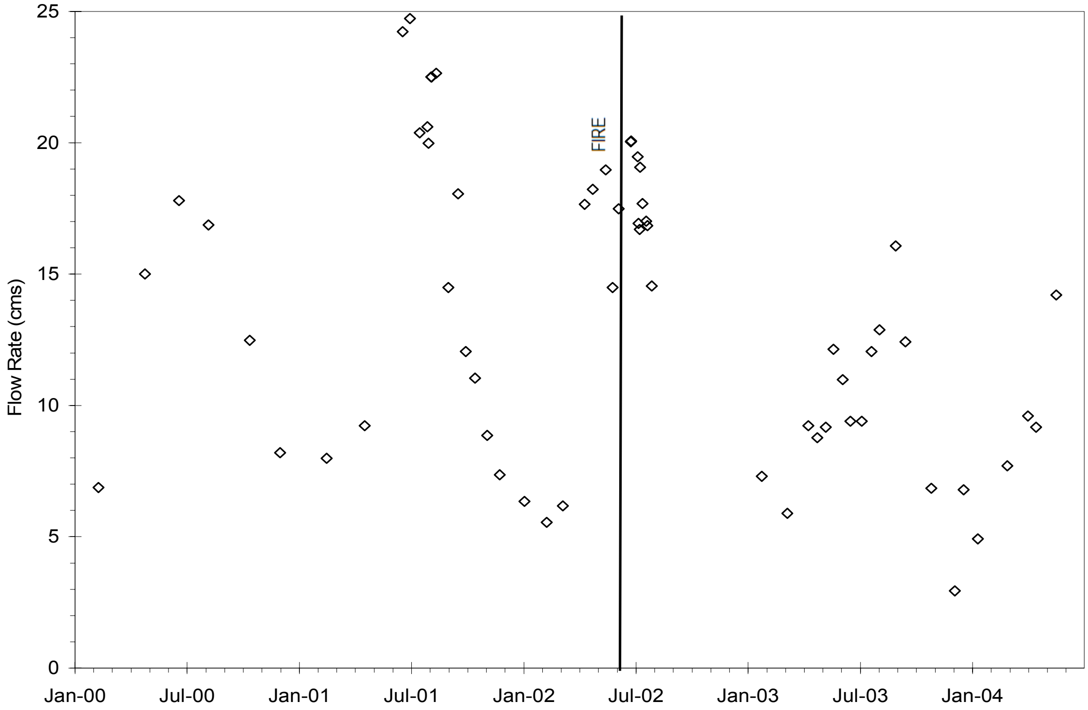

Figure 6.25 Flow rate at the confluence of the North Fork and South Platte River

Suspended Sediment

The suspended sediment concentration for NFSP is shown in Figure 6.26. Prior to the fire, particulate concentration ranged from 1mg/L–100mg/L. A peak concentration of 1200mg/L was achieved 1.5 months after the fire began. The concentration of the suspended sediment then returned to the pre-fire range. However, one year after the particulate concentration initially peaked following the fire, a concentration of 460mg/L was measured. This concentration is high relative to the typical suspended sediment levels in the river. Therefore, the increase in the suspended sediment concentration could be due to other events occurring within the North Fork River portion of the watershed. In addition, the high post-fire suspended sediment concentrations occur at stream flows less than the flow rate one year pre-fire.

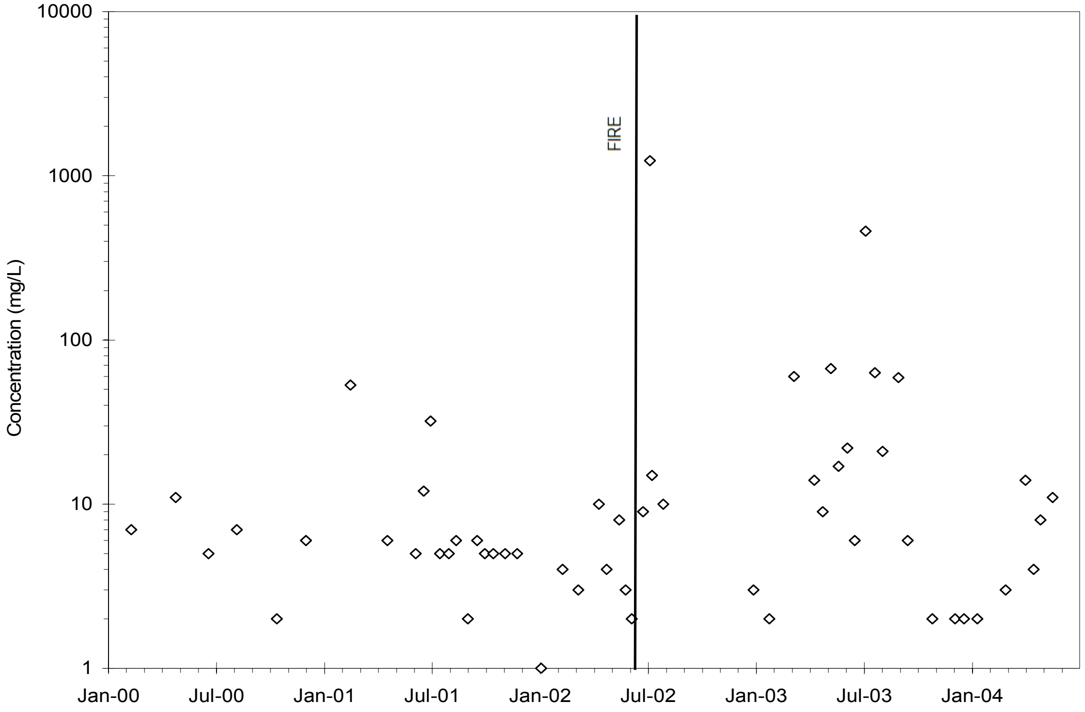

Figure 6.26 Total suspended sediment concentration at NFSP

Metals

The trace metals analyzed responded differently to the fire at NFSP. The concentrations of some metals remained within pre-fire concentration ranges, while other metals were elevated in concentration after the fire. However, most total trace metals did set a new maximum one year post-fire. This trend was observed for several dissolved trace metals as well.

The dissolved and total aluminum concentration displayed different behavior immediately after the fire and one year post-fire, as shown in Figures

6.27 and 6.28. The dissolved aluminum concentration stayed within the pre-fire range of 0.02mg/L-0.1mg/L immediately after the fire. However, one year after the fire the dissolved fraction of aluminum reached a maximum concentration of 0.28mg/L. This is one order of magnitude greater than the highest pre-fire concentration recorded. Unlike dissolved aluminum immediately after the fire, the total aluminum concentration deviated from the pre-fire range of 0.07mg/L-0.7mg/L immediately after the fire. The total aluminum concentration was elevated to 36mg/L, which is greater than two orders of magnitude higher than pre-fire values. This was due to an increase in the particulate fraction, which correlates with an increase in suspended sediment concentration during this time period. Unlike the dissolved aluminum concentration, total aluminum did not peak one year post-fire.

Post-fire trends in the dissolved and total barium were similar 1.5 months after the fire, but very different one year post-fire. The dissolved barium concentration stayed within a narrow range of 0.038mg/L-0.043mg/L before the fire, as shown in Figure 6.29. The post-fire dissolved barium concentration increased by 39%. Total barium also stayed within a narrow pre-fire concentration range of 0.044mg/L-0.05mg/L, as shown in Figure 6.30. However, immediately post-fire the particulate fraction caused total barium to reach 0.81mg/L, which is an increase of one order of magnitude. A similar increase in concentration occurred one year post-fire. In addition, dissolved barium was at

0.025mg/L one year post-fire, which is abnormally low. In general, the barium concentration is governed by the particulate fraction.

Total arsenic and total lead behaved similarly to total iron and total barium. Total arsenic concentration reached 0.01mg/L immediately after the fire, as shown in Figure 6.31. This same concentration was achieved one year post-fire. In all other instances, the total arsenic concentration remained at or below the MRL of 0.001mg/L. Total lead concentration was below the MRL of 0.002mg/L before the fire, as shown in Figure 6.32. However, after the fire the upper concentration limit shifted to 0.01mg/L. Also, the total lead concentration was elevated immediately after the fire to 0.24mg/L and to 0.31mg/L one year post-fire. Although the dissolved concentrations of arsenic and lead are not known, the elevated concentrations are likely due to the particulate fraction because the increases correlate with the post-fire suspended sediment trends.

Dissolved and total zinc responded differently to potential fire impacts. Prior to the Hayman fire, the dissolved concentration of zinc varied from 0.003mg/L-0.01mg/L, as shown in Figure 6.33. Starting two days after the Hayman fire began the dissolved concentration increased to 0.044mg/L. For this reason, the concentration increase cannot be attributed to the fire because NFSP is located downstream. After the fire, the upper concentration limit shifted to 0.015mg/L, the dissolved concentration has since remained below this limit. Similar to total iron and total barium, the total zinc concentration was elevated twice; once immediately after the fire and again one year post-fire, as shown in

Figure 6.34. The first elevated concentration was 0.58mg/L, which exceeds the pre-fire range of 0.01mg/L-0.02mg/L by one order of magnitude. The second elevated concentration was 0.84mg/L, a 44% increase beyond the previous maximum. The increase in the particulate fraction for total zinc is consistent with changes in suspended sediment concentration following the fire.

A strong correlation between dissolved and total manganese concentration exists. This correlation is displayed in Figure 6.35. Pre-fire dissolved manganese ranged from 0.005mg/L-0.03mg/L. The total manganese concentration range was 0.02mg/L-0.1mg/L. Immediately after the fire, the dissolved manganese concentration was 0.5mg/L, surpassing the upper limit by one order of magnitude. The total manganese concentration also increased by one order of magnitude to 4.2mg/L. One year post-fire another increase of one order of magnitude in the total manganese concentration occurred. This correlates with the elevated suspended sediment concentration during this time period.

Dissolved iron concentration displayed unique pre-fire behavior, while the total iron concentration showed trends similar to total barium. Dissolved iron increased more than 100% one year before the fire, as shown in Figure 6.36. Initially, the concentration range was 0.02-0.03mg/L. Within two months the concentration increased and a new range of 0.05mg/L-0.1mg/L was established. The dissolved iron concentration immediately after the fire did not deviate from the new concentration range. However, one year post-fire dissolved iron increased by 200%. The total iron concentration ranged from 0.1mg/L-1mg/L prior to the fire and strongly correlates with the suspended sediment concentration, as shown in Figure 6.37. Immediately following the fire and one year post-fire the total iron concentration was elevated to 59mg/L and 81mg/L, respectively. This is an increase of two orders of magnitude relative to pre-fire concentration limits.

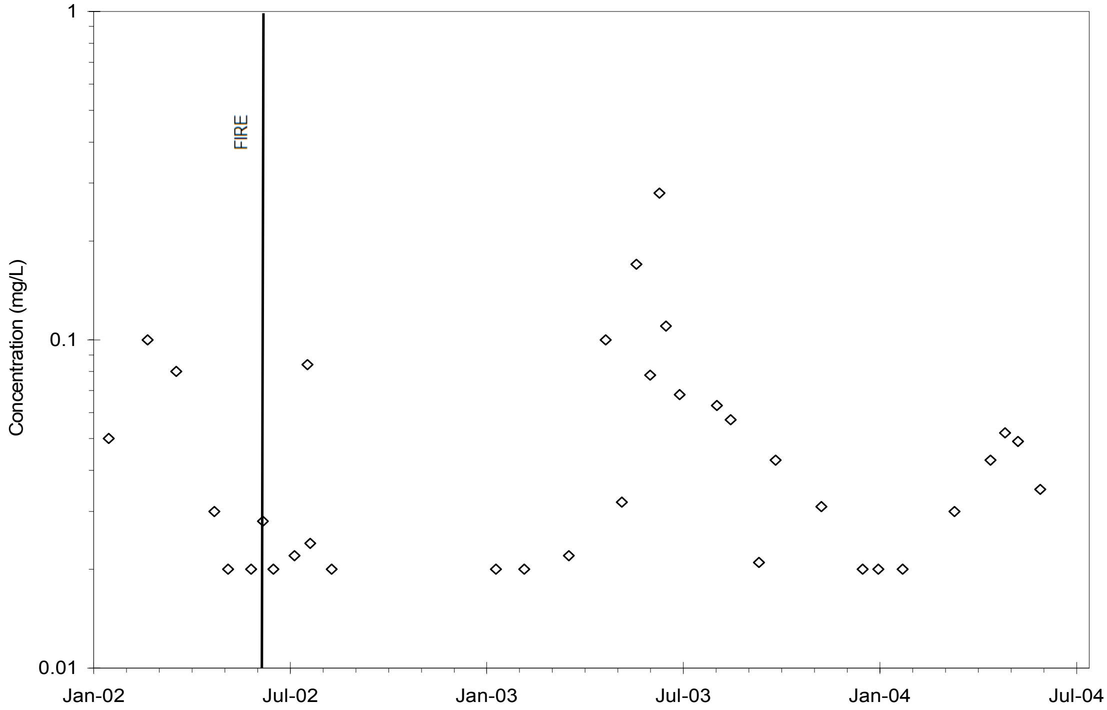

Figure 6.27 Dissolved aluminum concentration at NFSP

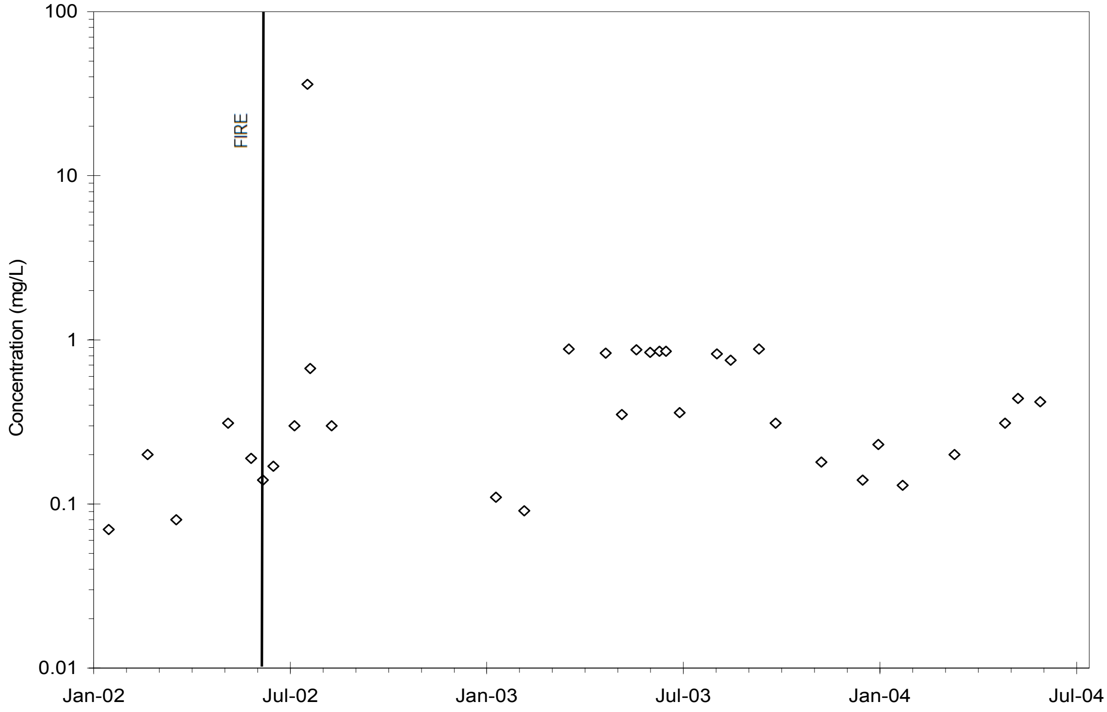

Figure 6.28 Total aluminum concentration at NFSP

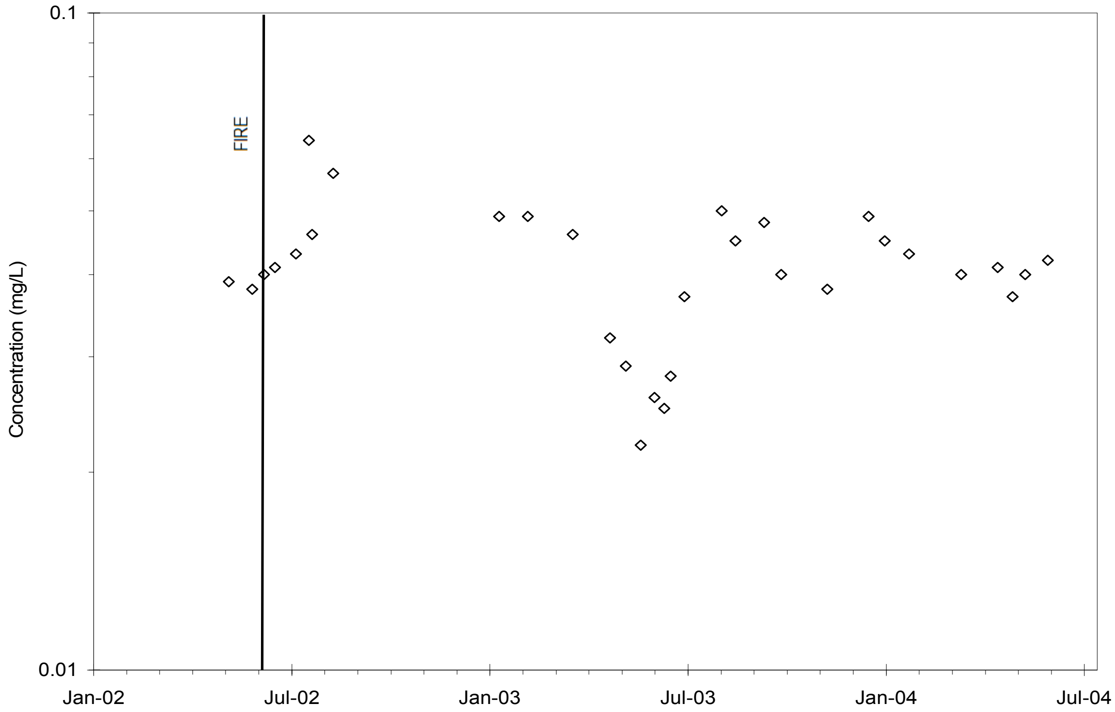

Figure 6.29 Dissolved barium concentration at NFSP

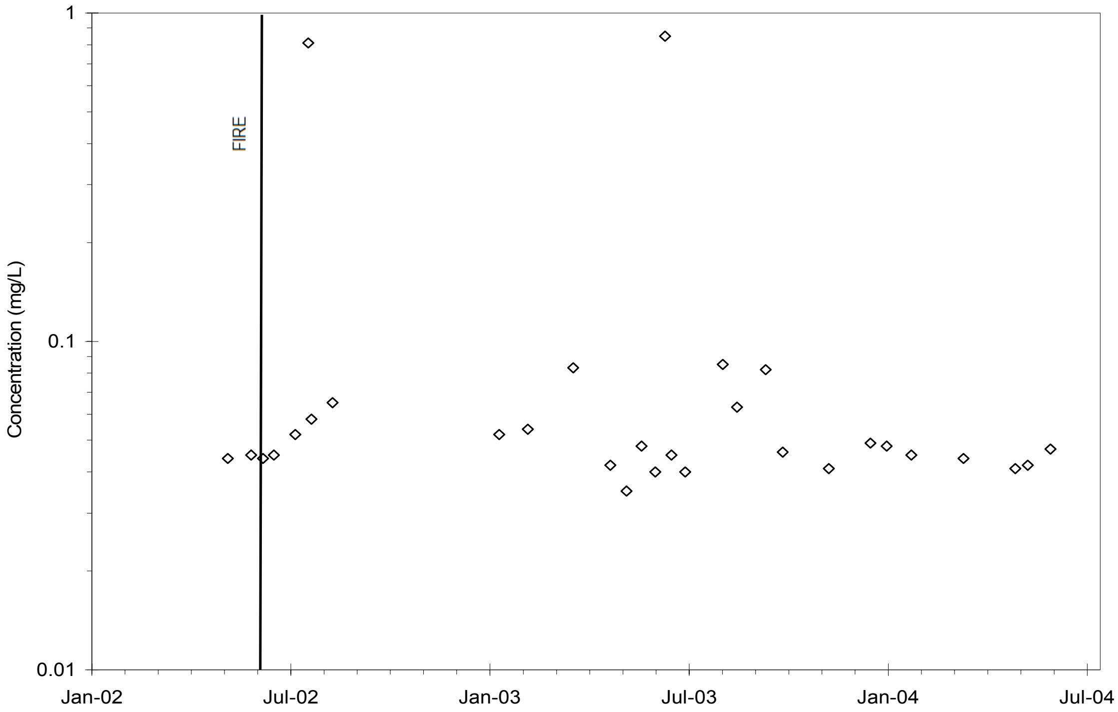

Figure 6.30 Total barium concentration at NFSP

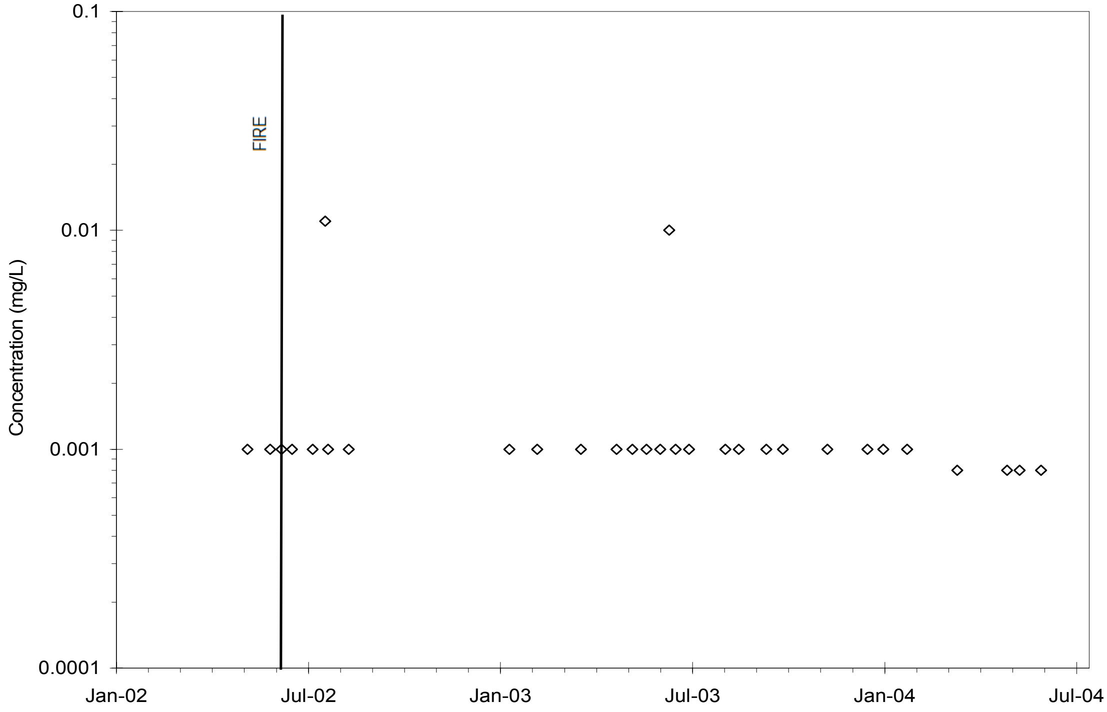

Figure 6.31 Total arsenic concentration at NFSP

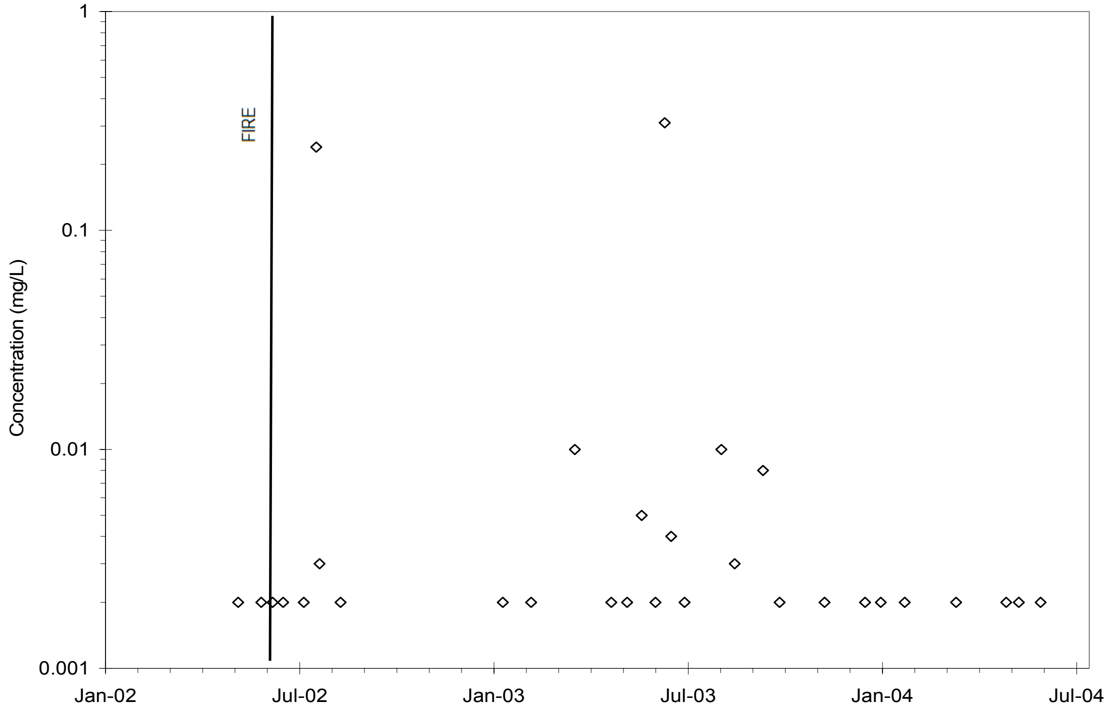

Figure 6.32 Total lead concentration at NFSP

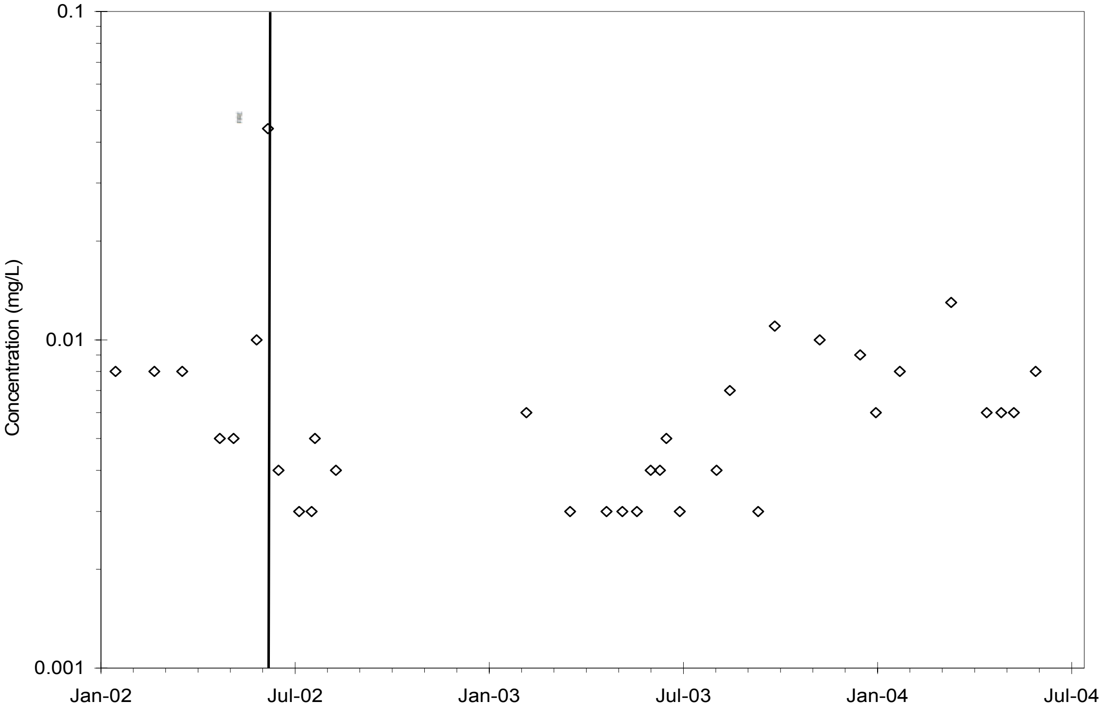

Figure 6.33 Dissolved zinc concentration at NFSP

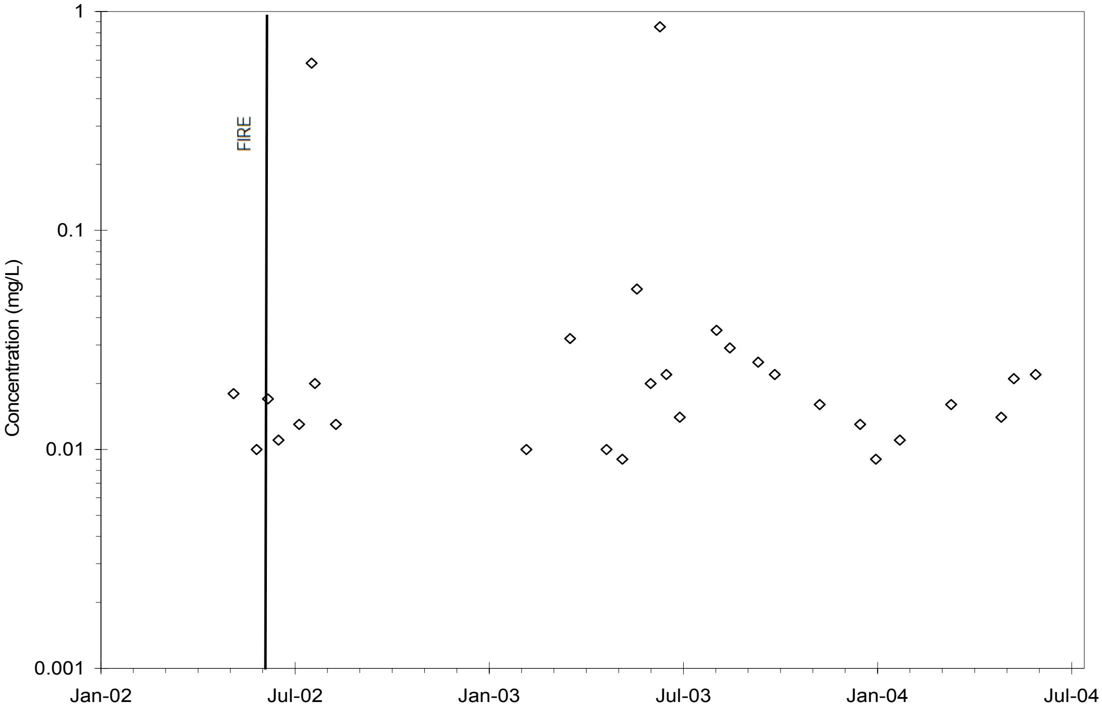

Figure 6.34 Total zinc concentration at NFSP

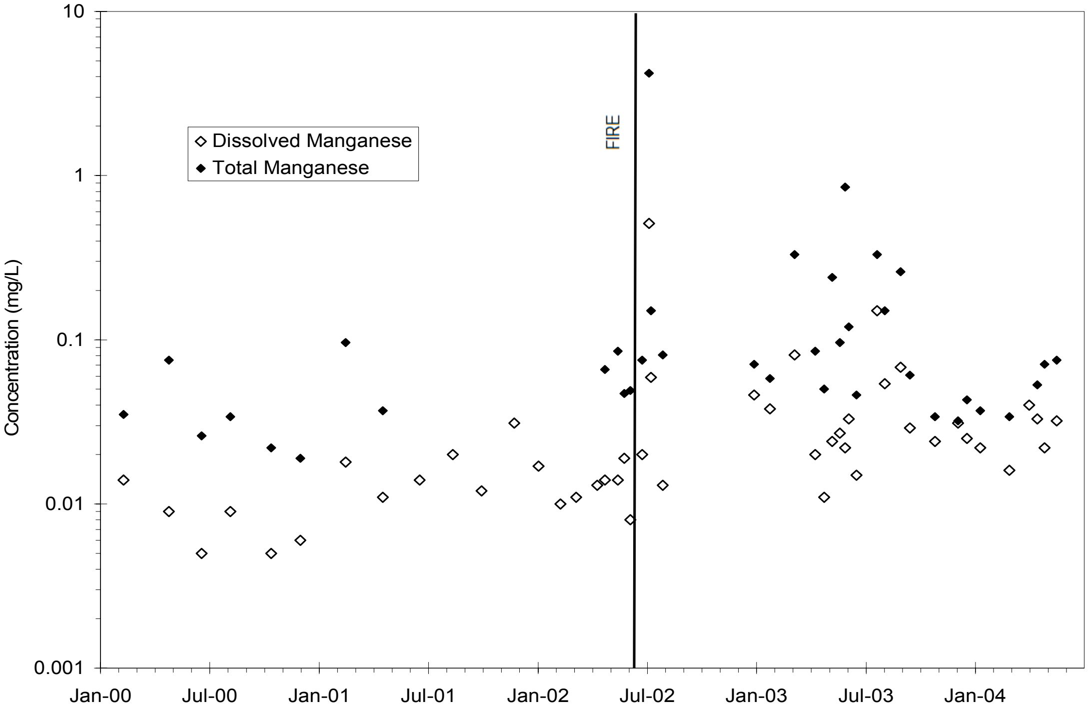

Figure 6.35 Dissolved and total manganese concentration at NFSP

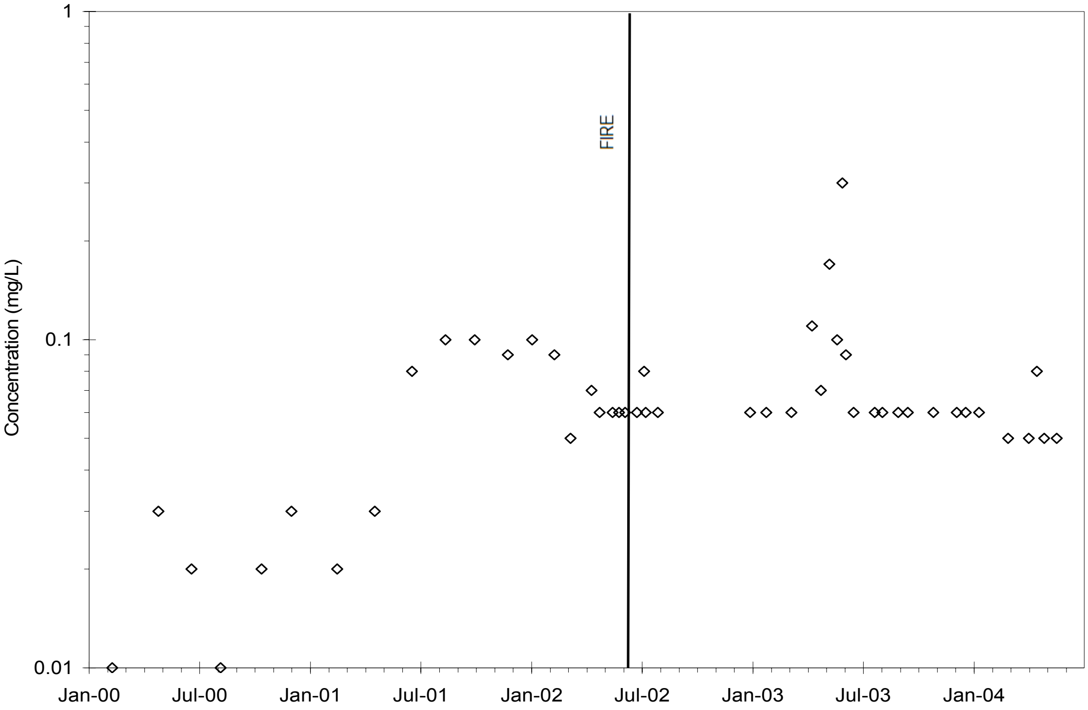

Figure 6.36 Dissolved iron concentration at NFSP

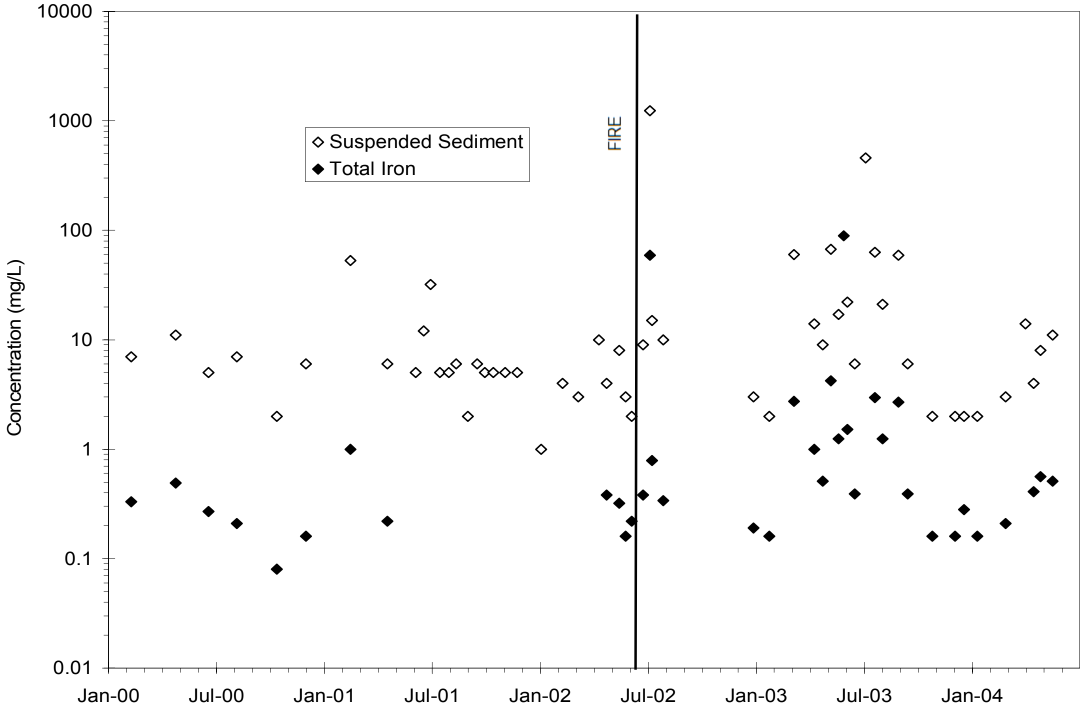

Figure 6.37 Suspended sediment and total iron concentration at NFSP

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Sampling Site Comparison

The influence of upstream concentration on downstream water quality was determined based on the configuration of the four sampling sites. The average concentration at each site was compared for each constituent listed in Table 6.1. This comparison was utilized to determine which constituents traveled through or were retained in Cheesman Lake and their downstream effects at NFSP. At sites with sufficient pre-fire data, pre- and post-fire trends were compared. Only post-fire trends were established at sites having insufficient pre-fire data.

Post-fire suspended sediment concentration was lower at CLO than GC and CLI, as shown in Table 6.3. This is because particulates settled to the bottom of Cheesman Lake post-fire. Also, the average concentration of the particulate fraction of aluminum, arsenic, barium, iron, manganese, lead, and zinc was lower at CLO than either inlet. These average concentration data are included in Tables 6.4-6.10. However, the average concentration for the particulate fraction of these constituents was elevated at NFSP. The elevated concentration is attributed to the North Fork River.

Constituents having sufficient pre-fire data were compared with post-fire data. The average post-fire suspended sediment, dissolved iron, and dissolved manganese concentrations increased at all sites, as shown in Tables 6.3, 6.7, and 6.9. However, the average post-fire dissolved aluminum concentration declined at all the sites, as shown in Table 6.4. The elevated average concentrations are most likely attributed to the Hayman fire. Dissolved aluminum was volatilized or immediately utilized by surviving vegetation, which is why no increase was observed..

Sufficient pre-fire data was available at NFSP for all constituents, except dissolved arsenic and dissolved cadmium. The average post-fire concentration was elevated for dissolved and total aluminum, total arsenic, total barium, total iron, total manganese, total lead, and total zinc, as shown in Tables 6.4-6.10. However, the average post-fire concentration for dissolved barium, dissolved manganese, and dissolved zinc declined. Higher than average post-fire concentrations for the total metals are consistent with the elevated suspended sediment concentration. Unlike the dissolved fraction of the other constituents listed, dissolved aluminum contributed to the total aluminum concentration. The uncharacteristically low average post-fire concentrations for the dissolved fraction are due to dilution of the South Platte River by the North Fork River.

The impacts of the fire cannot be evaluated for dissolved arsenic and dissolved cadmium. The average post-fire concentrations were consistently at or below the MRL, as shown in Table 6.5 and 6.11. In addition, dissolved cadmium and dissolved arsenic pre-fire data were available at GC and CLI, respectively. However, due to variable MRLs, the average concentration cannot be distinguished from the MRL.

Table 6.3 Total suspended sediment concentration comparison

Total Suspended Sediment

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	2	9	2-19	7.2	2	14	2-767	106
GC	1-2	9	<MRL-7	2.9	NR	13	3-44	16
CLO	1-2	9	<MRL-5	1.9	2	15	2-42	5
NFSP	1-5	28	1-53	8.04	2	27	<MRL-1234	78.7

Table 6.4 Dissolved and total aluminum concentration comparison

Dissolved Aluminum

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	0.020	3	<MRL-.14	0.087	0.02	14	.02-.056	0.027
GC	0.02	3	<MRL-1	0.41	NR	13	.023-.19	0.08
CLO	0.02	3	.02-.12	0.08	0.02	15	.02-.078	0.026
NFSP	.02-.1	8	<MRL	<MRL	.02-.05	27	<MRL-.28	0.057

Total Aluminum

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	-	-	-	-	NR	14	.065-20	2.2
GC	-	-	-	-	NR	13	.13-3.9	0.84
CLO	-	-	-	-	0.02	15	<MRL-3.4	0.321
NFSP	NR	7	.007-.31	0.17	0.05	26	.091-36	1.9

Table 6.5 Dissolved and total arsenic concentration comparison

Dissolved Arsenic

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	0.001	2	<MRL	<MRL	.0008-.001	14	.0008-.0020	0.0010
GC	-	-	-	-	.0008-.001	13	<MRL	<MRL
CLO	-	-	-	-	.0008-.001	15	<MRL-.001	0.001
NFSP	-	-	-	-	.0008-.001	27	<MRL	<MRL

Total Arsenic

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	-	-	-	-	0.001	14	<MRL-.005	0.0014
GC	-	-	-	-	.0008-.001	13	<MRL-.004	0.001
CLO	-	-	-	-	.0005-.001	15	<MRL-.002	0.001
NFSP	0.001	4	<MRL	<MRL	.0008-.001	26	<MRL-.011	0.0017

Table 6.6 Dissolved and total barium concentration comparison

Dissolved Barium

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	-	-	-	-	NR	14	.04-.09	0.053
GC	-	-	-	-	NR	13	.015-.05	0.025
CLO	-	-	-	-	NR	15	.044-.077	0.053
NFSP	NR	4	.038-.041	0.040	0.006	27	.022-.064	0.041

Total Barium

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	-	-	-	-	NR	14	.055-.41	0.1
GC	-	-	-	-	NR	14	.013-.36	0.065
CLO	-	-	-	-	NR	15	.049-.12	0.06
NFSP	NR	4	.044-.045	0.045	NR	26	.035-.85	0.11

Table 6.7 Dissolved and total iron concentration comparison

Dissolved Iron

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	.01-.07	9	.01-.14	0.044	.05-.06	14	.03-.14	0.069
GC	0.006	8	<MRL-.21	0.13	0.06	13	.13-.52	0.28
CLO	.01-.06	9	.01-.11	0.05	.05-.06	15	<MRL-.08	0.06
NFSP	.06-.1	20	<MRL-.1	0.05	.05-.06	27	<MRL-.3	0.078

Total Iron

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	NR	5	.19-.62	0.3	NR	14	.1-32	4.3
GC	0.06	8	.15-.66	0.3	NR	13	.13-.52	0.28
CLO	NR	4	.03-.05	0.04	.03-.06	15	.05-4	0.4
NFSP	NR	12	.1-1	0.32	NR	26	.16-89	6.6

Table 6.8 Total lead concentration comparison

Total Lead

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	-	-	-	-	0.002	13	<MRL-.01	0.005
GC	-	-	-	-	0.002	13	<MRL-.07	0.01
CLO	-	-	-	-	0.002	15	<MRL-.013	0.0027
NFSP	0.002	4	<MRL	<MRL	0.002	26	<MRL-.31	0.024

Table 6.9 Dissolved and total manganese comparison

Dissolved Manganese

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	0.01	9	.005-.02	0.01	NR	14	.013-.42	0.086
GC	0.005	8	<MRL-.026	0.009	NR	13	.042-.61	0.2
CLO	0.005	9	<MRL-.246	0.05	0.006	15	.006-.63	0.16
NFSP	.002-.005	20	<MRL-.031	0.0013	NR	27	.011-.51	0.054

Total Manganese

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	NR	5	.041-.082	0.052	NR	14	.02-1.7	0.3
GC	NR	5	.008-.028	0.018	NR	13	.042-.61	0.2
CLO	NR	5	.027-.26	0.094	NR	15	.059-.78	0.24
NFSP	NR	12	.019-.1	0.05	NR	26	.032-4.2	0.29

Table 6.10 Dissolved and total zinc concentration comparison

Dissolved Zinc

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	-	-	-	-	.003-.006	14	<MRL	<MRL
GC	NR	3	.005-.011	0.007	.003-.006	13	.003-.008	<MRL
CLO	-	-	-	-	.003-.006	15	<MRL	<MRL
NFSP	.003-.005	8	<MRL-.04	0.012	.003-.006	26	<MRL-.013	0.0056

Total Zinc

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	-	-	-	-	.003-.006	14	<MRL	<MRL
GC	-	-	-	-	NR	13	.003-.18	0.027
CLO	-	-	-	-	.003-.006	15	<MRL-.039	0.0067
NFSP	NR	4	.01-.018	0.014	NR	25	.009-.85	0.075

Table 6.11 Dissolved cadmium concentration comparison

Dissolved Cadmium

	Pre-fire				Post-fire			
Sampling Site	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**	**MRL**	**Number of Samples**	**Concentration Range (mg/L)**	**Average Concentration (mg/L)**
CLI	-	-	-	-	.0001-.0002	14	<MRL	<MRL
GC	.0001-.005	3	<MRL-.005	0.0017	.0001-.0002	13	<MRL-.0003	0.0002
CLO	-	-	-	-	.0001-.0002	15	<MRL	<MRL
NFSP	-	-	-	-	.0001-.001	27	<MRL-.001	<MRL

SUMMARY

The effects of the Hayman fire on area water quality are clearly defined. Concentration data was not available for a sufficient amount of time prior to the fire to establish a baseline for comparison. In addition, continuous flow rate data was not obtained pre-fire, except at CLO. A correlation between stream flow and concentration data would have been useful in determining seasonal trends. Also, continuous stream flow data was not available to perform an accurate mass loading analysis. In the future, the following recommendations for additional analysis should be considered.

- Collecting water quality data in areas susceptible to forest fires and at USGS gauging stations.
- Using instruments with the lowest detection limit for the analyte.

Although pre-fire data was limited for most of the constituents analyzed, potential impacts could still be established. Trends based on concentration data at each site, as well as between the sites were determined. The general conclusions are listed below.

- The post-fire average concentration of total suspended sediment increased at all four sites.
- A strong correlation was established between total suspended sediment concentration and the particulate fraction for all metals analyzed, with the exception of manganese at CLO and GC.

- The particulates settled to the bottom of the lake. As evidenced by the high total suspended sediment concentration observed at the inlets to Cheesman Lake (the South Platte River at Cheesman Lake inlet and the Goose Creek inlet), but was not observed at Cheesman Lake outlet.
- The post-fire dissolved concentrations of iron and manganese have steadily increased at the South Platte River Cheesman Lake inlet, Goose Creek inlet, and below the confluence of the North Fork and South Platte River.
- The average post-fire dissolved barium, dissolved manganese, and dissolved zinc concentration declined relative to the pre-fire average concentrations.
- Dissolved aluminum, barium, and zinc did not show a discernible response to the fire.
- The impacts of the fire cannot be evaluated for dissolved arsenic and dissolved cadmium because the concentrations were consistently at or below the MRL.
- The water quality of the North Fork River may explain increases in total and dissolved iron, and dissolved aluminum concentration at the sampling site located downstream of the confluence of the North Fork and South Platte River relative to Cheesman Lake outlet.

CHAPTER 7 : SYNTHESIS

Water quality data was analyzed for the Rodeo-Chediski fire (Arizona) and the Hayman fire (Colorado). The concentration data analysis was performed for the burned Arizona and Colorado watersheds and the unburned Verde River watershed. To determine the severity of the impacts of fire on drinking water quality, the burned Arizona watershed was compared to the unburned Verde River watershed and the burned Colorado watershed. Mass loading analysis was completed for the Salt River and Verde River, these results were compared as well. The objective of this chapter is to compile the results of all analyses completed and compare with the literature to determine potential long-term effects of forest fires on drinking water quality.

STREAM FLOW

Forest fires may cause a temporary increase in stream flow, while a drought period will decrease stream flow long-term. The Salt River post-fire peak stream flow, the first flush, occurred one month after containment of the Rodeo-Chediski. The Verde River stream flow was not elevated during the time of the first flush. The post-fire peak stream flow in the Salt River following the fire is consistent with studies conducted by Gottfried et al. (2003) and Elliott and Parker (2001).

The effect of the Hayman fire on stream flow is inconclusive. The discrete flow rate data sampling interval at GC, CLI, and NFSP was too large to determine the influence of individual storm events after the fire. However, an overall decreasing trend in the seasonal peak flow is noticeable at NFSP since the fire. This is due to the drought and the upstream dam at CLO. Continuous flow rate data is available for CLO, but the flow rate is controlled by humans. In streams that are controlled by a dam, the influence of storm events is not noticeable.

Drought has a prolonged impact on the Salt River stream flow relative to the Rodeo-Chediski fire. The Salt River stream flow remains low two years post-fire relative to pre-1996, which is when the drought began. Although flow rate data is limited, the streams within the drought-stricken South Platte River watershed are likely to be experiencing low stream flows as well.

EROSION AND SUSPENDED SEDIMENT

Post-fire erosion occurred following the Rodeo-Chediski and Hayman fires. The total suspended sediment concentration increased at all of the sampling sites within the burned watersheds. An increase in erosion after a fire was noted in studies conducted by Minshall et al. (2001), Gottfried et al. (2003), and Neary (2004). Due to both fires occurring in mountainous terrain, the steep slopes covered by burned vegetation and litter increased the potential for sediment transport. The grade of a slope having impacts on erosion is consistent with Tiedemann et al. (1979); increased erosion occurring in areas consisting of burned vegetative cover was reported by Wondzell and King (2003).

A mass loading analysis was completed in Arizona for the Salt River and Verde River watersheds. The particulate loading in the Salt River increased by

135% the first year post-fire. The Verde River suspended sediment load decreased by 81%. The fire caused the elevated Salt River load, while drought caused the depletion of the Verde River load.

NUTRIENTS

The dissolved and particulate fractions of nitrogen, phosphorus, and organic carbon were elevated in the Salt River after the first major storm event following the Rodeo-Chediski fire. Increased phosphate and nitrate concentrations in stream water, after a fire, were reported by Earl and Blinn (2003) and Spencer and Hauer (1991). Elevated nitrate concentrations in the runoff of a recently burned watershed were noted by Minshall et al. (2001) and Lewis (1974). Tiedmann et al. (1978) found higher total phosphorus concentrations in stream water after a wildfire. Debano and Conrad (1978) found that erosion caused nitrogen loss after a prescribed fire. The post-fire trends in nutrient concentrations for the Salt River are consistent with all of these studies.

The source of the elevated nutrient concentrations was determined by comparing Salt River concentration data with the adjacent, unburned Verde River watershed. Verde River data was collected and analyzed for ammonia, total nitrogen, dissolved organic carbon, and total phosphorus. The only nutrient to increase in the Verde River during the initial year after the fire was ammonia. The increase occurred in December, 2002 and could be a result of the drought the watershed has been in since the mid-1990s; during a drought nutrients accumulate on the sediment surface. Also, housing developments around Sedona, Arizona could be an ammonia source.

The Salt River watershed has experienced this same drought. Therefore, the increased concentrations of the post-fire nutrients may be magnified since the first flush mobilized newly released nutrients and the drought-induced build up of nutrients. Malmer (2004) reported an increase in post-fire nitrogen and nitrate concentrations in a burned watershed suffering from a drought relative to a burned watershed not in a drought. Although drought does impact nutrient concentrations, the fire is still responsible for further release of more nutrients. The first flush had a greater impact on the mobilization of nutrients due to the drought.

The results of the nutrient mass loading analysis were consistent with the concentration data analysis for the Salt River and Verde River. A comparison of these data is shown in Table 7.1. The first year after the Rodeo-Chediski fire, all nutrient loadings increased in the Salt River; dissolved organic carbon and ammonia increased on the Verde River. Although the increase in loading of dissolved organic carbon and ammonia in the Verde River is attributed to the drought, the fire had definitive impacts on the Salt River loadings. However, two years post-fire the mass loadings of all the constituents decreased.

Table 7.1 Mass loading comparison of the Salt River and Verde River

	% Change 1 year post-fire		% Change 2 years post-fire	
Constituent	**Salt River**	**Verde River**	**Salt River**	**Verde River**
Total Organic Carbon	↑↑↑	-	-	-
Dissolved Organic Carbon	↑↑	↑	↓	↓
Total Phosphorus	↑↑↑	↓	↓	↓
Dissolved Phosphorus	↑↑	↓	↓	↓
Total Nitrogen	↑↑↑	↓	↓	↓
Ammonia	↑↑↑	↑	↓	↓
Nitrate-Nitrite	↑↑		↓	-
Suspended Sediment	↑↑	↓	↓	-
Stream Discharge	↓	↓	↓	↓

↓ or ↑ < 100%	100%< ↑↑ <1000%	1000% < ↑↑↑

Similar to the concentration data, the drought magnified the increase in mass loading. The total phosphorus, total nitrogen, and suspended sediment loadings decreased in the Verde River, yet these loadings were elevated in the Salt River. The increased loadings in the Salt River during a drought contrast with Cordy et al. (2000). According to this report, nutrient loads increased in the upper Salt River when stream flow exceeded the historical average annual stream flow.

Since the Rodeo-Chediski fire, the average stream flow has remained low, but the mass loading increased. Figures 7.1 and 7.2 illustrate the effect of the decreased stream flow on nutrient and suspended sediment mass loadings. The mass loading of nutrients and suspended sediment was normalized to the volume of water. One year post-fire, the Salt River mass loadings of the nutrients were increased by at least one order of magnitude one year post-fire. Similarly the Verde River mass loading of dissolved organic carbon and ammonia were increased by less than one order of magnitude. In contrast, the Salt River suspended sediment, and Verde River total phosphorus, total nitrogen, and suspended sediment mass loads were decreased by less than one order of magnitude. Two years post-fire, the mass loading of all the constituents decreased in both rivers, with the exception of the mass loading of ammonia in the Verde River. The decrease in the Salt River ranged between one to three orders of magnitude; the decrease in the Verde River was less than one order of magnitude. The normalized the mass loadings provide further support that drought impacts water quality. However, the effects of fire on water quality dominate, at least, the first year following a forest fire.

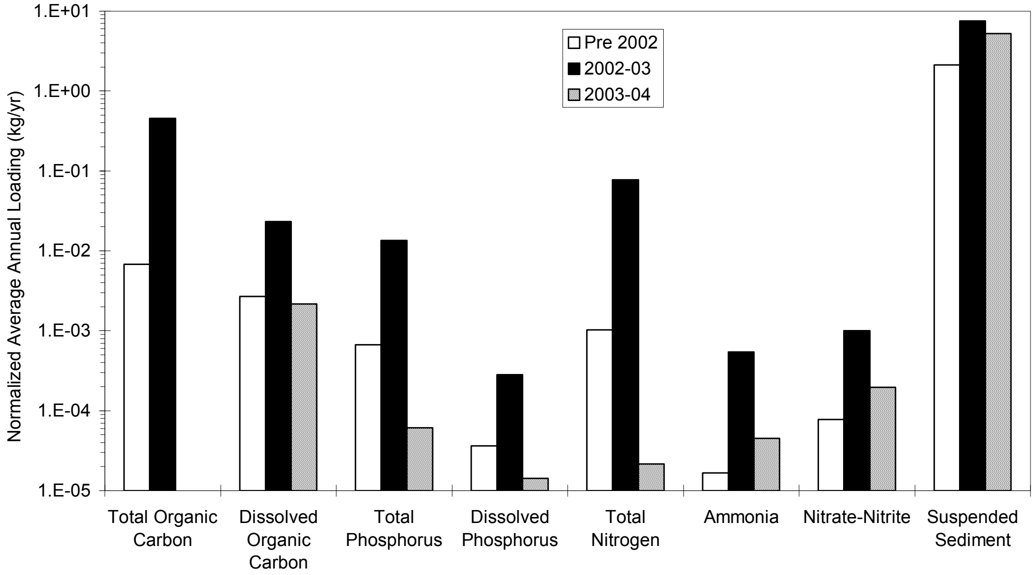

Figure 7.1 Normalized average annual loading of nutrients and suspended sediment for the Salt River at USGS gauging station 09498500

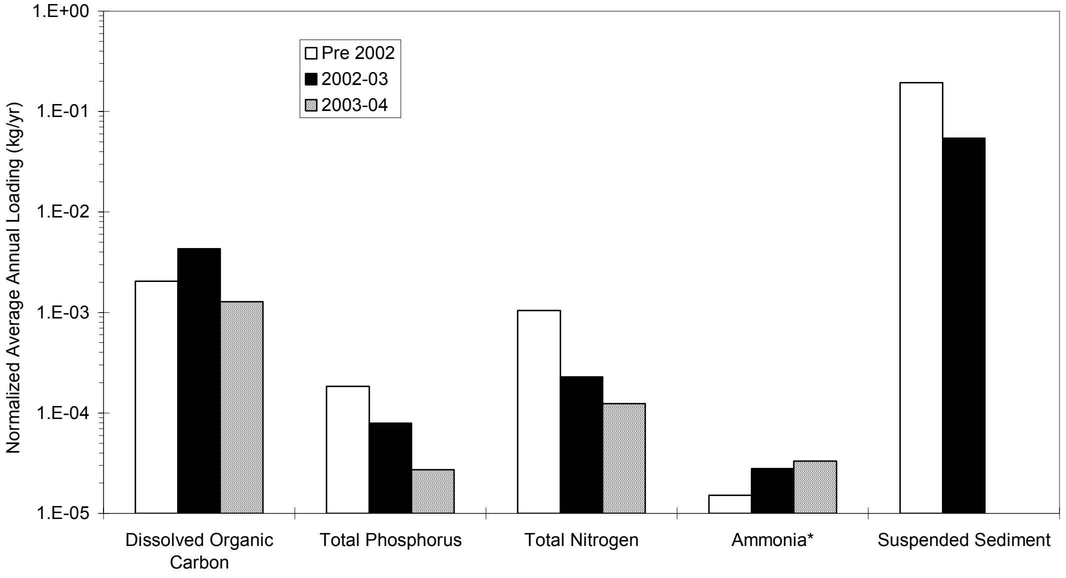

Figure 7.2 Normalized average annual loading of nutrients and suspended sediment for the Verde River at USGS gauging station 09508500

Although not compared to the Verde River, the total organic carbon, dissolved phosphorus, ammonia, and nitrate plus nitrite mass loadings were estimated for the Salt River. These nutrients also displayed an increase in the mass loading the year after the fire. However, all nutrient concentrations, except total organic carbon (data was not available), in both watersheds decreased two years post-fire. The initial mass loading increase is attributed to the fire.

METALS

The post-fire trends of metal concentrations were similar for the Salt River and the Cheesman Lake inlets and outlet. However, not all metals had the same post-fire response between the two wildfire sites. The complete list of metals and other constituents analyzed for the Salt River and the South Platte River watersheds are listed in Table 7.2.

Table 7.2 List of constituents analyzed for the Salt River and South Platte River watershed

Constituents	Pre 2002	2002-2003	2003-2004	Data collected by ASU
Nutrients				
Ammonia	X	X	X	X
Dissolved Organic Carbon	X	X	X	X
Dissolved Phosphorus	X	X	X	X
Nitrate-Nitrite	X	X		X
Total Nitrogen	X	X		
Total Organic Carbon	X	X		
Total Phosphorus	X	X	X	X
Trace Metals				
Total Arsenic	X	X		
Dissolved Arsenic	X	X	X	X
Total Barium	X	X		
Dissolved Barium	X	X		
Total Copper	X	X		
Dissolved Copper	X	X		
Total Iron	X	X		
Dissolved Iron	X	X		
Total Lead	X	X		
Dissolved Lead	X	X		
Total Manganese	X	X		
Dissolved Manganese	X	X		
Total Mercury	X	X		
Dissolved Mercury	X	X		
Dissolved Selenium	X	X		
Total Selenium	X	X		
Total Zinc	X	X		
Dissolved Zinc	X	X		
Ions				
Potassium	X	X		
Manganese	X	X		
Calcium	X	X		
Sodium	X	X		
Chloride	X	X		
Bicarbonate	X	X		
Other				
Suspended Sediment	X	X	X	X
Stream Discharge	Continuous data			

Post-fire metal concentrations were elevated in the receiving water bodies of both burned areas. This is a strong indication the release of the metals was fire-induced. Dissolved arsenic and dissolved manganese concentrations were elevated in the Salt River and South Platte River inlet at Cheesman Lake (CLI). However, the pre-fire dissolved arsenic concentration data at CLI was limited. Total manganese concentrations were higher at all sites in Colorado and Arizona. Although the dissolved manganese concentration in the Salt River was impacted by mining, the post-fire concentration was elevated relative to the concentrations five years prior to the fire; dissolved manganese concentration was elevated at all the Colorado sites. The elevated post-fire concentrations at the burned sampling sites indicate the fire impacted the release of dissolved arsenic and total and dissolved manganese.

Total arsenic, dissolved and total barium, total lead, and total zinc concentrations were elevated post-fire in the Salt River and below the confluence of the North Fork River and South Platte River (NFSP). These elevated concentrations at NFSP were not expected based on the concentrations of these metals at the outlet at Cheesman Lake (CLO). The higher concentrations are probably a result of the confluence of the South Platte River with the North Fork River. The North Fork River does not drain the area burned by the Hayman fire; however, this watershed was burned by the Hi Meadow fire in June, 2000. Therefore, the Hayman fire cannot be the designated source for the higher concentrations of these metals. In addition, the dissolved aluminum concentration was not impacted by the fire; this is based on available concentration data for CLO, CLI, and NFSP.

Iron was transported from the burned South Platte River watershed. The dissolved iron concentrations increased at CLI, Goose Creek (GC), and CLO after the Hayman fire. The total iron concentration was only elevated at CLI and GC. No concentration increase was observed at CLO due to the settling of the particulate fraction to the bottom of Cheesman Lake. In contrast, the total and dissolved iron concentrations in the Salt River were not altered by the Rodeo-Chediski fire. The contrast between the two fires could be due to geological differences; more iron may naturally exist in the South Platte River watershed than the Salt River watershed.

Other metal concentrations were elevated at one site only. The total copper and total mercury concentrations were higher post-fire relative to background levels in the Salt River. Dissolved zinc and total aluminum concentrations increased at NFSP in Colorado, this could be a result of the Hi Meadow fire that burned the North Fork River watershed in June, 2000. The dissolved cadmium concentration surpassed detection limits only at GC. Since ample pre-fire concentration data exists for the Salt River, the elevated total copper and total mercury concentrations are attributed to the Rodeo-Chediski fire. However, pre-fire concentration data for these metals was insufficient at the Colorado sites. Therefore, the elevated concentrations cannot be directly attributed to the Hayman fire.

Selenium was only analyzed at the Salt River site and was the only EPA regulated metal to exceed the MCL. Selenium impacts human health and aquatic life. In the aquatic environment, bioaccumulation and biomagnification can occur, as well as reproductive impairment in fish (Hamilton and Lemly 1998, Andrews and King 1997). In addition, selenium can cause high sediment concentrations in a lake 11 months after a short-term pulse (Hamilton and Lemly 1998). Therefore, pulse of selenium transported by the first flush may have prolonged effects on the aquatic environment and water quality.

As a result of the increased suspended sediment load in the Salt River, the particulate fraction of the metals analyzed was influenced. Total zinc, total manganese, total barium, total arsenic, total selenium, and total lead mass loadings all increased after the fire. The total mercury loading was minimally depleted, while total iron and total copper loadings remained steady. These trends indicate that some metals are released and mobilized as a result of fire.

In contrast to the increased particulate loadings of metals, the dissolved loading was depleted post-fire, except for barium, arsenic, and selenium. The dissolved manganese, dissolved iron, dissolved copper, dissolved zinc, dissolved lead, and dissolved mercury mass loadings all decreased one year post-fire. The dissolved barium, dissolved arsenic, and dissolved selenium loadings increased; however, the dissolved arsenic load decreased two years post-fire. The results of the loading analysis of these constituents indicate that the fire-induced the release of barium, arsenic, and selenium. If the fire did cause the release of the other metals, then volatilization is a likely reason for the decreased loadings. In addition, the opportunity for mineral weathering to occur after a fire is depleted. The decrease in infiltration increases overland flow, therefore the particulate fraction is mobilized before the mineral weathering process occurs.

MAJOR IONS

Analysis of the major ionic concentration was completed for the Salt River. The cations, magnesium and potassium, did not increase in concentration after the fire; sodium and calcium concentration increases were minimal. The anionic concentration of bicarbonate and chloride was not impacted by the fire. These results are consistent with the literature. Malmer (2004) concluded the elevated calcium concentrations in a stream draining a burned watershed were due to a long-term drought; therefore the calcium concentration increase is attributed to the drought. In addition, calcium was the only ion to increase in concentration during the first flush. The minimal sodium concentration increase was also noted in a study conducted by Tiedemann et al. (1978); although calcium, magnesium, and potassium concentrations were also elevated in this study, as well. Davis (1989) concluded that the ionic concentration in stream water was not significantly affected by a forest fire. Similarly, Belillas and Roda (1993) found that the anionic concentration in stream water was not responsive to fire. Based on this analysis, sodium is the only ion that was impacted by the fire. The other ionic concentrations remained within background levels.

The ionic loadings for the Salt River were impacted by the fire. As discussed in Chapter 6, the calcium, magnesium, potassium, and bicarbonate loadings were high relative to the flow rate the first year after the fire. The chloride and sodium loadings increased one year post-fire; however no correlation with flow rate was observed. The minimal increase in the ionic mass loading is a result of post-fire concentrations remaining near the upper limits of the background levels.

FACTORS AFFECTING FIRE IMPACTS ON DRINKING WATER UTILITIES

The impact of a forest fire on drinking water utilities depends on two major factors.

- A reservoir located upstream of a WTP.
- The proximity of the forest fire to the surface water source.

A reservoir upstream of a water treatment plant (WTP) can minimize the effects of increased turbidity and ash flows after a forest fire; the particulates would settle to the bottom of the lake. If the WTP pumped water directly from a river within a burned watershed, then the ash and particulates would need to be removed by the WTP. In addition, reservoirs may minimize the effects of post-fire nutrient fluxes. The nutrients stimulate the growth of algae and bacteria, therefore causing potential taste and odor problems. MIB is one taste and odor metabolite produced by algae and bacteria. The MIB profile for Saguaro Lake, which is the final reservoir in the Salt River system, is shown in Figure 7.3. The post-fire MIB concentration remained within background levels.

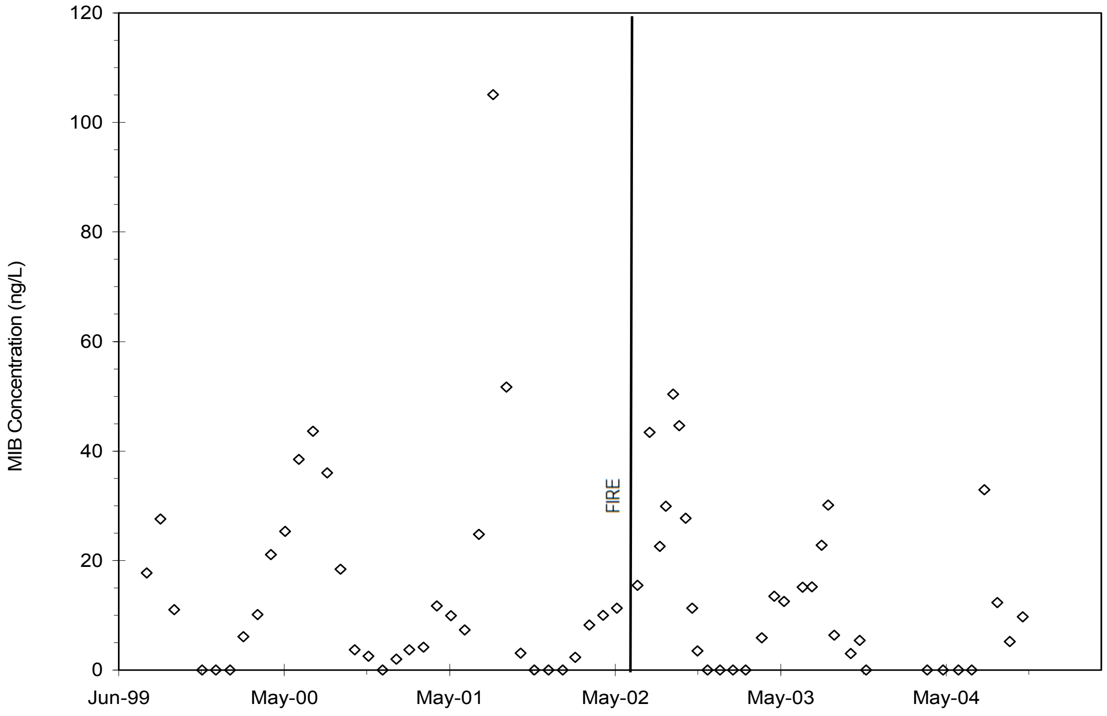

Figure 7.3 Saguaro Lake MIB concentrations
Source: Westerhoff, et al. (2004)

The proximity of the forest fire to surface water will impact influent water quality at the WTP. A fire that burns the area surrounding the drinking water source will have more pronounced effects than a fire burning the tributaries of the drinking water source. The runoff from a fire that burned tributaries will be intercepted by natural obstacles such as plants and algae within the stream and be deposited on the stream bed. However, runoff from a fire located at the drinking water source will not be intercepted.

The Rodeo-Chediski fire is an example of a fire that burned the tributaries to a drinking water source. The Hayman fire is an example of a fire that burned the area surrounding the drinking water source. However, at least one reservoir exists upstream of the WTPs utilizing the fire-impacted drinking water source. Therefore, the impacts of the forest fires on WTPs were minimized.

ECOLOGICAL IMPACTS OF FOREST FIRES

After a forest fire, re-growth of vegetation begins. The plant succession occurs in stages (Forest and Range 2004).

1. Bare ground
2. Annual plants and grasses
3. Perennial plants and grasses
4. Shrubs
5. Trees

The amount of time for plant succession to occur depends on the severity of the fire. A low intensity fire does not burn all of the vegetation and seeds, whereas a severe fire burns plants, seeds, and organisms. A more severe fire will require more time for plants and trees to re-establish growth (Kormondy 1996).

Similarly, aquatic life is impacted by forest fires as well. Initially, fish kills due to increased stream water temperature and water chemistry changes can occur (Spencer et al. 2003). Spencer et al. (2003) also observed an increase in the density of algae blooms in fire-impacted streams. Long-term impacts result from the post-fire shift in vegetation and land cover. Allan et al. (1997) and Griffith (2001) observed that vegetation can affect stream habitat and water quality. Spencer et al. (2003) noted a shift in consumer food sources following a forest fire. The reliance of consumers on allochthonous food sources decreased while the consumption of autochthonous food sources increased. This is likely due to the increase in autochthonous production that can occur following a forest fire (Spencer et al. 2003).

SIGNIFICANCE OF FINDINGS

The elevated loadings and concentrations of nutrients, metals, and suspended sediment impact drinking water quality. Although the ionic loads increased, the effects on drinking water quality are minimal. The particulates can fill or damage reservoirs and infiltration basins, therefore increasing the level of water treatment necessary (Meixner and Wohlgemuth 2004). The increase in turbidity prevents efficient disinfection; for example, disinfection by ultraviolet radiation is hindered at high turbidity levels (AWWA 1999). Therefore, water treatment plants using turbidity-sensitive disinfection equipment, such as ultraviolet radiation, will need to modify the disinfection process to provide adequate disinfection.

Nutrients and metals impact drinking water quality as well. Metals can be toxic to humans; therefore the EPA has set MCLs. Selenium is one metal regulated by the EPA. The post-fire dissolved concentration of selenium exceeded the MCL. In addition, the particulate concentration was elevated post-fire. The dissolution of particulates into the drinking water source and the negative human health impacts make selenium the most important constituent. In addition, metals and nutrients promote bacterial and algae growth, causing taste and odor problems.

Long-term drought combined with forest fire impacts water quality. The Salt River and Verde River concentration data comparison lead to this conclusion. Increases in the nutrient concentration data for the Verde River are likely a result of the mobilization of accumulated material due to the drought. The Rodeo-Chediski fire added to this accumulated material in the Salt River watershed. The material was transported into the Salt River during the first flush. Since both watersheds had drought conditions prior to the summer of 2002, the South Platte River watershed probably experienced elevated nutrient concentrations after the Hayman fire.

The long-term effects of forest fires on drinking water quality will be caused by the particulates. The Hayman fire water quality analysis resulted in a definitive trend of particulates settling to the bottom of Cheesman Lake. A similar scenario likely exists for Roosevelt Lake. The particulates will slowly be degraded and enter the dissolved phase. In addition, the suspended sediment organic content after the Rodeo-Chediski fire increased from 2% to 11%. The high organic content is the result of the fire.

CHAPTER 8 : CONCLUSIONS

The purpose of this research was to determine the long-term impacts of forest fires on drinking water quality. The fate of the nutrients, metals, and ions as they travel through the water system will ultimately determine those effects. Hence, the concentration data and mass loading analysis was performed. The following conclusions were reached based on the analyses.

- Long-term drought affects forests and water quality.
    - Drought causes a forest to become susceptible to massive forest fires.
    - Drought conditions cause an accumulation of soluble material on the sediment surface.
    - Storm events mobilize the accumulated material and influence water quality.
- The mass loading analysis on the Verde River showed the effects of the drought on nutrients and suspended sediment.
    - Suspended sediment load decreased by 81% one year post-fire.
    - Total nitrogen load decreased by 86% one year post-fire.
    - Total phosphorus load decreased by 72% one year post-fire.
    - Dissolved organic carbon load increased by 39% one year post-fire.
    - Ammonia load increased by 22% one year post-fire.
    - Two years post-fire the load for these constituents declined.

- The mass loading analysis on the Salt River showed the Rodeo-Chediski fire had short-term effects on the water quality.
- The mass loading analysis on the Salt River showed that the effects of forest fires on nutrients, suspended sediment, and metals were greater than drought effects. Several impacted constituents are listed below.
  - Suspended sediment load increased by 135% one year post-fire.
  - Total nitrogen load increased by 4,935% one year post-fire.
  - Total phosphorus load increased by 124% one year post-fire.
  - Dissolved organic carbon load increased by 470% one year post-fire.
  - Ammonia load increased by 2,059% one year post-fire.
  - Total barium load increased by 1,991% one year post-fire.
  - Total arsenic load increased by 484% one year post-fire.
  - Dissolved lead load decreased by 84% one year post-fire.
  - Dissolved iron load decreased by 80% one year post-fire.
  - Two years post-fire dissolved arsenic, suspended sediment, and nutrient loads decreased to near background levels.
- Selenium was the only metal regulated by the EPA to exceed the MCL, therefore it is an important constituent to monitor post-fire.
  - The maximum post-fire concentration was 2mg/L. The MCL is 0.05mg/L.

- Infiltration normally decreases following a fire; therefore groundwater recharge is decreased which leads to lower base flow.
- The Salt River system remains limited by phosphorus regardless of the drought and Rodeo-Chediski forest fire.
- Ionic concentrations in runoff from the Salt River watershed were minimally impacted by the Rodeo-Chediski fire.
- Post-fire concentrations returned to background levels within two years for most constituents analyzed in Arizona and Colorado.
  - Dissolved iron and dissolved manganese have steadily increased at the South Platte River at Cheesman Lake inlet, Goose Creek inlet, and below the confluence of the North Fork and South Platte River.
  - Pulses of various nutrients and metals were observed within the two years after both fires; in the Salt River watershed, this is attributed to drought conditions.
- Increased erosion in Colorado and Arizona was a direct result of the Hayman fire and Rodeo-Chediski fire, respectively.
- In general, the particulate fraction of the constituents analyzed was impacted to a greater extent than the dissolved fraction. This is due to the increase in erosion.
- The fate of the particulate fraction of the constituents will determine the long-term impacts.

  - The high particulate concentrations entering Cheesman Lake were typically depleted before exiting. This is due to the particulates settling to the bottom of the lake. This trend most likely existed for Roosevelt Lake, in Arizona, as well.

FUTURE RECOMENDATIONS

- Baseline water quality data and continuous stream flow data need to be collected in areas that are at high risk for wildfires. Therefore, a solid and complete assessment of the impacts of fire on water quality can be determined.
- In areas susceptible to forest fire sediment traps could be constructed to prevent sediment from entering the surface water.
- The effects of the particulate fraction settling to the bottom of lakes and reservoirs need to be determined. This is a source for long-term water quality problems.
- Studies focusing on the long-term impacts of forest fires on drinking water quality need to be conducted.

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APPENDIX A

LITERATURE REVIEW

WILDFIRE EFFECTS ON WATER QUALITY

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December 6, 2002

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WILDFIRE EFFECTS ON WATER QUALITY

Extensive forest fires in northeastern Arizona marked the summer of 2002. The Rodeo fire began on June 18, 2002 and merged with Chediski fire, started June 20, on June 23. The combined fires burned nearly one third of the Tonto National Forest, an area just under 500,000 acres. Concern for the impact of the fires on water quality arose when streams in the effected watersheds flowed black with ash and soot from burned areas. This literature review summarizes research work conducted in relationship to wildfires and their effect on water quality. The focuses of this review include general water quality parameters, polycyclic aromatic hydrocarbons, dioxins, heavy metals and algal response to fire induced water quality changes.

General Water Quality

Forest fires produce immediate effects on nutrients in ecosystems, which in turn impact effected water bodies in the watershed. These effects include the volatilization of nitrogen as gas from plant materials, mineralization of calcium, potassium, and magnesium as oxides followed by conversion to soluble carbonate salts[1-4]. Water quality parameters examined in the literature following wildfires included nutrients, organics, and ions.

Tiedemann et al[1] studied water quality changes in receiving streams following a fire in central Washington in 1970. They found that the maximum nitrate-nitrogen concentrations increased primarily due to increased nitrification following the fire. This nitrification was likely due to reduced demand for nitrate nitrogen by vegetation, as well as increased soil pH and electrolyte concentration as a result of the fire[2]. Total phosphorous concentrations also increased 1.5 to 3 times concentrations in unburned watersheds. Concentrations of calcium, magnesium, potassium and sodium decreased below concentrations measured previous to the forest fire. The study concluded that changes in nitrate nitrogen, phosphorous, and ion concentrations produced negligible effects on water quality for municipal use[1].

Carignan et al [5] noted that few studies have focused on the effects of fire on lake chemistry and hydrology. Four major wildfires occurred in Quebec during the summer of 1995. Carignan et al conducted research to study the effects of these fires on impacted lakes. Burnt lake water quality was compared with reference lakes to determine fire impacts. Burnt lakes had higher concentration of total phosphorus (2 to 3 fold), total organic nitrogen (2 fold), and K^+, Cl^-, and Ca^{2+} (up to 6 fold). The ions K^+, Cl^-, SO_4^{2-}, NO_3, and some dissolved organic carbon (DOC) were rapidly flushed through the watersheds, showing a 50% decrease over three years. Other parameters, total phosphorous, total organic nitrogen, Ca^{2+} and Mg^{2+} were still increasing or stayed the same three years following the fire. Possible explanations for low DOC concentrations in burnt lakes included the loss of organic carbons sources (leaves, shrubs, vegetation), the combustion of the organic topsoil due to fire, and loss of some DOC through sorption to ash and charcoal left behind by the fire[5].

Water quality studies of the Pecos and Gallinas rivers in New Mexico were studied following the 29,000 acre Viveash wildfire in May 2000. Hopkins[6] saw a large increase in turbidity, total suspended solids (TSS), total dissolved solids (TDS), total phosphrous (TP), sulfate and chemical oxygen demand (COD) in the Pecos River in succeeding months. Several metals were also measured in the Pecos river. Mercury and aluminum concentrations exceeded chronic standards (12 ng/L and 87 ig/L, respectively) at least ones in the months immediately following the fire. Cyanide was also measured in the rivers and will be discussed under the Heavy Metals section of this review. Hopkins' research did not include effects of the fire on receiving reservoirs at the time his report was printed.

Garcia and Carignan[7] documented water quality effects following forest fires in Canadian lakes in Boreal forests. Concentrations of sulfate, nitrate-nitrogen, TP, and chlorophyll-a were significantly higher in burned lakes than in reference lakes ($P > 0.01$). Dissolved oxygen concentrations, pH, alkalinity, and temperature were similar between burned and reference lakes.

Polycyclic Aromatic Hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons (PAHs) are a group of different chemicals that are formed during the incomplete burning of oil and gas, coal, garbage or other organic substances like wood[8]. The Center of Disease Control (CDC) provides a summary of the environmental occurrence of PAHs and health effects associated with these compounds. PAHs may enter the air when released due to forest fires. PAHs do not dissolve easily in water and stick to solid particles that settle to the bottom of lakes or may move through soil to contaminate groundwater. PAHs may break down by reacting with sunlight or other chemicals, or some microorganisms can break down PAHS in soil or water over weeks or months. PAHs may be transferred through drinking water and have been shown to cause birth defects in mice and some PAHs may reasonably be expected to be carcinogens[8].

Research was conducted in Malaysia to determine the source of atmospheric PAHs, using isotopic composition, following a 1997 fire in Indonesia[9]. Polycyclic aromatic hydrocabon source identification was examined because of differing toxicity and genotoxicity of PAHs[10]. The research indicated that the fires did not significantly impact PAHs levels in Malaysia. However, Okuda et al, were able to identify compound specific carbon-isotope analysis of PAHs for source identification; a technique that may be useful in future PAH research[9].

Dioxins

Dioxins are persistent organic pollutants that are by-products of natural processes such as forest fires, and industrial processes[11]. The World Health Organization fact sheet on dioxins explains that dioxins consist of a family of structurally and chemically related polychlorinated dibenzo-para-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and polychlorinated biphenyls (PCBs).

Dioxins dissolve in fats and have a half-life in the body of several years. Environmentally, dioxins bio-accumulate in the food chain. Short-term human exposure to dioxins may result in skin lesions and altered liver function. Long-term exposure may lead to impairment of the immune system, nervous system, endocrine systems and reproductive functions[11].

Following extensive forest fires in Alberta Canada (1998), polynuclear aromatic hydrocarbons, PCDD/Fs and PCBs in sediment were studied[12]. Gabos et al found a predominance of alkylated polynuclear aromatic hydrocarbon derivatives over parent polynuclear aromatic hydrocarbons. The presence of the polynuclear aromatic hydrocarbons was attributed to wood combustion. Very low concentrations of PCDD/Fs and PCBs were measured; the presence of these compounds was attributed to atmospheric contributing sources[12].

Heavy Metals

Forest fire ash may contain heavy metals. Living plants can extract and concentrate metals from their environments. In addition to trace metals utilized by plants for growth, these heavy metals may include cadmium, chromium, lead, cobalt, silver, selenium, and mercury[13]. The transport of mercury, cyanide and cadmium following forest fires has been documented. Cadmium transport is further discussed as it relates to algae growth.

Mercury

Mercury is deposited into catchments via atmospheric deposition[14]. Typically watersheds retain deposited mercury until it is mobilized following watershed disturbances such as forest fire[7, 15]. Increased runoff due to fire, leads to an increase of mercury and organic carbon in water reservoirs. Organic and inorganic matter provide a carbon source for methylation processes in lake sedimentation. Monomethylmercury may be produced following fire runoff events through methylation processes. Monomethylmercury is the most toxic mercury compound[16]. Caldwell et al[17] studied the effects of forest fire on the distribution of mercury and monomethylmercury in an arid land reservoir in south-central New Mexico. They found that fire had a twofold effect on mercury concentrations. One, mercury trapped in a catchment was mobilized and found in sediments downstream from the burned area. Two, allochthonous organic carbon may promote the production of monomethylmercury through methylation. Hopkins[6] also identified elevated levels of mercury in the Pecos River (400 ng/L) following the Viveash fire in New Mexico, but the mercury source was undetermined.

Garcia and Carignan [7]explain that forest fires may influence the transport of mercury to aquatic ecosystems. After a medium-intensity fire, the top layer of organic soil is exposed to erosion and transfers particulate and dissolved organic carbon to aquatic systems[18]. The DOC in soil is a vector of some metals including mercury[19] and thus the concentration of mercury is expected to increase in effected waters. High-intensity fires are more likely to cause the organic matter in the soil to mineralize[20] and elements associated with the DOC may volatilize. Garcia and Carignan's [7] research focused on the effects of fire on the methyl mercury contamination of the zooplankton community. Methyl mercury was chosen because it accumulates and is bioamplified along the food web. The DOC loading in the burned lakes observed did not increase even when soil exposure increased following fire. Consequently, little change in methyl mercury concentrations was observed. The methyl mercury concentration may not have increased due to loss through volatilization, or dilution of mercury levels associated with biota due to the nutrient and productivity pulse following fire. For this study DOC was the most important predictor of methyl mercury concentrations, but Garcia and Carignan concluded that this relationship is still not well understood [7].

Cyanide

Increased levels of cyanide may be found in water following forest fires. Some cyanide may be produced naturally during forest fires[21], however a more common cyanide source is fire retardant slurries. Little and Calfee[22] explain that sodium ferrocyanide is often used as an anticorrosion agent in fire retardant. They further explain that sodium ferrocyanide has been shown to release Cn^- ions when exposed to high temperatures or UV radiation. Fire retardant containing sodium ferrocyanide was used in combating the Viveash fire in New Mexico. Hopkins reported levels of cyanide in both water and sediment at the Pecos River at levels up to 120 ug/L[22].

Algal Response

Forest fires impact algal communities in effected watersheds. Fires lead to increased flow, silt loads, and chemical concentrations in watersheds[23]. Water quality is impacted and productivity may be modified due to increased nutrient concentrations and decreased light penetration[5, 24]. Increased nutrients and changes in light penetration effect algae as well as increased heavy metal loads in the watershed following fires[13].

Planas et al [25] examined overall algal response in nine lake watersheds burned (50-100% of watershed area) in 1995. The impacted lakes were sampled for three years following wildfires. In the burnt lakes, benthic chlorophyll-a concentrations were as high as 100 mg/m^2 two years following the fire, with the highest corresponding reference lakes chlorophyll-a concentration of 30 mg/m^2. On average the chlorophyll-a concentrations increased 1.4- to 3- fold in impacted lakes when compared with sixteen reference lakes. Phytoplankton was sampled three times per year, during the spring, summer and fall. Photoautotrophic diatoms increased in burnt lakes. During the three-year study, the most significant algal response was measured during the second year following the fires. Biomass increased and major taxa shifts were seen in burnt lakes. Algal biomass was proportional to the perturbed fraction of the catchment's area divided by the catchment's lake surface area[25].

Pace and Cole [26] assessed respiration following whole lake manipulations. Nutrients were added in the form of inorganic nitrogen and phosphorous. In general, planktonic respiration increased with nutrient addition, and was independent of DOC. Dissolved organic carbon was correlated with respiration in untreated lakes, but not with respiration responses in nutrient treated lakes. There were differences noted in respiration measurements of lakes treated with similar nutrient loads. These differences were attributed to zooplankton grazing determined by food web structure in the lakes [26].

Shin et al [13] examined the impact of cadmium (Cd) transferred by forest fire ash on micro- and macroalgae. Burned wood samples of the impacted watershed showed high concentrations of Cd and were considered to be the main source of Cd observed in receiving waters. Large quantities of Cd have been reported in several forest tree species, particularly in conifers[27, 28]. Cadmium is toxic to laboratory microalgae cultures [29]. The purpose of Shin et al's study was to determine the effects of forest fire ash containing Cd on the growth of the micoralga, *N. oculata* and the macrophyte *Ulva pertusa*. Their research documented a 20% reduction in the growth rate of *N. oculata* at low dosages of Cd (0.224-0.448 mg/L). At higher concentrations of Cd (1.792 mg/L), pigments were completely leached from both the micro- and macroalgal cultures. Bioaccumulations factors for the cultures indicated excessive bioconcentration of Cd[29].

Conclusion

In conclusion the effects of wildfire on water quality include increases in nitrogen, phosphorous, and ion concentrations. Polycyclic aromatic hydrocarbon and dioxin concentrations are anticipated following forest fires. However, limited literature was available on PAH and dioxin concentrations following fires. Malaysian research provided a means of distinguishing PAHs released by combustion of wood from PAHs produced by fossil fuel combustion. Some dioxins have been monitored following fires, but no significant changes were observed. Based on the potential health hazards associated with PAHs and dioxins additional research in these areas may be appropriate in subsequent years following the Rodeo-Chediski fire. Forest fires also lead to increases heavy metal concentrations. Specific heavy metals discussed in the literature included mercury, chromium and cyanide. Finally, forest fires have been shown to effect algal growth through changes in nutrient loading, light penetration, increase in trace metals required for growth, and the addition of some heavy metals that are toxic to algae.

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APPENDIX B

ANNUAL MASS LOADING AND FLOW RATE FOR THE SALT RIVER AND VERDE RIVER

SALT RIVER ANNUAL MASS LOADING AND STREAM FLOW

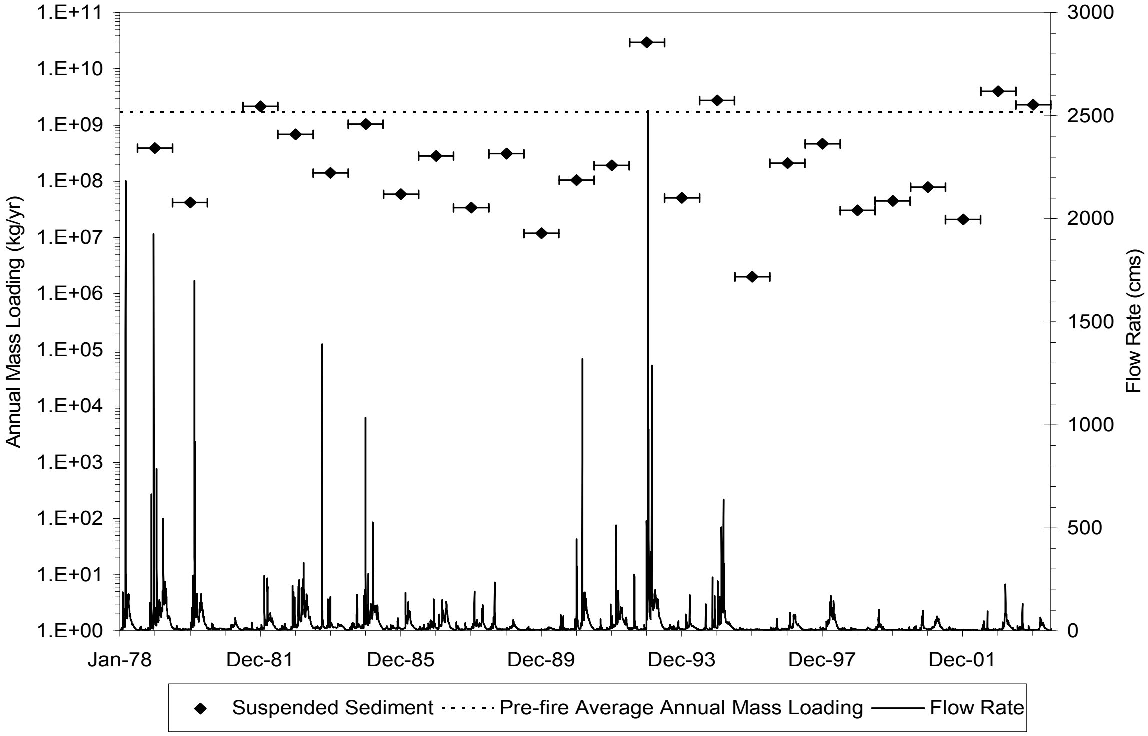

Figure B.1 Annual suspended sediment mass loading and flow rate in the Salt River at USGS gauging station 09498500

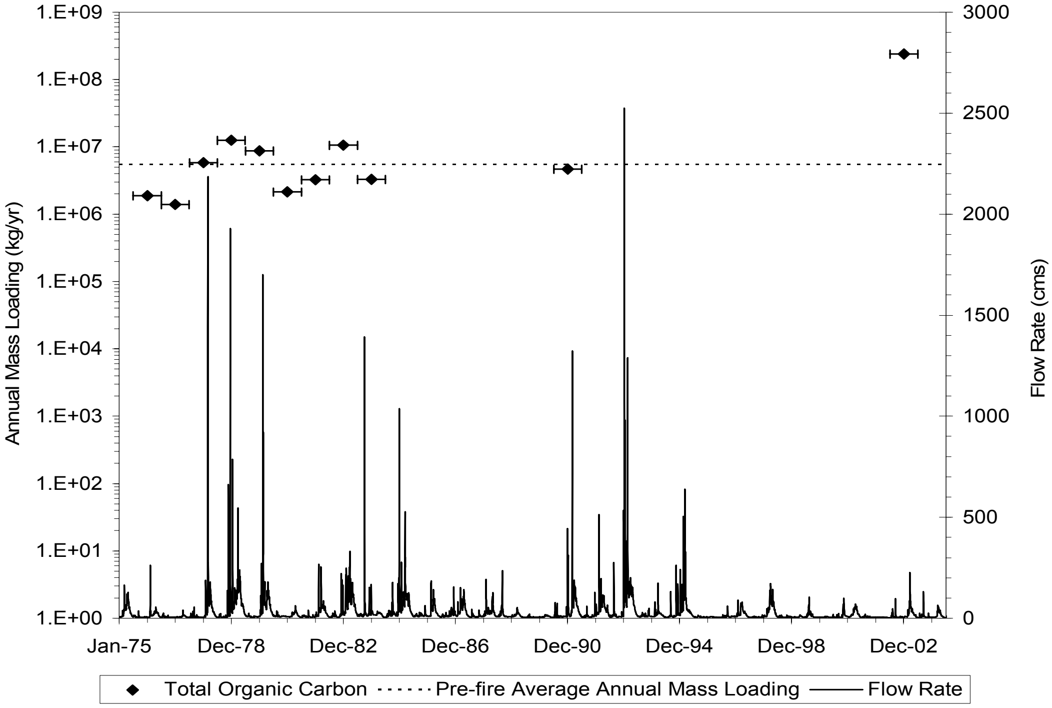

Figure B.2 Annual total organic carbon mass loading and flow rate in the Salt River at USGS gauging station 09498500

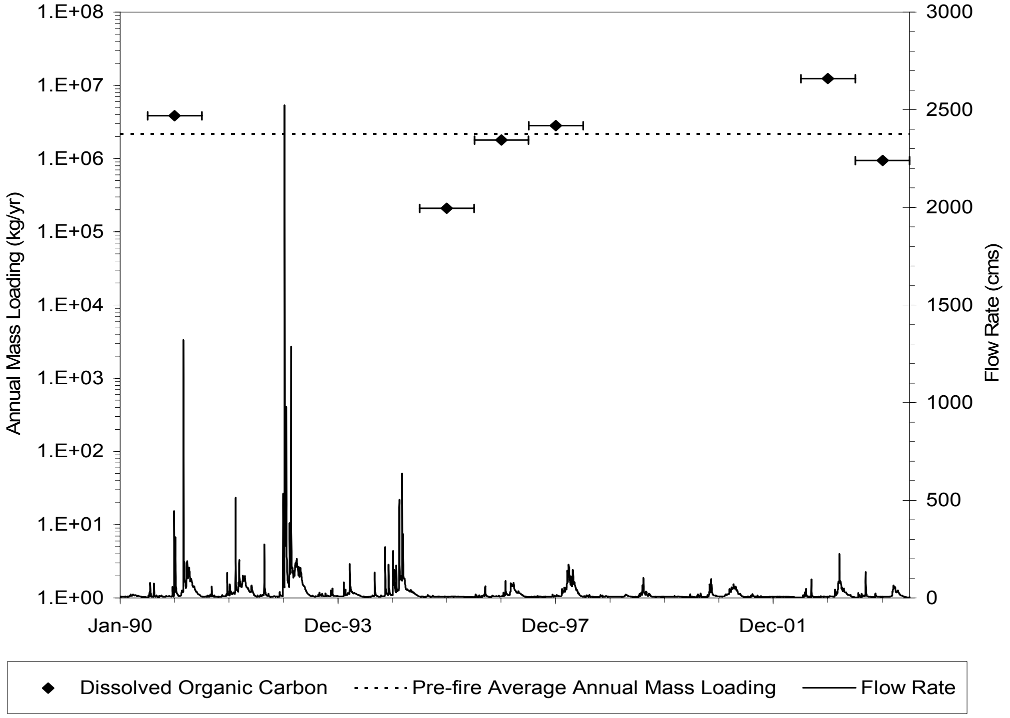

Figure B.3 Annual dissolved organic carbon mass loading and flow rate in the Salt River at USGS gauging station 09498500

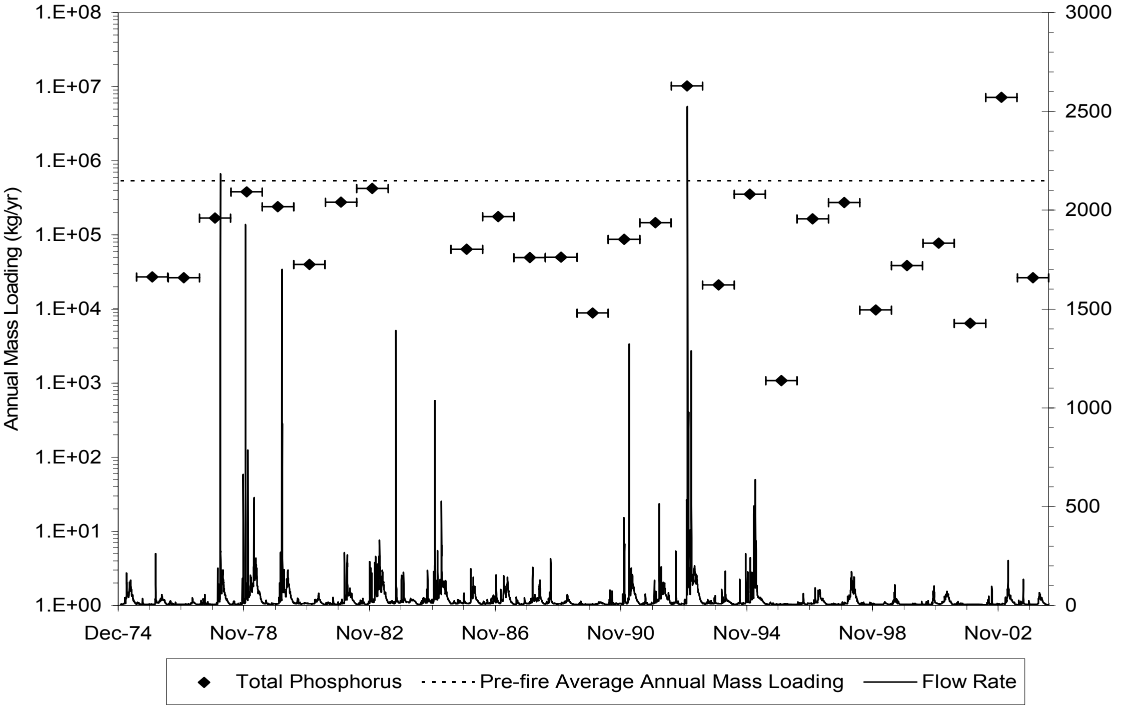

Figure B.4 Annual total phosphorus mass loading and flow rate in the Salt River at USGS gauging station 09498500

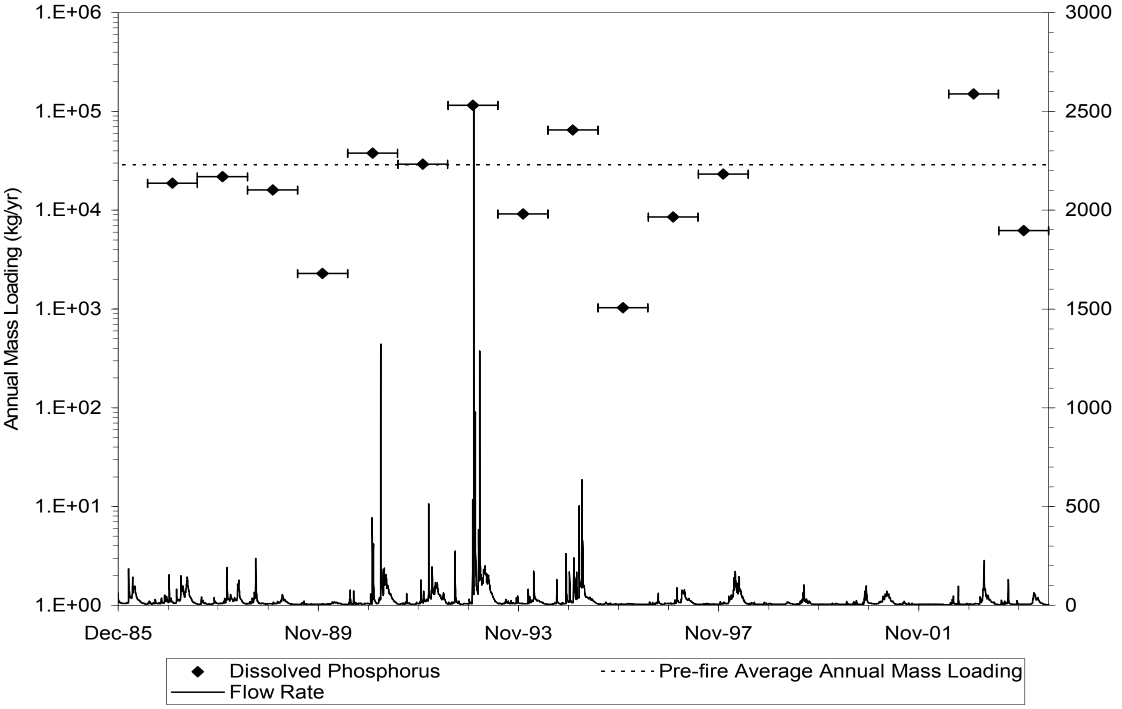

Figure B.5 Annual dissolved phosphorus mass loading and flow rate in the Salt River at USGS gauging station 09498500

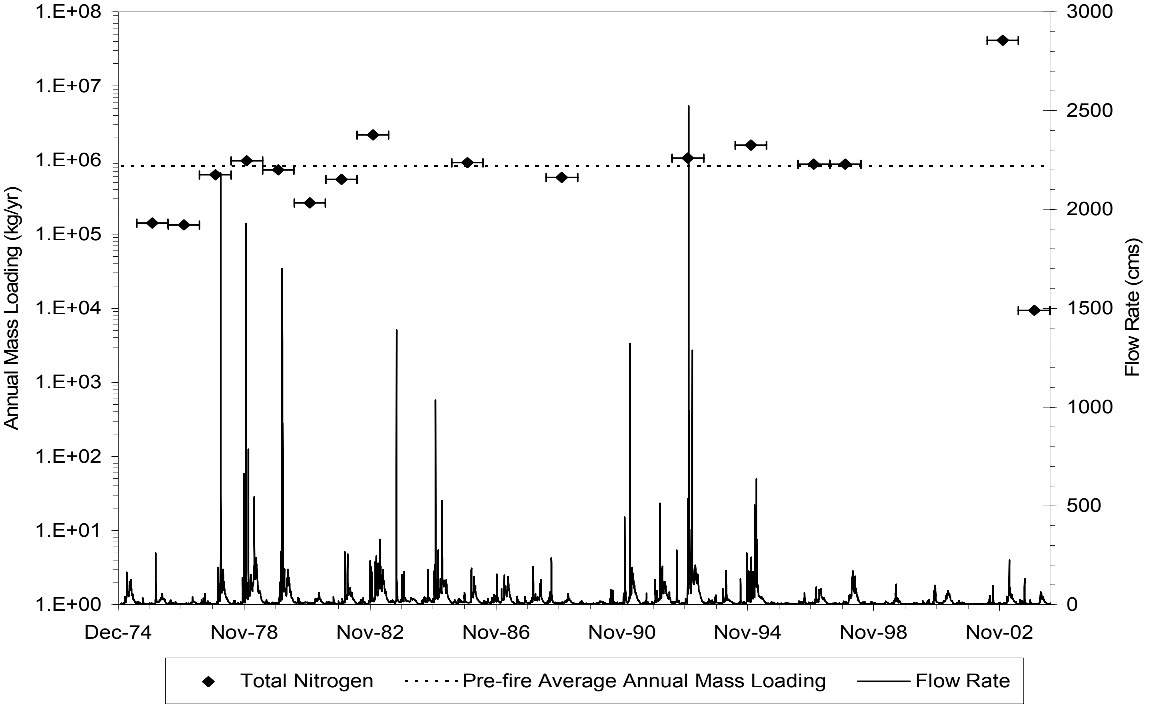

Figure B.6 Annual total nitrogen mass loading and flow rate in the Salt River at USGS gauging station 09498500

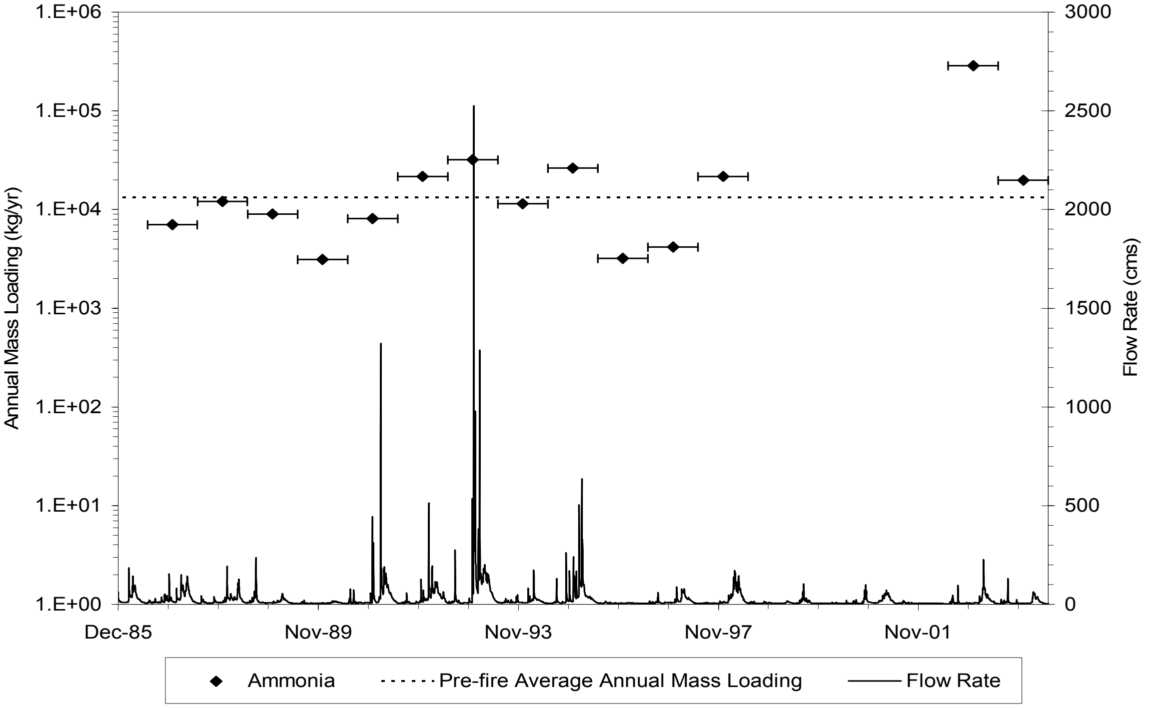

Figure B.7 Annual ammonia mass loading and flow rate in the Salt River at USGS gauging station 09498500

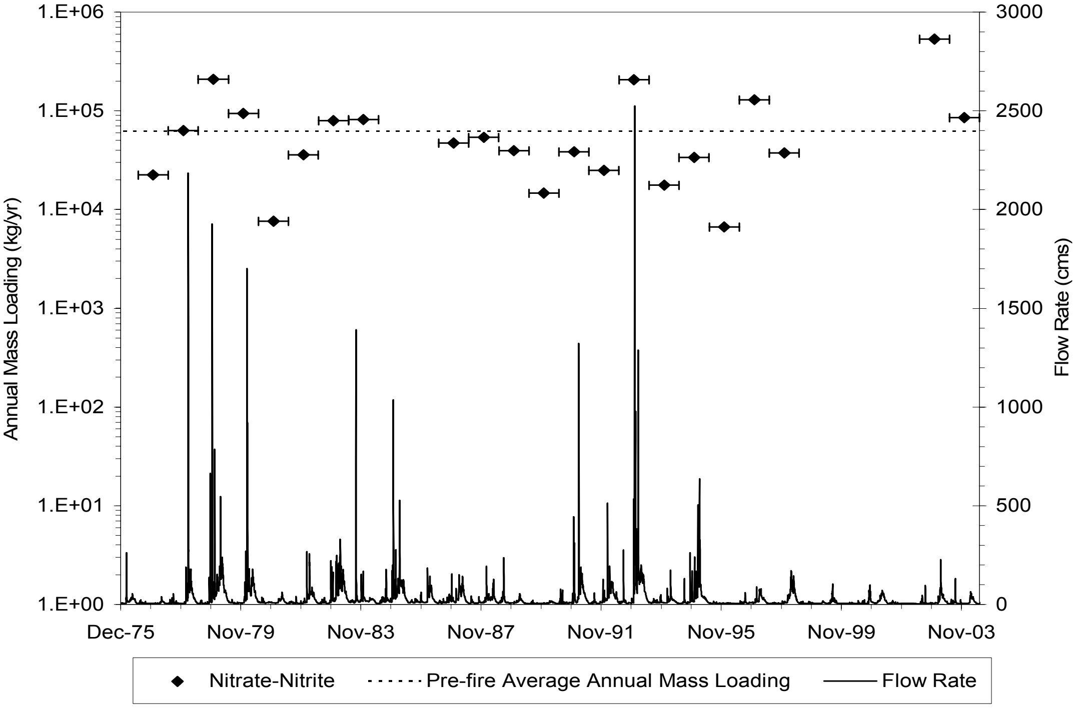

Figure B.8 Annual nitrate-nitrite mass loading and flow rate in the Salt River at USGS gauging station 09498500

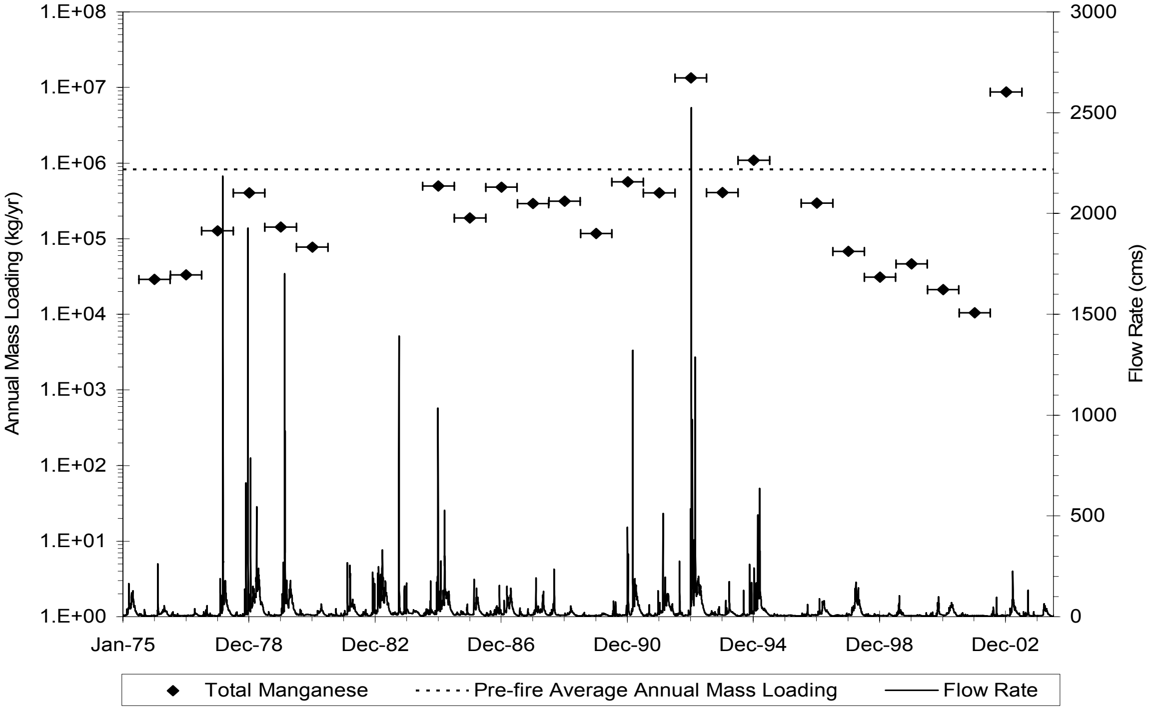

Figure B.9 Annual total manganese mass loading and flow rate in the Salt River at USGS gauging station 09498500

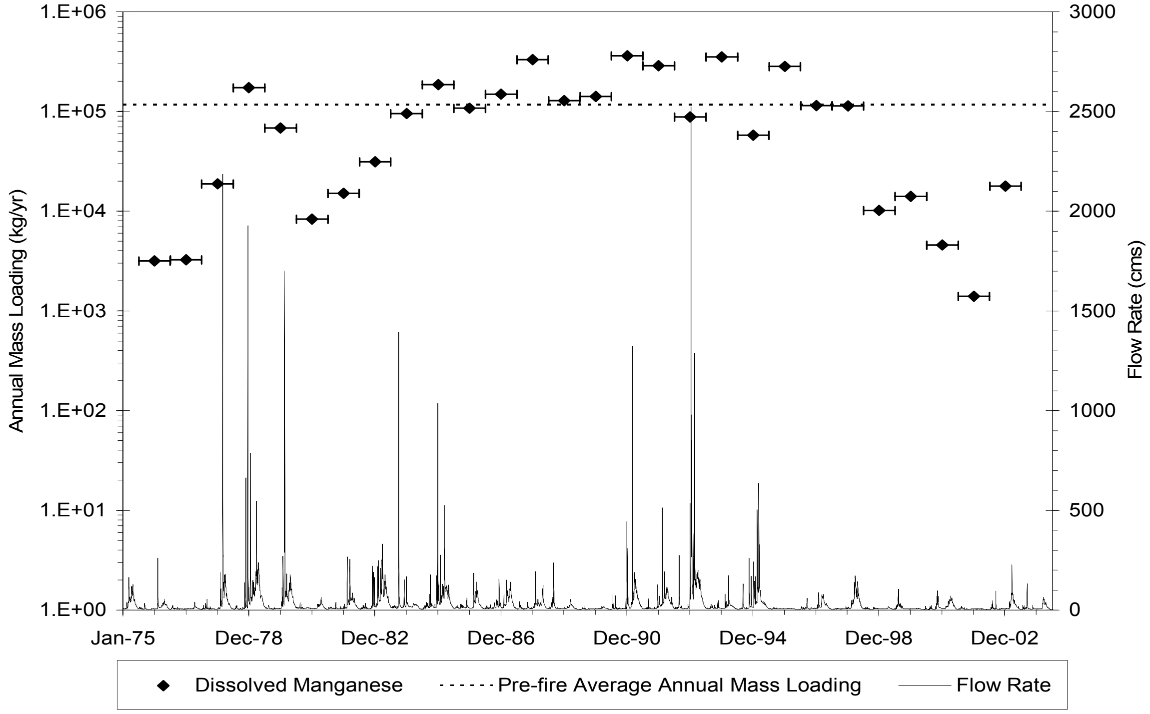

Figure B.10 Annual dissolved manganese mass loading and flow rate in the Salt River at USGS gauging station 09498500

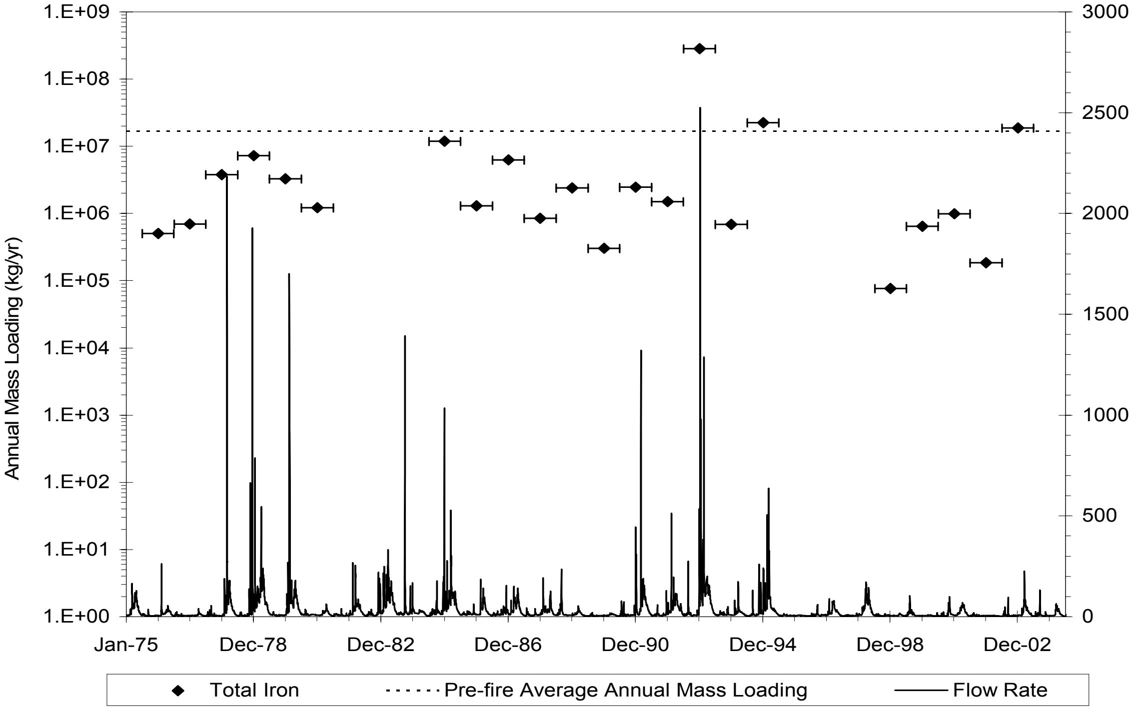

Figure B.11 Annual total iron mass loading and flow rate in the Salt River at USGS gauging station 09498500

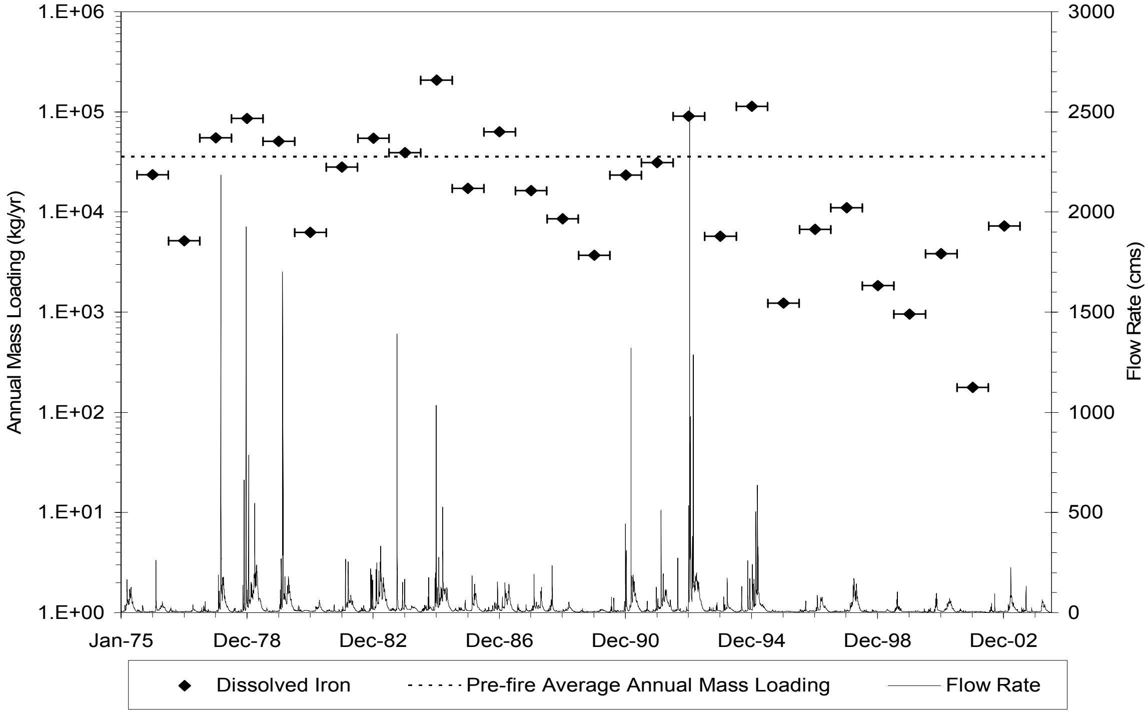

Figure B.12 Annual dissolved iron mass loading and flow rate in the Salt River at USGS gauging station 09498500

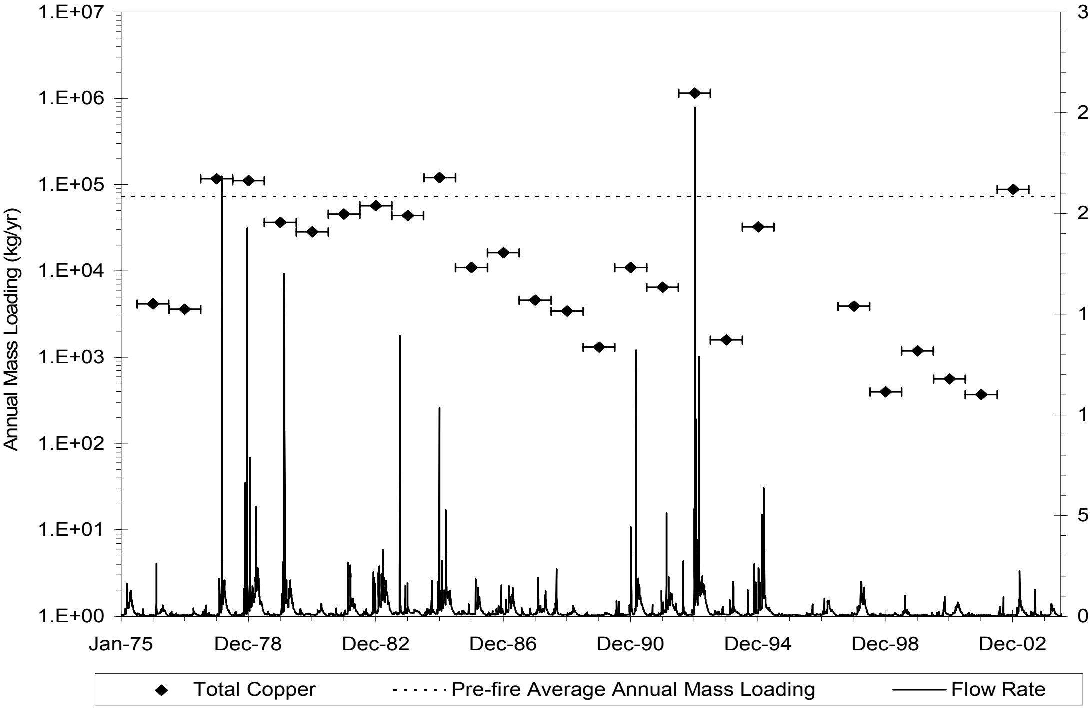

Figure B.13 Annual total copper mass loading and flow rate in the Salt River at USGS gauging station 09498500

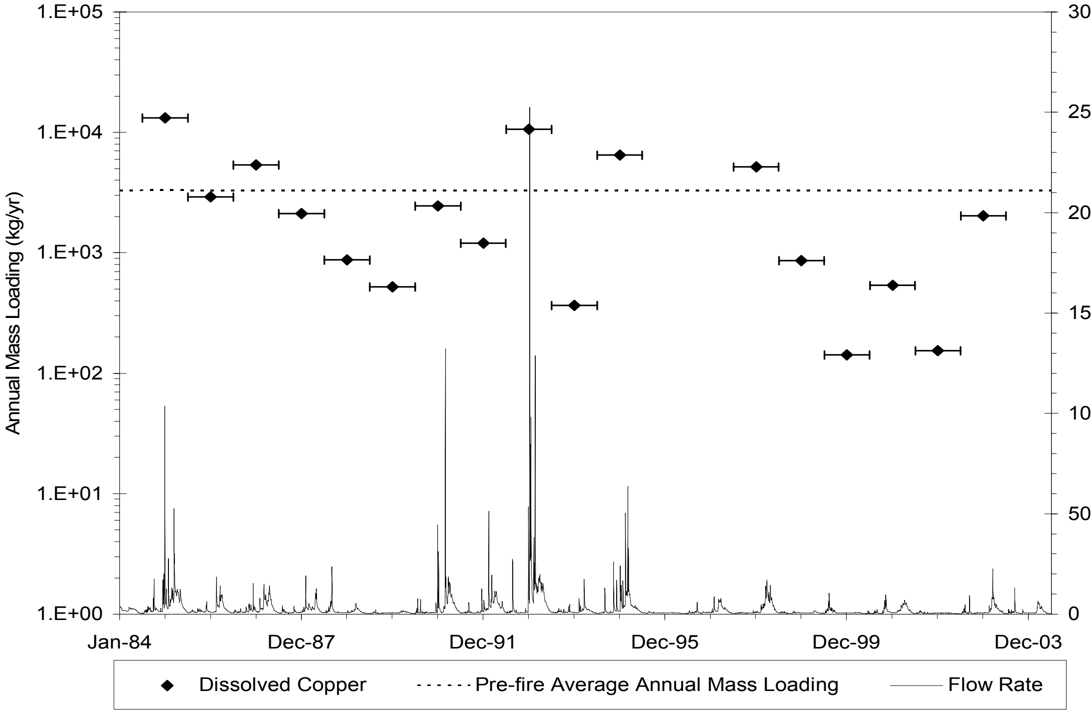

Figure B.14 Annual dissolved copper mass loading and flow rate in the Salt River at USGS gauging station 09498500

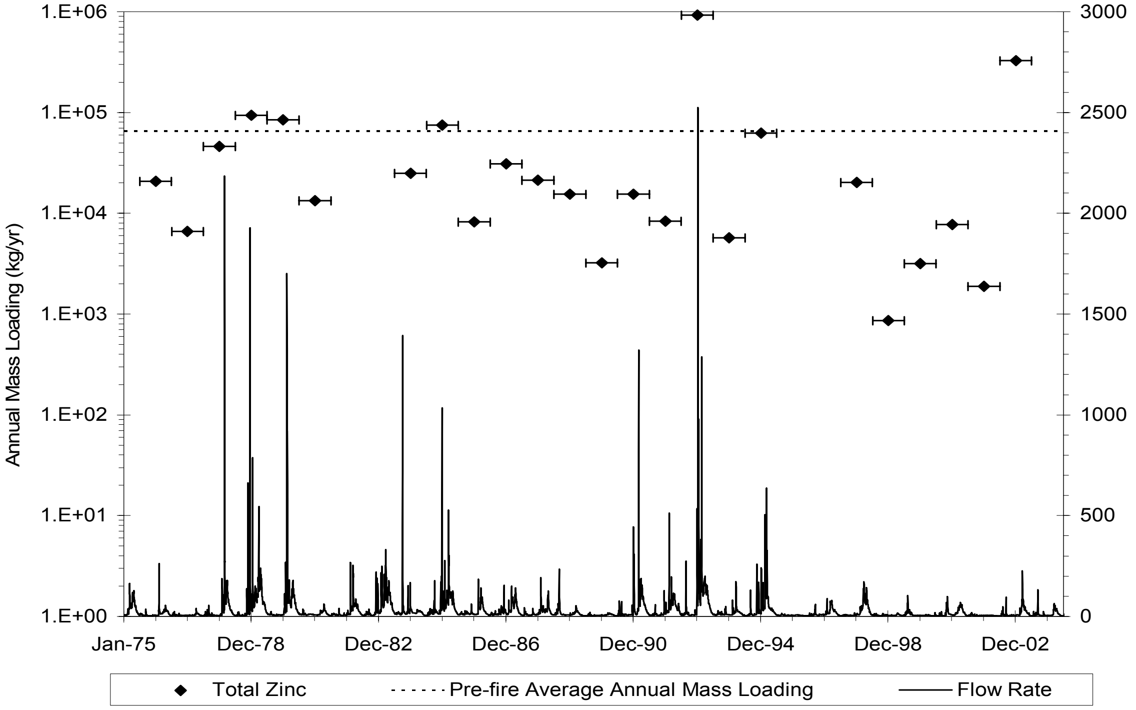

Figure B.15 Annual total zinc mass loading and flow rate in the Salt River at USGS gauging station 09498500

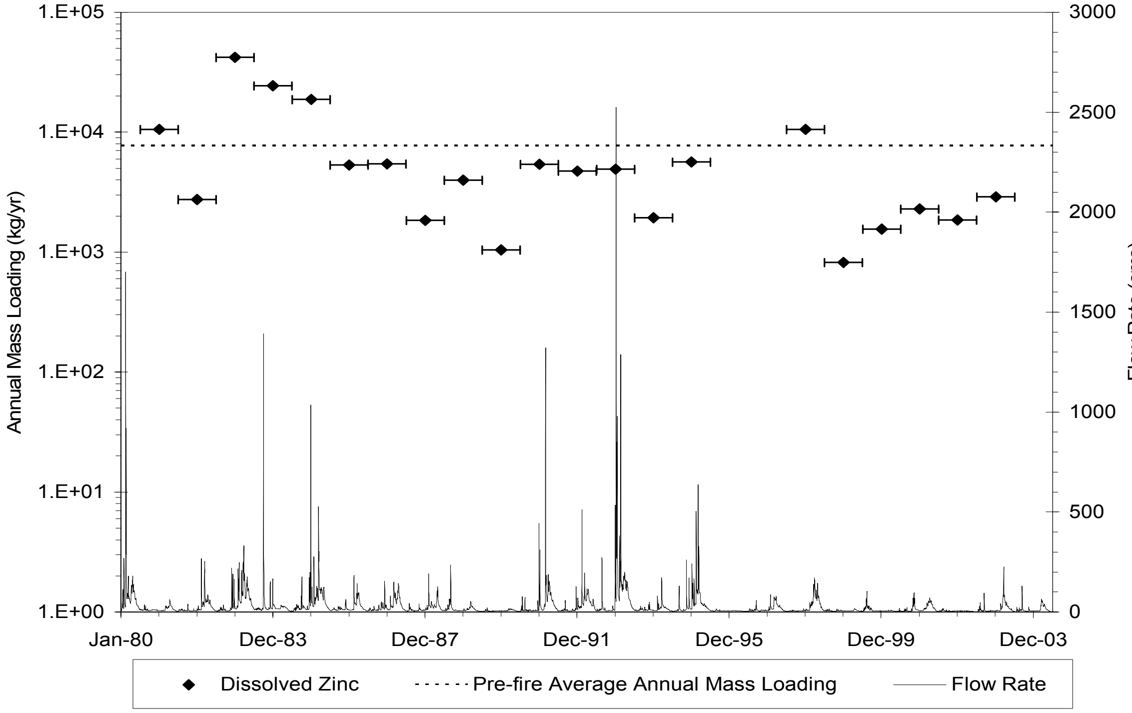

Figure B.16 Annual dissolved zinc mass loading and flow rate in the Salt River at USGS gauging station 09498500

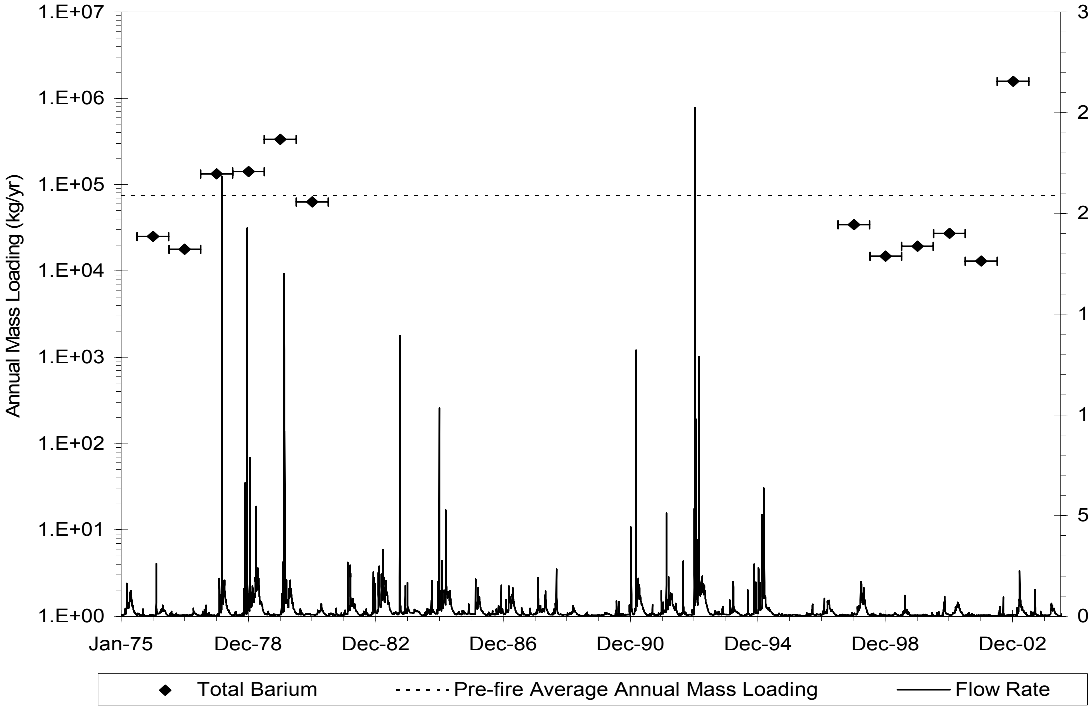

Figure B.17 Annual total barium mass loading and flow rate in the Salt River at USGS gauging station 09498500

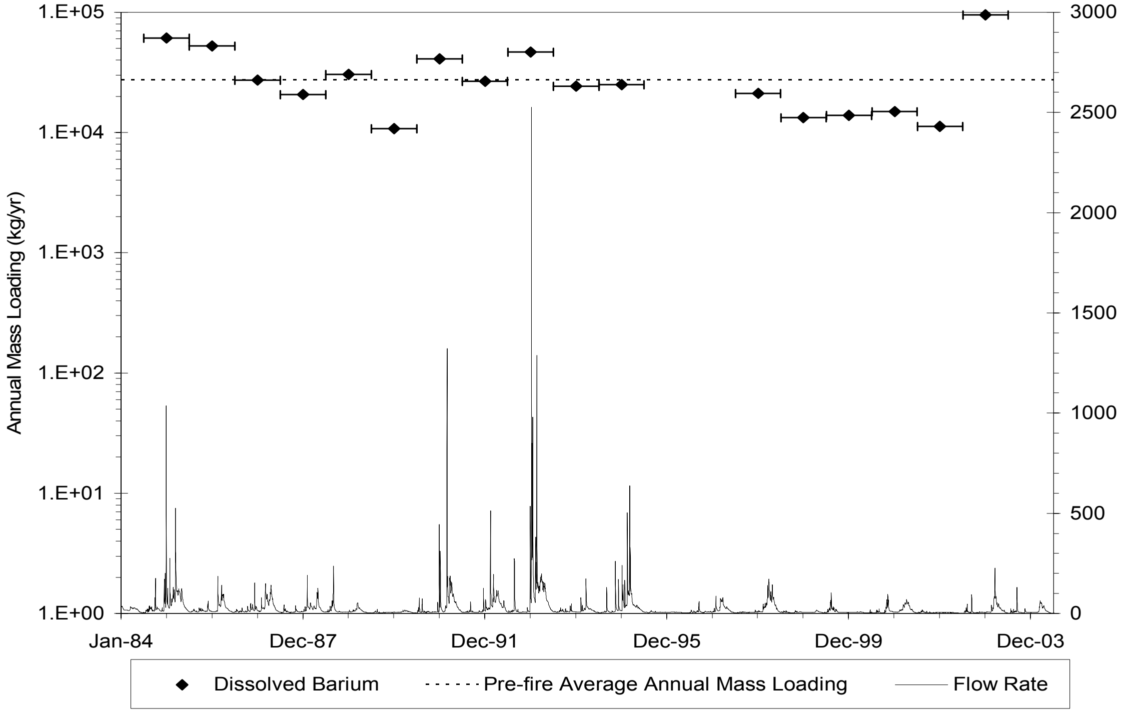

Figure B.18 Annual dissolved barium mass loading and flow rate in the Salt River at USGS gauging station 09498500

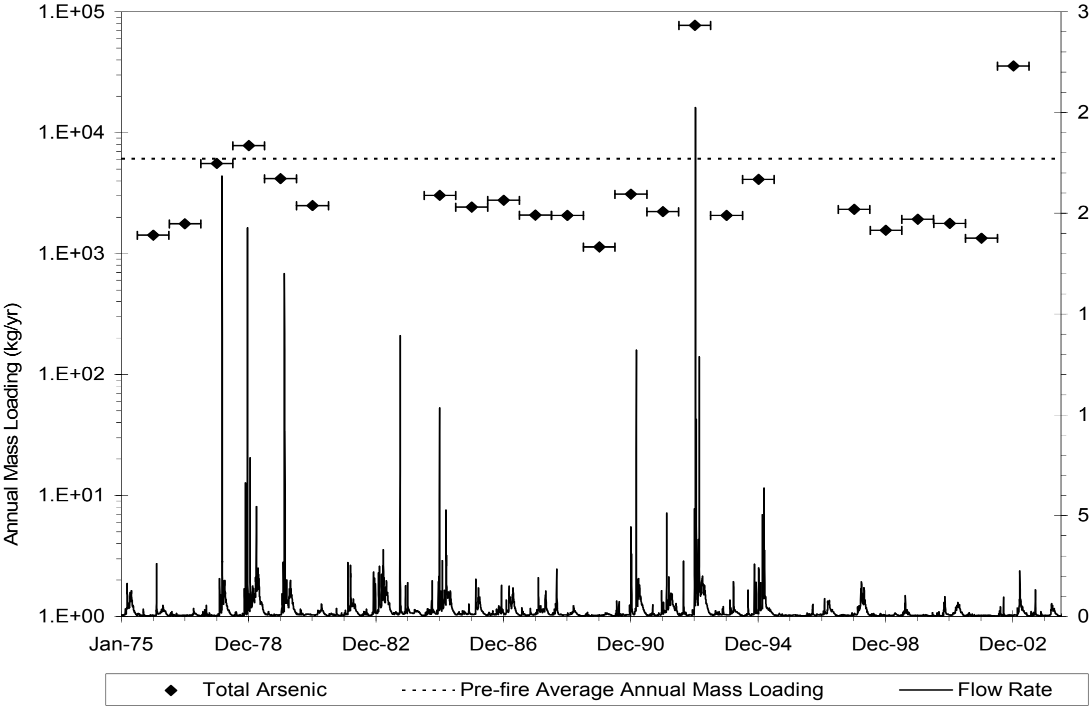

Figure B.19 Annual total arsenic mass loading and flow rate in the Salt River at USGS gauging station 09498500

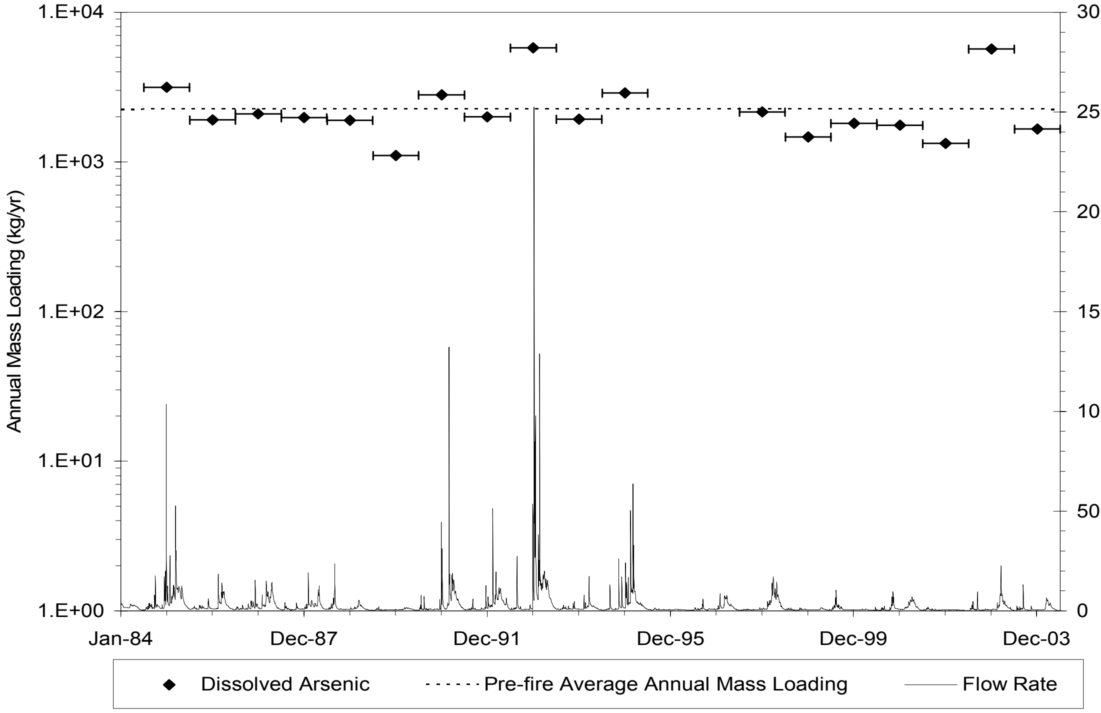

Figure B.20 Annual dissolved arsenic mass loading and flow rate in the Salt River at USGS gauging station 09498500

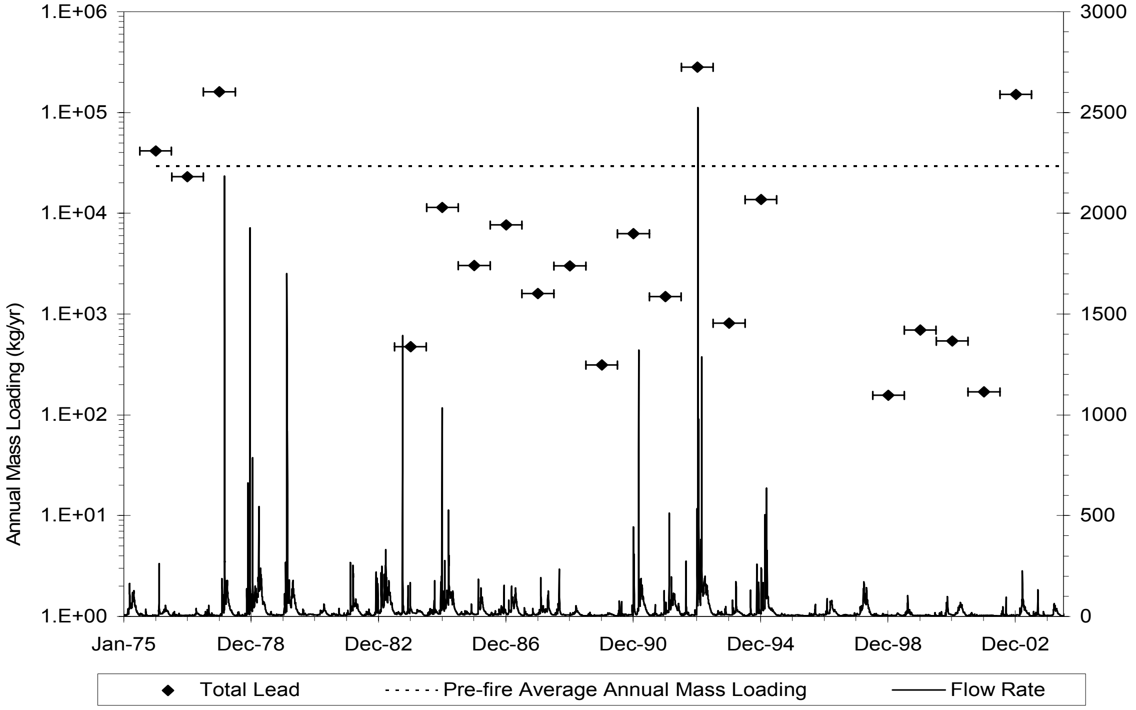

Figure B.21 Annual total lead mass loading and flow rate in the Salt River at USGS gauging station 09498500

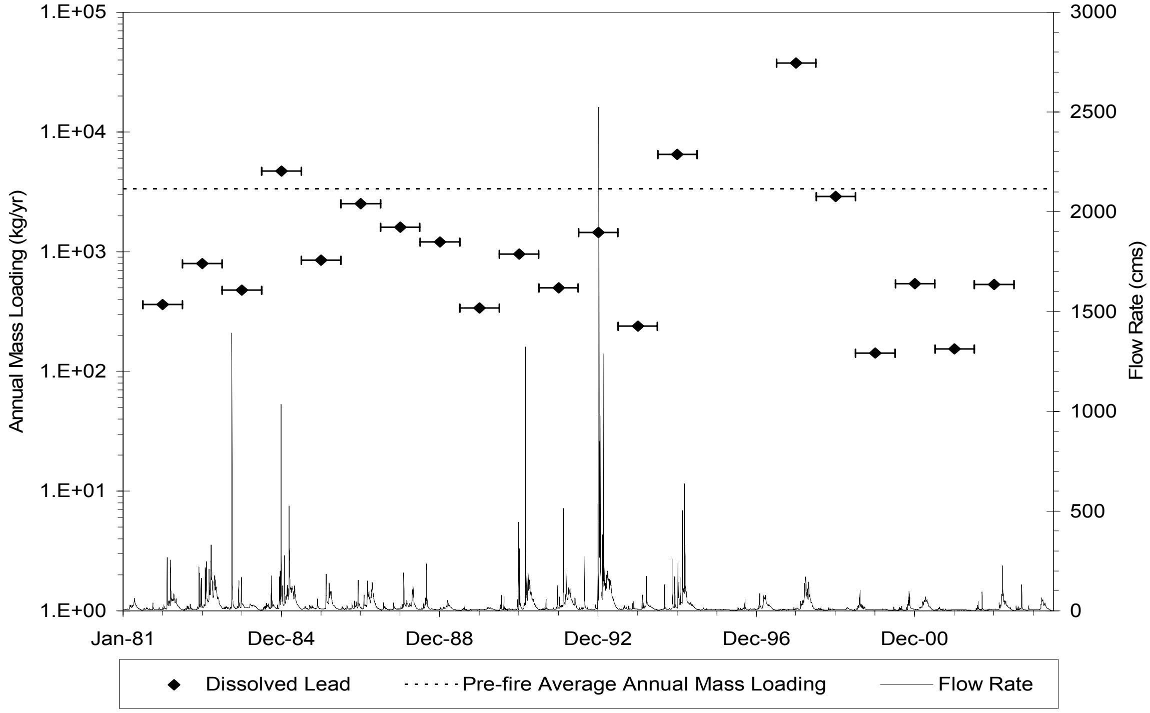

Figure B.22 Annual dissolved lead mass loading and flow rate in the Salt River at USGS gauging station 09498500

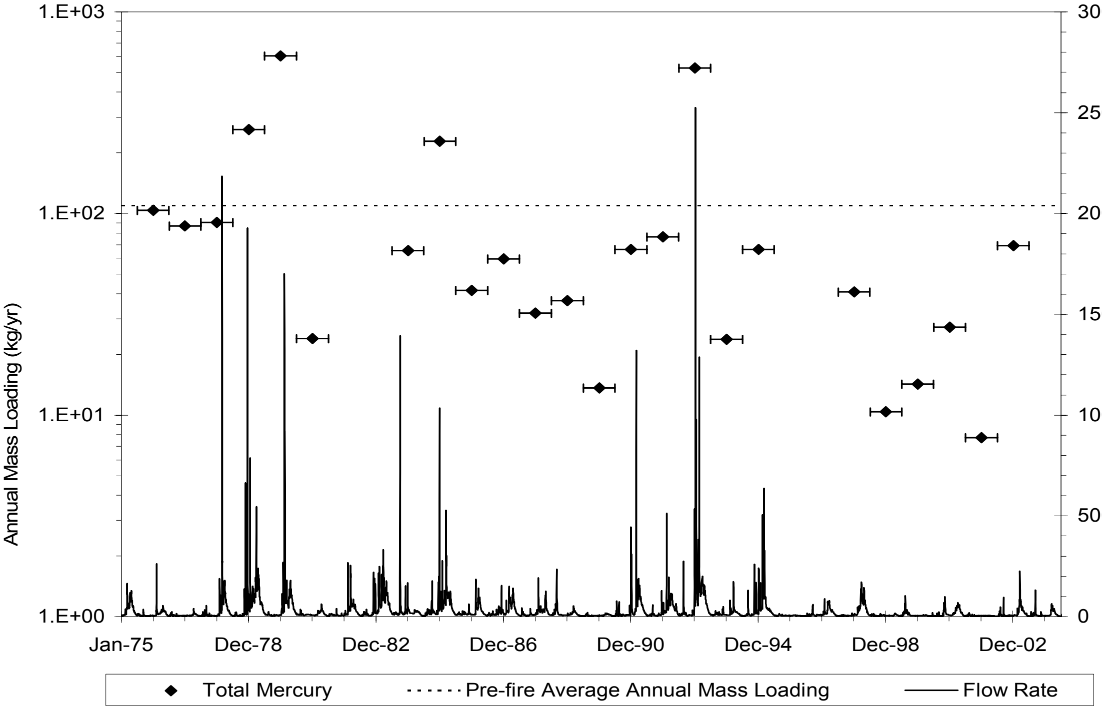

Figure B.23 Annual total mercury mass loading and flow rate in the Salt River at USGS gauging station 09498500

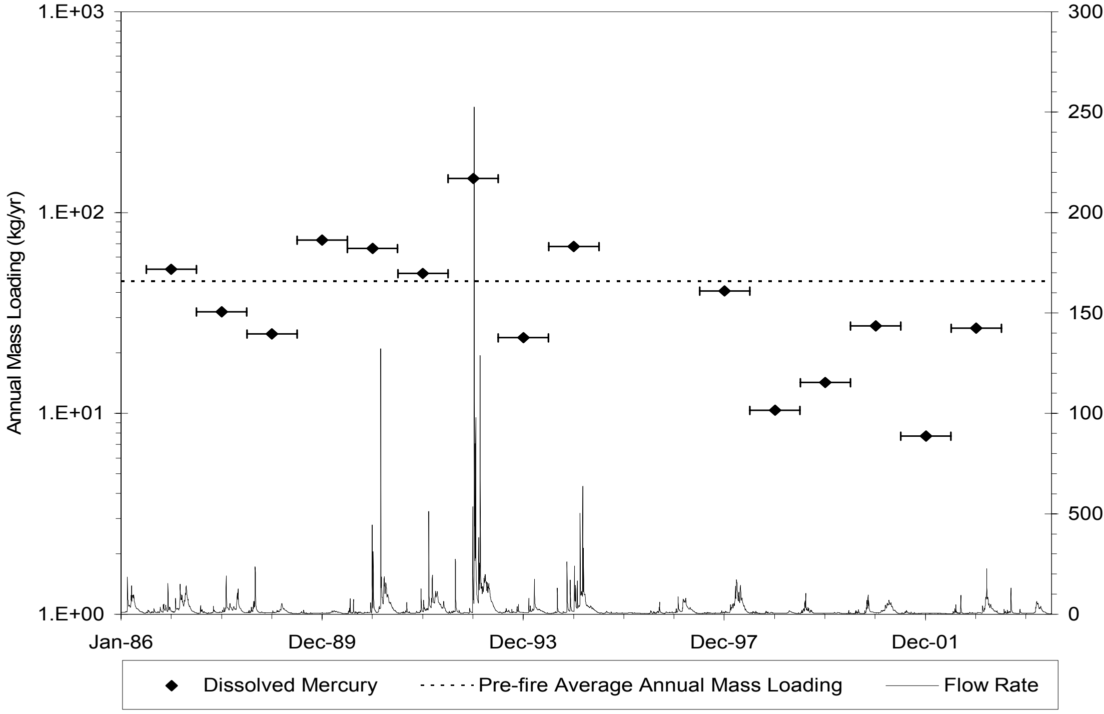

Figure B.24 Annual dissolved mercury mass loading and flow rate in the Salt River at USGS gauging station 09498500

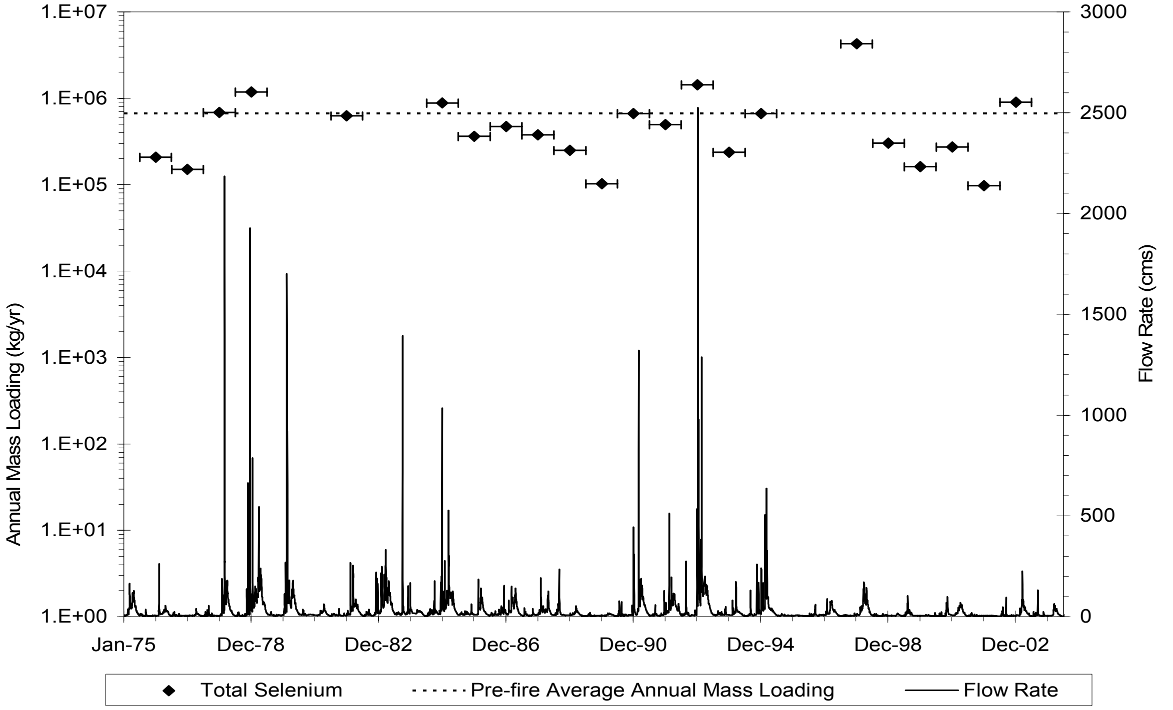

Figure B.25 Total selenium annual mass loading and flow rate in the Salt River at USGS gauging station 09498500

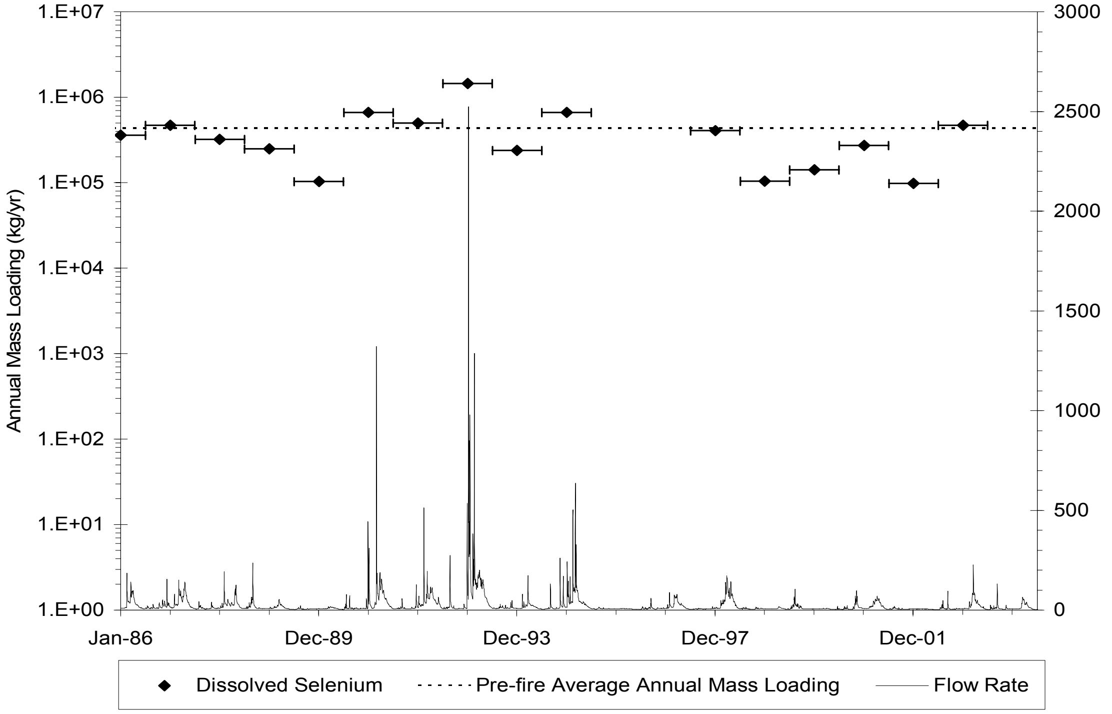

Figure B.26 Dissolved selenium annual mass loading and flow rate in the Salt River at USGS gauging station 09498500

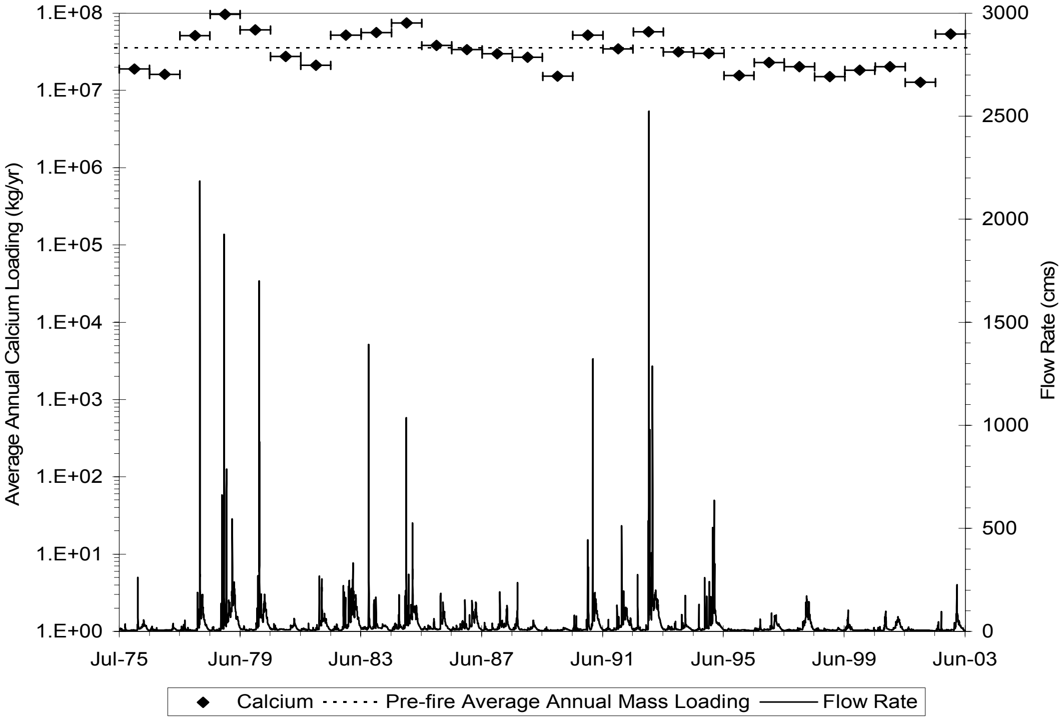

Figure B.27 Annual calcium mass loading and flow rate in the Salt River at USGS gauging station 09498500

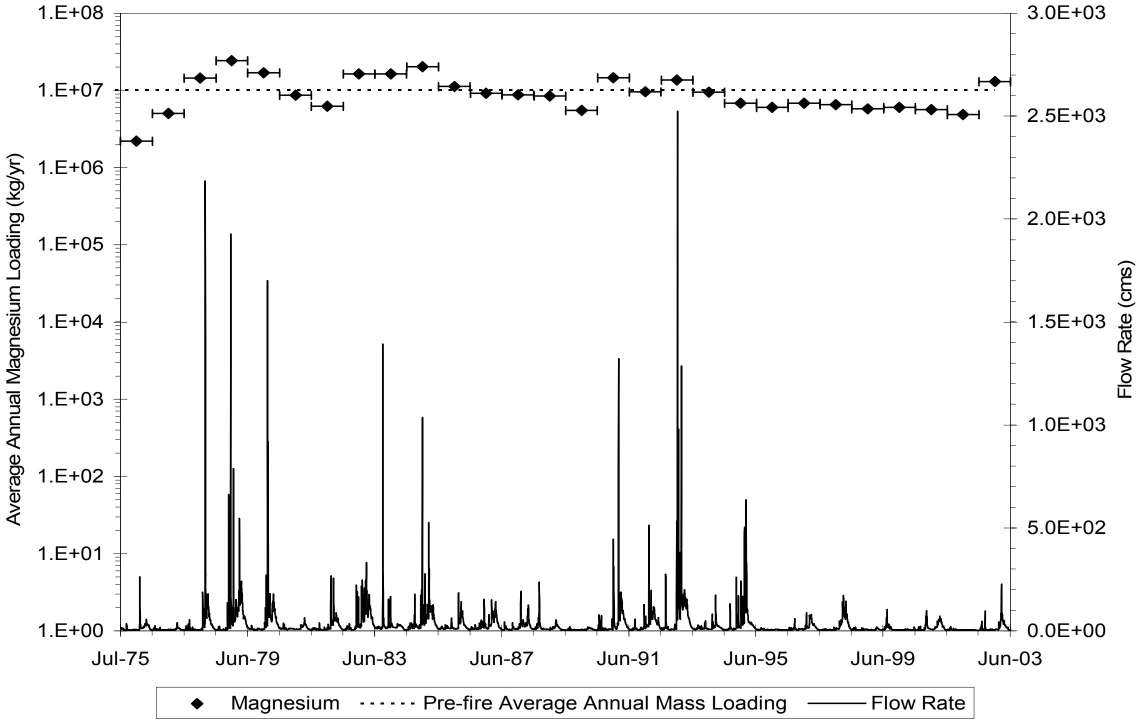

Figure B.28 Annual magnesium mass loading and flow rate in the Salt River at USGS gauging station 09498500

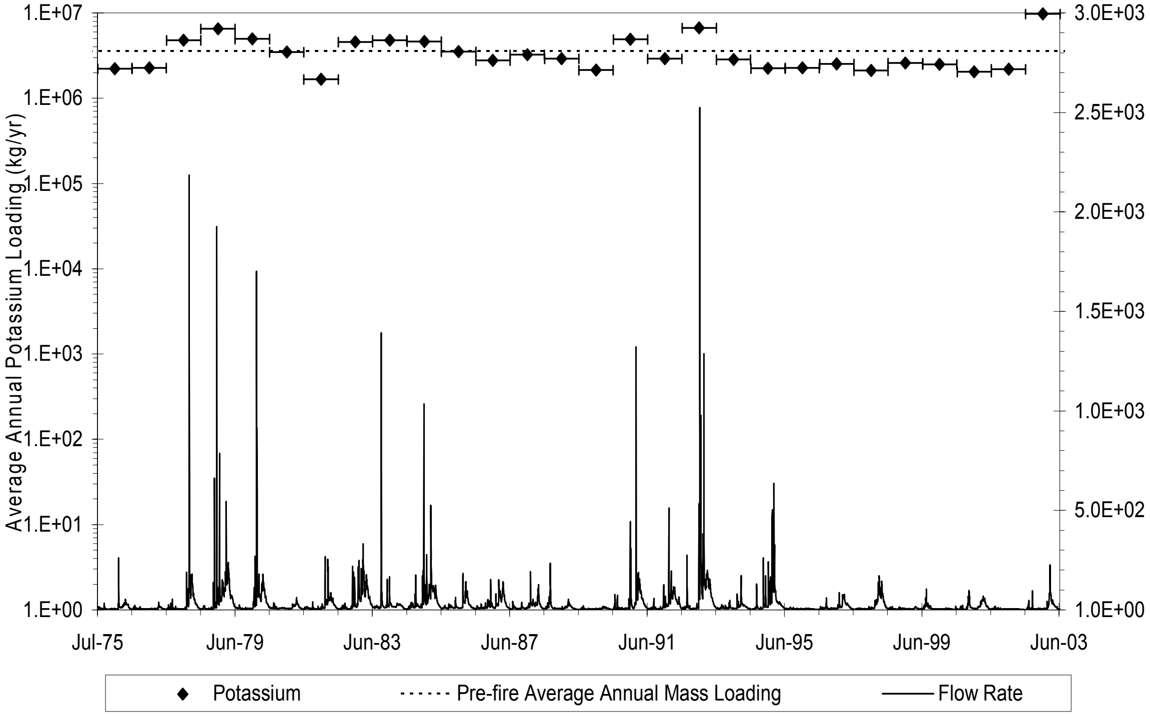

Figure B.29 Annual potassium mass loading and flow rate in the Salt River at USGS gauging station 09498500

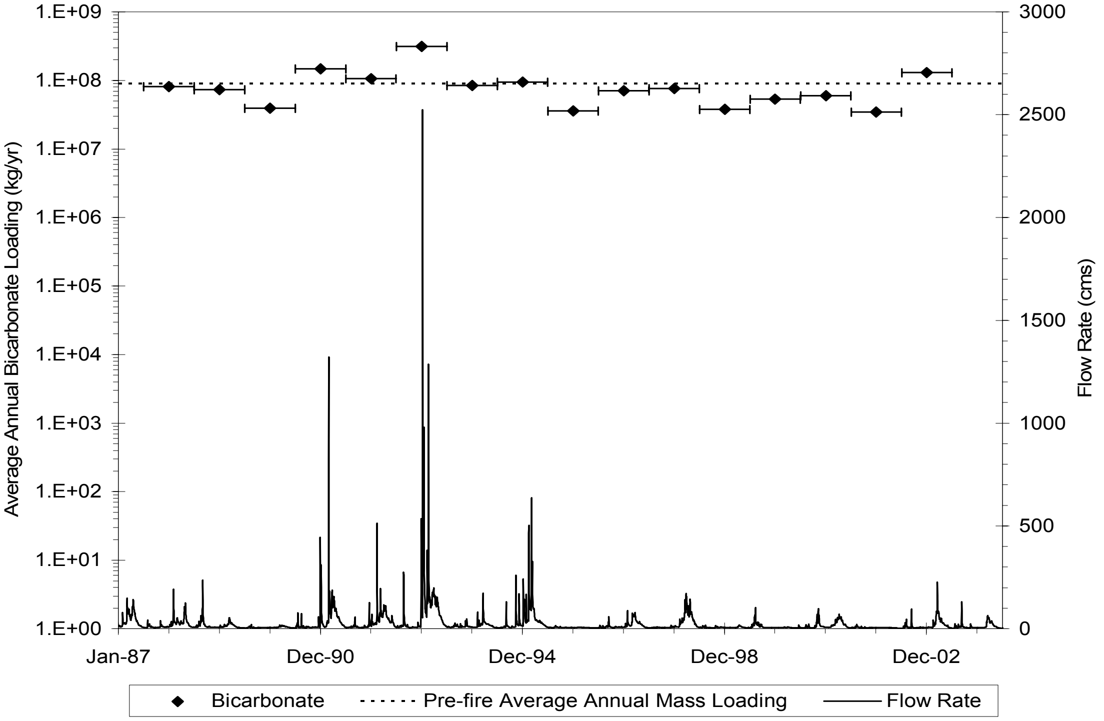

Figure B.30 Annual bicarbonate mass loading and flow rate in the Salt River at USGS gauging station 09498500

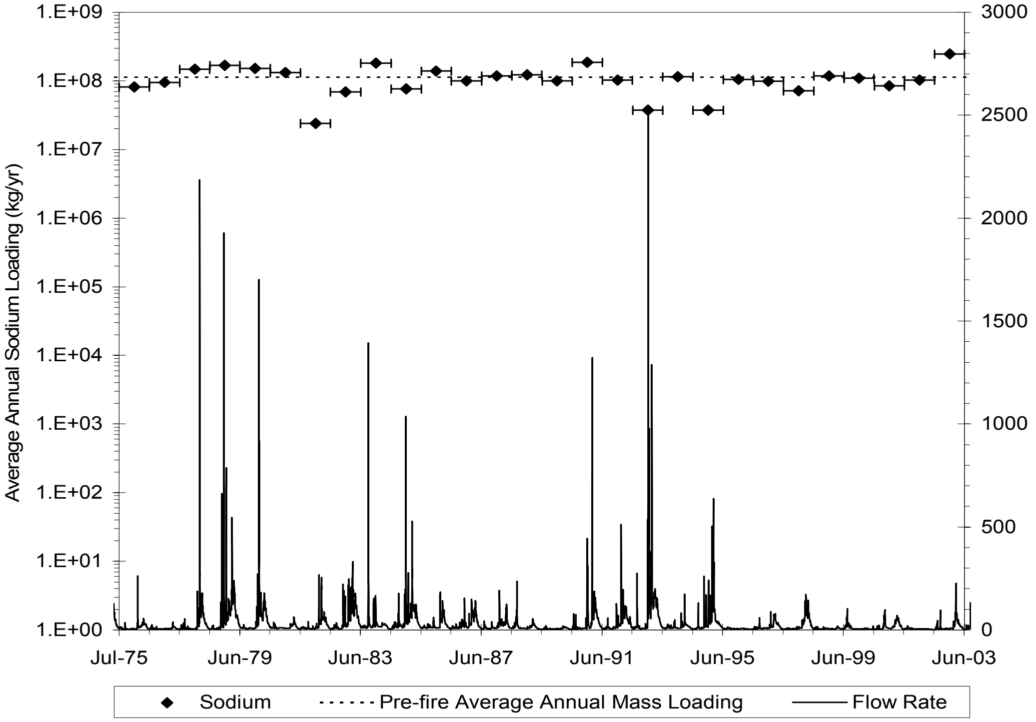

Figure B.31 Annual sodium mass loading and flow rate in the Salt River at USGS gauging station 09498500

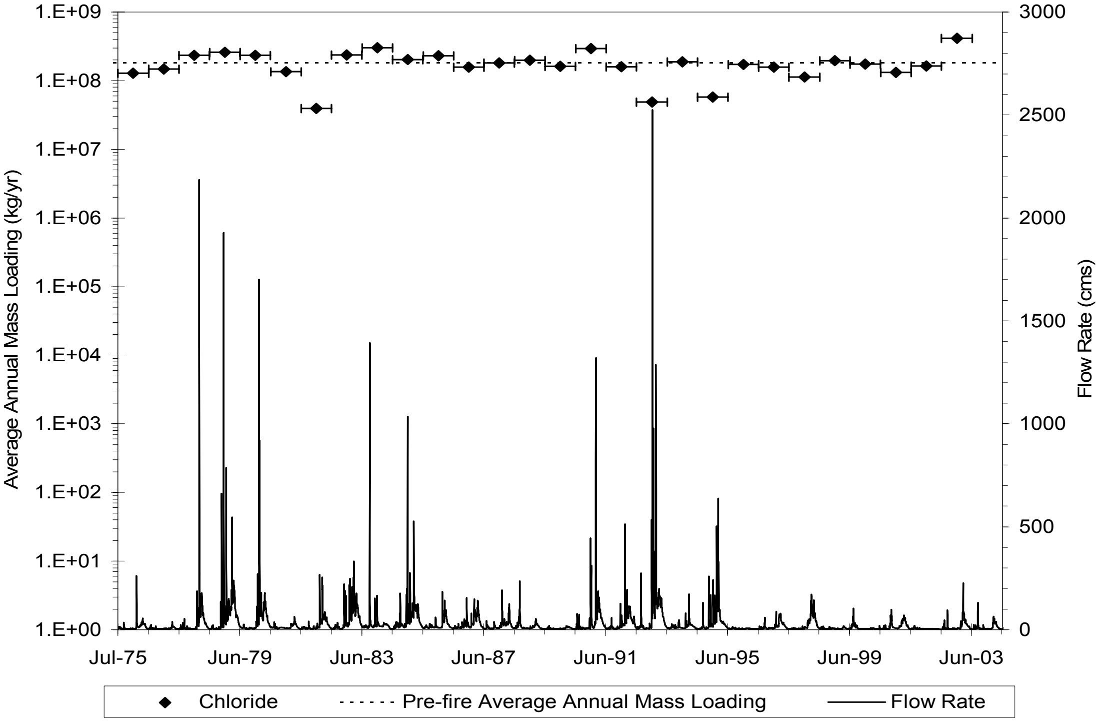

Figure B.32 Annual chloride mass loading and flow rate in the Salt River at USGS gauging station 09498500

VERDE RIVER ANNUAL MASS LOADING AND STREAM FLOW

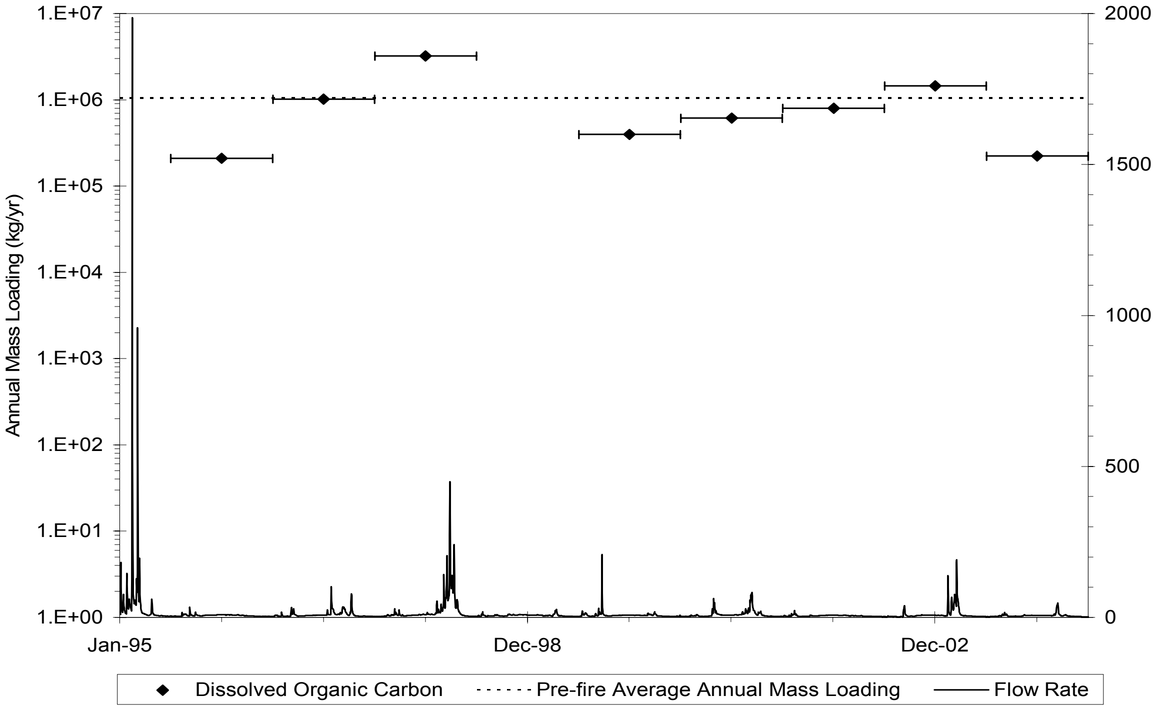

Figure B.33 Annual dissolved organic carbon mass loading and flow rate in the Verde River at USGS gauging station 09508500

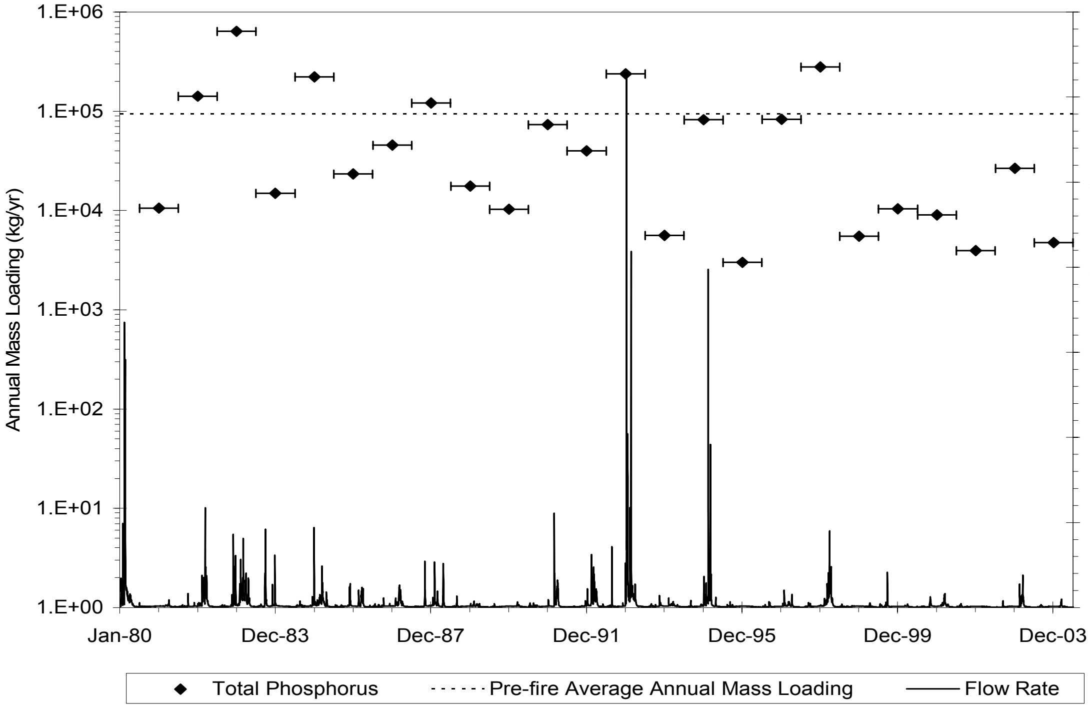

Figure B.34 Annual dissolved phosphorus mass loading and flow rate in the Verde River at USGS gauging station 09508500

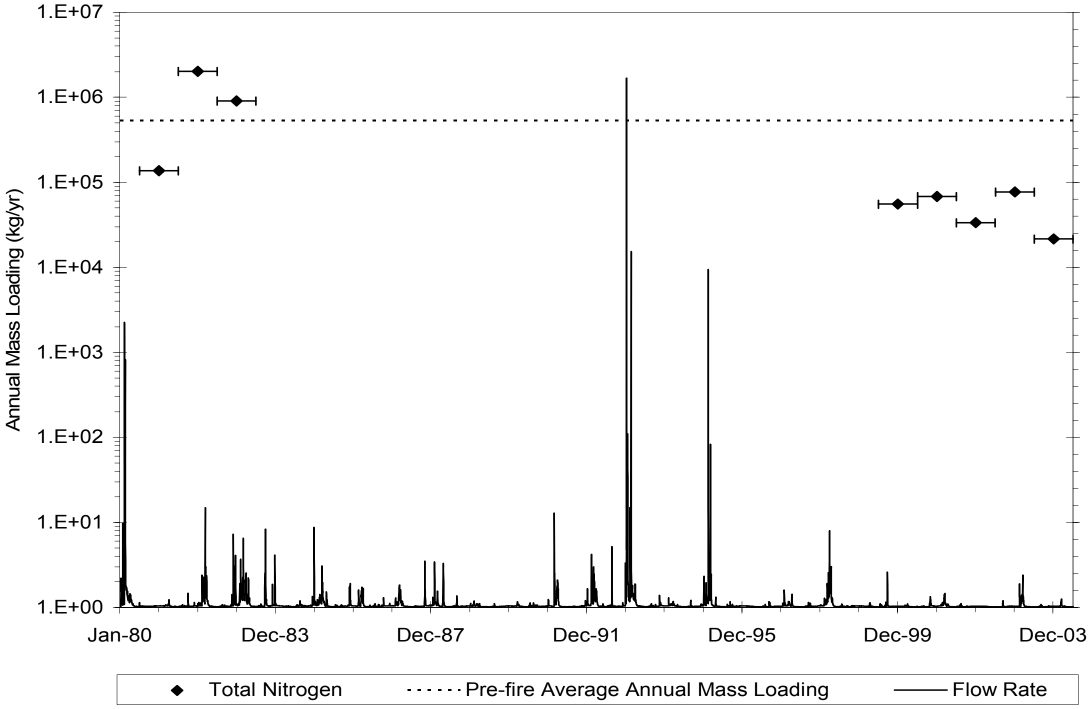

Figure B.35 Annual total nitrogen mass loading and flow rate in the Verde River at USGS gauging station 09508500

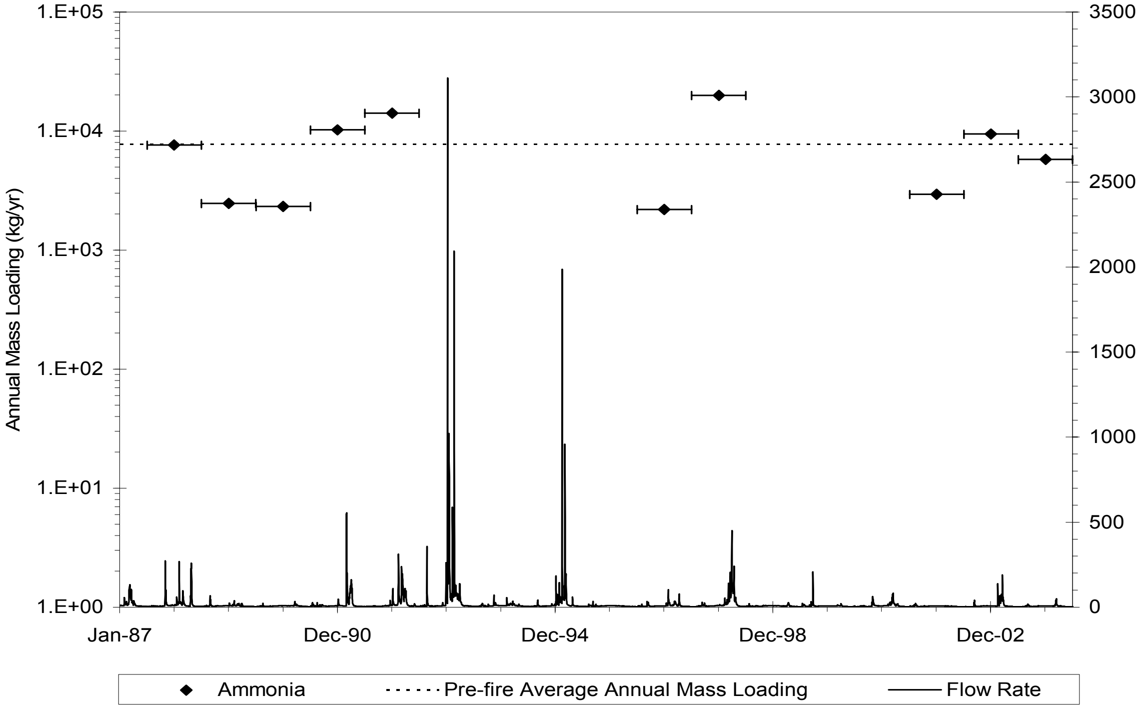

Figure B.36 Annual ammonia mass loading and flow rate in the Verde River at USGS gauging station 09508500

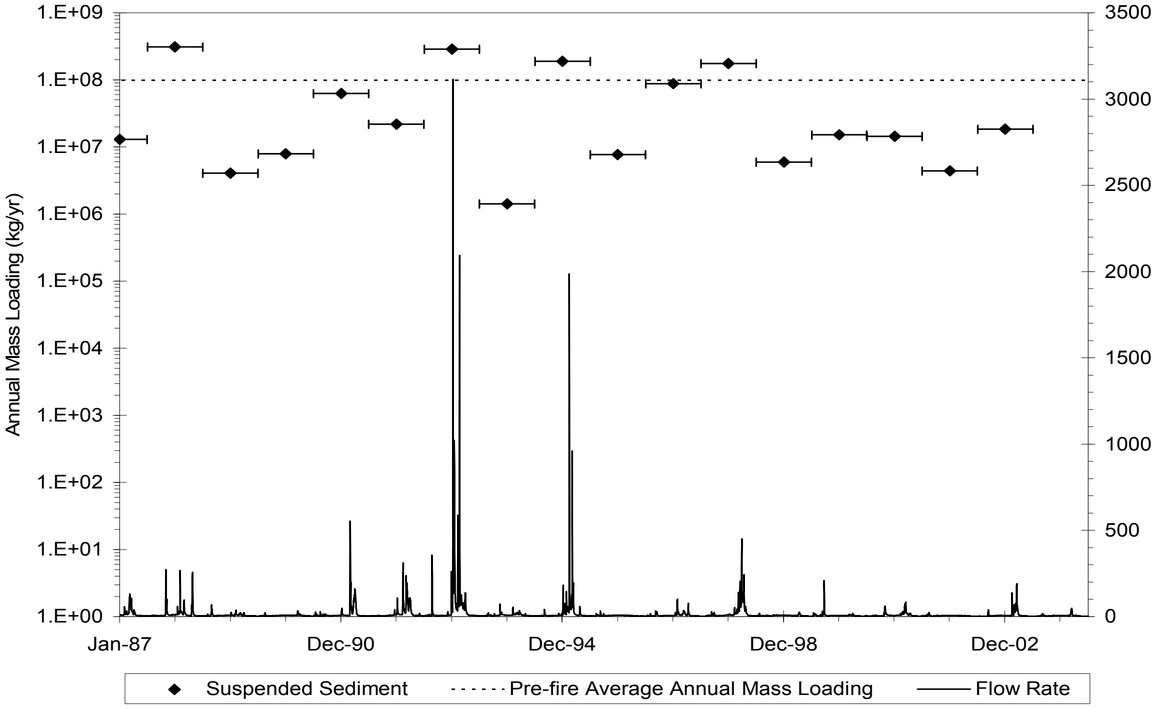

Figure B.37 Annual suspended sediment mass loading and flow rate in the Verde River at USGS gauging station 09508500

APPENDIX C

ADSORPTION OF BIS-PHENOL A BY CHARCOAL FROM A FOREST FIRE

Bisphenol A (BPA) is an industrial chemical that is found in the environment and a known endocrine disruptor. It is used to create epoxy resins used for adhesives and industrial protective coatings, and polycarbonate plastic used for water bottles and cell phones (American Plastics Council 2004). BPA is considered non-carcinogenic and can be degraded in the environment.

The objective of this experiment was to determine the adsorption capacity of BPA by charcoal from a forest fire relative to powder activate carbon (PAC). A previous study was completed by Yoon et al. (2003) to determine the adsorption of BPA by PAC. The methods presented in the study conducted by Yoon et al. (2003) were followed. However, a contact time of 7 days was selected for this experiment and the type of PAC used was Calgon WPM.

The results of this experiment are presented in Figures C.1 and C.2. The HPLC results for the adsorption of BPA by PAC were excluded for all PAC doses greater than 6mg/L due to pressure fluctuations. In addition, the standard deviation of samples ran in triplicate are shown in Table C.1. The standard deviation of the triplicate samples was less than 5%, with the exception of sample for the charcoal concentration of 6mg/L. The high standard deviation is potentially due to minor pressure fluctuations or inadequate mixing of the sample prior to running the analysis.

Table C.3 Standard deviation of samples ran in triplicate

$C_{charcoal}$ (mg/L)	$C_{BPA, final}$ (nmol)	Standard Deviation
1	66.2	1.69
6	65.4	6.92
10	69.8	2.74
40	53	1.53

The PAC is a better adsorbent than charcoal. At a PAC dose of 4mg/L only 5% of the BPA remained in solution. However a charcoal dose of 250mg/L was required to obtain the same level of removal. Therefore, in the natural environment charcoal resulting from a forest fire can adsorb contaminants, however high charcoal concentrations are required.

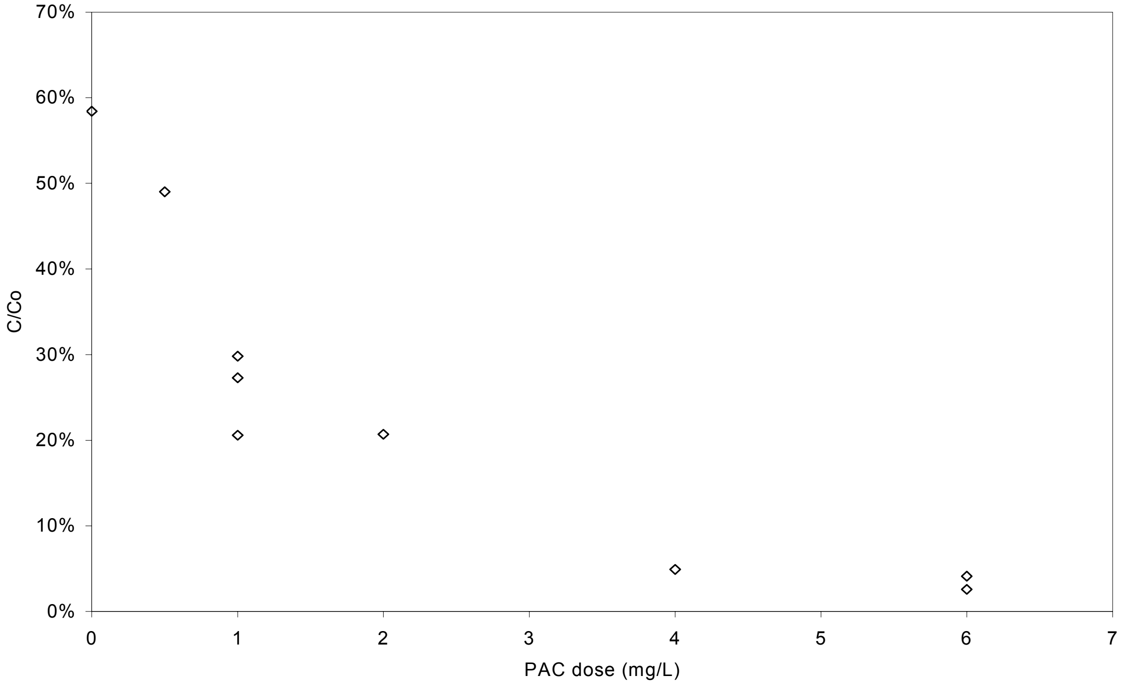

Figure C.1 Percent of BPA remaining versus PAC dose

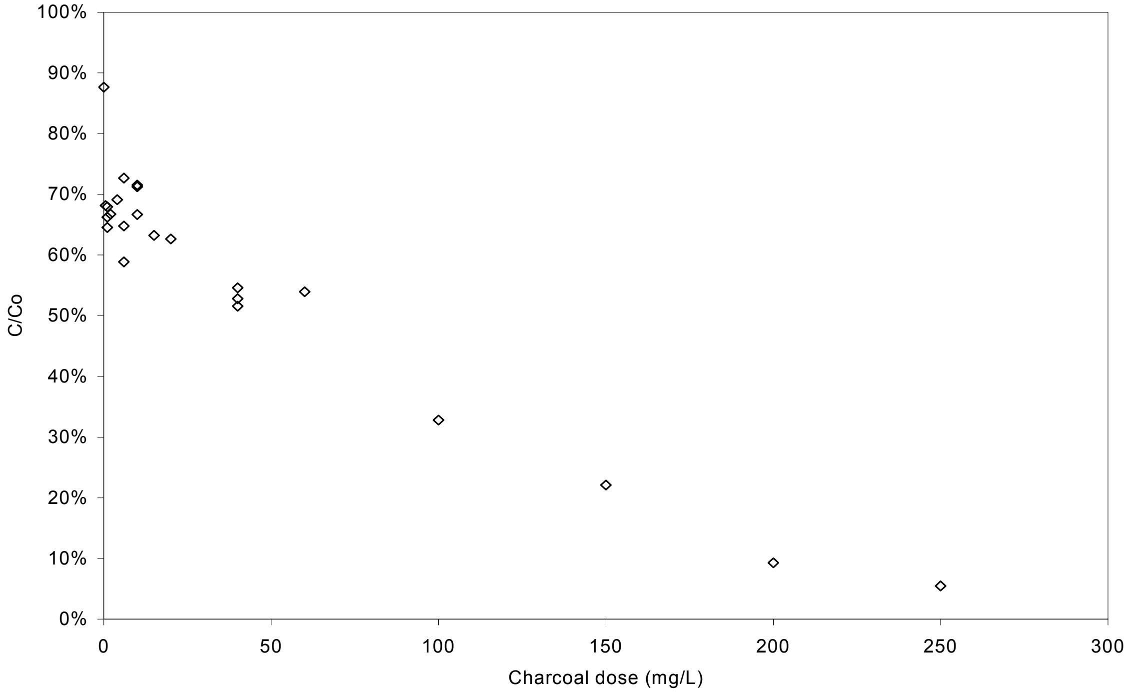

Figure C.2 Percent of BPA remaining versus charcoal dose