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THE EFFECTIVENESS OF BEST MANAGEMENT PRACTICES IMPLEMENTED AT LAKE GREGORY, CRESTLINE, CA

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THE EFFECTIVENESS OF BEST MANAGEMENT PRACTICES IMPLEMENTED AT LAKE GREGORY, CRESTLINE, CA

A Thesis

Presented to the

Faculty of

California State University,

San Bernardino

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

in

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by

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ABSTRACT

Water quality deterioration of recreational surface waters from excess pollution inputs is a significant concern for the health of the public and aquatic ecosystem. This study will examine the effectiveness of Best Management Practices (BMPs) in reducing pollution inputs into Lake Gregory, Crestline, CA. The effectiveness of the BMPs was examined by testing for water quality parameters including dissolved oxygen, temperature, pH, conductivity, turbidity, nitrate (NO_3-), ammonium (NH_4+), total coliform, and Escherichia coli on weekly to bi-weekly basis for a period of 1 year. A statistical analysis involving descriptive statistics, Pearson's correlation, the Efficient Ratios (ER) and Percent Removal Rates (PRR) equation, and comparisons to water quality criteria/objectives further assisted in examining the water quality of the lake. The findings of this study can be used by recreational managers in determining what BMPs to implement based on what pollutants were best controlled for by the BMPs specified in this study. On a more local level, the findings may assist recreational managers at Lake Gregory in determining if the maintenance of the BMPs is adequate for the BMPs to meet their design purpose of improving the water quality for recreational users. Recreational users can refer to the results of this study in determining if the lake is safe for recreational uses based on the number of exceedances of water quality criteria/objectives put in place to protect the public health.

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CHAPTER ONE

INTRODUCTION

Literature Review

The degradation of surface water quality in water resources used for recreation is a serious concern for both the public health and aquatic life. Anthropogenic sources such as changes in land use from natural landscapes to urbanized areas are known to be primary contributors to the water quality deterioration of surface water bodies (Paerl et al., 2016; Tong et al., 2002; Meyer et al., 2005). As areas become more urbanized, the percentage of land covered by impervious surfaces increases. Impervious surfaces (e.g., paved streets, parking lots, rooftops) are any surface that prevents the infiltration of stormwater into the subsurface. Studies have shown that water quality degradation of streams can occur at 10-20 percent impervious surface coverage and the impacts become nearly irreversible at 30 percent impervious surface coverage (Arnold & Gibbons, 1996; Schueler, 1994). Water quality degradation occurs at such low percentages of impervious surface coverage due to the fact that increases in impervious surface coverage contributes to increases in the volume and velocity of stormwater runoff (Arnold & Gibbons, 1996; Dunne & Leopold, 1978; Jacobson, 2011). Stormwater is a leading cause of water quality deterioration due to its ability to mobilize and transport pollutants (e.g., nutrients, sediments, microorganisms) into waterways (Booth & Rhett, 1997; Hathaway et al, 2012; Paerl, 2017; Phillips et al., 2018; Russel et al., 2019). As a result, impervious surfaces have been identified at various spatial contexts as a primary factor contributing to impaired surface water resources (Brabec et al., 2002).

The types of pollutants present in stormwater are dependent upon what land uses and impervious surfaces the stormwater flows across, in addition to the amount of stormwater discharge. Common pollutants in urban landscapes with varying land uses include nutrients, chemicals, bacteria, and sediments, deriving from fertilizers, pesticides, yard waste, pet feces, failing septic systems, and atmospheric deposition (Cahoon et al., 2006; Carpenter et al., 1998; Mallin et al., 2009; Pitt et al., 1995; USEPA, 2016). In a study by Yazdi et al., (2021) that examined multiple land uses designations, high concentrations of Nitrate (NO_3-) and Total Phosphorus (TP) were found at low density residential areas, high concentrations of Total Nitrogen (TN) were associated with transportation land use destinations, and open space areas (parks) had elevated levels of Total Suspended Solids (TSS). Relatively high concentrations of Total Phosphorus (TP) and Phosphorus (P) were also found in urban areas with high percentages of tree cover and forested areas (Brett et al., 2005; Janke et al., 2017). The addition of nutrients (i.e., NO_3-, NH_4+ TP, P, TN, etc.) in surface water bodies is important to monitor since it can contribute eutrophication, a process in which excess nutrients contributes to an increase in plant and algae matter that block sunlight and through decomposition deplete Dissolved Oxygen (DO) levels vital to the health of aquatic ecosystems (Rabalais et al., 2009).

Urban areas are efficient in the transport of pollutants to waterways since they are specifically engineered to efficiently move the largest volume of surface runoff possible into receiving water bodies in order to prevent flooding (Ellis & Marsalek, 1996; Simperler et al., 2020). Most impervious surfaces in urban areas are designed to direct stormwater runoff through storm drains/pipes which serve as conduits transporting stormwater and its contents directly to surface water bodies such as rivers, lakes, streams, or the ocean (SWRCB, 2021; USEPA, 2019). Through this system, virtually any pollutant on urban streets, rooftops, parking lots, etc., that can be transported by stormwater, will end up in a local surface water body, without any treatment, shortly after a precipitation event.

In more rural and suburban landscapes pollution inputs of surface water bodies may originate from small point sources upstream such as septic tanks, where pollution may be transferred by groundwater flow via seepage of septic effluent. Withers et al., (2011) observed high levels of nutrients including soluble N, P, Ammonium-N (NH_4N) and nitrite-N (NO_2N) originating from septic tank effluent discharging from a pipe directly into a headwater stream, contributing to eutrophication downstream. A study by Sowah et al., (2014) that analyzed undeveloped land uses, low density suburban areas, and high-density suburban areas covering multiple watersheds in Georgia, revealed that septic system density and median distance of septic systems to streams were strongly correlated with elevated counts of fecal contamination indicators including Escherichia coli (E. coli) and enterococci. This can lead to negative downstream effects, as pollutants originating from rural and undeveloped land uses (e.g. forests) can adhere to sediments and be further transported to water bodies down gradient (Anderson et al., 2005; Davies et al., 1995; Pachepsky & Shelton, 2011).

One area of particular interest from a public health perspective is the water quality of recreational waters used for various uses such as swimming, fishing, boating, etc. It was reported in national USEPA assessments from 2016, that over 293,650 acres of lakes and reservoirs that support contact recreation were listed as impaired (USEPA, 2016). Water quality impairments occur when state and/or federal standards/criteria are exceeded for one or more water quality parameters. With the input of pollution from stormwater contributing to water quality impairments of recreational waterbodies that support contact recreation, there are various health risks for public users. These health risks include irritations of the skin, eyes, and ears, and gastrointestinal (GI) illnesses that have varying symptoms (e.g., diarrhea, nausea, vomiting, abdominal discomfort, etc.) (CDC, 2020; USEPA, 2012). In the study by Perkins and Trimmer (2017), it was noted that most recreational water illnesses tend to be underreported because recreational users often become exposed at private properties or under supervised venues and tend to disperse before the illness occurs. Through the use of CDC data, it was also reported in the study that most recreational water illnesses occur in summer months (June-August), when water based recreational use is at its highest.

From the period 2000–2014, public health officials from 35 states and Guam, reported 140 untreated recreational water (e.g., lakes, rivers, coastal waters) associated outbreaks that resulted in 4,958 illnesses and two deaths (Graciaa et al., 2018). The Center for Disease Control (CDC) reported that of the 140 outbreaks, 80 were caused by enteric pathogens including norovirus and E. coli with fecal contamination being the likely mode of transmission. Recreational waters may become exposed to fecal contamination through various sources including human and animal waste transmitted by stormwater runoff, seepage of septic systems, and wastewater discharges (Cahoon et al., 2006; Glassmeyer et al., 2005; Gitter et al., 2020; Meyer et al., 2005). While rural areas are known to contribute to fecal contamination, there has also been an overall positive correlation with increases in urban development and contamination of fecal bacteria in freshwater streams (Young and Thackston, 1999). In a study by Cahoon et al., 2006, faulty septic tanks were the main source of fecal contamination to coastal waters in North Carolina, resulting in shell-fishing closures. A study by Meyer et al., 2005, linked the highest levels of fecal contamination in a recreational lake in Iowa directly to both human and animal sources after storm events.

Another phenomenon that greatly affects the water quality of recreational waters on a national scale, is the formation of harmful algal blooms (HABs) (e.g., cyanobacteria/blue-green algae). HABs are likely to occur in both marine and freshwaters when there is an excess of nutrients input (e.g., Nitrogen and

Phosphorus), increases in surface water temperatures, and slow moving or stagnant water (Paerl & Scott, 2010; USEPA, 2019; Wells et al., 2015). In addition to water quality deterioration, HABs can have a multitude of negative effects such as; mass mortalities of fish; human illness and death from toxic seafood or from exposures to toxins; and closures of fisheries and beaches resulting in substantial economic losses (Anderson et al., 2002; Koreivienė et al., 2014; Carmichael & Boyer, 2016). A HAB event in 2015 caused Dungeness crab fishery openings to be delayed in Washington, Oregon, and California contributing to a decrease in revenue of $97.5 million (National Marine Fisheries Service, 2016; Ritzman et al., 2018). It is estimated that cyanobacteria HABs (cHABs), a type of HAB that commonly occurs in freshwater, results in losses of recreational water resources and waterfront real estate worth up to $2 billion annually in the United States alone (Dodds et al., 2009).

In relation to public health, cyanobacteria HABS are also known to cause skin, eye, nose, and throat irritations, gastrointestinal (GI) illnesses, and liver damages in humans through water contact, ingestion, or inhalation of cyanotoxins (CDC, 2017; Funari & Testai, 2008; Manganelli et al., 2012). Ecologically, HABs can harm aquatic life through ingestion of cyanotoxins, suffocation of algae and/or cyanobacteria biomass, and through the depletion of dissolved oxygen upon decomposition, which can result in mass destruction for aquatic habitats (Sayer et al., 2016; Scavia et al., 2014; Wang & Zhang, 2020). Large scale fish kills are an ecological phenomenon known to be associated with the presence of HABs (Landsberg et al., 2020). Due to climate change and increasing global temperatures, the frequency of HAB occurrences is expected to increase, highlighting the need for more intensive recreational management strategies (Moore et al., 2008; Nwankwegu et al., 2019; Pearl et al., 2017). Oxygen depleting cyanobacteria HABs capable of producing cyanotoxins are known to occur at the study area, Lake Gregory (Figure 1), where fishing and swimming advisories have been previously issued (CDWR, 2021; OEHHA, 2016).

The USEPA and individual states are responsible for the development of water quality objectives/criteria and standards of the nation's recreational waters, however direct management often is the responsibility of county or local resource managers. With the increasing number of public users at recreational waters, recreational resource managers are being challenged on their ability to adequately employ management strategies and monitor recreational waters (Clow et al., 2011; White, 2016). Studies have shown that in some cases, the frequency of water quality sampling of recreational waters is too low and that a high frequency of sampling would better protect the public health from recreational water illnesses (Laws, 2014). Additionally, there is a lack of recreational monitoring requirements and national regulatory water quality standards/criteria for recreational managers to follow and adhere too (USEPA, 2018). Despite being found in one-third of 1,098 assessed lakes by the USEPA in 2007, there are still no national regulatory water quality standards for cyanotoxins (Otten & Paerl, 2015; USEPA, 2009). Yet, there are a multitude of non-regulatory water quality standards/criteria for recreational managers to use as guidelines, including standards for cyanotoxins. Regulatory standards do exist for water quality parameters associated with fecal contamination (e.g., fecal coliforms and E. Coli). In order to adequately monitor recreational waters, recreational managers must follow both regulatory and non-regulatory water quality standards/criteria through the use of frequent sampling and the implementation of management strategies including stormwater Best Management Practices (BMPs).

Stormwater Best Management Practices (BMPs) are mitigation measures used to reduce pollutants in stormwater, control for erosion, and prevent water quality degradation of waterbodies from occurring. BMPs are diverse, can be either structural or non-structural (e.g. vegetation, educational materials), and are selected based on site specific conditions including land use, storm and runoff characteristics, soil, topography, geography, and vegetation (Gautum et al., 2010). The effectiveness of BMPs are dependent on the type of BMP chosen and its design purpose, as some BMPs are more efficient at removing pollutants than others, while other BMPs are better at preventing erosion and stormwater peak flows (Aceves & Musandji, 2013; Davis et al., 2011; Yu et al, 2013). Common BMPs used to reduce stormwater pollution include detention ponds, bioretention areas, rain barrels, filter strips and fibers, rip-rap (rock material), weirs, and vegetated swales. Extensive research has been done, showing the effectiveness of BMPs in meeting water quality objectives/criteria (Amatya & Gilliam, 2003; Gabriel & Bodensteiner, 2011; Mallin et al., 2002; Mallin et al., 2016; Thompson et al., 2018).

In a study by Mallin et al., (2016), BMPs including grassed swales, and rain gardens installed in municipal areas of Wrightsville Beach, North Carolina resulted in fecal coliform bacteria reductions of 57 percent, and Total Suspended Solids (TSS) reductions of 99 percent. Another study by Mallin et al., (2002), revealed the implementation of wet detention ponds resulted in significant pollutant reductions of nitrate, ammonium, total phosphorus, conductivity, and fecal coliform (Mallin et al., 2002). Thompson et al., (2018) discovered that the implementation of BMPs comprised of a mixture of sand, gravel, and woodchip fibers stabilized by rock weirs, resulted in the removal of 48 percent ammonium, 25 percent nitrate, 49 percent total nitrogen, and 73 percent suspended sediments directly downstream of BMPs. It was reported by Amatya & Gilliam, (2003) that weir structures installed at ditch outlets to reduce peak drainage rates during high flows resulted in 13 percent average annual flow reductions, 54 percent total sediment reductions, and 30 percent total phosphorus reductions. Gabriel & Bodensteiner, (2011) identified increased dissolved oxygen levels and lower turbidity (increased water clarity) associated with the installment of rip-rap BMPs in shallow Wisconsin lakes.

Study Purpose and Objectives

The purpose of this study is to analyze the effectiveness of the BMPs implemented at Lake Gregory in mitigating various pollution inputs. This will be achieved by using water quality data from samples taken from the lake consisting of multiple water quality parameters, including parameters associated with recreational water illnesses and the formation of HABs. To further understand the effectiveness of the BMPs, physicochemical relationships among water quality parameters will be assessed and seasonal trends (e.g., storm events producing stormwater) will be analyzed based on how water quality parameters are affected. Water quality parameters will also be compared to water quality criteria/standards in determining how often exceedances of the water quality criteria/standards occur. Additionally, the water quality parameters will be compared against different sites including an inlet BMP site, an outlet BMP site and a non-BMP site. The results of this study may be used by the public in determining whether or not the lake is safe for recreation on a regular basis. Recreational managers of Lake Gregory may also benefit from the results of this study, as it will provide insights into whether or not more mitigation efforts are needed at the lake to increase surface water quality, protecting the public health and aquatic life. Finally, the methods and results of this study may be referenced by future studies determining the effectiveness of BMPs at recreational lakes.

Lake Gregory

Lake Gregory is a small recreational lake located within the San Bernardino National Forest (SBNF) and San Bernardino Mountains near the town of Crestline, California (CA), 14 miles North of San Bernardino, CA. As an artificial lake, the purpose behind the creation of Lake Gregory was to increase real estate values and tourism for Crestline, two stable sources of the town's economy (DataUSA, 2021). The lake was created in the 1930's when a large area around the nearby headwater stream, Houston Creek, was excavated and a dam was constructed downstream to the Northeast. Since the lake is located in a mountainous region, it has a relatively high surface elevation of 4,550 feet above mean sea level (City of Crestline, 2005). The lake is predominantly surrounded by various land uses including resource conservation (forest), single family residential, and rural living areas, with a pocket of general commercial areas at the West portion (Crest Forest Community Plan, 2007). Average summer temperatures at Crestline range from the mid 70's to the low 80's °F, attracting high volumes of visitors from all around Southern California to visit the lake (Weather Underground, 2021). The total water surface area of Lake Gregory is nearly 84 square acres, used mainly for recreational purposes (ino, 2021).

Lake Gregory offers a variety of water-based recreational opportunities and amenities including designated fishing and swimming areas, a water park consisting of a water slide and several inflatable structures, and kayak, paddle board, and other non-motorized boat rentals. Fishing and swimming remain two of the most popular recreational activities at the lake, emphasizing the importance of proper lake management in preventing pollution inputs that may cause recreational water illnesses (RWI). Those who decide to swim in Lake Gregory may contract RWI through contact or ingestion of the water, likewise fishermen may contract RWI through ingestion of contaminated fish. The recurrence of HABs is an ongoing issue at the lake, with reports of HABs most recently occurring in 2019 and 2020, resulting in the issuance advisories, warning the public to be cautious of and/or avoid water contact recreation (CAWQ, 2020). HAB occurrence in Lake Gregory can be attributed to a variety of factors previously noted including climatic factors (i.e., increasing temperatures, precipitation) slow-moving water, and pollution inputs.

During the study period, Crestline received a high frequency of precipitation in the form of rain and snow, primarily occurring between the end of fall and the beginning of spring. The precipitation serves as an efficient transport mechanism of pollutants into the lake once it has been converted to stormwater or snowmelt. The flows runoff either urban land uses such as commercial or residential areas composed of impervious surfaces (e.g., roofs, roads, and parking lots), or the natural forested landscape, before entering the lake through storm drains or inlet streams. Storm pipes near the West portion of the lake bring in flows that primarily traverse commercial areas and recreational areas, while an intermittent stream called Houston Creek near the Southeast portion directs flows into the lake from residential and forested landscapes through a culvert. With the aging infrastructure of Crestline, seepage of septic and sewer systems in combination with stormwater and snowmelt are potential inputs of fecal contamination into the lake. Lake Gregory also has a natural flow gradient towards the North end of the lake, where pollutants can accumulate, water quality deterioration can occur, and HABs can form in the summer and fall seasons, when conditions are favorable.

In an effort to prevent water quality deterioration, a series of BMPs were implemented at the Southeast portion of Lake Gregory in a nearly 900-foot channel down slope of the Houston Creek that directs flows in from the culvert. The BMPs consist of a cement/stream bed sediment slab, spillover weirs, rip-rap (rock material), and geotextiles. All of the BMPs were added as part of a sediment management plan completed in 2018 by the San Bernardino County Regional Parks Department (Regional Parks), where over 36,598 cubic yards of sediment and debris that accumulated within the channel were removed and the BMP channel was created (San Bernardino County, 2016). The intended design purpose of the BMPs is to enhance the water quality of the lake by filtering out and capturing pollution inputs in stormwater and flows from Houston Creek and preventing channel erosion that would result in additional sediments/debris into the lake.

Figure 1. Images taken from the shores of Lake Gregory in 2019 showing a Harmful Algal Bloom (HAB) and the advisory sign advising the public to be cautious of the HAB.

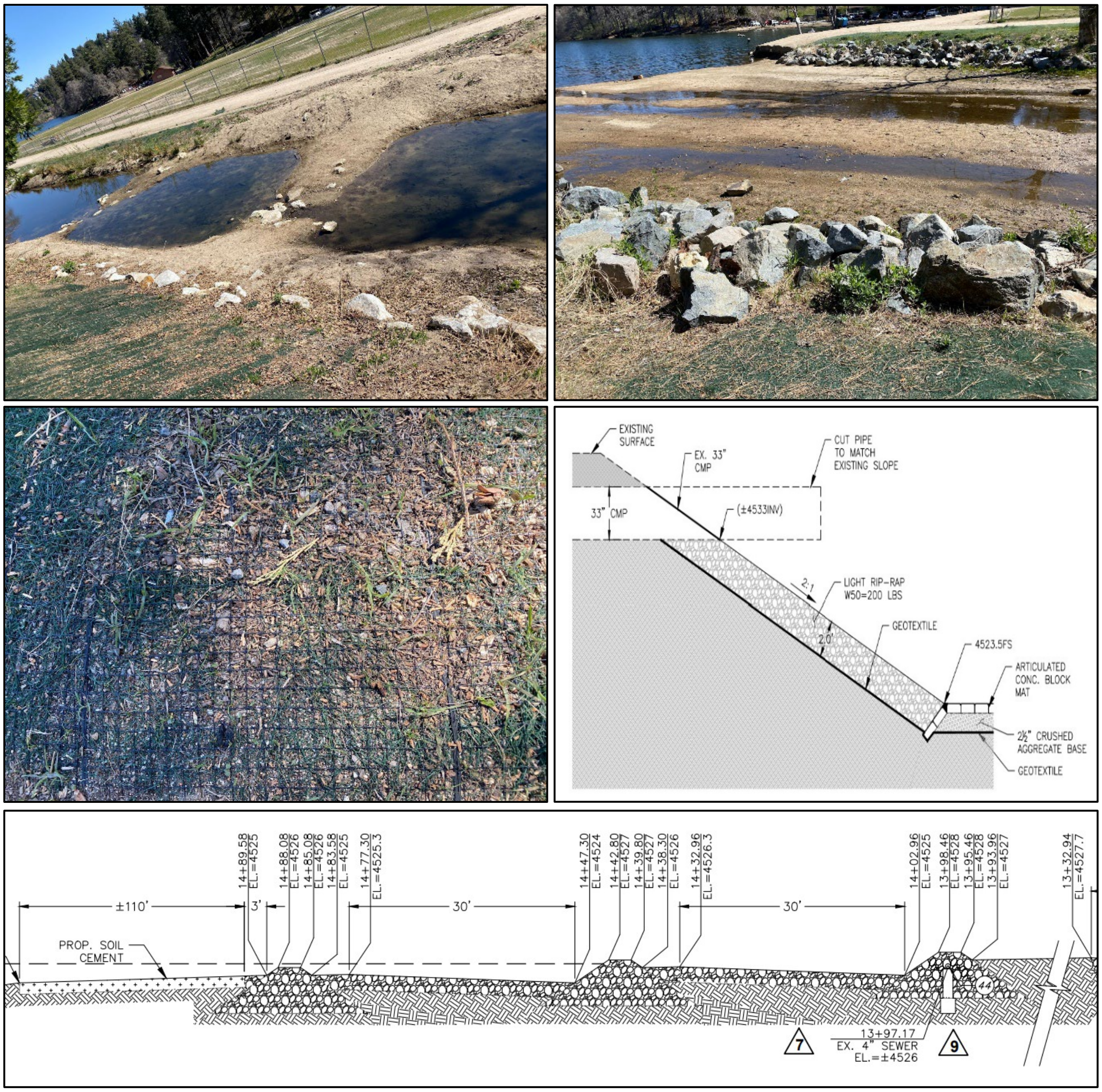

Figure 2: Images and construction designs of the Lake Gregory BMPs. The top left image of the figure depicts the spillover weirs including geotextiles and rip-rap on the sides of the channel. The top right image depicts the cement/stream sediment slab including geotextiles and rip-rap. The middle-left image shows a close-up view of geotextiles. The middle right image is the construction design of the geotextiles and rip-rap, and the bottom image is a construction design of the spillover weirs and cement/stream sediment slab (SB County, 2018).

CHAPTER TWO

STUDY SITES

There are a total of three sampling sites used in this study to determine the effectiveness of the BMPs implemented at Lake Gregory, including two sites in close proximity to the BMPs and one other non-BMP shoreline location. By selecting these sites, comparisons can be made of the water quality near the BMP sites versus the water quality at the non-BMP site located at opposite ends of the lake. Site 1 is a non-BMP site located on a beach in Lake Gregory Regional Park at the West portion of Lake Gregory, near a storm pipe that brings in stormwater flows from the park parking lot and surrounding communities. An inlet stream where stormwater discharges are directed from commercial areas, also flows into the lake to the West of the site. Site 1 is significant since it is where most of the water-contact recreation occurs, and it can provide a view of how stormwater directly impacts water quality without BMP implementation. Site 2 is located on a shoreline on the Southeast portion of the lake downstream of the channel containing the BMPs (after BMP implementation). Since Site 2 is located downstream the BMPs, it is expected to have the best overall water quality. Site 3 is located upstream of the channel before BMP implementation, near where Houston Creek flows through forest and rural residential areas then down to the culvert and into the lake. Site 2 and Site 3 will be used to analyze the water quality in the channel before BMP implementation and the water quality that is discharging into the lake from the channel after BMP implementation. All of the sampling sites and the general location of the BMPs can be viewed on Figure 2, which was created using ArcGIS Online and ArcGIS 10.6 software.

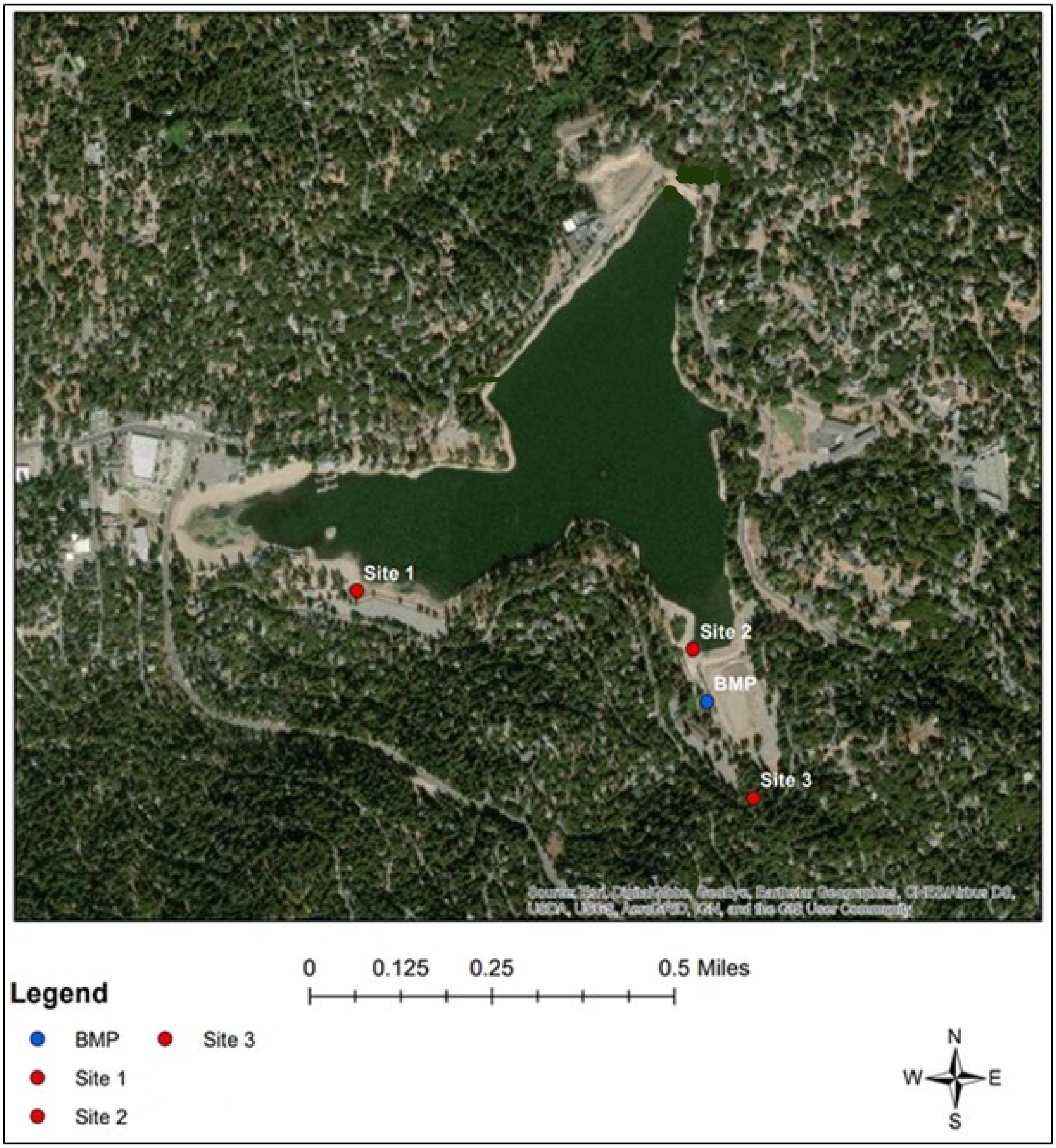

Figure 3. Lake Gregory sampling sites and BMP channel location.

CHAPTER THREE

METHODOLOGY

Water Quality Sampling Techniques

The water quality sampling for this thesis was conducted by myself, Dr. Jennifer Alford, and other students in her research team. To get an in-depth understanding of the water quality at the lake, the water quality parameters that were tested for included dissolved oxygen (DO) (mg/L), temperature (˚C), pH, conductivity (µm/cm), turbidity (NTU), nitrate (NO_3-) (mg/L), ammonium (NH_4+) (mg/L), total coliform (TC) (MPN/100mL), and Escherichia coli (E. coli) (MPN/100mL). Several studies such as Khatoon et al., (2013); Vega et al., (1998); Zheng et al., (2015), have all been successful in using the same and similar parameters.

Testing at each site required submerging one of six Vernier LabQuest 2 instrument probes into the lake water from a time period between 30 seconds to 1 minute, until the stabilization of readings for each parameter on the Vernier LabQuest 2 data collection device. Once stabilized, the readings were recorded for each of the six non-bacteria water quality parameters. Testing for bacteria (e.g., E. coli and TC) and turbidity required different methods that had to be conducted in a research lab. When testing for bacteria, water samples were taken from the lake at each site using sterilized 1 (L) brown opaque HDPE plastic bottles, placed on ice, transported to the lab, and poured into IDEXX 100mL bottles for further analysis. Following the standard USEPA IDEXX methods, Colilert reagents were mixed in the 100mL IDEXX bottles containing the water samples, which were then poured into a Quanti-Tray/2000 to be sealed and incubated at 4 (°C) for a period of 24 hours. After incubation the samples were analyzed for bacteria by recording the number of positive well readings for E. coli and TC and reporting the appropriate values in MPN from the corresponding IDEXX MPN table (IDEXX, 2003). The results reported for E. coli and TC in the most probable number per 100 milliliters (MPN/100mL) of water, are interchangeable with USEPA colony forming units (cfu). Testing for turbidity was completed in the lab, using the grab samples from the lake collected in the 1 (L) brown opaque HDPE plastic bottles and pouring the samples into small turbidity bottles. The bottles were then placed in a LabQuest turbidity sensor to get an accurate reading of turbidity in NTU.

When examining the effectiveness of the BMPs, water quality criteria/objectives will be used from the United States Environmental Protection Agency (USEPA) and the California State Water Resources Control Board (SWRCB), Lahontan Region 6 (USEPA, 2012; CASWB, 2015). These specified criteria/objectives will be used for determining whether each parameter is meeting or exceeding federal and state standards. By examining the effectiveness of the BMPs, for the selected water quality parameters, it can be determined what pollutants are best controlled for by the BMPs to meet specified water quality criteria/objectives, in addition to what pollutants are not adequately controlled for and need further mitigation efforts to improve water quality.

Statistical Analysis

By applying methods similar to Alford et al., (2016), Alford and Caporuscio (2020), and Varol et al., (2012), descriptive statistics including the mean, median, standard deviation, and variance were calculated for each parameter by using Microsoft Excel. Each parameter was tested for normality by using Shapiro-Wilks tests, in addition to analyzing kurtosis and skewness values in IBM SPSS version 27. As referenced in McNett et al., (2010) and Koch et al., (2014), the efficiency ratios (ER), and percent removal rates (PRR) for each applicable parameter were calculated by using the ER equation (ER= (inflow concentration - outflow concentration) / inflow concentration) and multiplying by 100 to get the percent removal rate. By using methods similar to Barakat et al. (2016), and Khatoon et al., (2013), SPSS will also be used to observe potential associations between parameters by generating a Pearson's correlation matrix at significance levels of $p<0.05$, $p<0.01$, and $p<0.001$. To understand seasonal trends and the effect of precipitation on each water quality parameter, time series line graphs were created in Microsoft Excel.

Table 1. Water quality criteria/objectives.

Water Quality Metric	Criteria/Objective	Source
Temperature	<25℃	SWRCB Objectives (Fact Sheet) (2004a)
Dissolved Oxygen (DO)	>4 mg/L	SWRCB, Lahontan Region Objectives (2015)
pH	6.5-8.5	SWRCB, Lahontan Region Objectives (2015)
Turbidity	<100 NTU	SWRCB Objectives (Fact Sheet) (2004b)
Conductivity	150-500 Range <336 µS/cm	USEPA Criteria for stream fisheries (2012a) SWRCB Objectives
Nitrate (NO_3-)	0.8-2.5 mg/L	SWRCB Grass Valley Creek Objectives (2015)
Ammonium (NH_4+)	0.02-0.4 mg/L	USEPA Aquatic Life Criteria (2013)
Total Coliform (TC)	1,000 cfu/100mL	SWRCB State Water Board Objectives (2004b)
Escherichia coli (E. coli)	<126 cfu/100mL	USEPA 2012 Recreational Water Quality Criteria (2012b)

Sources: SWRCB, 2004a, 2004b, 2015; USEPA, 2012a, 2012b, 2013

Timeline

The lake water quality at all three sites was tested for each of the water quality parameters weekly during the wet season (November 2018 to February 2019) and bi-weekly during the dry seasons (May 2018 to November 2018 and February 2019 to June 2019), with additional testing related to rain events for a period of 1 year. As several studies have shown, seasonal fluctuations can contribute to large variations in water quality, especially temperature and dissolved oxygen (Bello et al., 2017; Butcher & Covington 1995; He et al., 2011). By frequently testing for a period of 1 year, the changes in water quality can be seen from week to week and from season to season, in addition to changes caused from increasing and/or decreasing atmospheric temperatures, and precipitation events. The water quality data that has been collected will be used to complete the methods, results, and conclusion portions of this thesis.

CHAPTER FOUR

RESULTS

Descriptive Statistics and Exceedances

To get a better understanding of the water quality at each sampling site, the descriptive statistics and the count and percent of exceedances for each parameter are included in Table 2, Table 3, and Table 4. Figure 4 was designed to show graphically the percent of exceedances for each parameter at each site. As Table 1 represents the non-BMP control site (Site 1), its descriptive statistics and exceedances for each parameter provide an example of the baseline shoreline water quality and may be compared to Tables 2 and 3. Since Table 3 represents the BMP outlet site (Site 2) and Table 4 represents the BMP inlet site (Site 3), these tables in combination with Figure 4, provide insights into which specific parameters the BMPs are effective in controlling for and not controlling. Tables 3-4 and Figure 4 do so by displaying the mean and median values, in addition to the count and percent of exceedances for each parameter and the applicable water quality objectives/criteria.

Site 1 Water Quality Descriptive Statistics and Exceedances

The statistics and exceedance values for Site 1, the non-BMP control site for this study, are included in Table 2. Both nutrient parameters had mean values that exceeded water quality objectives/criteria for each corresponding parameter, NH_4+ (1.1 mg/L) and NO_3- (4.7 mg/L). NH_4+ and NO_3- also exhibited a low variability of 5.5 and 17.9, indicating the values were not widely dispersed from the mean. There were 18 individual NH_4+ samples or 45% of the total 40 NH_4+ samples that exceeded USEPA water quality criteria. For NO_3- there were 26 individual samples or 65% of the total 40 NO_3- samples exceeded SWRCB water quality objectives.

Conductivity had a mean value of 194 µS/cm and a median value of 185.6 µS/cm, which were both below the USEPA Criteria and SWRCB Objectives for Conductivity. However, there were 9 of 44 or 22.7% of the total conductivity samples that did exceed the USEPA Criteria and/or SWRCB Objectives. There was a high variance for conductivity (2961.7), which can be explained by the wide range exhibited by conductivity from 122.4 µS/cm to 444 µS/cm. pH both had a mean value (7.1) and median value (7.0) not exceeding the SWRCB Lahontan Region Objective for pH. Yet, there were 14 of 44 or 31.8% of the total pH samples that exceeded the SWRCB Lahontan Region Objective. The low variability for pH (0.7) indicated there was insignificant influence by outliers.

The two bacteria parameters, Total Coliform (TC) (1479.9 MPN) and E. coli (217.5 MPN), both had mean values exceeding the water quality criteria/objectives. E. coli and TC both had a high variance since the bacteria count was calculated using the Most Probable Number (MPN) approach which yields a counting range of 1–2,419 at a high confidence limit of 95% (IDEXX, 2020). There were 24 individual TC samples or 61.5% of the total 39 TC that exceeded SWRCB objectives. E. coli had 11 individual samples or 28.2% of the total 39 E. coli samples that exceeded USEPA criteria.

Table 2. Descriptive Statistics, Count, and Percent of Exceedances at Site 1

Site 1 Descriptive Statistics and Exceedances										
Parameter	**Mean**	**Med-ian**	**Min**	**Max**	**Stand. Dev.**	**Variance**	**Kurtosis**	**Skewness**	**Criteria**	**# & % Exceeding**
DO (mg/L) (N=43)	9.8	9.9	6.0	14.7	1.7	3.0	0.5	-0.1	>4 mg/L	0.0 (0%)
NH_4+ (mg/L) (N=40)	1.1	0.3	0.0	13.0	2.4	5.5	17.5	3.9	0.02-0.4 mg/L	18.0 (45.0%)
NO_3- (mg/L) (N=40)	4.7	3.7	0.2	23.5	4.2	17.9	9.0	2.4	0.8-2.5 mg/L	26.0 (65.0%)
Cond. (µS/cm) (N=44)	194.1	185.6	122.4	444.0	54.4	2961.7	9.3	2.3	150-500 Range or <336 µS/cm (Avg)	9.0 (22.7%)
Temp °C (N=44)	13.9	12.4	4.1	29.5	7.4	54.9	-0.5	0.7	<25°C	5.0 (11.4%)
pH (N=44)	7.1	7.0	5.6	9.3	0.8	0.7	0.6	1.0	6.5-8.5	14.0 (31.8%)
Turbidity (NTU) (N=42)	30.3	19.7	1.3	141.6	30.9	957.5	6.1	2.3	<100 NTU	2.0 (4.5%)
TC (MPN) (N=39)	1479.9	1553.1	76.7	2419.6	917.1	841116.6	-1.7	-0.2	1,000 cfu/ 100mL	24.0 (61.5%)
E. coli (MPN) (N=39)	217.5	28.1	1.0	2419.6	469.5	220392.9	13.0	3.4	<126 cfu/ 100mL	11.0 (28.2%)

Site 2 Water Quality Descriptive Statistics and Exceedances

The descriptive statistics, count, and percent of exceedances for each parameter at Site 2, the outlet shoreline site, are located in Table 3. For each nutrient parameter, NH_4+ (1.2 mg/L) and NO_3- (3.8 mg/L), the mean was above specified water quality criteria/objectives. NH_4+ had 19 individual samples or 47.5% of the total 40 NH_4+ samples that exceeded USEPA water quality criteria and NO_3- had 20 individual samples or 50% of the total 40 NO_3- samples that exceeded SWRCB water quality objectives. Both NH_4+ and NO_3- expressed a low variability of 5.9 and 10.4, indicating the mean was not strongly influenced by outliers.

Both the mean value (6.9) and median value (6.7) for pH, were below the SWRCB objective, however there were 14 individual samples or 31.8% of the total 44 pH samples that exceeded the SWRCB, Lahontan Region Objective. pH also had a low variability of 0.5 indicating limited influence by outliers. Conductivity had a mean value of 178.6 µS/cm and a median value of 174.2 µS/cm, which were both below the USEPA Criteria and SWRCB Objectives for conductivity. Yet, there were 9 individual conductivity samples or 20.5% of the total 44 conductivity samples that did exceed the USEPA Criteria and/or SWRCB Objectives. There was a high variance for conductivity (1854.4), which may be attributed to the wide range of conductivity.

TC had a mean value of 1042.5 MPN, which is above the SWRCB objective. Similarly, E. coli had a mean value of 126.9, which is above the specified USEPA criteria. The variances of both bacteria parameters remained high due to the MPN approach used to provide an estimate of the bacteria. There were 17 individual samples or 43.6% of the total 39 TC samples that exceeded SWRCB objectives and 8 individual samples or 20.5% of the total 39 E. coli samples that exceeded USEPA criteria.

Table 3. Descriptive Statistics, Count, and Percent of Exceedances at Site 2

Site 2 Descriptive Statistics and Exceedances										
Parameter	**Mean**	**Med-ian**	**Min**	**Max**	**Stand .Dev.**	**Variance**	**Kurtosis**	**Skewness**	**Criteria**	**# & % Exceeding**
DO (mg/L) (N=43)	8.9	8.7	5.0	12.5	1.8	3.1	-0.4	0.2	>4 mg/L	0 (0%)
NH_4+ (mg/L) (N=40)	1.2	0.4	0.0	11.9	2.4	5.9	10.5	3.2	0.02-0.4 mg/L	19.0 (47.5%)
NO_3- (mg/L) (N=40)	3.8	2.4	0.2	13.3	3.2	10.4	0.9	1.3	0.8-2.5 mg/L	20.0 (50.0%)
Cond. (µS/cm) (N=44)	178.6	174.2	75.2	252.0	43.1	1854.4	-0.6	-0.3	150-500 Range or <336 µS/cm (Avg)	9.0 (20.5%)
Temp °C (N=44)	13.3	11.3	4.4	28.6	6.8	45.7	-0.5	0.8	<25°C	4.0 (9.1%)
pH (N=44)	6.9	6.7	5.8	9.2	0.7	0.5	2.5	1.5	6.5-8.5	14.0 (31.8%)
Turbidity (NTU) (N=42)	30.0	21.5	2.0	182.7	37.2	1381.0	10.9	3.2	<100 NTU	2.0 (4.8%)
TC (MPN) (N=39)	1042.5	648.8	18.6	2419.6	918.2	843071	-1.5	0.5	1,000 cfu/100 mL	17.0 (43.6%)
E coli. (MPN) (N=39)	126.9	12.1	1.0	2419.6	394.3	155462.3	32.0	5.5	<126 cfu/100 mL	8.0 (20.5%)

Site 3 Water Quality Descriptive Statistics and Exceedances

The descriptive statistics and exceedance values for each parameter at Site 3, the inlet stream site, are included in Table 4. NH_4+ had a mean value (0.7 mg/L) that was above the USEPA water quality criteria. Similarly, NO_3- had a mean value (8.5 mg/L) above the SWRCB objective. Both nutrient parameters exhibited low variability (2.4 and 33.4) from the mean. There were 5 individual NH_4+ samples or 29.4% of the total 17 NH_4+ samples that exceeded the USEPA water quality criteria, while 15 individual NO_3- samples or 83.3% of the total 18 NO_3- samples exceeded the SWRCB objective.

Conductivity had a mean value (137.5 µS/cm) and median value (143.7 µS/cm) that was below the USEPA and/or SWRCB water quality criteria/objectives for the conductivity. However, there were 15 of individual or 75% of the total conductivity samples that exceeded the USEPA and/or SWRCB water quality criteria/objectives for conductivity. For pH, the mean and median values did not exceed the SWRCB Lahontan Region Objective and were the same value (6.9) resulting in a low variance of 0.3. Yet, there were 6 individual pH samples or 30% of the total pH samples that exceeded the SWRCB Lahontan Region Objective.

TC had a mean value of 606.2 MPN, that was below its SWRCB objective, while E. coli had a mean value of 184.7 MPN that was above its USEPA criteria. However, it is important to note, the median values for TC and E.

coli of 292.9 MPN and 30.2 MPN were far below the water quality criteria/objectives for each parameter due to the high variances and low sample count (N=20), allowing for outliers to influence the mean. Out of the 20 total samples for each bacteria parameter, there were 4 individual TC samples or 20% that exceeded the SWRCB objective and 2 individual E. coli samples or 10% that exceeded the USEPA criteria.

Table 4. Descriptive Statistics, Count, and Percent of Exceedances at Site 3

Site 3 Descriptive Statistics and Exceedances										
Parameter	**Mean**	**Med-ian**	**Min**	**Max**	**Stand Dev.**	**Variance**	**Kurtosis**	**Skewness**	**Criteria**	**# & % Exceeding**
DO (mg/L) (N=19)	10.3	9.8	7.1	14.1	1.7	2.9	0.1	0.3	>4 mg/L	0.0 (0%)
NH_4+ (mg/L) (N=17)	0.7	0.1	0.0	5.8	1.6	2.4	8.1	2.9	0.02-0.4 mg/L	5.0 (29.4%)
NO_3- (mg/L) (N=18)	8.5	9.1	0.4	18.3	5.8	33.4	-1.3	0.1	0.8-2.5 mg/L	15.0 (83.3%)
Cond. (µS/cm) (N=20)	137.5	143.7	100.3	169.5	22.5	504.7	-1.1	-0.3	150-500 Range or <336 µS/cm (Avg)	15.0 (75.0%)
Temp °C (N=20)	8.4	8.4	4.2	13.8	2.3	5.5	0.2	0.4	<25°C	0.0 (0%)
pH (N=20)	6.9	6.9	5.8	8.2	0.6	0.3	0.8	0.2	6.5-8.5	6.0 (30.0%)
Turbidity (NTU) (N=19)	15.5	8.8	0.5	93.1	23.4	546.3	6.9	2.6	<100 NTU	0.0 (0%)
TC (MPN) (N=20)	606.2	292.9	51.2	2419.6	718.5	516309.2	2.6	1.8	1,000 cfu/100 mL	4.0 (20.0%)
EC (MPN) (N=20)	184.7	30.2	4.1	2419.6	544.3	296215.6	17.0	4.1	<126 cfu/100 mL	2.0 (10.0%)

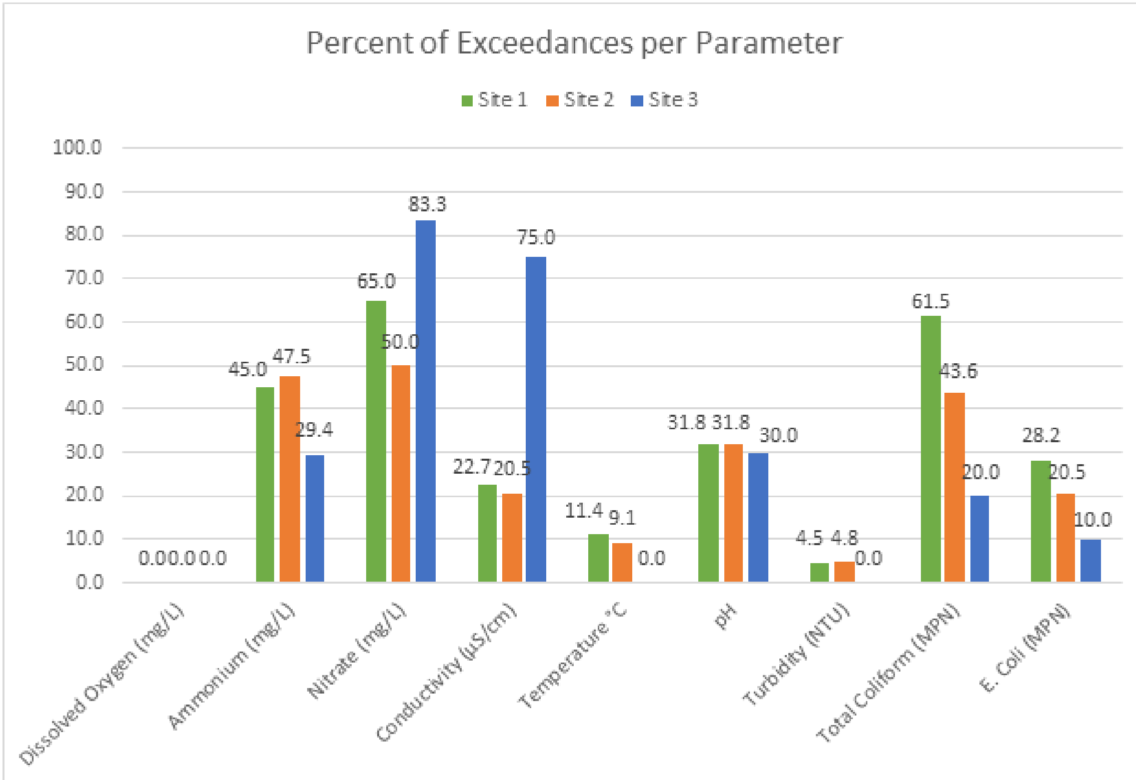

Figure 4. Percent of exceedances of the water quality criteria/objectives for each parameter at each sampling site.

Seasonal and Water Quality Parameter Trends

Precipitation

Across the entire study period from May 29, 2018, to June 10, 2019, the average precipitation was 0.99 centimeters (cm) based on precipitation data gathered during sampling, and 24 hours, 48 hours, and 72 hours prior to sampling. Most of the precipitation occurred in the winter season from the end of November 2018 to the middle of February 2019, which is considered the wet season for the study period. The precipitation event with the highest frequency of precipitation (14.66 cm) occurred 48 hours prior to the sampling event on January 1, 2019. Precipitation occurred in the form of both rain and snow, contributing to the flow of Houston Creek at Site 3 that flowed into the lake from the end of November 2018 to early June 2019. As shown in Figures 5-13, the water quality parameters were impacted after precipitation events, due to the ability of stormwater and snow melt to mobilize and transport pollutants across impervious surfaces into the lake. The dry season for the study period began in early May 2018, was followed by the winter season, began again at the end of February 2019, then continued past June 2019, with only two precipitation events exceeding 2 cm. During the dry season there was no precipitation recorded within 72 hours of any sampling event from March 18, 2019 to April 16, 2019. Due to the lack of precipitation and snowmelt in the dry season, the flow of

Houston Creek ceased after June 2019 and samples could no longer be obtained from Site 3.

Water Quality Parameter Trends

The water quality trends for each parameter can be viewed in Figures 5-13. Individually, the figures provide an overview of how each parameter responded to seasonal variations and precipitation events. The impacts of the flow of Houston Creek (November 2018 to June 2019) and the lack thereof (May 2018 to November 2018) can also be understood by reviewing the trendlines of Site 3 (inlet) and Site 2 (Outlet). When taken collectively, the water quality trends can be compared against other parameters to see direct and inverse relationships, that can be confirmed with a Pearson's correlation.

Dissolved Oxygen and Temperature. As shown in Figure 5, dissolved oxygen (DO) (mg/L) was generally lower in the dry season than in the wet season. Increases in DO also appeared to be related to precipitation events, as each site experienced increases in DO shortly after precipitation events that produced more than 5 cm of precipitation. Higher DO values were consistent at Site 3, which may be traced back to the flow of Houston Creek as opposed to the stagnant lake water at the shoreline sites. Figure 6 shows water temperature (°C), which as expected is largely influenced by seasonality. In the dry season water temperatures were consistently higher than in the wet season. Site 3 also had the lowest water temperature due to the flow of moving water. When comparing Figure 5 and Figure 6, a slight inverse relationship can be noticed between DO and water temperature, since DO values are generally lower in the dry season and higher in the wet season, while water temperature experiences higher values in dry season and lower in the wet season.

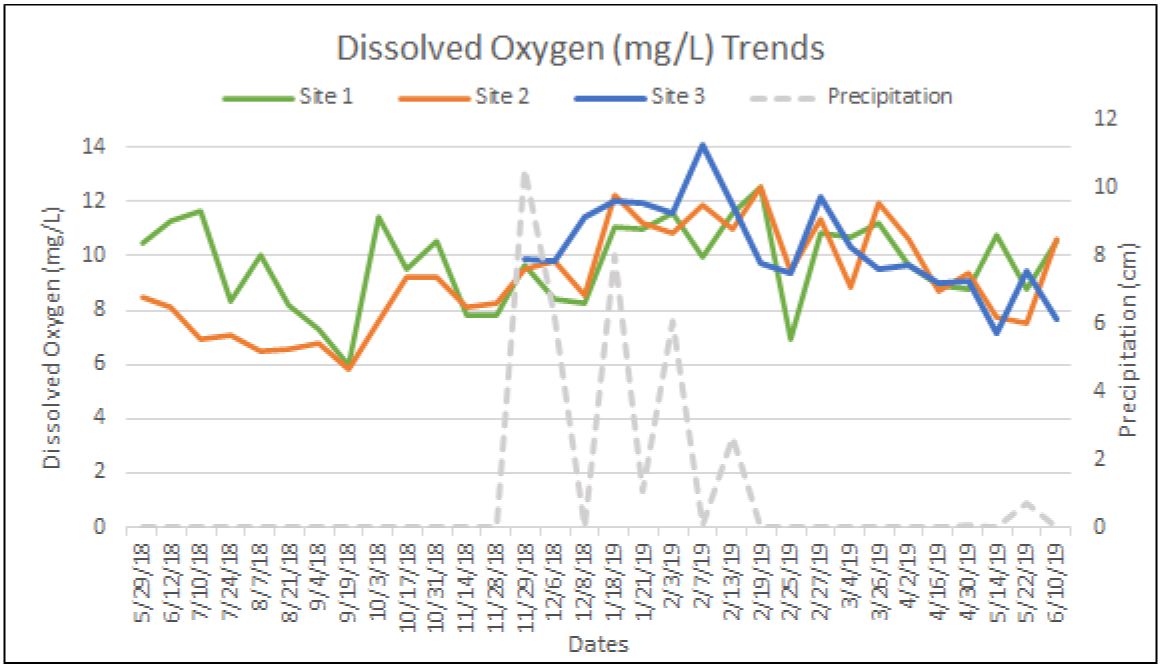

Figure 5. Dissolved Oxygen (mg/L) and precipitation (cm) trends for all sites

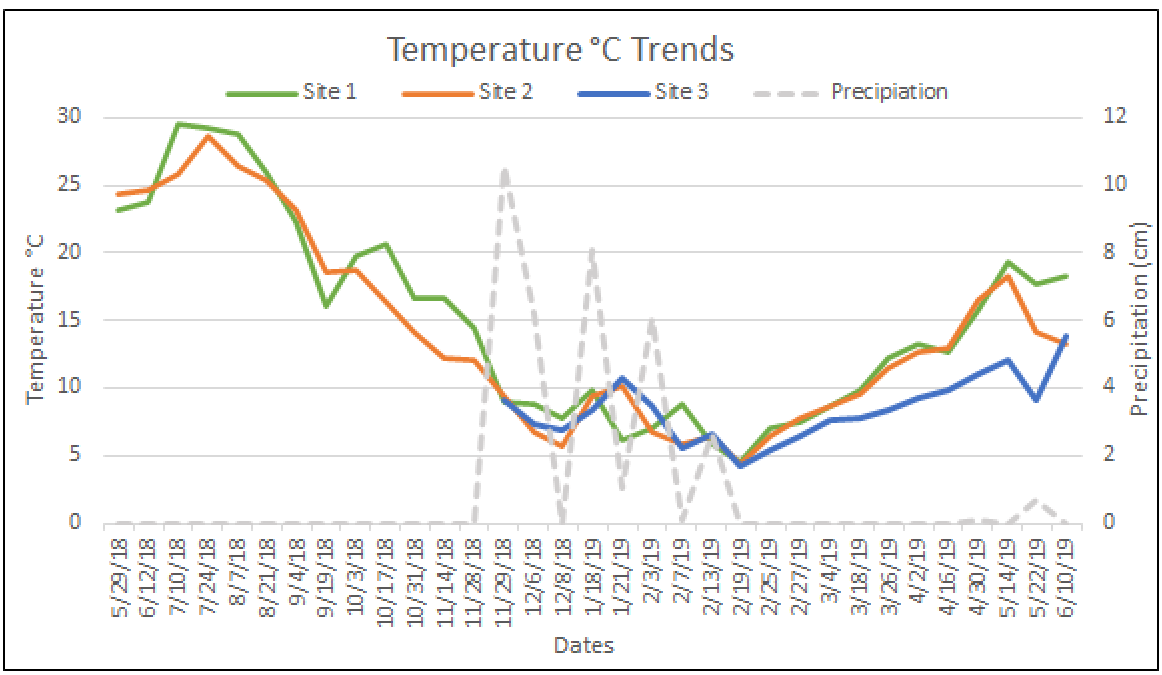

Figure 6. Temperature (°C) and precipitation (cm) trends for all sites

Conductivity and pH.. As shown in Figure 7, conductivity values experienced a slight decreasing trend overtime and were not significantly affected by precipitation. Conductivity (μm/cm) values for Site 1 and Site 2 were the highest during the early dry season from late May 2018 to late November 2018. The conductivity value of 444 (μm/cm) recorded on July 10, 2018, was the only conductivity value across the study period to exceed the upper limit water quality criteria/objective for conductivity of 336 (μm/cm). Figure 8 illustrated the pH values, which ranged from 5.8 to 8.2. The pH values did not have any general trends related to precipitation or seasonality, with the exception that some of the highest pH values happened to occur in the dry season.

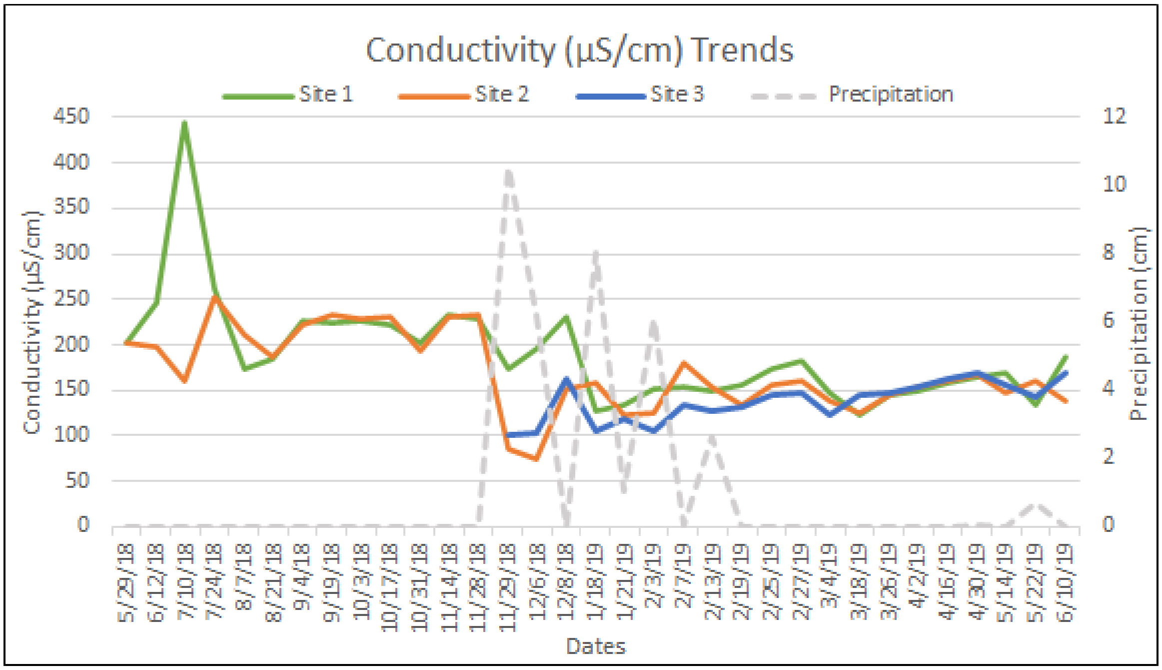

Figure 7. Conductivity (μm/cm) and precipitation (cm) trends for all sites

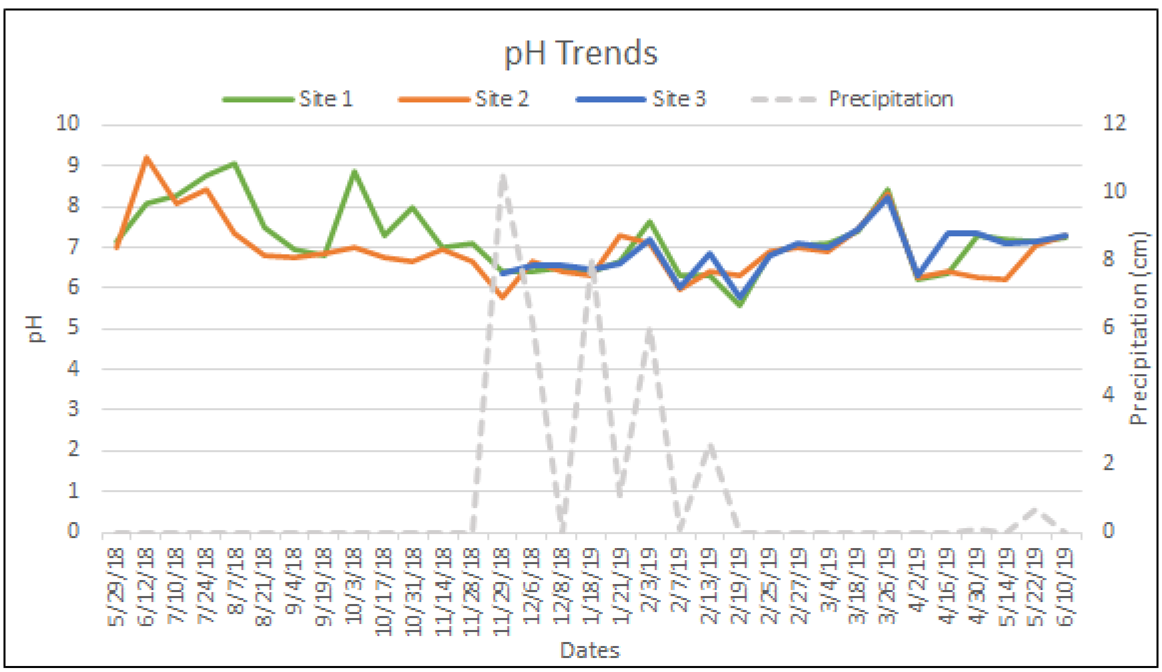

Figure 8. pH and precipitation (cm) trends for all sites

Ammonium and Nitrate. Ammonium (NH_4+) (mg/L) trends can be viewed in Figure 9, which shows that NH_4+ values were influenced by seasonality. Specifically, the NH_4+ values were generally lower in the dry season and consistently higher in the wet season. Sharp increases in NH_4+ can also be seen shortly after the precipitation events on November 29, 2018 and December 6, 2018. NH_4+ values at Site 3 were consistently lower than at Site 2. Figure 10, shows Nitrate (NO_3-) (mg/L) values did not follow a consistent trend from dry season to wet season. Rather, NO_3- values were lower in the early dry season beginning in May 2018, began to increase in the wet season beginning November 2018, and continued to fluctuate until the end of the late dry season in June 2019. NO_3- trends show that NO_3- was impacted by precipitation, as multiple spikes of NO_3- at each site occurred shortly after precipitation events. When Houston Creek was flowing, Site 3 had consistently had the highest NO_3- values.

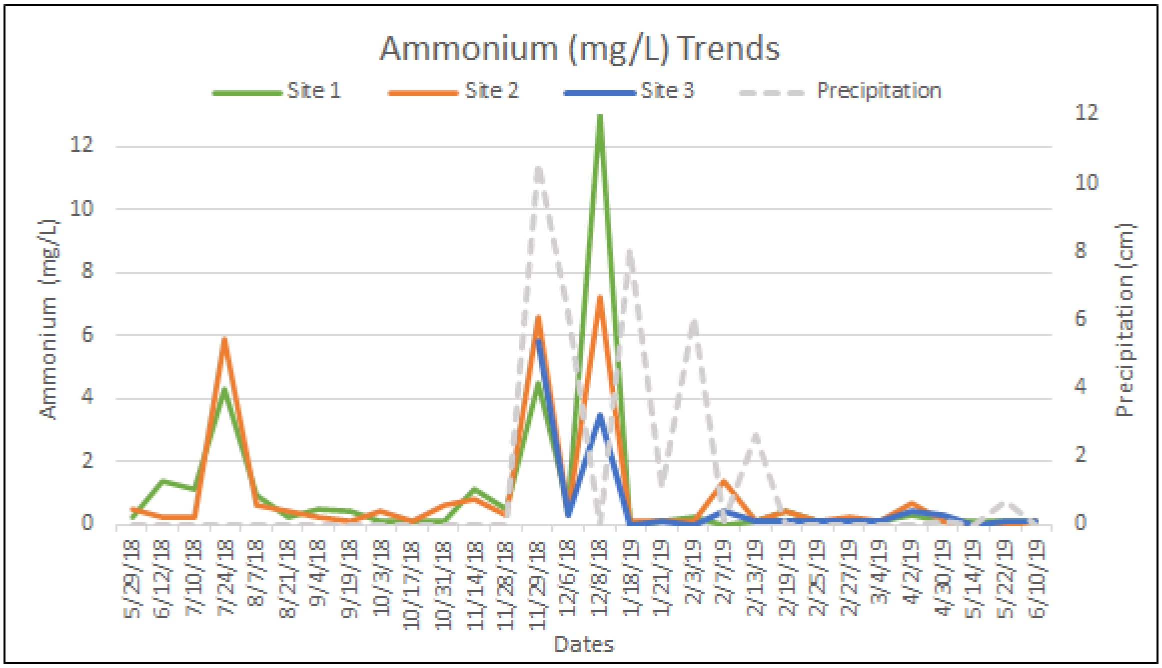

Figure 9. Ammonium (mg/L) and precipitation (cm) trends for all sites

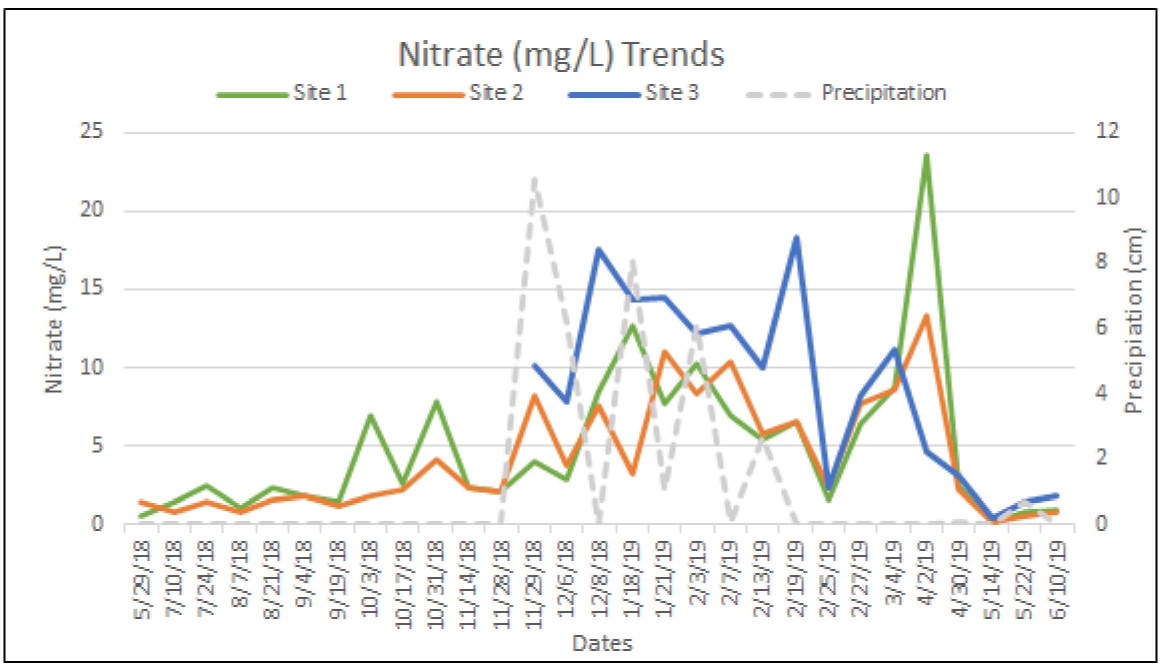

Figure 10. Nitrate (mg/L) and precipitation (cm) trends for all sites

Total Coliform, E. coli, and Turbidity. The trends of Total Coliform (TC) (MPN) can be viewed in Figure 11. TC values were highly variable throughout the wet and dry seasons with a series of spikes in the TC results (≥2419.6 MPN) and a series of low values (≤500 MPN) occurring at each site during each season. However, TC values did correspond with precipitation events since multiple spikes in TC at each site occurred shortly after precipitation events. As shown in Figure 12, E. coli (MPN) values were generally low (≤500 MPN) in the early and late dry seasons and were high in the wet season. E. coli values were also affected by precipitation events, as E. coli values increased shortly after precipitation events occurred. Spikes in E. coli values (≥2419.6 MPN) for each site occurred shortly after precipitation events. Turbidity (NTU) values can be seen in Figure 13, which shows fluctuating values both in the wet and dry season at site. However, the highest values for turbidity occurred after precipitation events. This trend may be attributed to the washout of sediments into the lake and inlet streams shortly after precipitation events. A correlation between turbidity and/or TC or E. coli may exist as each parameter experienced sharp increases in concentration shortly after precipitation events.

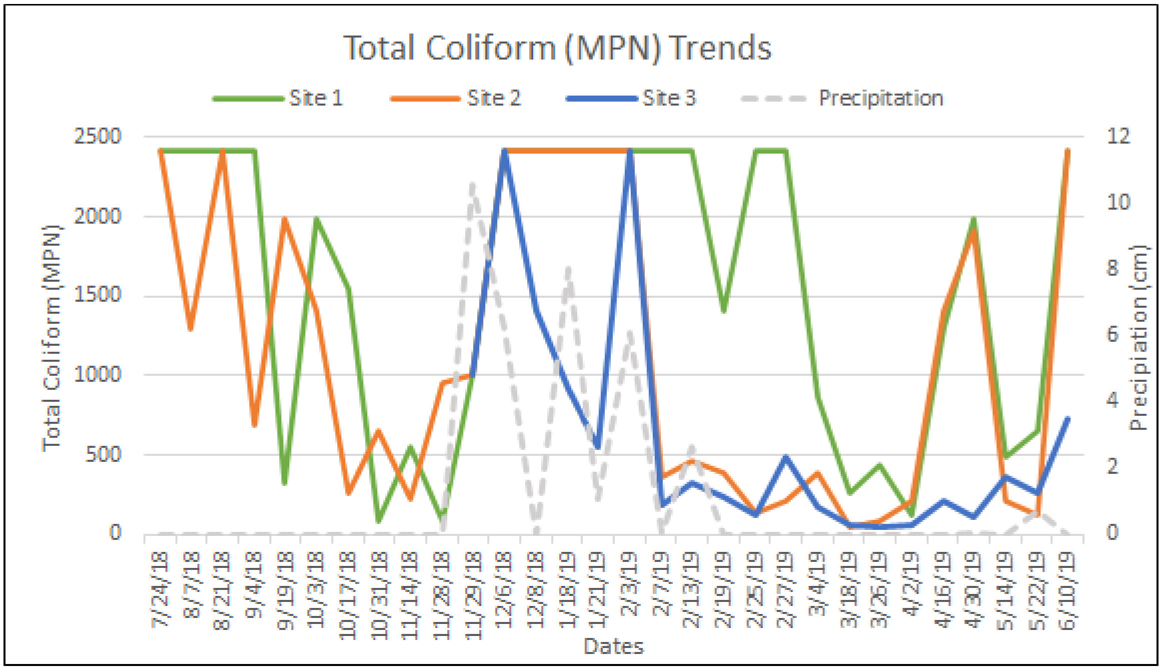

Figure 11. Total coliform (MPN) and precipitation (cm) trends for all sites

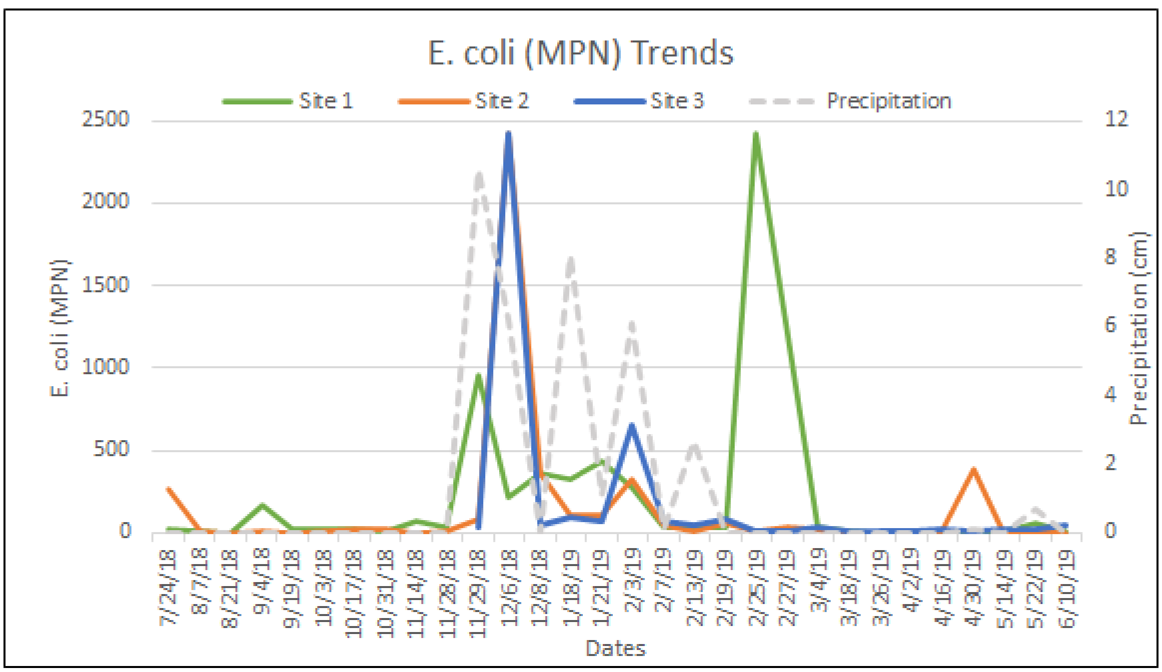

Figure 12. E. coli (MPN) and precipitation (cm) trends for all sites

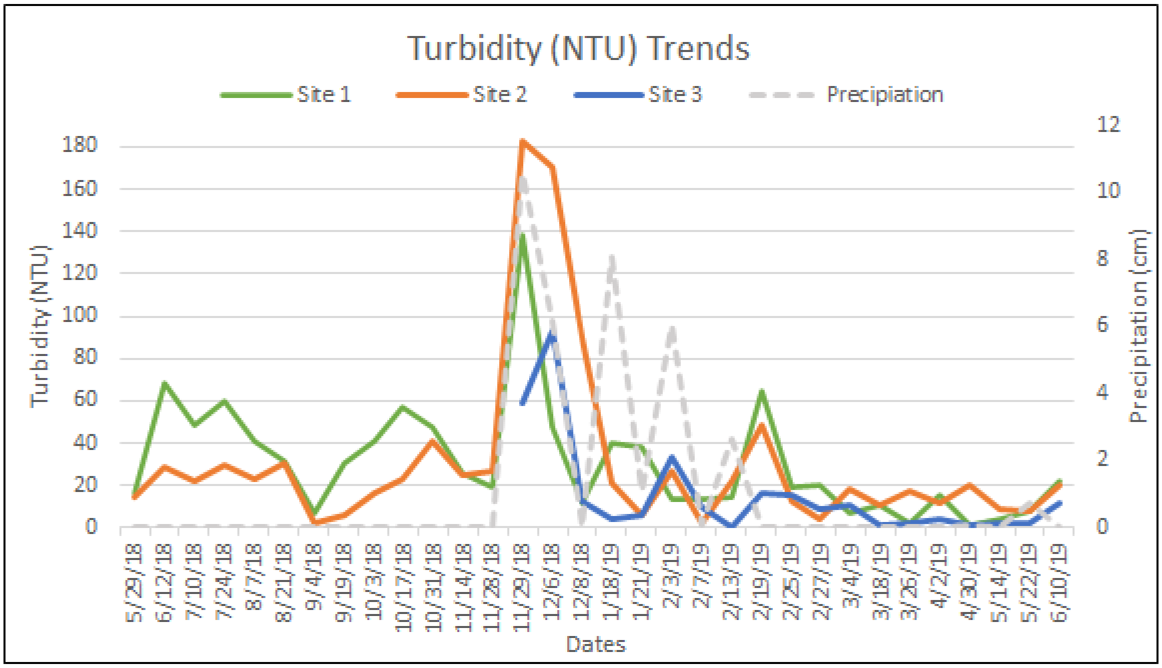

Figure 13. Turbidity (NTU) and precipitation (cm) trends for all sites

Pearson's Correlation Coefficient

Site 1 Pearson's Correlations between each Parameter

The Pearson's Correlation Coefficient used to determine positive or negative relationships between each parameter at Site 1 is included in Table 5. The table shows that conductivity and NH_4+ had a statistically significant relationship ($p<0.01$) and were positively correlated ($r=0.40$), implying that as conductivity levels increased, the concentration of NH_4+ also increased. Temperature had a statistically significant relationship with pH ($p<0.001$) and a positive correlation ($r=0.68$), suggesting that a rise in surface water temperature resulted in a rise in pH levels. Temperature also had a statistically significant relationship ($p<0.001$) and a negative correlation with NO_3- ($r=-0.54$) and E. coli ($r=-0.45$), meaning that as the surface water temperature increased, NO_3- and E. coli concentrations decreased. E. coli had a statistically significant ($p<0.05$) and a positive relationship with turbidity ($r=0.38$) and TC ($r=0.35$), suggesting that high E. coli concentrations were consistent with turbid waters and high TC concentrations.

Table 5. Pearson's Correlation Matrix for Water Quality Parameters at Site 1

Site 1 Pearson's Correlation Matrix									
	DO (mg/L)	*NH4+ (mg/L)*	*NO3- (mg/L)*	*Cond. (µS/cm)*	*Temp. °C*	*pH*	*Turbidity (NTU)*	*TC (MPN)*	*EC (MPN)*
DO (mg/L)	1								
NH_4^+ (mg/L)	-0.18	1							
NO_3^- (mg/L)	0.29	0.14	1						
Cond. (µS/cm)	-0.10	**0.40****	-0.17	1					
Temp.°C	-0.01	-0.10	**-0.54*****	0.34	1				
pH	0.23	-0.07	-0.31	0.31	**0.68*****	1			
Turbidity (NTU)	0.00	0.33	0.11	0.31	0.05	0.08	1		
TC (MPN)	-0.02	-0.09	-0.07	-0.07	-0.01	0.03	0.11	1	
E. coli (MPN)	-0.20	0.14	0.23	0.22	**-0.45*****	-0.31	**0.38***	**0.35***	1

*** Correlation is significant at the 0.001 level (2-tailed).

** Correlation is significant at the 0.01 level (2-tailed).

* Correlation is significant at the 0.05 level (2-tailed).

Site 2 Pearson's Correlations between each Parameter

The Pearson’s Correlation Coefficient for each water quality parameter at Site 2 is included in Table 6. NO_3- had a statistically significant relationship with dissolved oxygen ($p<0.001$) and a positive correlation ($r=0.57$), indicating that high concentrations of NO_3- were associated with elevated dissolved oxygen levels. Dissolved oxygen also had a statistically significant relationship ($p<0.001$) and negative correlation with conductivity ($r=-0.52$) and temperature ($r=-0.54$), meaning that elevated dissolved oxygen levels were associated with lower conductivity levels and surface water temperatures. Temperature and pH had a statistically significant relationship ($p<0.001$) and a positive correlation ($r=0.54$), suggesting that a rise in surface water temperature was associated with a rise in pH levels. However, temperature also had a statistically significant relationship with NO_3- ($p<0.001$) and E. coli ($p<0.05$) and a negative correlation with both NO_3- ($r=-0.62$) and E. coli ($r=-0.35$), indicating that a decrease in surface water temperatures was associated with an increase in NO_3- levels and E. coli concentrations. E. coli had a statistically significant ($p<0.01$) and a positive relationship with turbidity ($r=0.44$) and TC ($r=0.42$), suggesting that high E. coli concentrations were consistent with turbid water and high TC concentrations.

Table 6. Pearson's Correlation Matrix for Water Quality Parameters at Site 2

Site 2 Pearson's Correlation Matrix									
	DO (mg/L)	*NH4+ (mg/L)*	*NO3- (mg/L)*	*Cond. (µS/cm)*	*Temp. °C*	*pH*	*Turbidity (NTU)*	*TC (MPN)*	*EC (MPN)*
DO (mg/L)	1								
NH_4+ (mg/L)	-0.24	1							
NO_3- (mg/L)	**0.57*** **	-0.03	1						
Cond. (µS/cm)	**-0.52*** **	0.13	-0.32	1					
Temp.°C	**-0.54*** **	-0.08	**-0.62*** **	**0.38** **	1				
pH	-0.13	-0.22	-0.31	0.13	**0.54*** **	1			
Turbidity (NTU)	-0.11	0.31	0.13	-0.20	-0.17	-0.10	1		
TC (MPN)	-0.14	-0.19	0.00	-0.08	0.30	-0.01	0.32	1	
E. coli (MPN)	0.14	0.12	**0.43** **	-0.26	**-0.35***	-0.17	**0.44** **	**0.42** **	1

*** Correlation is significant at the 0.001 level (2-tailed).

** Correlation is significant at the 0.01 level (2-tailed).

* Correlation is significant at the 0.05 level (2-tailed).

Efficiency Ratio and Percent Removal Rate

The Efficient Ratios (ER) and Percent Removal Rates (PRR) for each parameter with a measurable removal concentration can be seen in Table 7. DO had an ER of 0.01 mg/L and a PRR of 1.0 percent, indicating that DO levels had a 1 percent decrease from the inlet site (Site 3) to the outlet site (Site 2). NO_3- had an ER of 0.33 mg/L and a PRR of 32.6 percent, suggesting that there was a 32.6 percent decrease of the NO_3- concentration entering the lake from inlet to the outlet. NH_4+ had an ER of -0.54 mg/L and a PRR of -53.9 percent, meaning there was an increase of NH_4+ concentrations by over 50 percent at the outlet site. The ER for conductivity was -0.03 µm/cm and the PRR was -3.3 percent, suggesting a slight increase in conductivity from the inlet to the outlet. Turbidity had an ER of -1.38 NTU and a PRR of -138.5 percent, indicating that the water was over twice as turbid in the outlet than the inlet. The ER for TC was -0.77 MPN, while the ER for E. coli was -0.08 MPN. TC had a PRR of -77.3 percent and E. coli had a PRR of -7.9, indicating that TC concentrations increased over 77 percent and E. coli concentrations increased nearly 8 percent from the inlet to the outlet site in the lake.

Table 7. Efficiency Ratios and Percent Removal Rates

Efficiency Ratio (ER) and Percent Removal Rate							
	DO (mg/L)	NH_4+ (mg/L)	NO_3- (mg/L)	Cond. (µS/cm)	Turb. (NTU)	TC (MPN)	EC (MPN)
Efficiency Ratio (ER) ER= $\frac{\text{Inflow Concentration-Outflow Concentration}}{\text{Inflow Concentration}}$	0.01	-0.54	0.33	-0.03	-1.38	-0.77	-0.08
Percent Removal Rate (PRR) PRR=ER x 100%	1.0	-53.9	32.6	-3.3	-138.5	-77.3	-7.9

CHAPTER FIVE

DISCUSSION

Water Quality Parameter Source Contributions

Fluctuations in water quality parameter trends can be explained by various factors including seasonality, site specific conditions, and parameter specific sources. In order to understand the results of this study, explanations will be provided on the source contributions for each water quality parameter. Understanding what sources are contributing to exceedances of water quality criteria/objectives, will provide more insight if the implementation of the BMPs is effective and applicable for specific parameters.

Dissolved Oxygen and Temperature

Water temperatures at Lake Gregory displayed similar trends to the air temperature and seasonal fluctuations in the region. In the dry season (May 2018 to November 2018 and February 2019 to June 2019) when air temperatures in Crestline were the highest, water temperatures were also the highest. In the wet season (November 2018 to February 2019) when air temperatures were the lowest, water temperatures were also the lowest. Water temperatures were generally higher at Site 1 and Site 2 since the flow rate of the water in the lake is lower than of Site 3, which has a high flow rate from Houston Creek. In contrast to temperature, DO was generally lower in the dry season and generally higher in the wet season. This is due to the inverse relationship between DO and temperature observed in this study and many others (Wilhelm & Adrian, 2008; Zhang et al., 2015). The presence of HABs impacted DO levels at Site 1 and Site 2 after HABs were identified on August 21, 2018, and December 12, 2018, causing the DO to decline due to the eutrophic conditions created by HABs. Future studies are needed to determine the type of cyanotoxins associated with the HABs that occur at Lake Gregory.

Since DO and temperature are mainly influenced by seasonality and site-specific conditions (e.g., Lake Gregory water vs Houston Creek water), it can be concluded that the BMPs do not have a substantial impact on either parameter. Additionally, the trends and percent exceedances of each parameter indicate that neither parameter is considered a threat to the overall water quality of Lake Gregory. If the baseline conditions of DO and temperature remain the same to what was observed during this study, then no additional BMPs will need to be implemented for temperature and DO.

Conductivity and pH

Conductivity relates to the measure of water's ability to conduct an electrical current and is affected by the amount of inorganic dissolved solids present. As revealed in several studies, a potential source contributing to the increased conductivity values is the application of de-icing road salts near the lake (Kaushal et al. 2005; Corsi et al. 2010). As reflected in this study, water temperature also affects conductivity; increases in temperature corresponded with increases in conductivity (Hayashi, 2004; Sorensen & Glass, 1987). More so than conductivity, pH values were greatly affected by water temperature, as confirmed by Pearson's correlation between the parameters and the pH trends that show higher pH levels in warmer months. Another potential pollutant source contribution affecting both pH and conductivity is atmospheric deposition, although determination is beyond the scope of this study (Carling et al., 2017; Marty et al., 2021)

Since conductivity is influenced by de-icing road salts and Houston Creek flows through residential neighborhoods of Crestline that use de-icing salts, conductivity values may be affected by the BMP implementation. However, since the BMPs are specifically designed to control for the influx of sediments (e.g., total suspended solids) in the lake, the effectiveness in controlling for conductivity (associated with total dissolved solids) is expected to be low per the BMP design. This is because the total dissolved solids will remain in a dissolved state and will not be deposited as the water of Houston Creek flows through the BMPs into Lake Gregory. Similarly, pH is not expected to be substantially impacted by the implementation of the BMPs, since the BMPs were not designed to control for pH.

Ammonium and Nitrate

With the percent of NO_3- exceedances being over 50 percent at each site and the percent of NH_4+ exceedances being over 29 percent at each site, there are multiple suspected sources. The most probable source of NO_3- and NH_4+ is the aging septic and sewer systems in Crestline. Many studies show that septic and sewers can contribute to elevated levels of NO_3- and NH_4+ (Jung 2020; Tao et al., 2020; Withers et al., 2011). A sewer system near the West shoreline of the lake was identified in other studies of Lake Gregory as a contributor to the NO_3- exceedances due to potentially leaking pipes below the surface (Margullis et al., 2007). As Crestline is a vacation destination, there are many seasonal homes and rentals with aging septic systems of unknown condition and maintenance, that are designed to gradually leach effluent into the ground percolating into the soil and groundwater. This septic effluent is another likely source of NO_3- and NH_4+ contamination into the lake, since the groundwater has a downward flow towards the lake (Kochary et al., 2017; Schneeberger et al., 2015). Historical accounts from the Crest Forest Historical Society and research from prior CSUSB students suggest that the remains of an outhouse from 1938 may still be at the bottom of the Lake Gregory, which can contribute to increases in the NO_3- and NH_4+ concentrations (Margullis et al., 2007). Other potential sources that may contribute to the high NO_3- and NH_4+ values include animal and fish waste, and even dynamite used for the construction of the Lake Gregory dam that was left over when the Lake Gregory rapidly filled up. While future studies including groundwater studies of the area are needed to confirm the contributions of each potential pollutant source, the high NO_3- and NH_4+ values are themselves indicators of fecal contamination.

Since some of the residential neighborhoods Houston Creek flows through have septic systems that are susceptible to seepage of effluent overtime, the creek upgradient the BMPs may serve as a transport mechanism of the nutrients into the lake. This excess of nutrients entering the lake is one of the primary factors that can contribute to the formation of the HAB, which form at the North portion of Lake Gregory where the lake water naturally flows down slope (Anderson et al, 2002; Paerl et al., 2016). The BMPs implemented are expected to control for the excess nutrients entering the lake via Houston Creek and the effectiveness will be examined in the next section.

Total Coliform, E. coli, and Turbidity

Turbidity measures the relative clarity of the water and can be representative of the presence or absence of the total suspended materials or sediments in the water affecting water clarity. After rain events turbidity values in Lake Gregory increased due to the influx of sediments transported into the lake by stormwater through various inlets including the storm drain near Site 1 and increased storm flows in Houston Creek near Site 2. The strong positive correlation observed between turbidity and E. coli in this study is likely attributed to the fact that E. coli can adhere to the sediments entering the lake that are increasing the turbidity values (Anderson et al, 2005; Davies et al, 1995; Pachepsky & Shelton, 2011). TC can also adhere to sediments, and although there was not a strong correlation between TC and turbidity, the strong positive correlation between TC and E. coli, in combination with the high frequency of exceedances per parameter, provides more supporting evidence of fecal contamination. The potential nutrient pollutant sources of the aging septic and sewer systems, outhouse, human and animal waste would all contribute to the E. coli and TC exceedances (Gitter et al., 2020; Jung 2020; Meyer et al., 2005; Sowah et al., 2014). With fecal contamination of Lake Gregory evident, it is imperative that the public be informed about recreational water illnesses associated with fecal contamination.

Since turbidity, E. coli, and TC can all be associated with total suspended sediments, each parameter would be impacted by the BMPs as sediments from Houston Creek are transported through the BMP channel. Per the BMP design, the spillover weir system promotes the deposition of sediments by reducing stream flow, and the cement/stream bed sediment slab allows for natural sediment build-up to occur until the sediments can be excavated out. If maintained, the BMPs are expected to be effective in reducing the amount of fecal contamination from Houston Creek.

BMP Effectiveness

Based on the results of the study and source contributions per each parameter, the BMPs implemented at the Southeast portion of Lake Gregory consisting of a cement/stream bed sediment slab, spillover weirs, rip-rap (rock material), and geotextiles, are not effective in reducing additional pollution inputs of NH_4+, NO_3-, conductivity, turbidity, TC, and E. coli into the lake. A final determination can be made by comparing the water quality across the non-BMP site (Site 1), the BMP site downstream of the BMPs (Site 2), and the site upstream the BMPs (Site 3). It was hypothesized that the water quality at Site 2 would be of higher quality than that of Site 1 and Site 2. However, as the results showed, this was not the case.

It was expected that the overall water quality at Site 2 downstream of BMPs would be of much higher quality than Site 1, since Site 1 has no BMPs implemented to prevent water quality deterioration and polluted stormwater may flow freely through an input into the lake near the site. With the exception of NO_3- (15 percent difference) and TC (17.9 percent difference), there was within an 8 percent (+ or -) difference in regard to the percent of exceedances per parameter at Site 1 and Site 2. When comparing specific parameters to their corresponding water quality criteria/objectives, the results were similar for Site 1 and Site 2. At both sites, the mean values of NO_3-, NH_4+, TC, and E. coli were all above their corresponding water quality criteria/objectives. However, it should be noted that while Site 1 and Site 2 are both shoreline sites near storm drain inputs to the lake, the water quality at both sites is still representative of the background surrounding lake water quality. The background lake water quality could be influenced by site specific conditions including other pollutant source contributions not related to pollutant contributions from the nearby inlets (e.g., other inlets that drain into the lake from areas of high impervious service cover).

For the BMPs to be effective, it was also expected that there would be an improvement of the overall water quality from Site 3 upstream the BMPs (inlet) to Site 2 downstream the BMPs (outlet). Based on the Efficiency Ratio (ER) and Percent Removal Rate (PRR), the BMPs were not effective in reducing the pollutant concentrations of NH_4+, conductivity, turbidity, TC, and E. coli. It can be stated that the BMPs were not effective in reducing the pollutant concentrations of NH4+, conductivity, turbidity, TC, and E. coli since the ER and PRR for each parameter increased from Site 3 to Site 2 downstream the BMPs. Yet, a potential limitation to using the ER and PRR method is that it does not account for the background water quality of the surrounding lake water at the output site (Site 2), that as previously mentioned could have been influenced by other pollutant source contributions.

It should be noted that the effectiveness of the BMPs is not attributed to the specific design and design purpose of each BMP, but rather the lack of maintenance required to keep the BMPs functioning properly. Since the installation of the BMPs in 2018, the scheduled annual maintenance has not occurred on the spillover weirs and cement/stream bed sediment slab that continue to actively build up sediment (San Bernardino County, 2016). The spillover weirs and cement/stream bed sediment slab BMPs are designed to protect the water quality of the lake by reducing the volume and velocity of stormwater flow and thus increasing deposition allowing for sediments and pollutants to settle and be physically removed overtime. As shown in Osouli et al., 2017, the efficiency of BMPs designed for sediment build-up decreases overtime as sediment build-up continues to occur without removal. In order for the BMPs to continue to function properly, regular maintenance involving the removal of accumulated sediment is required for the BMPs at Lake Gregory.

Recommendations

In order to prevent the water quality deterioration of Lake Gregory from stormwater pollution entering the lake via Houston Creek and surrounding areas, the BMPs already implemented must be properly maintained. Annual maintenance of the BMPs were proposed in Initial Study/Mitigated Negative Declaration for the BMP installation project, however, based on visual observations during the weekly to bi-weekly testing during the study period, the maintenance has not occurred (County of San Bernardino, 2016). Based on several studies, the frequency of annual maintenance is appropriate for the BMPs designed for sediment retention and removal (spillover weirs and cement/stream sediment slab) (Erickson et al., 2010; Kang et al., 2008; Houle et al., 2013; Weiss et al., 2005). Geotextile and rip-rap replacement/maintenance for the bank erosion control should occur on an as needed basis dependent on the condition of the materials. Regular inspections of the BMPs will be needed in the future to determine when maintenance is required and to ensure they are operating at functional capacities.

In combination with scheduled maintenance of the BMPs, frequent sampling near Site 2 and Site 3 is also recommended to ensure the BMPs are working efficiently, once properly maintained. The public should also have access to view the sampling results when exceedances of water quality criteria/objectives occur, so that they can make the final determination of whether or not to recreate in the lake shortly after the exceedances occur. By informing the public, resource managers of Lake Gregory may be able to prevent recreational water illness contracted from the lake water.

In order to protect the public health and aquatic life, future studies of the fecal contamination at Lake Gregory are recommended. This future work may include specific source contributions of the fecal contamination. More comprehensive studies noting the density of septic systems in the area vs. sewer sources are needed to determine where the fecal contamination is originating. Geological and groundwater studies focusing on the downward groundwater flow gradient, as well as the Houston Creek flow gradient, toward the lake in relation to the location of septic and sewer systems and fecal bacteria concentrations, can confirm whether the septic systems or sewer systems are contributing to higher concentrations of fecal bacteria. Similar to Meyer et al., 2005 and Silkie and Nelson, 2009, fecal bacteria concentrations may be even further analyzed through the use of different testing methods to determine what percentage of fecal bacteria is from human origin and what percentage if any is from animal origin. Finally, additional inlets into Lake Gregory may be analyzed for BMP applicability to determine if additional BMPs will improve the water quality of the lake through pollutant reductions of various parameters.

CHAPTER SIX

CONCLUSION

This study examined the effectiveness of the BMPs implemented at Lake Gregory to prevent water quality degradation. The effectiveness of the BMPs was evaluated by reviewing water quality sampling results for DO, temperature, pH, conductivity, turbidity, NO_3^-, NH_4^+, TC, and E. coli, that were taken over a period of 1 year across three different sites around Lake Gregory. Each sampling point served a different purpose in the evaluation of the BMPs. The shoreline non-BMP site (Site 1) was meant to be used as a control site that represented the baseline shoreline water quality. The shoreline site after BMP implementation (Site 2), was the main site used to draw comparisons about the water quality at the site upstream the BMPs prior to implementation (Site 3), and the non-BMP site (Site 1). The sampling results were first evaluated on an individual basis in relation to exceedances of water quality criteria/objectives. The overall trends were then examined across different sites and in relation to seasonality. The results per parameter were then compared using a Pearson's correlation to understand relationships between parameters. Finally, the results from Site 2 and Site 3 were examined through Efficient Ratios (ER) and Percent Removal Rates (PRR) to review differences before and after BMP implementation. Based on the results, a final pollutant source assessment was completed to determine the applicability of the BMP implementation for each parameter.

Through a collective review of the results of this study, it was concluded that the BMPs are not effective in reducing pollution inputs related to NH_4+, NO_3-, conductivity, turbidity, TC, and E. coli. This conclusion can be drawn based on two main reasons; 1) the specified parameters exhibited similar trends at the site after BMP implementation (Site 2) and the site without any BMPs implemented (Site 1) and 2) concentrations of NH_4+, conductivity, turbidity, TC, and E. coli, did not decrease from the site upstream the BMPs (Site 3) and the site down-stream the BMPs after implementation (Site 2). With the implementation of the BMPs, the water quality at Site 2 was expected to be highest, but since the water quality is of similar or poorer condition than the water quality at Sites 1 and Site 3 in relation to the two reasons described above, the BMPs can be considered ineffective.

In order to increase the effectiveness of the BMPs, they must be frequently inspected for maintenance, and maintenance must occur when required. The cement/stream sediment slab and spillover weirs are designed to promote the sedimentation of sediments latent with pollutants, and therefore require the sediments to be manually removed overtime. Results from future water quality samples from Sites 2 and Site 3 can future assess the effectiveness of BMPs once maintained and allow the public to make informed decisions of when to recreate in Lake Gregory.

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