EPA Applications to Identify Headwater Watershed Changes and Best Management Practices in the San Bernardino National Forest

Sandra Ukaru

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EPA APPLICATIONS TO IDENTIFY HEADWATER WATERSHED CHANGES
AND BEST MANAGEMENT PRACTICES IN THE SAN BERNARDINO
NATIONAL FOREST

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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
Earth and Environmental Sciences:
Geology

____________________

by
Sandra Ukaru
May 2021
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May 2021
Approved by:

Jennifer Alford, Committee Chair, Geography
Kerry Cato, Committee Member
Bo Xu, Committee Member
ABSTRACT

The human-environmental landscape influences water quality across highly variable spatiotemporal scales. One such example is the human-environmental landscape in California, which has changed dramatically over the past several decades leading to a multitude of environmental issues including the reduction and impairment of water resources. This study compares surface water quality at four headwater streams in the San Bernardino National Forests using multiple assessment tools including in situ data, the Environmental Protection Agency (EPAs) Model My Watershed Tool and multiple databases within a GIS framework. These collective efforts enabled headwater watershed landscape changes and the extent of impacts to soil erosion and surface water quality to be modeled. The primary objectives of this include: 1. to determine the type of physical changes that will occur if the watershed landscape is altered; 2. to model these changes especially as they relate to soil erosion and changes in surface runoff related to precipitation events; and 3. to use findings to identify and recommend appropriate stormwater and watershed best management practices that reduce soil and water impacts and promote the protection and conservation of water resources that support community resilience. The results show the impact of climate change and also indicate that land use changes in the landscape draining to and forming headwater streams is a primary factor of hydrological variations in the catchments.
Key Words: Landscape changes, watershed modeling, surface water resource management, soil erosion and surface runoff.
DEDICATION

My deepest gratitude goes to my amazing committee chair, Dr. Jennifer Braswell Alford who guided me through my graduate education with her help, guidance, and motivation. Her unwavering support, mentoring and encouragement kept me engaged with my research and helped with the completion of this thesis.

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## TABLE OF CONTENTS

ABSTRACT ........................................................................................................................................ iii

LIST OF TABLES ........................................................................................................................ vii

LIST OF FIGURES ........................................................................................................................ viii

LIST OF ABBREVIATIONS ........................................................................................................ xi

CHAPTER ONE: INTRODUCTION

Literature Review ......................................................................................................................... 1

Watersheds .................................................................................................................................... 7

Land Use/Land Cover (LU/LC) and Water Quality ................................................................. 10

Climate Change ........................................................................................................................... 20

Water Quality Management .......................................................................................................... 24

Best Management Practices ......................................................................................................... 27

Geospatial Applications to Watershed Management ............................................................ 31

Arid Environments: A Southern California Case Study ......................................................... 35

Study Purpose and Objectives ...................................................................................................... 40

Study Site .................................................................................................................................... 42

CHAPTER TWO: METHODS

Modelling ...................................................................................................................................... 48

Water Quality Sampling .............................................................................................................. 48

Statistical Analysis ..................................................................................................................... 51

CHAPTER THREE: RESULTS

Watershed Landscape Characteristics/Description ................................................................. 52

Watershed Landscape Characteristics ...................................................................................... 53
LIST OF TABLES

Table 1. Water quality criteria include the EPA Recreational Criteria for E. coli, Lahontan Region Objectives for DO and pH, Hooks Creek Objectives for NO$_3^-$ and TDS ......................................................................................................................... 50

Table 2. Descriptive Statistics of Water Quality Data for Seeley Creek (SC) Samples ........................................................................................................................................ 62

Table 3. Covariance Correlations Matrix for Seeley Creek ........................................ 63

Table 4. Descriptive Statistics of Water Quality Data for Little Bear Creek (LBC) Samples ........................................................................................................................................ 66

Table 5. Covariance Correlations Matrix for Little Bear Creek ................................ 67

Table 6. Descriptive Statistics of Water Quality Data for Burnt Mill Creek (BMC) Samples ........................................................................................................................................ 70

Table 7. Covariance Correlations Matrix for Burnt Mill Creek ................................ 71

Table 8. Descriptive Statistics of Water Quality Data for Orchard Creek (OC) Samples ........................................................................................................................................ 74

Table 9. Covariance Correlations Matrix for Orchard Creek .................................. 75
LIST OF FIGURES

Figure 1a. Map of Seeley Creek drainage basin .......................................................... 46

Figure 1b. Map of Little Bear Creek, Burnt Mill Creek and Orchard Creek drainage basins ........................................................................................................... 47

Figure 1c. Map of the study area .................................................................................. 47

Figure 2a. Map of Seeley Creek showing land cover distribution .............................. 54

Figure 2b. Seeley Creek LU/LC chart ........................................................................ 54

Figure 3a. Map of Burnt Mill Creek showing land cover distribution ..................... 55

Figure 3b. Burnt Mill Creek LU/LC chart ................................................................. 56

Figure 4a. Map of Little Bear Creek land cover distribution ..................................... 57

Figure 4b. Little Bear Creek LU/LC chart ................................................................. 57

Figure 5a. Map of Orchard Creek showing land cover distribution ....................... 58

Figure 5b. Orchard Creek LU/LC chart ................................................................. 59

Figure 6. Dry Season Percentage Exceedance for SC, LBC, BMC and OC: DO, Temperature, Conductivity, NO₃, NH₄⁺, Turbidity, pH, TC, and E.coli .......... 77

Figure 7. Wet Season Percentage Exceedance for SC, LBC, BMC and OC: DO, Temperature, Conductivity, NO₃, NH₄⁺, Turbidity, Ph, TC, and E.coli ............. 78

Figure 8a. Developed low land cover for BMC, OC, LBC, SC: Percent increase in TSS ....................................................................................................................... 81

Figure 8b. Developed low land cover for BMC, OC, LBC, SC: Percent increase in TN ......................................................................................................................... 82
Figure 8c. Developed low land cover for BMC, OC, LBC, SC: Percent increase in TP ........................................................................................................................................................................82

Figure 9a. Developed medium land cover for BMC, OC, LBC, SC: Percent increase in TSS ........................................................................................................................................................................84

Figure 9b. Developed medium land cover for BMC, OC, LBC, SC: Percent increase in TN ........................................................................................................................................................................85

Figure 9c. Developed medium land cover for BMC, OC, LBC, SC: Percent increase in TP ........................................................................................................................................................................85

Figure 10a. Developed high land cover for BMC, OC, LBC, SC: Percent increase in TSS ........................................................................................................................................................................87

Figure 10b. Developed high land cover for BMC, OC, LBC, SC: Percent increase in TN ........................................................................................................................................................................88

Figure 10c. Developed high land cover for BMC, OC, LBC, SC: Percent increase in TP ........................................................................................................................................................................88

Figure 11a. Forested land cover for BMC, OC, LBC, SC: Percent decrease in TSS ........................................................................................................................................................................90

Figure 11b. Forested land cover for BMC, OC, LBC, SC: Percent decrease in TN ........................................................................................................................................................................91

Figure 11c. Forested land cover for BMC, OC, LBC, SC: Percent decrease in TP ........................................................................................................................................................................91

Figure 12a. Dry Season for SC: Runoff, Evapotranspiration, Infiltration.................................................................93

Figure 12b. Wet Season for SC: Runoff, Evapotranspiration, Infiltration.................................................................93
Figure 13a. Dry Season for LBC: Runoff, Evapotranspiration, Infiltration........... 94
Figure 13b. Wet Season for LBC: Runoff, Evapotranspiration, Infiltration........... 94
Figure 14a. Dry Season for BMC: Runoff, Evapotranspiration, Infiltration........... 95
Figure 14b. Wet Season for BMC: Runoff, Evapotranspiration, Infiltration........... 95
Figure 15a. Dry Season for OC: Runoff, Evapotranspiration, Infiltration ........... 96
Figure 15b. Wet Season for OC: Runoff, Evapotranspiration, Infiltration........... 96
<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>AGWA</td>
<td>Automated Geospatial Watershed Assessment Tool</td>
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<td>BMC</td>
<td>Burnt Mill Creek</td>
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<td>BMP</td>
<td>Best Management Practices</td>
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<td>BOD</td>
<td>Biological Oxygen Demand</td>
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<td>CFU</td>
<td>Colony Forming Units</td>
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<td>CLAWA</td>
<td>Crestline-Lake Arrowhead Water Agency</td>
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<td>COD</td>
<td>Chemical Oxygen Demand</td>
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<td>DO</td>
<td>Dissolved Oxygen</td>
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<td>DWR</td>
<td>Department of Water Resources</td>
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<td>EPA</td>
<td>Environmental Protection Agency</td>
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<td>GIS</td>
<td>Geographic Information Systems</td>
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<td>HAWQS</td>
<td>Hydrologic and Water Quality System</td>
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<td>LA</td>
<td>Lake Arrowhead</td>
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<td>LBC</td>
<td>Little Bear Creek</td>
</tr>
<tr>
<td>LU/LC</td>
<td>Land Use/Land Cover</td>
</tr>
<tr>
<td>MAF</td>
<td>Million-Acre Feet</td>
</tr>
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<td>MMW</td>
<td>Model My Watershed</td>
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<td>MPN</td>
<td>Most Probable Number</td>
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<td>NLCD</td>
<td>National Land Cover Database</td>
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<td>NPSP</td>
<td>Non-Point Source Pollution</td>
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<td>Acronym</td>
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<td>OC</td>
<td>Orchard Creek</td>
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<td>SAR</td>
<td>Santa Ana River</td>
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<td>SBM</td>
<td>San Bernardino Mountains</td>
</tr>
<tr>
<td>SC</td>
<td>Seeley Creek</td>
</tr>
<tr>
<td>SLAMM</td>
<td>Source Loading and Management Model</td>
</tr>
<tr>
<td>SWAT</td>
<td>Soil and Water Assessment Tool</td>
</tr>
<tr>
<td>SWRCB</td>
<td>State Water Resource and Control Board</td>
</tr>
<tr>
<td>TC</td>
<td>Total Coliform</td>
</tr>
<tr>
<td>TDS</td>
<td>Total Dissolved Solids</td>
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<td>TN</td>
<td>Total Nitrogen</td>
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<td>TP</td>
<td>Total Phosphorus</td>
</tr>
<tr>
<td>TSS</td>
<td>Total Suspended Solids</td>
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<td>USGS</td>
<td>United States Geological Survey</td>
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<td>USFS</td>
<td>United States Forest Service</td>
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<td>WQMA</td>
<td>Water Quality Management Agency</td>
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<td>WRPI</td>
<td>Water Resources and Policy Initiative</td>
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CHAPTER ONE

INTRODUCTION

Literature Review

The growing world population places increasing demands on water resources, especially as they relate to water quality and quantity. Increasing global populations will further strain water resources needed to adequately sustain the quality of life, including food and energy resources that collectively impact the environment, social and economic characteristics of a given community (Cosgrove and Loucks, 2015). Additionally, nutrients and sediments that are collected via water flowing through land surfaces are transported to adjoining surface water bodies. The depletion in the quality and quantity of water resources from the surface and ground coupled with the diversion of water resources from its source and climatic changes has created a multitude of highly variable issues related to water resource management across multiple geographical scales. The biological, physical, and chemical properties of headwaters, downstream to receiving waters are negatively influenced over a period of time by these processes (Tong and Chen, 2002; Meyer et al., 2007; Mallin et al., 2009; Brabec, 2009).

Point source and nonpoint source pollution (NPSP) loadings to estuaries have been found to be a great contribution to surface water quality impairment in regional and local watersheds in the last few decades (Carpenter et al., 1998; Mallin et al., 2009; Brabec, 2009; Alford et al., 2016). This is largely linked to
urban and rural landscapes as well as changes to the landscape including deforestation and increases in impervious surfaces. More recent observations note that climatic changes including prolonged droughts or intense rainfall events and flooding, also contribute to water impairment and quantity uncertainties (Levy et al., 2016).

One spatial context to observe landscape and surface water relationships including watersheds. Watersheds represent the area of land where water drains from higher elevation to a central point such as a river, lake or the ocean (Brooks et al., 2012). Observing relationships between the extent to which landscape characteristics influence the physicochemical characteristics of water resources is primarily observed at the watershed scale which enables one to understand what factors influence hydrological networks from the headwaters to the main stem and the mouth or discharge point of the watershed. The variable nature of landscape changes and pollution inputs warrants the need to understand how specific changes to watersheds, especially in headwater streams, serving as the start of the hydrologic network, create physiochemical changes to water resources at the site where these changes occur as well as downstream (Edwards et al., 2015). Despite the vast literature highlighting pollution inputs in surface waters throughout the United States, federal and state agencies have examined only about 19% (US EPA 2013; Levy et al., 2016). As a result, there is limited literature observing the spatial amplitude of what factors influence the physicochemical characteristics of headwater streams and how these factors
influence stream quality across the entire hydrological network. To resolve gaps in knowledge, more recent studies (Cosgrove et al., 2015; Wang and Zhang, 2018; Jabbar, 2019; Yuan et al., 2020) have observed surface water quality from a watershed scale to better identify and understand the spatiality of pollution inputs and how the locality of these inputs can impact the longitudinal reach of the hydrological network. Within a watershed setting, headwater streams are essential to observe because they represent the largest portion of the total hydrologic network and activities in these reaches can create or compound impairments downstream (Dodds and Oakes, 2008; EPA, 2013; Edwards et al., 2015; Alford et al., 2016). Observing these hydrological-terrestrial relationships is especially important in arid to semiarid environments where water resources are often insufficient to seasonal rainfall and drought conditions are becoming more prevalent reducing both the quality and quantity of water resources needed to sustain public and ecological health (Newman, 2006; EPA, 2008).

Although several pollution inputs have been identified with anthropocentric activities, nutrients including ammonium, phosphorus and nitrogen have a significant impact on water resources including eutrophication and hypoxic conditions (i.e. low dissolved oxygen) (Chislock et al., 2013). Nitrogen based fertilizers used in agriculture production, for example, is a major pollutant of both surface and groundwater (Gao et al., 2012). Nitrogen, pesticides, plant nutrients and heavy metals are all pollutants caused by both rural and urban storm related runoff (Khatari and Tyagi, 2015). As a contaminant, it appears in many forms
which includes ammonia (NH4 +), nitrate (NO3 -) and nitrite (NO2-) (Ghaly and Ramakrishnan, 2015). Nitrate is the most monitored form of nitrogen because of the solubility in surface/groundwater and adverse health effects to both humans and ecological services. An upper limit was set on nitrate in drinking water at 10mg per liter by the United States Public Health Service in 1962, however, environmental, and ecological impacts occur well below this standard often creating eutrophic conditions in surface waters (Oram, 2014; Dodds et al., 1998). One such example is the increasing occurrences of eutrophication in surface water bodies across the United States. Eutrophication is the result of excessive nutrient (i.e. nitrogen, phosphorus, ammonia) richness in a water body often from the application of fertilizers and livestock waste on the landscape as well as failing wastewater infrastructure (i.e. septic and sewer systems) entering waterways. Over time, excessive nutrient inputs lead to a dense growth of algae and plants that eventually displaces oxygen from the water. This type of extreme algal growth is a primary issue related to water impairments across the United States and globally (Dodds and Smith, 2016).

A majority of nutrient applications to the landscape are associated with agricultural production such as crop and livestock production on rural landscapes, however, some studies have noted this in relations to fertilizer applications in urban and suburban areas (Dale and Polasky, 2007; Kanianska 2016; Stubbs 2016). Agriculture activities including the presence of livestock on the landscape and the use of machinery to till soil in preparation for planting and
related harvesting, coupled with wind and precipitation events leads to widespread erosion (Zalidis et al., 2002; Montgomery, 2007). In urban and suburban areas, soil erosion and nutrient applications are often associated with the clearing of natural vegetation for buildings and roads as well as landscaping practices associated with grass lawns and ornamental vegetation (Muller et al., 2013; Issaka and Ashraf, 2017). For example, soil mounds formed due to the movement of soil and displacement of vegetation during urban construction are highly susceptible to wind and water erosion, therefore being a significant source of eroded soils entering waterways because of the rise of economic growth and construction projects in urban areas (Hu et al., 2001).

Collectively, urban, and agricultural activities represent a significant amount of the total eroded soils across the globe warranting a significant amount of attend to determine the extent to which eroded soils adversely impair surface water resources as observed by (Bai et al., 2008; Emam et al., 2015; Issaka and Ashraf, 2017; Aslam et al., 2020; Borelli et al., 2020 and others). Across both settings, nutrients often adhere to soil particles and during storm events, both the eroded soils and nutrients are transported to nearby waterways (Sthiannopkao et al., 2006; Issaka and Ashraf, 2017). This further leads to considerable changes in the amount of sediment reaching the streams as well as the amount and concentrations of nutrients residing in waterways (Mateo-Sagasta et al., 2017; Sthiannopkao et al., 2006). Once in waterways, nitrate and phosphate affect the ecological processes in rivers and lakes by increasing the level of fertility in the
stream, fluctuations in oxygen concentrations and species diversity. Owens and Walling (2002) assessed the amount of phosphorus in the fluvial sediment in a watershed that had high percentages of industrial activities and rural land types. Results highlight that the amount of phosphorus found in the sediment was between 500 and 1500 μg -1. The authors stressed that the rural and industrial build up was a significant factor in the high amount of total phosphorus in the basin indicating the need for reductions across the entire watershed.

Other characteristics of excessive nutrients in waterways are the presence of cyanobacteria or blue-green algal blooms often called toxic algal blooms. Blue-green algae in water bodies release toxic substances thereby reducing turbidity and limiting the amount of sunlight entry. This negatively impacts aquatic life and ecosystem structure including disruption to food chains, webs and aquatic species reproduction (Chang 2005; Koraley and Kara, 2018). Given the highly variable longitudinal physicochemical characteristics of surface waters as well as the diverse sources of pollution inputs a watershed context is needed to identify ways in which pollution inputs can be mitigated across the watershed, but especially in headwater streams. The exclusion of headwater streams in modeling human-environmental relationships is an essential component to developing comprehensive, collaborative, and sustainable watershed management strategies.
Watersheds

In order to understand the complexities related to water resources impairments, it is vital to understand and identify the hydrologic network (Omernik et al., 2017). Watersheds are hydrological units defined by topographical changes on the landscape (i.e. hills and ridges) that drain precipitation in the form of runoff to receiving water bodies (Walter et al., 2007). These natural landscape characteristics often cause soil erosion along slopes and impairs water quality as a result of the peak rate of runoff of water traversing the landscape during precipitation events. This association causes soil erosion and impairs water quality as a result of the peak rate of runoff (Carroll et al., 2000; Neary et al., 2009; Rickson, 2014). Studies have revealed that the quantity and quality of water that flows in a watershed is affected by climatic factors and the physical characteristics (such as geology, land use/land cover, soil type) of the watershed (Huang et al., 2013; Cho, 2016). The water storage and permeability of a watershed can be changed with vegetation removal by natural disasters such as fire (Bladon et al., 2014). This is because landscapes affected by fire lack vegetation thereby increasing runoff which eventually leads to soil erosion (Neary et al., 2005). There is a high risk of water quality impairment and flooding as a result of an increase in runoff and erosion during high precipitation events. In this case, there is slope instability causing water to flow through slippery slope while collecting and depositing sediments and rock particles which produces stream corridors. One of the factors that affects the development of the
stream corridor is the physical processes of sediment transport (Vandas et al., 2002). In order to reduce this, a good comprehension of the hydrological processes, climatic conditions, land use characteristics, soil types and water quality of the watershed are important.

One of the vital processes that support the shape of the surface of the Earth is the transport of sediment within and from a watershed, while also providing essential nutrients downstream (Munn et al., 2018). It can also be the primary mechanism by which pollution inputs enter and are transported downstream (Alexander et al., 2007). The amount of soil transported through surface flows are often determined by the topography of the landscape, soil types, frequency and intensity of precipitation events and landscape features. In undeveloped watersheds, soil erosion is primarily a function of slope steepness and precipitation regiments. However, in developed watersheds, human activities associated with impervious surfaces related to development activities that remove vegetation from the land are often primary drivers of soil erosion (Kouli et al., 2011; Markogianni et al., 2016; Gan et al., 2020). During precipitation events, raindrops often break up soil particles enabling them to be eroded through overland flow once soil reaches a saturation point (i.e. enable it to absorb more water through infiltration). Impervious surfaces within watersheds create barriers to infiltration causing stormwater flows that contribute to nearby surface waters (Holz et al., 2015). As the water flows across the landscape it will transport eroded soil particles that often include
natural (i.e. organic nitrogen, ammonium, phosphorus) and anthropogenic inputs (i.e. bacteria, metals, fertilizers, etc.) (Lee et al., 2003; Shen et al., 2020). Once in waterways, suspended sediments (i.e. eroded soils) may remain stationary or move downstream (Antoine et al., 2020). When pollution inputs adhere to eroded soils, they are transported to surface waters resulting in longitudinal variations in the types and concentrations of pollution inputs at the point of entry and downstream (Horowitz, 2009). During these processes excessive nutrients entering waterways are a primary concern. Nutrients may be natural (i.e. organic leaf matter, wildlife waste) or anthropogenic (i.e. human and livestock waste, fertilizers, septic or sewer failures) leading to algal blooms that endangers aquatic, wildlife and human health (Foulon et al., 2020). Within the watershed landscape, anthropogenic activities in headwater influence the amount of sediments and related pollution inputs entering the stream at the site of erosion as well as downstream. When pollution inputs adhere to eroded soils and are transported into surface water systems, they can influence the longitudinal physicochemical characteristics in headwaters and downstream influencing the entire hydrological network (Wallace and Eggert, 2015). As a result, primary headwater streams that are intact are like the support system network of healthy larger streams and rivers (Alexander et al., 2007). Suspended sediment at the watershed scale is critical in analyzing water quality degradation, sediment pollution, and the impairment of riparian ecosystems (Gao, 2008).
Land Use/Land Cover (LU/LC) and Water Quality

Surface water quality can also be affected by land use within its watershed as well as where specific land use types are located within the watershed (Camara et al., 2019). According to Sahu and Gu, 2009 there is a direct association between human activities and land-use types. The same studies also showed that there is a positive correlation between urban land use types, farmland and water quality pollution parameters and a negative correlation between grasslands, forested land use types and water quality pollution parameters (e.g., nutrients such as nitrogen and phosphorus) (Sahu and Gu, 2009). Less common is the observation of the extent to which specific activities impact water resources. For example, Kolpin et al., (2002), observed that housing and population density affect nitrate, chloride, and pesticides in streams that empty out urban and suburban settings. Additionally, water supplies in various high-density cities have proved that a minimal amount of chloride and nitrate concentrations transpire in water from forested land dominated areas and with higher concentrations occurring in urban areas with high density of housing and septic sewer systems. Wintertime application of road salt also increases the high concentrations of chloride in streams increasing conductivity that may impact downstream aquatic species (Jiang et al., 2014; Clough et al., 2016).

In order to understand how landscape changes over time by human activities impact water quality, Hicks and Larson (1997) generated a habitat assessment protocol showing that change in land cover percentage affects water
quality. The results of the study showed that increase in impervious surfaces and decrease in forest covers deteriorates water quality. It was observed that there was no noticeable humane impact with (>50%) forested cover, (10%) wetlands, and (<4%) impervious surface. Low impacts with (30-50%) forested cover, (6-10%) wetlands, and (4-9%) impervious surfaces. Moderate impacts with (10-29%) forest cover, (2-5%) wetlands, and (10-15%) impervious surface. High impacts with (<10%) forested cover, (<2%) wetlands, and (>15%) impervious cover. This suggests that water quality is affected by a reduction in natural landscapes and alterations to various land use types (Alford, 2014). According to Arnold and Gibbons, (1996), based on Schueler 1995, (10%) of impervious surfaces cause deterioration and (30%) cause critical deterioration to the landscape. Arnold and Gibbons (1996) also noted that there are issues in water quantity in the form of reduction in infiltration rates and soil percolation for recharge of groundwater as a result of high impervious covers. This also changes the hydrology of watersheds, thereby causing an increase in runoff volumes, flow rates and erosion. While land use categories are helpful, more specifically we need to understand the activities within these land use categories. Several studies (Ahearn et al., 2005; Huang et al., 2013; Henderson et al., 2014; Tahiru et al., 2020) have noted that it is not just the overarching land use covers that make the difference, it is the activities within those land use categories. For example, urban areas can be defined by highly compact development with large percentage of impervious surfaces, but it can also be sprawling residential areas.
that are only 20% impervious surfaces. Urban landscapes include commercial, industrial and residential. With all these going on in the urban watersheds, we are particularly interested in impervious surfaces as they are the main contributors to excess stormwater runoff that otherwise impair water quality. Agricultural landscapes are highly variable ecosystems and could be seasonal, phenological and inter annual. Agricultural land having livestock on the landscape is going to be more impactful than crop production. Forested landscapes could be recreation, it could be tambourine, it could be left alone and so on. With different activities defining these landscapes, it is vital to understand the connections between the surrounding landscapes and headwater stream conditions (Alexander et al., 2007).

**Urban Watersheds – The Role of Impervious Surfaces.** Human activities and processes can cause adverse effects on both water quality and quantity, though land types provide a benchmark on what type of pollutants to expect. Impact of human activities are important to a drainage basin because the local, regional and global differences in water flow and climate are notably large in size which cause inconsistent effects of human activities on water quality and land. This also is largely dependent on location within a watershed, biology, climate, geology, and topography. Human activities are controlled by these natural characteristics which alter the natural composition of water. The impairment of water quality in one portion of a watershed causes adverse consequences on downstream end users (Peters and Meybeck, 2009). Human activities such as
chemical applications to the landscape, waste disposal and treatment systems, and recreational activities pollute surface water features in urban environments (Mallin et al., 2000).

With urban sprawl being on the rise, impervious surfaces have become a primary problem in watershed planning and growth management as a result of their negative effects on habitat health (Arnold and Gibbons, 1996). The built-up urban areas such as parking lots, roads, buildings and pavements create surfaces that increase stormwater runoff rates that collect and move pollution inputs to nearby water features, while reducing soil infiltration opportunities that support groundwater recharge (Frazer, 2005). These impervious surfaces impact the natural hydrologic systems because they form barricades that prevent rainfall from infiltrating into the soil thereby recharging groundwater. Impervious surfaces are typically linked to streams by a stormwater pipe where water is not treated before entering surface waterways (Sabouri et al., 2013). The number and extent of impervious surfaces in an area is one of the most important elements in determining the negative effects of development on water quality (Moglen, 2009).

Researchers have proposed that water quality decline starts when 10% to 20% impervious surface in the watershed area (Holland et al., 2004; Schueler, 2009). Additionally, the concentrations of various stormwater pollutants increase with an increase in impervious cover while there is a decrease in stream biodiversity. Stormwater pollutants found in urban areas are connected with nutrients, suspended solids, trace metals and bacteria (Kunhikrishnan et al.,
Arnold and Gibbons (1996) identified four fundamental attributes of imperviousness that make it a major criterion of environmental quality: pollutants are transported into the waterways via the direct piping of stormwater by impervious surfaces; an impervious surface represents typical urbanization; even though impervious surface does not directly cause pollution, there is an obvious connection between impervious surface and changes to hydrologic processes that impair water quality; an impervious surface averts the development of natural pollutant in the soil by obstructing filtration. Studies reveal that one of the main water quality issues globally is microbiological contamination which is spatially linked with human activities as a major contribution. Toxic and carcinogenic chemicals such as biological contaminants and metals mostly found in urban and industrial wastes impair water quality and is dangerous to ecological and human health (Ashraf et al., 2019). Faecal pollutants are known to be associated with pathogens and when found in water resources, endangers human and ecological health (Reischer et al., 2008). In order to ensure the success of a quantitative microbial evaluation, it is required to carry out a rigorous evaluation of catchment hydrology and pollution dynamics; and continuous monitoring of water quality and seasonal changes (Reischer et al., 2008).

In an attempt to understand the relationship between urban areas and water resource quality, Permatasari et al., (2017) examined the different LULC types on the water quality in the Ciliwung Watershed using remote sensing data and water
quality monitoring data in 2010 and 2014. Water quality parameters that were
tested were dissolved oxygen (DO mg/L), total suspended solids (TSS, mg/L),
chemical oxygen demand (COD, mg/L), Total phosphorus (TP, mg/L), and
biological oxygen demand (BOD, mg/L). They reported that these water quality
parameters displayed remarkable disparities between the forest-dominated and
urban-dominated sites. The percentage of urban land had a strong positive
correlation with total nitrogen and ammonia nitrogen concentrations primarily
driven by the presence and amount of impervious surfaces across the
watershed.

**Forested Watersheds.** Forested watersheds provide access to essential
natural resources which includes, drinking water supply, agricultural, industrial,
hydro-electric and transportation (Larsen, 2017). These watersheds provide
numerous benefits to the ecosystem. They enhance water quality and increase
the supply in water storage, reduce flood damages and stormwater runoff,
balance stream flows and groundwater recharge amongst others. Although
forested and vegetated areas have little or no disturbance from human activity on
land, it is still vital to pay attention to activities that are likely to occur. Humans
still come in contact with these landscapes through hiking, kayaking. In the case
of natural forests, permits are granted to carry out activities such as mining,
timbering, amongst others. These activities can contribute to nonpoint source
pollution (Laney and Coleman, 2018). As a result of this, the effects of forestry
activities on water quality have been extensively researched with Brown and
Binkley (1994) and Tobin et al. (2007), observing that forestry operations have an adverse effect on water quality if best management practices (i.e. erosion and stormwater controls) are not adequately executed. A lack of proper management is likely to cause forestry operations to negatively affect water quality properties in surface water bodies that drain from the forests (Calder, 2007; Van Dijk and Keenan 2007; Sun and Vose 2016). Sediment concentrations can increase due to the removal of vegetation for hiking and biking trails; lack of riparian shade can cause a rise in temperature; harvesting and application of fertilizers and pesticides can cause an increase in organic and inorganic chemical accumulations; and reduction in dissolved oxygen due to accumulation of organic (Dismeyer, 2000; Kaffer and Martins, 2013; Orndorff, 2017). Climatic conditions and the type of forest practices applied have an impact on silvicultural nonpoint-source pollution.

Past literature on paired watershed approach have revealed that there is an increase in water yield with a reduction in forest cover, a decrease in water yield with afforestation, and nitrate nitrogen does not increase in streamflow after partial or complete clear-cutting (Neary, 2016). The effect of fire, invasive species and disease have all been a major concern of paired watershed studies (Stednick, 2008; Neary et al., 2009). Forest management activities such as fertilization of existing forest and harvesting have been observed to cause water quality impairment by creating changes in stream temperature, dissolved oxygen, nutrients, and sediment loads. It has also been observed that impairment of
water quality is generally short-lived and diminishes speedily as vegetation is restored and takes place at irregular intervals because in the course of a forest rotation, forest practices in a particular location is likely to occur about once or twice (Schoenholtz, 2004).

**Agricultural Watersheds.** Agricultural land is a man-made land area intended for the purpose to grow biological products for human consumption or use. There are three types which include arable land (cropland and fallows), land under permanent crops, and pastures and hay fields (Ritchie and Roser, 2013). Although agricultural land has lower runoff rates than impervious land cover, it adds the most nutrients to water bodies that impervious land use covers which is as a result of fertilizer application and intensive land use (Brabec et al., 2002). Intensive land use often results in nutrients accrual and agrochemicals in the soil which represents possible risk for the quality of surface and ground waters (Hekstra, 1995; Rozemeijer and Broers, 2007; Lockhart et al., 2013; Hunke et al., 2014; Mateo-Sagasta, 2017; Serpa, 2017). As a result of increase in population, more arable land for agriculture will be required to meet the population growth related food resource needs. This will cause the need to convert more natural lands to agricultural land use. This agricultural land use intensification has raised questions for the perpetual durability of agroecosystems (Liebig et al., 2004). Intensive farming such as pastures and livestock grazing, orchards and irrigated arable crops can cause different types of stress to the ecosystem health (Reidsma et al., 2006). Expansion of these agricultural systems leads to
environmental degradation such as depletion of water resources, soil erosion, water and soil contamination, flooding and landslides, and loss of biodiversity (Beaufoy, 2001). A decrease in river discharge as a result of overutilization of water could alter the river hydrology, increase siltation, reduce habitat heterogeneity and negatively impact aquatic biota (Meador, 2011). Water and soil resources are typically contaminated by frequent uses of synthetic fertilizers which are used to improve and increase the fertility of land. Effects of downstream runoff from such systems lead to high nutrient accumulation in the water bodies which result in exhaustion of dissolved oxygen, water eutrophication and loss of fish fauna (Matono et al., 2013).

In an effort to comprehend the association between agricultural systems and water quality, Meissner et al. (1999) identified that more than 50% of nitrogen discharged into the waterbodies is from agricultural systems which occur via a diffuse soil-groundwater-surface water pathway. Pionke and Urban (1985) noted that the main consequence of agricultural land use on ground water quality is through fertilization and manuring. Huang et al. (2013) notably reported in a study that agricultural landscapes that drain to rivers and lakes also impair water quality. The results revealed that there was a positive correlation of ammonia nitrogen and dissolved oxygen in cultivated land due to use of chemical fertilizers and agricultural practices. They were able to prove that agricultural land absorbs pollutants and reduces nutrient salts while improving water quality, leading to higher DO levels and lower TN and TP levels (Huang et al., 2013).
In the wet seasons, there is a weak association between water quality variables and various land use types compared to the dry seasons, whereas there is a stronger association between water quality variables and agricultural lands in both the wet and dry seasons.

Jarusianas, 2016 made an observation that water quality is highly influenced by agricultural land uses in steeper slopes compared to the flatter slopes. Yu et al. (2015) made a conclusion that land use types found near stream water is one of the best ways to test out the effectiveness of water quality. A global problem with agricultural fields is soil erosion which not only influences water quality but also reservoir sedimentation and soil fertility. It causes loss of fertile topsoil in the long run, decreases the land yield which also negatively impacts food security worldwide. The problem of soil erosion in agricultural lands has also been extensively recorded as a major factor affecting land degradation especially in Mediterranean watersheds (Pimental and Burgess, 2013). In California, the extensive period of droughts accompanied by heavy rainfalls on steep slopes and low vegetation increases the likelihood of soil loss. Watershed managers and hydrologists are mostly faced with the problem of soil loss from productive watersheds which usually have on-site and off-site effects. On-site effects consist of reduction in soil nutrients, soil moisture and organic matter; and a disintegration of soil structure (Mosbahi, 2013). These effects cause a loss of productivity which can lead to a reduction in the utilization of natural resources on the watershed landscapes. Off-site effects include nutrient loss and
sedimentation. Concentration of sediments in headwaters watershed causes a decline in water quality downstream and poses as a health risk for human consumption. There is an interruption in the flow of water via irrigated systems, a high risk of flooding in river basins and reduction of the life expectancy of downstream reservoirs (Ffolliot et al., 2013). Increase in sediment loading and soil erosion also threatens the efficiency of ecosystem services by the watersheds (EPA, 2018). Ecosystem services such as preservation of ecological diversity, climate control, nutrient cycling, ground water recharge and water purification are all essential in effective watershed management (Ffolliot et al., 2013). Human related activities over a long period of time also contributes to soil erosion challenges. The physical landscape altered by anthropogenic forces cause considerable soil erosion which in turn have negative effects on surface water bodies thereby requiring sediment control as a major aspect of management planning (Olsson and Barbosa, 2019).

Climate Change

Land use/land change and human activities are not the only factors contributing to water quality degradation. Climatic factors also play a huge role in this water quality degradation as an indirect consequence of these activities (EPA, 2017). Climate change comes in many forms, but the general trend is that precipitation events are becoming less predictable. The main effects of climate change on availability of water and surface water quality remains flood and drought. Drought, which is caused by land and water temperature, is a period of
time below-normal precipitation is experienced (Mosley, 2015). Drought reduces flow, increases pollution concentrations, reduces soil moisture leading to dryer vegetation increasing the conditions for fires. This creates an alteration of surface and ground water quality which leads to a limitation of water supply (Pena-Guerrero et al., 2020). Flood is a temporary overflow of water onto dry land. It moves pollution on landscape downstream in a short time, expanding pollution in water waters across the hydrologic/watershed network, impacting economic activities, agricultural and urban land, reducing aquatic diversity in waterways, and removing riparian habitat. Increased storm frequency and intensity commonly associated with climate change are intensified by the rise in floodplains, silted-up drainage, insufficient waste management and increased runoff from hard surfaces (Douglas et al., 2008). Although non-point source pollution could be mitigated, effects of climate change could likely intensify the distributed pollution with urban and agricultural run off for example. The primary cognitive factors of climate change impacting water quality are the increase of severe hydrological events and ambient temperature, and the reduction in precipitation that concentrates pollutants in waterways as the water evaporates and or is lowered in flows. Other factors are increase in solar radiation, soil drying-rewetting cycles, and immediate effects of dilution or concentration of dissolved substances. The major effect of water quality in low river flow rates is an increase in concentration of dissolved substances in water, a decrease in
concentration of dissolved oxygen and an increase in temperature (Prathumratana et al., 2008; Van Vliet and Zwolsman, 2008).

Water quality parameters such as pathogens, micropollutants and dissolved organic matter, are vulnerable to increase in concentration as a result of heavy rainfalls and temperature increase (water, air and soil) in temperate countries (Delpla et al., 2009). It is predicted that the number of precipitation events will be decreased by climate change in temperate regions, but the average volume of infrequent rainfall events will be increased (Brunetti et al., 2001; Bates et al., 2008). This will result in drought-rewetting cycles which could affect the quality of water while amplifying the decomposition and flushing of organic matter into streams (Evans et al., 2005). In the 1960s in North America, Europe and Asia, it was noted that there was a rise in surface water temperatures (0.2–2 °C), which was a consequence of atmospheric warming related to increase in solar radiation (Bates et al., 2008). Negative significant correlations were found between conductivity (from 0.2 to 0.9) and pH, dissolved oxygen and precipitation, in a study conducted to understand the surface water quality in the lower Mekong River.

Different lakes in North America and Europe experienced an extensive stratified period of two – three weeks, water temperatures rose from 0.2 to 1.5 °C. This had an impact on hydrodynamics of lakes (Bates et al., 2008) and thermal stratification (Komatsu et al., 2007). Van Vliet and Zwolsman (2008), Zwolsman and Van Bokhoven (2007) observed that in European rivers there was
a modal rise in water temperature by $2 \, ^\circ C$ after the drastic drought in 2003, an increase in pH with a decrease in CO$_2$ concentration, a decrease in dissolved oxygen solubility under higher water temperatures. A decrease in dissolved oxygen is sometimes associated to a rise in dissolved oxygen absorption of biodegradable organic matter by microorganisms (Prathamratana et al., 2008).

An increase in temperature of $2^\circ C$ has been estimated by computer models in European lakes by 2070. This will be as a result of the season and lake characteristics (George et al., 2007; Malmaeus et al., 2006). A rise in mean soil temperature is likely to increase N mineralization in soil (Ducharne et al., 2007). An increase in soil temperature could lead to an increase in enzymatic activity. Changes that occur in enzymatic activity are associated with direct effect of soil warming which increases N availability and energizes biological activity (Sardans et al., 2008).

One of the major influences on nutrient patterns and loadings is alternating weather conditions which plays a major role in the quality of surface water (Zhu et al., 2005). During warmer temperatures, there is a reduction in flow rate thereby increasing nutrient loadings in surface and groundwater (Van Vliet and Zwolsman, 2008). This prevents the effectiveness of policies put in place to control nutrient loadings in water bodies (Wilhelm and Adrian, 2008). An increase in mineralization and carbon from soil organic matter is often observed with a rise in temperature. However, during periods of high precipitation after a season of drought, there is an increase in erosion and runoff which also increases
transportation of pollutants. Drought causes an increase in ammonium loadings in water bodies that have a reducing dilution capacity (Zwolsman and Van Bokhoven, 2007; Van Vliet and Zwolsman, 2008). As a result of a decrease in the concentration of oxygen in bottom waters, there is an expected increase in the release of phosphorus in stratified lakes (Wilhelm and Adrian, 2008). These studies further prove that shallow lakes are mostly susceptible to climate change.

Across a watershed context, headwater streams are an important factor to consider with climatic changes because they are an integral part of the watersheds and critical in sustaining human society and culture (Colvin et al., 2019). With frequent drought seasons, headwater streams that are not perennial streams and lack a direct linkage year-round to groundwater, stand a risk of drying out naturally (EPA, 2017). With flooding, there will be an increase in stormwater runoff from urban and agricultural land covers, collecting pollutants from the landscape and transporting them to waterways linking up to the headwaters (Alexander et al., 2007).

**Water Quality Management**

There are a lot of chemical and biological agents that affect surface water quality and render them unsafe for human consumption. The most harmful being the dissolved pollutants which are invisible to the human eye. These include nutrients, heavy metals, agricultural chemicals, pathogens, human and municipal wastes and visible pollutants like suspended sediments. Nutrient sources could
be from animal waste and fertilizers (Bhagwat, 2019). In the United States, water quality has been a call for concern for an extended period of time. The United States addressed this issue through the Federal Legislation in 1900 along with the Oil Pollution Act of 1912, Public Health Act enacted in 1912 and the Refuse Act enacted in 1899 (Dzurik, 1996). The Federal Water Pollution Control Act of 1972 was the first federal legislation act that addressed the United States agricultural systems by identifying relationships between human activities and non-source point pollution like nutrients and pesticides which are the major agricultural problems of concern. When the concentration forms of nutrients exceed a critical level, they become harmful and can lead to eutrophication (USEPA 2002, Ashford and Caldart, 2008).

In order to successfully apply methods that aid in the reduction of pollutant transport from land to waterbodies, it is important to have a good knowledge of how soil properties interact with management practices. This helps to know the types, amounts and factors controlling the pollutants in the water bodies. Well managed soil has a high infiltration rate making it an effective receiver of rainwater, while poorly managed soil has a low infiltration rate causing runoff on land surface and conveying soil particles with it (Crouse et al., 2015). Dry and wet periods in arid environments tend to have an indistinguishable amount of small precipitation events but the wet periods mostly have a few large precipitation events. In a study carried out by Yahdjian and Sala, 2008 in the Patagonian steppe of Argentina, the different response to
soil processes in both the wet and dry seasons were examined. They compared the responses of litter decomposition and soil N mineralization to excess rainfall with responses to periods of droughts that were once recorded for the ecosystem. The results showed that environmental controls are likely to be saturated by the accessibility of substrate on longer time scales, and that the individual microorganisms that are in charge of ammonification and nitrification have dissimilar susceptibility to the availability of water (Yahdjian and Sala, 2008).

In order to properly manage and maintain water quality within a watershed, a holistic watershed management approach is required. By taking a holistic approach on watersheds, stakeholders have the ability to effectively assess pollution source inputs that are responsible for water quality impairment. Management steps such as setting criteria for surface water quality; frequent monitoring of surface water conditions; data analysis that supports the identification of impairments; validating the sources of pollution and establishing measures for water quality restoration. Collectively, these strategies are necessary to identify and implement the appropriate best management practices (Willett and Porter, 2001; Hunt et al., 2006; Alexander et al., 2007, EPA, 2020).

**Best Management Practices**

Best Management Practices' (BMPs), first originated in the field of Environmental Engineering in the 20th century and was used to describe
supporting functions for pollution control systems (Weaver, 2018). The term later came to be used about 35 years ago, to classify sustainable applications that would help in protecting water quality (NCFS, 2018). Simply put, they are measures that are used to lessen the quantity of pollution entering surface waters, air, land or ground waters. They are used to treat stormwater runoff dependent on the type and concentration of pollutants and their results vary in correlation to the size of the drainage areas. BMP selection is primarily based on certain site criteria such as landscape, quantity of stormwater and type of pollutant (Meals et al., 2014). According to Gautam et al. (2010), stormwater BMP design is affected by factors such as climate, topography, land use, soil type, vegetation, and geology. There is a limited amount of water in the arid regions of the desert southwest which makes BMP selection to be dependent on both surface water quality and groundwater quality. BMPs in arid regions are important because of the runoff patterns and high intensity of rainfall. Guatam (2010) stated that suitable best management practices (BMP) for arid regions should be modified for arid watershed characteristics, groundwater quality promotion, minimization of sediment and channel erosion, and avoidance of irrigation. In the past, stormwater management was targeted towards reduction of runoff peak flow from developed watersheds with secondary consideration given to water quality. BMP practices used to address this goal involved design of detention ponds which reduced peak flows and an insignificant amount of solid particles. Studies have shown that this method is not adequate in
addressing ecological stream degradation (Booth et al., 2002). Presently there is so much focus on water quality issues and other stormwater management outlook are being executed.

In addition to the implementation of BMPs, alternative stormwater management strategies have also been utilized to mitigate impacts to water resources from human activities. This approach employs small, localized infiltration frameworks to imitate the predevelopment hydrologic regime (US EPA, 2005). Infiltration trenches, bioretention systems, rain gardens, green roofs and constructed wetlands are all examples of substitute best management practices that can be put in place. Infiltration trenches are reservoirs below ground that accumulates stormwater runoff and allows for infiltration into the soil. Bioretention systems are shallow reservoirs that slows the flow of stormwater and diverts stormwater runoff through a soil medium for removal of pollutants and sediments (NJDEP, 2009). Rain gardens are depressions that collect and direct stormwater for recharge into the ground (EPA, 2020). Green roofs are covered with vegetation and are planted on the roof of buildings. They reduce the flow rate of runoffs and allow for evapo-transpiration to occur (Teemusk and Mander, 2007). Constructed wetlands are systems designed to mimic natural vegetated processes and filter beds to treat stormwater runoff and industrial wastewater (Vymazal, 2011). The above-mentioned processes are often known as stormwater BMPs in protecting water quality (Clary et al., 2002). A greater number of stormwater BMPs require stormwater to permeate into the soil for
groundwater recharge and underground flow providing a decrease in the quantity of runoff while promoting pollutant removal efficiency through physical, chemical and biological processes. Although, other management infiltration methods may not use infiltration as their main treatment approach, they are still effective in the infiltration of stormwater. An example of this is a standard detention pond which uses sedimentation for pollutant removal while infiltration takes place through the bottom and sides of the pond (Weiss et.al., 2008).

In the Western United States, water quality degradation has primarily been as a result of nonpoint source pollution and large-scale land use changes which has resulted in increases in stormwater runoff from impervious surfaces entering surface waterways (EPA, 2000, Corrao et al., 2015). Stormwater BMP implementation in the western United States have improved water quality efficiently by getting rid of pollutant inputs. This is proven by Piza et al. (2011) in his study of how the detention basin performed in Colorado with a 50.5% impervious surface. The water quality parameters that were tested were total suspended solids, metals, and nutrients and they were examined using paired t-tests and Wilcoxon signed-rank tests. The results of the study showed a decrease in nitrite-nitrate, total nitrogen and total copper. Barret (2005) observed different BMPs pollutant removal efficacy in Southern California. These included vegetated buffer strips and swales, detention basins, wet basins, infiltration trenches and basins. Water quality was tested from highway runoff for metals, nutrients and total suspended solids. In vegetated buffer strips and detention
basins, mean runoff volume was reduced by 30%, and 47% in vegetated swales. There were high concentrations of nitrate and a mass reduction of runoff volume. Stormwater BMPs have been effective in pollution control and water quality improvement to a large extent but have certain limitations. Such limitations are pollutant efficacy, catchment area, type of soil, slope, regional climate and groundwater depth (Jia et al., 2013). According to Booth and Jackson (1997), there are other issues which are a factor of design including cost, operation, maintenance and point discharges.

Research also observed that bioretention systems are among the best BMPs for stormwater quality control (Liu et al., 2014). They have a finite capacity for heavy metal removal. They effectively remove stormwater volume and increase the duration of stormwater discharge which helps to decrease the peak discharge of the storm event (Davis et al., 2009). Bioretention is not recommended for areas with slopes greater than 20% and where the water table is within 6 feet of the ground surface. Pollutant removal efficiencies for bioretention are as follows: Total Suspended Solids – 90%, Total Phosphorus – 70-83%, Total Kjeldahl Nitrogen – 70-80%, Metals – 93-98%, Bacteria – 90%, Organics – 90% (Wang et al., 2017). The design criterion for bioretention systems are the permeability, character, storage volume, and thickness of its planting soil bed, permeability of its subsoil or hydraulic capacity of its underdrain. When selecting a bioretention system and location, topography, ecology and geology of the proposed system site and bordering areas should be
considered. The best location would be upland from inlets that receive sheet flow from graded areas, areas that will be excavated and areas without karst topography (MSM, 2020). Best Management Practices (BMPs) are essential to mitigate human impacts on water resources, however, it is essential to know where within the watershed to place them highlighting the need to understand and apply geospatial techniques that observed impacts to water resources across a watershed, especially as they relate to landscape draining to and forming headwater streams.

**Geospatial Applications to Watershed Management**

Erosion processes are controlled by certain components which are climate, soil, vegetation and man's activities as well as the primary mechanism by which pollution inputs enter waterways. A lot of efforts have been made to examine how rate or erosion and key components relate to each other by using different types of models (Loucks, 2017). Models could be based on empirical data from plot and slope studies; simple or multiple regressions; or deterministic models that are directed at quantitative analysis of the physical process of entrainment of particles (Senanayeke et al., 2020). Computer modeling is growing in popularity in speculating soil erosion for land use and management practices. Geographic Information Systems (GIS) coupled with water quality models can be used either jointly or independently in estimating watershed erosion. Some of the water quality models developed to tackle soil erosion are: Soil and Water Assessment Tool (SWAT); Automated Geospatial Watershed
Assessment Tool (AGWA); the Hydrologic and Water Quality System (HAWQS); Agricultural Non-Point Source Pollution model, the Water Erosion Prediction Project; and the Areal Nonpoint Source Watershed Environment Response Simulation amongst others (Yuan et al., 2020).

The primary benefit of these models is that they can practically illustrate the spatial variability of catchment properties and related landscape changes that impact runoff and erosion rates potentially impacting water resource quality. Additionally, the models can assess the effectiveness of BMPs, and climate change on streamflow and water quality. Watershed models are created to be used over a compass of scales including field versus watershed, and environmental conditions with model structure complexity and a varying level of input data. A lot of studies have been put together to compare certain elements and complete modeling tools for different subcategories of current water quality models (Borah et al., 2006; Breuer et al., 2008; Refsgaard et al., 2010; Daniel et al., 2011). Over time, the SWAT model has emanated as one of the most frequently used water quality watershed modeling tool globally (Gassman et al., 2007; Williams et al., 2008; Arnold et al., 2012). The SWAT tool is a basin-scale model that is designed to help with the management of water resources, monitor the impacts of land use on the environment and impacts of climate change. It is designed to assess sediments, nutrient inputs, and surface water runoff from watersheds. However, the SWAT model is mostly used for agricultural areas and has less ability to model water quality parameters (Gassman et al., 2007).
Model My Watershed, a new tool developed by the Stroud Watershed Research Center in 2013 and recommended by the EPA (EPA, 2017), provides a platform that helps to delineate watersheds, helps understand the topography where the hydrologic lines are located, process changes that are likely to occur in different rain events related to the physicochemical parameters and provides a method for observing landscape and hydrological relationships, especially in smaller, headwater watersheds. Although it has not been widely used by researchers, it has various components that makes it a powerful tool. Some of the many advantages include the following:

● It is an interactive and online tool
● Easy to navigate/user friendly
● Allows users to calculate water budgets for various storm intensities, generic land-cover types, and soil textures for a generic patch of land and an area of defined watershed
● Allows users to gain an understanding of how soil and land use both control soil infiltration, stormwater run off, evaporation and transpiration by plants
● It investigates water management best practices in urban/suburban neighborhoods
● It models stormwater runoff and water quality for multiple parameters such as Total Suspended Solids, Total Nitrogen and Total Phosphorus
● Analyzes real land cover, soil and other geospatial data
It compares conservation or development scenarios in your watershed. The application computes stormwater runoff and how it affects water-quality with the use of professional-grade models and it compares possible land use/land cover development scenarios to existing conditions. Its numeric predictions of water pollution are known to be accurate and practical for calculating how land use and land management changes affect water quality. The Model My Watershed tool currently has two models to choose from which are Watershed Site Storm Model and Multi-year Model (EPA, 2017).

“The Model My Watershed Site Storm Model simulates a single 24-hour storm by applying a hybrid of the Source Loading and Management Model (SLAMM), TR-55, and the simplest of the Food and Agriculture Organization of the United Nations evaporation models for runoff quantity and EPA’s STEP-L model for water quality over the selected Area of Interest within the continental United States. The TR-55 component is used to calculate runoff for all natural and land-use types while the SLAMM component is used to calculate runoff for urban land-use types. The results are calculated based on actual land cover data from the USGS National Land Cover Database 2011, NLCD2011 and actual soil data from the USDA Gridded Soil Survey Geographic Database, gSSURGO, 2016 for the selected land area of interest. After which the model output pane is filled with predicted amounts of runoff and stream water quality. The runoff quantities are calculated using a combination of the TR-55 runoff model (USDA, 1986) developed by the U.S. Department of Agriculture and the Small Storm
Hydrology Model for Urban Areas developed by Robert Pitt for a single 24-hour rain storm. The water quality parameters are calculated using the EPA’s STEP-L water quality model (EPA, 2014). The model runs with real data on a chosen custom area” (Model My Watershed Technical Documentation by Stroud Water Research Center, 2020).

Arid Environments: A Southern California Case Study

Arid and semi-arid regions face multiple and highly variable challenges related to water resource management, particularly where surface waters support economic, social, and environmental activities and related ecosystems (Ragab and Prudhomme, 2001). These areas are identified by low precipitation which leads to drought, and flooding when precipitation occurs as a result of high extremities. Managing and developing water resources in watersheds located in arid regions require a unique approach due to the extreme seasonal differences in hydrologic flows resulting in wet and dry seasons (Achour, 2016; Chen et al., 2020). There are certain attributes with lasting effects on the watersheds which are not limited to, low humidity which leads to high incident radiation and temperature that cause rapid evapotranspiration, losses in river beds, strong winds and rare vegetation, poorly leached soils with high mineral content leading to low levels of infiltration and salt runoff, water yield which decreases with increase in size of watershed as a result of large losses in line and small unit size of producing runoff precipitation (Sahrawat et al., 2010; Achour, 2016). Arid areas tend to be adversely affected by these demands as evaporation rates
exceed natural recharge making the use of reservoirs and other water management techniques challenging to maintain over time. Hence, continuous depletion of water resources will occur (Stonestrom et al., 2007; Lund, 2018). The basic need for food, inflated cost of water supply, water budget deficit, and related energy value in arid areas should be addressed scientifically to proffer relatable solutions to the global current and future water scarcity (Abdalla and Chen, 2017). One such example of a dynamic landscape that includes semi-arid environments is California. The central and southern reaches of California have challenging water resource management issues.

Given its geographical reach, California has a highly variable climate including a Mediterranean climate in southern California which is warm, dry summers and mild winters receiving about 5 – 10 inches of precipitation per year while the northern parts of California receive 100 inches of rain or more annually. In other words, 75% of water available to California is in the northern third of the state while 80% of water demands are in the southern two-thirds of the state (WEF, 2014). This creates a stretch between the inadequate and volatile water supplies and the water needs of a growing economy and population (Xiao, 2017).

Agriculture has long been a major economic source for California as it provides food resources on a local, regional, national, and global scale. However, agriculturally based activities, not only consume the most water resources, but they are also the primary source of pollution related to surface runoff (Wilson et al., 2017; 2018; Eric et al., 2018; Lund et al., 2018). Runoff from agricultural
landscapes often contains sediments from soil erosion, nutrients (i.e. nitrogen, phosphorus and ammonia), and harmful bacteria and pharmaceuticals from livestock waste that flows into adjoining water bodies during precipitation events and excess irrigation which, over time, enters both surface waters and leaches into groundwater contamination both resources (Johnson and Cody, 2015; EPA, 2020). As a result of this, water bodies throughout the state that include 8,000 miles of rivers and streams and 300,000 acres of lakes, bays and wetlands have become impaired. Scientific literature proves that streams have a significant impact on downstream waters (Liebowitz et al., 2018; EPA, 2015).

According to USGS, about 60% of water use and withdrawal is for agricultural sectors in the state, while DWR stipulates that about 40% of water supplies is for agricultural irrigation. However, there is an adjustment in agricultural usage which is being reduced and traded to cities like Imperial Valley and San Diego. This results in economic changes to each of the affected regions (Hanak et al., 2008). USGS evaluates water use in California to be about 42.6 million-acre feet (MAF) in 2010 which is 38 billion gallons per day. These figures represent water taken from surface and groundwater sources (Johnson and Cody, 2015). During drought periods, a decrease in groundwater reserves is anticipated because there is availability of less surface water which results from aquifers not being recharged in wetter years. The long-term reduction in California’s groundwater reserves shows the lack of proper groundwater management and surface-water scarcity (Lund, 2016). In any case, the
consolidation and improvement of the water infrastructure, with appropriate management, amounts to vital conditions for long lasting solution of the problem, serving as a basic element for inland development (Cirilo, 2008). California is most likely to face additional water challenges in the future. California Department of Finance made some population projections in 2013, that by 2030, California will gain a population of 44.1 million, which is a growth of over 5 million from the estimated 38.7 million in 2015 (PPIC, 2016; Xiao, 2017). If water conservation efforts reach their limit, the rise in population growth coupled with a booming economy will probably propel an increase in water demand (Xiao, 2017). To mitigate these impacts, it is important to understand the extent to which watershed landscape changes, especially in perennial headwater streams that provide water resources year-round (i.e. wet and dry seasons) impact surface water hydrology and related quality, which includes soil erosion since eroded soils are a major contributor to impaired surface water resources (Escriva et al., 2016).

Southern California has already been affected by climate change especially where regional rise in temperature and vegetation shifts have been observed (Bachelet et al., 2015). For the past few decades, Southern California has experienced prolonged, extreme drought conditions with only a few major precipitation events highlighting the need to implement more sustainable resource management strategies and related policies. The changing climate has increased the need for water but reduced the supply. As the climate warms
leading to increase in temperatures, evapotranspiration increases (PPIC 2015; Thomas et al., 2017; Swain et al., 2018; Stewart et al., 2020). Transpiration also increases, making irrigated farmland require more water. These rise in temperatures and decrease in rainfall in nearby states cause the reduction of water flow in the Colorado River, which is a major source of irrigation water in southern California (EPA, 2016; Swain et al., 2018). More than half of the annual precipitation will either evaporate, be used up by vegetation, plunge into the subsurface or flow to the ocean. The rest of it is what becomes available to meet California’s agricultural, urban and Southern California environmental needs of which most of it is used up by agriculture for irrigation purposes (Javadinejad et al., 2020; Osaka et al., 2020).

Applying techniques that enable communities to understand potential impacts allows communities to become more knowledgeable about how such changes impede their ability to become resilient to ensuring that water resources are protected for current and future generations as well as supporting highly variable ecological services. The water quality modeling tool is an important tool to support identifying land use and water quality relationships in this study area. It will also help to support with source water assessment and protection initiatives, locate areas that are likely to have adverse impacts and determine where to concentrate management efforts in the affected communities.
Study Purpose and Objectives

This study compares surface water quality data collected in situ, sponsored by Water Resources and Policy Initiatives (WRPI); the Environmental Protection Agency’s My Watershed Tool; and multiple databases within a GIS framework to model and project the effects of landscape changes and the extent of impacts to soil erosion and surface water quality. The study area encompasses four watersheds with headwater streams that experience various anthropogenic activities (i.e. tourism, residential, commercial development) that contribute to multiple recreational lakes and water resources in the San Bernardino National Forest. These sites include: Lake Arrowhead/Little Bear Creek (HUC 22658693); Lake Arrowhead /Burnt Mill Creek (HUC 180902080103); Lake Arrowhead/Orchard Creek (HUC 22660093); and Silverwood Lake/Seeley Creek (HUC 22660085).

To determine the extent to which landscape changes (i.e. increases in impervious surfaces, removal of forest vegetation, grading of natural landscapes) may impact surface hydrology, recreational (i.e. tourism) and water resources that support human and ecological activities, this study will apply the Stroud Water Research Center’s Model My Watershed Web Application. The primary objectives of this application to this study include:

1. to determine the type of physical changes to soil and water resources and the location within the watershed of these changes (headwaters vs. mouth of
watershed) that will occur if the watershed landscape is altered (i.e. increases in impervious surfaces, vegetative removal, etc.);

2. to model the temporal aspect of these changes especially as they relate to soil erosion and changes in surface runoff related to precipitation events; and

3. use findings to identify and recommend appropriate best management practices that reduce soil and water impacts and promote the protection and conservation of water resources that support community resilience.

The results of this study will further be made available to the public and decision makers (i.e. resource and planning agencies) to encourage the implementation of sustainable water resource management at the watershed scale so that communities reliant on these resources can predict and become proactive in achieving community resiliency. The Model My Watershed tool was chosen for this study because it incorporates various important elements and operations for simulating the water balance, sediment loss, climate change, agricultural and land management practices. The model can be used in both large and small watershed scales.

**Study Site**

The San Bernardino National Forest encompasses the San Bernardino and San Gabriel Mountains on the eastern portion and the San Jacinto and Santa Rosa Mountains on the northern portion. The forest includes seven wilderness areas. The forest covers 823,816 acres of land and is the most visited in Southern California. The San Bernardino mountains are the primary
headwaters of the Santa Ana River (SAR) which happens to be the largest river in southern California. They also serve as headwaters of the Mojave River in the desert (Miles et al., 1998). SBNF has 15 management areas which are established on: desert areas, possible desert areas and a mixture of watersheds that have homogenous properties (RMC Water & Environment, 2015). As one of the 18 national forests in California, it is inclusively recognized as Region 5 of the US Forest Service (USFS). In 1981, Region 5 and the State Water Resources Control Board (SWRCB) got into a Management Area Agreement in accordance with the Clean Water Act Section 208. Based on the agreement, Region 5 of the USFS was designated to be the Water Quality Management Agency (WQMA) for SBNF. This designation also requires that Region 5 takes charge of the appropriate positioning, performance, and servicing of the State- and EPA-authorized BMPs in SBNF. Region 5 is also responsible for ensuring that water quality issues in SBNF are rectified, appropriate BMPs are applied, and executing of maintenance services for BMPs. In order to effectively resolve water quality problems, SBNF and the Upper SAR watershed work together with the Regional Water Quality Control Board (RWQCB) (RMC Water & Environment, 2015).

In an effort to understand how different watershed characteristics influence headwater streams across this reach of the San Bernardino National Forest, it is essential to properly manage water resources across the hydrologic networks in the Santa Ana and Mojave river basins (Izbicki, 2007). This is
important because the headwater streams in the San Bernardino National Forest provide perennial surface flows year-round to the largest two river basins in Southern California and also probably the largest two river basins in California. The Mojave river basin being a desert basin is highly limited in water resources and Santa Ana is the most populous basin in Southern California. As a result of this, there is a strain on downstream water resources (Alford and Caporuscio, 2020).

**Lake Arrowhead.** Lake Arrowhead serves as the primary drinking water resource for Lake Arrowhead residents. Secondary purposes include recreational opportunities such as fishing, boating and swimming, landscape, tourism, and providing water resources for ecological services. This makes it a multifunctional lake. Lake Arrowhead (LA) is surrounded by the San Bernardino Mountains (SBM) with a water level at an elevation of 5,174 ft above sea level. Lake Arrowhead acts as a catchment for the surrounding tributaries year-round. While some water is purchased and imported to LA from the Crestline-Lake Arrowhead Water Agency (CLAWA), most water is introduced to LA from the surrounding headwater stream that drains diverse watersheds. Some of which are just residential, some are mixed residential and commercial. The LA outlet creates a headwater to Deep creek, a national scenic river that converges with the outlet to Silverwood lake downstream creating the headwater or the start of the Mojave river. The difference in surrounding the topography and elevations determine where the water will flow and accumulate (Edwards, 2015). The water that
accumulates in LA from surface runoff contains pollutants and other water quality indicators including nitrogen, phosphorus, nitrates, Total Dissolved Solids (TDS), as well as turbidity (Lake Arrowhead CSD, 2012) and is tested bi-annually to determine the water quality of LA. Headwater streams have been found to have the biggest influence on water quality. One study found that the smallest streams (first order) contributed more to water quality than any other factor. The study determined this was because the larger numbers of smaller streams tend to have high nutrient loads which influences water quality downstream (Dodds and Oakes, 2008).

Seasonal variations also affect physicochemical characteristics of headwater stream quality as well. Pollutants and pathogens continuously collect throughout dry spells and when high precipitation events occur, stormwater runs off and spills into local waterways (Gershunov et al., 2018). The San Bernardino National Forest watershed’s topography channels water to LA where it discharges into the lake and the differing experiences during wet and dry seasons influences the over stream water quality. In the winter, LA and the surrounding mountains receive on average 41.7 inches of rain a year (US Climate Data). More than half the total precipitation falls between December-March. During the winter months, high precipitation events wash accumulated debris, nutrients, pollutants into headwater streams that discharge into LA. Most pollutants and excessive nutrients enter headwaters during the first few inches of rainfall. However, during warmer months, many of the streams that contribute to
headwaters begin to dry up, the evaporation that takes place concentrates pollutants and nutrients that continue to discharge into LA, during the summer, the San Bernardino mountains experience little to no rain. The streams in the higher elevations begin to evaporate as the summer progresses, causing concentrated levels of pollutants and nutrients. Eventually some streams dry up completely during the dry season and the concentrations of pollutants and nutrients remain in the stream beds. Southern California does receive precipitation during the dry season, and in the SBM’s the two main events that cause precipitation are monsoons and orographic lifting. Monsoons occur in the summer months and produce intense rainfall. Summer thunderstorms initially form along the mountain ridges in Southern California and produce high precipitation events that cause flash flooding in the region (Moore et al., 2015) and can contribute up to 50 percent of total annual rainfall for the southwestern United States (Pascale et al., 2016). While the SBM’s only see a fraction of the precipitation, it still contributes to headwater quality. Current scientific research suggests that climate change will continue to affect climate patterns, and as a result, droughts prone to Southern California will become longer, precipitation events will become less, but intensity and amount will increase, increasing the risk of headwater pollution (Marion et al., 2017).

**Seeley Creek.** Seeley Creek is a creek in Crestline, located in Silverwood Lake State Recreation Area, San Bernardino National Forest, California. Seeley Creek is accessible through Forest Service Road from Highway 138 to Crestline.
Seeley Creek is a hot spot for recreation, swimming and hiking and it is mostly called Heart Rock located in the Valley of Enchantment. Seeley Creek is the only perennial creek in the SBM’s lake that flows all year round and drains into Silverwood Lake. Silverwood Lake is used for recreation, it is a primary drinking water resource for mountain communities and San Bernardino communities including the city of San Bernardino and Rialto. Silverwood Lake is also a part of the State Water Project. Silverwood Lake has excessive algal bloom issues, so it threatens public and ecological health (DWR, 2019). For this reason, it is important to study and test the water quality at Seeley Creek.

Figure 1a. Map of Seeley Creek drainage basin.
Figure 1b. Map of Little Bear Creek, Burnt Mill Creek and Orchard Creek drainage basins.

Figure 1c. Map of the study area.
CHAPTER TWO

METHODS

Modelling

Model My Watershed tool was used to analyze and model the watersheds. Model My Watershed is a web application developed by Stroud Water Research Center, that allows users to integrate a variety of data sets, such as land cover and soil type, to calculate stormwater runoff for a specific area. It models how water, sediment, nitrogen, and phosphorus move across and underneath the Earth’s surface.

Water Quality Sampling

Water quality was monitored for ammonium (NH$_4^+$, mg/L), conductivity (μS/cm), dissolved oxygen (DO, mg/L), pH, nitrate (NO$_3^-$, mg/L), turbidity (NTU), and water temperature (°C) using ion selective electrode probes and a Vernier Labquest 2 monitor comparable to Vega et al. (1998), Varol et al. (2012), Khatoon et al. (2013), and Schraga and Cloern (2017). Additional grab samples were collected, immediately placed on ice, and transported to California State University, San Bernardino to test for total coliform (TC, cfu/100mL) and Escherichia coli (E. coli, cfu/100mL). Grab samples were collected in 1 (L) brown opaque HDPE plastic bottles that were acid washed using EPA protocols (EPA, 2003). Samples were collected bi-weekly during the dry season (i.e. April to October) and weekly during the wet season (i.e. November to April) to identify
physicochemical and surface flow trends related to climatic and seasonal changes. Though the EPA approved IDEXX testing procedures for total coliform and E. coli are reported in most probable number (MPN), IDEXX indicates that results align with the EPA colony forming units (cfu), so both units can be used interchangeably (IDEXX, 2018). Individual sampling events and the means of samples were compared to federal, state and regional water quality objectives and standards to determine the frequency in which samples met or exceeded these requirements as outlined in Table 1.
Table 1. Water quality criteria include the EPA Recreational Criteria for E. coli, Lahontan Region Objectives for DO and pH, and Hooks Creek Objectives for NO$_3$ and TDS.

<table>
<thead>
<tr>
<th>Water Quality Metric</th>
<th>Standard</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature (°C)</td>
<td>&lt; 25</td>
<td>CA State Water Board</td>
</tr>
<tr>
<td>Dissolved Oxygen (DO) (mg/L)</td>
<td>&gt; 4</td>
<td>CA State Water Board, Lahontan Region</td>
</tr>
<tr>
<td>pH</td>
<td>6.5-8.5</td>
<td>CA State Water Board, Lahontan Region</td>
</tr>
<tr>
<td>Turbidity (NTU)</td>
<td>&lt; 100</td>
<td>CA State Water Board (Fact Sheet)</td>
</tr>
<tr>
<td>Conductivity (µS/cm)</td>
<td>150-500</td>
<td>EPA (Range)</td>
</tr>
<tr>
<td></td>
<td>Range</td>
<td>CA State Water Board (Average)</td>
</tr>
<tr>
<td></td>
<td>&lt; 336</td>
<td>(Average)</td>
</tr>
<tr>
<td>Nitrate (NO$_3^-$) (mg/L)</td>
<td>0.8-2.5</td>
<td>San Bernardino Mountains Hooks Creek Objectives</td>
</tr>
<tr>
<td>Ammonium (NH4+) (mg/L)</td>
<td>0.02-0.4</td>
<td>EPA Aquatic Life Criteria</td>
</tr>
<tr>
<td>Total Coliform (TC) (cfu/100mL)</td>
<td>1,000</td>
<td>CA State Water Board Objectives</td>
</tr>
<tr>
<td>E. Coli (cfu/100mL)</td>
<td>&lt; 126</td>
<td>EPA Recreational Standards</td>
</tr>
<tr>
<td>Total Dissolved Solids (TDS) (mg/L)</td>
<td>&lt; 127</td>
<td>San Bernardino Mountains, Hooks Creek Objectives</td>
</tr>
</tbody>
</table>

Source: CWT, 2004; EPA, 2018, 2019b; WB, 2002; WQCP, 2015
Statistical Analysis

Data for physicochemical parameters of water samples were analyzed using descriptive analysis. Descriptive statistics such as mean, variance, and standard deviation, for each water quality parameters were calculated across all sites during the study period using SPSS version 24. Applying similar methods by Alford et al. (2016), and Khatoon et al. (2013), water quality parameters were tested for normality using skewness, kurtosis and shapiro-wilk’s tests. Parameters that were not evenly distributed were log transformed using Microsoft Excel (Mallin et al., 2016). After which Pearson’s correlation coefficient for each sampling location was created in order to understand the statistical relationship or association as well as direction between continuous variables. A p-value threshold of 0.05 and 0.01 were considered statistically significant.
CHAPTER THREE
RESULTS

Watershed Landscape Characteristics/Description

In natural settings, when trying to compare hydrologic responses, we deduce that watersheds situated in the same or corresponding ecozones found within a physiographic region, have variable amounts and configurations of land use types as one moves from the headwaters to the mouth of the hydrological unit. In this study, the sites range from first-order headwater streams and catchments with similar land use types, but variation in land type configurations and resulting percentages in specific land uses including impervious surfaces.

When considering soil and water relationships, all watersheds observed are characterized by slow infiltration rates related to similar soil types, vegetation and slopes. The culmination of high runoff potential, very slow rate of water transmission, shallow over nearly impervious material, clay pan or clay layer near the surface, clays with high swell/shrink potential, permanent water table. This means that with any form of development in the area, there is high runoff risk in sloping areas which can lead to on site and downstream erosion and flooding. As a result, there is a need in this region to consider how increasing development or changes to the landscape may increase water quality impairments. Utilizing both in situ and modeled data, the results of this study may assist with identifying stormwater best management practices to reduce excessive runoff rates,
promote infiltration and to collectively improve water quality in headwater systems and downstream.

**Watershed Landscape Characteristics**

*Seeley Creek.* Seeley Creek (SC) also known as Heart Rock is located 2.4 miles Northwest from Crestline. Collectively, the watershed characteristics of Seeley Creek has an evergreen forest space of 54.86%, 27.25% developed open space (Figure 2b) and has a slow infiltration rate. There is an open lot, a camp (Camp Seeley) and the village of the Valley of Enchantment. This supports a perennial stream flow that traverses a mixed watershed landscape characterized by spatially dense residential and commercial areas. Within this watershed, the area that drains and forms the headwaters is characterized as developed open space (i.e. a parking lot, Seeley Creek waste treatment plant). As the stream traverses the watershed from upstream to downstream, that landscape of the drainage areas increases to include roadways and developed medium intensity developed areas with housing, a school and a small commercial business district (i.e. county services, restaurants, storage units, stores). Lastly, Seeley Creek traverses a United States National Forest recreational hiking area before it converges with the east fork of the Mojave right and Silverwood Lake, a primary drinking and recreational water source for the mountain, High Desert and the City of San Bernardino and surrounding communities.
Figure 2a. Map of SC showing land cover distribution.

Figure 2b. Seeley Creek LU/LC chart.
Burnt Mill Creek. Burnt Mill Creek (BMC) is a watershed with a perennial stream that drains the upper ridge of California Highway 18 and a down sloping area that terminates into Lake Arrowhead (Figure 3a). It primarily drains residential land use development types, churches, schools, and some commercial storage areas are mixed within the watershed. Additionally, a yacht club bay and a 75 car parking lot along a gentle slope ¼ from the beach at the terminus of the old highway 138 are present before the creek terminates into the lake. It is characterized by 59% evergreen forest and 40% developed open space (i.e. compact development). It also has a slow infiltration rate. The main facility of the area is the Burnt Mill Beach Club which has 18 picnic units terraced on the hillside under the spread of a group of sycamores which all make up the developed open space.

Figure 3a. Map of BMC showing land cover distribution.
Little Bear Creek. Little Bear Creek (LBC) is characterized by perennial stream flow, with headwaters developing from drainages from a forested area. As the stream flows downstream, it traverses commercial business areas, residential homes and a second larger commercial business area before terminating into Lake Arrowhead. The waters of LBC feed Lake Arrowhead and it is a tributary of Deep Creek. It contains 64.36% evergreen forest and has a slow infiltration rate. LBC is west to east whereas burnt mill creek is south to north. This is important because north facing slopes have more shadows than the south facing slopes so when it snows, it melts sooner on south facing slopes than the north facing slopes causing variability in stream flow regiments.
Figure 4a. Map of LBC showing land cover distribution.

Figure 4b. Little Bear Creek LU/LC chart.
Orchard Creek. Orchard Creek (OC) is a stream situated southwest of Cedar Glen close to Orchard Bay and drains into Lake Arrowhead. It has some housing units in the area with low levels of impervious surfaces. It is characterized by 70% evergreen forest and 26% developed open space (a few scattered residential buildings mixed with evergreen and a business district) and has a slow infiltration rate.

Figure 5a. Map of OC showing land cover distribution.
Prior to running the model, in situ data collected by Dr. Alford in the San Bernardino National Forests was observed to understand general watershed landscape and water quality characteristics across both wet and dry season periods to develop a baseline knowledge of the landscape-water quality relationships that can be related to the EPA’s Model My Watershed to determine appropriate stormwater Best Management Practices (BMPs) (Alford, 2021). In situ surface water quality monitoring for 2019-2020 occurred during extreme weather patterns that included prolonged dry periods. Hydrologically, this created low base flow events, precipitation periods marked by heavy rains and smaller
events that resulted in higher stream flows. It was observed that during heavy and prolonged precipitation rainfall accumulations, there was a spike in NO$_3^-$ across all sites with some individual samples exceeding regional water quality objectives in Table 1. This proposes that during precipitation events, surface flows contribute higher accumulations of pollution inputs to the potentially impacting groundwater quality. It was also observed that spikes in water quality parameters were associated with both precipitation events and extended dry periods which explain that the impact is year-round.

Various parameters had a mean that exceeded the criteria and objectives of state regulations (e.g. NO$_3^-$, NH$_4^+$, TC, and others) while other parameters had individual samples that exceeded the criteria and objectives (e.g. Conductivity, pH, E. coli, and others).

**Seeley Creek**

Seeley Creek (SC) has four water quality parameter means exceeding the criteria stated in Table 1. These are NO$_3^-$ (2.59 mg/L), NH$_4^+$ (0.44 mg/L), TC (1040.41 cfu/100mL), E. coli (200.59 cfu/100 m/L). However, a majority of parameters have individual samples that failed to meet their criteria and objectives including conductivity, NO$_3^-$, NH$_4^+$, turbidity, pH, TC and E. coli. Conductivity has a mean of (221.47 μS/cm) which does not exceed the CA State Water Board mean objective (<336 μS/cm), but 8.3% of the samples (i.e 3 individual samples) did not meet the EPA range standards (150-500 μS/cm) with values of 106μS/cm, 19.1μS/cm and 141.5μS/cm with no recorded rain event on
those days. NO$_3^-$ has a mean of (2.59 mg/L) and 61% of the samples (i.e. 22 individual samples) which did not meet the San Bernardino Mountains Hooks Creek objectives of (0.8-2.5 mg/L). High individual NO$_3^-$ values were recorded during the wet season and low individual values were recorded during the dry season. NH$_4^+$ has a mean of (0.44 mg/L) and 40% of the samples (i.e. 14 individual samples) that did not meet the EPA Aquatic Life Criteria (0.02 – 0.4 mg/L). High individual NH$_4^+$ values were recorded during the wet season and low individual values were recorded during the dry season. Turbidity has a mean of 12.03 NTU, but 3.33% of the samples (1 sample) did not meet the CA State Water Board objective (< 100NTU). The individual sample was recorded on a day with no precipitation event. pH has a mean (6.66) that was within the CA State Water Board objective (6.5-8.5), but 36% of the samples (i.e. 13 individual samples) that did not meet the objectives. Low individual samples of pH were recorded on days with little to no precipitation events. Total Coliform has a mean (1040.41 cfu /100 mL) and 20% of the samples (i.e. 4 individual samples) that exceeded the CA State Water Board objective. These high individual samples were recorded during high precipitation events. E. coli has a mean of (200.59 cfu/100mL) and 15% of samples (i.e. 3 individual samples) that exceeded the EPA standards (<126 cfu/100mL). These individual samples were recorded during high precipitation events. Total coliform (721402.63 cfu/1000mL) and E. coli (286481.45 cfu/1000mL) have the greatest variance followed by conductivity (3976.35 μS/cm) and turbidity (1091.98 NTU).
Table 2. Descriptive Statistics of Water Quality Data for Seeley Creek (SC) Samples.

<table>
<thead>
<tr>
<th>Descriptive Statistics SC</th>
<th>N</th>
<th>Min</th>
<th>Max</th>
<th>Mean</th>
<th>Std. Deviation</th>
<th>Variance</th>
<th>Criteria/ Standards</th>
<th># and % Exceeding</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow m/s</td>
<td>27</td>
<td>0.02</td>
<td>1.5</td>
<td>0.55</td>
<td>0.39</td>
<td>0.15</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>DO mg/L</td>
<td>34</td>
<td>6.84</td>
<td>10.87</td>
<td>8.79</td>
<td>0.93</td>
<td>0.86</td>
<td>&gt;4 mg/L</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>Temp. C</td>
<td>34</td>
<td>4.1</td>
<td>17.7</td>
<td>9.99</td>
<td>3.31</td>
<td>10.93</td>
<td>&lt;25 C</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>Conductivity µS/cm</td>
<td>36</td>
<td>19.1</td>
<td>376</td>
<td>221.47</td>
<td>63.06</td>
<td>3976.35</td>
<td>150-500 Range</td>
<td>3 (8.3%)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>&lt;336 µS/cm (mean)</td>
<td></td>
</tr>
<tr>
<td>NO₃ mg/L</td>
<td>36</td>
<td>0.2</td>
<td>6.3</td>
<td>2.59</td>
<td>1.72</td>
<td>2.94</td>
<td>0.8-2.5 mg/L</td>
<td>22 (61%)</td>
</tr>
<tr>
<td>NH₄ mg/L</td>
<td>35</td>
<td>0.0</td>
<td>1.9</td>
<td>0.44</td>
<td>0.49</td>
<td>0.24</td>
<td>0.02-0.4 mg/L</td>
<td>14 (40%)</td>
</tr>
<tr>
<td>Turbidity NTU</td>
<td>33</td>
<td>0.3</td>
<td>190</td>
<td>12.03</td>
<td>33.05</td>
<td>1091.98</td>
<td>&lt;100 NTU</td>
<td>1 (3.33%)</td>
</tr>
<tr>
<td>pH</td>
<td>36</td>
<td>5.49</td>
<td>7.45</td>
<td>6.66</td>
<td>0.47</td>
<td>0.22</td>
<td>6.5-8.5</td>
<td>13 (36%)</td>
</tr>
<tr>
<td>TC cfu/100 mL</td>
<td>20</td>
<td>156.5</td>
<td>2419.6</td>
<td>1040.405</td>
<td>849.35</td>
<td>721402.63</td>
<td>&lt;1,000 cfu/100mL</td>
<td>4 (20%)</td>
</tr>
<tr>
<td>E. coli cfu/100 mL</td>
<td>20</td>
<td>1</td>
<td>2419.6</td>
<td>200.59</td>
<td>535.24</td>
<td>286481.45</td>
<td>&lt;126 cfu/100mL</td>
<td>3 (15%)</td>
</tr>
</tbody>
</table>

Source: Alford, 2021.
Table 3 illustrates correlation between the physicochemical parameters and samples for Seeley Creek.

Table 3 shows that DO was positive and statistically significant with NO$_3^-$ ($r = 0.601; p < 0.01$) and negatively significant with temperature ($r = -0.677; p < 0.01$). This explains that as DO increased, nitrate increased and as DO increased, temperature decreased and vice versa. Nitrate has a negative correlation with temperature ($r = -0.607; p < 0.01$). As nitrate increased, temperature decreased. Conductivity was negatively correlated with E. coli ($r = -0.536; p < 0.05$). pH was negatively correlated with turbidity ($r = -0.538; p < 0.01$).

<table>
<thead>
<tr>
<th></th>
<th>Flow</th>
<th>DO</th>
<th>NH$_4^+$</th>
<th>NO$_3^-$</th>
<th>Cond.</th>
<th>Temp.</th>
<th>pH.</th>
<th>Turb.</th>
<th>TC</th>
<th>E. coli</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>DO</td>
<td>.316</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NH$_4^+$</td>
<td>-.070</td>
<td>.182</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO$_3^-$</td>
<td>.308</td>
<td>.601**</td>
<td>.176</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cond.</td>
<td>-.396*</td>
<td>-.008</td>
<td>.198</td>
<td>.029</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Temp.</td>
<td>-.084</td>
<td>-.677**</td>
<td>-.113</td>
<td>-.607**</td>
<td>.033</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>pH.</td>
<td>.150</td>
<td>-.011</td>
<td>-.223</td>
<td>.233</td>
<td>.109</td>
<td>-.088</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Turb.</td>
<td>.091</td>
<td>.068</td>
<td>-.102</td>
<td>-.207</td>
<td>-.039</td>
<td>.141</td>
<td>.538*</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TC</td>
<td>-.615</td>
<td>-.229</td>
<td>-.188</td>
<td>-.247</td>
<td>.192</td>
<td>.135</td>
<td>-.363</td>
<td>.354</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>E. coli</td>
<td>.467</td>
<td>.221</td>
<td>-.166</td>
<td>-.082</td>
<td>-.536*</td>
<td>-.099</td>
<td>-.345</td>
<td>.360</td>
<td>.394</td>
<td>1</td>
</tr>
</tbody>
</table>

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

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

Little Bear Creek

Little Bear Creek (LBC) does not have any water quality parameter means exceeding the criteria stated in Table 1. However, there are a majority of
parameters that have individual samples that failed to meet their criteria and objectives including Conductivity, NO$_3^-$, NH$_4^+$, pH, TC and E. coli. Conductivity has a mean of (200.67 μS/cm) which does not exceed the CA State Water Board mean objective (<336 μS/cm), but 50% of the samples (i.e. 14 individual samples) did not meet the EPA range standards (150-500 μS/cm). These individual samples were lower than the EPA range standard and were measured during days of no precipitation events. NO$_3^-$ has a mean of (0.85 mg/L) which does not exceed the San Bernardino Mountain Hooks Creek objectives of (0.8-2.5 mg/L) but 50% of the samples (i.e. 14 individual samples) which do not meet the objectives. These individual NO$_3^-$ samples were lower than the San Bernardino Mountain Hooks Creek objectives and were all recorded during both dry and wet seasons. NH$_4^+$ has a mean of (0.40 mg/L) which does not exceed the EPA Aquatic Life Criteria (0.02 – 0.4 mg/L), but 35.71% of the samples (i.e. 10 individual samples) that exceeded the criteria. These individual samples were recorded during the wet sampling season. pH has a mean (6.59) that was within the CA State Water Board objective (6.5-8.5), but 46.43% of the samples (i.e. 13 samples) did not meet the objectives. The thirteen individual pH samples were all lower than the CA State Water Board objectives and were recorded during both the dry and wet sampling season. Total Coliform has a mean (510.95 cfu /1000 mL) which does not exceed the CA State Water Board objective, but 35.71% of the samples that exceeded the objectives, and these samples were recorded during the wet season. E. coli has a mean of (113.14 cfu/100mL) which meets
the EPA standards of (<126 cfu/100mL), but 31.25% of the samples were higher than the EPA standards and were recorded during the wet season. Total coliform (477577.85 cfu/100ML) has the greatest variance followed by E. coli (24304.7 cfu/100ML) and conductivity (20622.89 μS/cm).
Table 4. Descriptive Statistics of Water Quality Data for Little Bear Creek (LBC) Samples.

<table>
<thead>
<tr>
<th></th>
<th>N</th>
<th>Min</th>
<th>Max</th>
<th>Mean</th>
<th>Std. Deviation</th>
<th>Variance</th>
<th>Criteria/ Standards</th>
<th># and % Exceeding</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow m/s</td>
<td>29</td>
<td>0.12</td>
<td>1.39</td>
<td>0.522</td>
<td>0.36</td>
<td>0.13</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>DO mg/L</td>
<td>28</td>
<td>7.28</td>
<td>11.79</td>
<td>9.88</td>
<td>1.10</td>
<td>0.16</td>
<td>&gt;4 mg/L</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>Temp. C</td>
<td>29</td>
<td>3.1</td>
<td>15.2</td>
<td>7.03</td>
<td>2.98</td>
<td>8.89</td>
<td>&lt;25 C</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>Conductivity μS/cm</td>
<td>28</td>
<td>7.6</td>
<td>846.1</td>
<td>200.67</td>
<td>143.60</td>
<td>20622.89</td>
<td>150-500 Range &lt;336 μS/cm (mean)</td>
<td>14 (50%)</td>
</tr>
<tr>
<td>NO₃ mg/L</td>
<td>28</td>
<td>0.0</td>
<td>2.4</td>
<td>0.85</td>
<td>0.64</td>
<td>0.42</td>
<td>0.8-2.5 mg/L</td>
<td>14 (50%)</td>
</tr>
<tr>
<td>NH₄ mg/L</td>
<td>28</td>
<td>0.0</td>
<td>1.8</td>
<td>0.40</td>
<td>0.48</td>
<td>0.23</td>
<td>0.02-0.4 mg/L</td>
<td>10 (35.71%)</td>
</tr>
<tr>
<td>Turbidity NTU</td>
<td>28</td>
<td>0.5</td>
<td>33.7</td>
<td>10.82</td>
<td>8.58</td>
<td>73.62</td>
<td>&lt;100 NTU</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>pH</td>
<td>28</td>
<td>5.84</td>
<td>7.62</td>
<td>6.59</td>
<td>0.41</td>
<td>0.16</td>
<td>6.5-8.5</td>
<td>13 (46.42%)</td>
</tr>
<tr>
<td>TC cfu/100 mL</td>
<td>14</td>
<td>65.7</td>
<td>2419.6</td>
<td>510.95</td>
<td>691.07</td>
<td>477577.85</td>
<td>&lt;1,000 cfu/100mL</td>
<td>5 (35.71%)</td>
</tr>
<tr>
<td>E. coli cfu/100 mL</td>
<td>16</td>
<td>6.3</td>
<td>579.4</td>
<td>113.14</td>
<td>155.89</td>
<td>24304.74</td>
<td>&lt;126 cfu/100mL</td>
<td>5 (31.25%)</td>
</tr>
</tbody>
</table>

Source: Alford, 2021.
Table 5 illustrates correlation between the physicochemical parameters and samples for LBC. Table 5 shows that DO is negatively correlated with temperature ($r = -0.759; p < 0.01$). This explains that as DO increased, temperature decreased and as DO decreased, temperature increased. Nitrate is negatively correlated with temperature ($r = -0.400; p < 0.05$) indicating that as nitrate increased, temperature decreased. Total coliform was positively correlated to E. coli ($r = 0.622; p < 0.05$), which indicates a strong association and a 95% confidence level. This means that as total coliform increases, E. coli increases too.

Table 5. Covariance Correlations Matrix for Little Bear Creek.

<table>
<thead>
<tr>
<th></th>
<th>Flow</th>
<th>DO</th>
<th>NH$_4^+$</th>
<th>NO$_3^-$</th>
<th>Cond.</th>
<th>Temp.</th>
<th>pH..</th>
<th>Turb.</th>
<th>TC</th>
<th>E. coli</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>DO</td>
<td>.288</td>
<td>.261</td>
<td>.296</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>NH$_4^+$</td>
<td>.53</td>
<td>.249</td>
<td>.336</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>NO$_3^-$</td>
<td>.028</td>
<td>.168</td>
<td>.166</td>
<td>-.154</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Cond.</td>
<td>.028</td>
<td>.168</td>
<td>.166</td>
<td>-.154</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Temp.</td>
<td>-.137</td>
<td>-.759*</td>
<td>.237</td>
<td>-.400</td>
<td>.113</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>pH..</td>
<td>.263</td>
<td>.263</td>
<td>.032</td>
<td>.114</td>
<td>.079</td>
<td>.062</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Turb.</td>
<td>.155</td>
<td>.176</td>
<td>-.130</td>
<td>.236</td>
<td>-.043</td>
<td>-.067</td>
<td>.070</td>
<td>1</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>TC</td>
<td>.094</td>
<td>.421</td>
<td>-.301</td>
<td>.149</td>
<td>-.486</td>
<td>.349</td>
<td>-.263</td>
<td>.103</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>E. coli</td>
<td>.066</td>
<td>.091</td>
<td>.143</td>
<td>.139</td>
<td>-.213</td>
<td>.219</td>
<td>.272</td>
<td>.056</td>
<td>.622*</td>
<td>1</td>
</tr>
</tbody>
</table>

** Correlation is significant at the 0.01 level (2-tailed).
* Correlation is significant at the 0.05 level (2-tailed).
**Burnt Mill Creek**

Burnt Mill Creek (BMC) (Table 6) does not have any water quality parameter means exceeding the criteria stated in Table 1. However, there are some parameters that have individual samples that failed to meet their criteria and objectives. These are Conductivity, NO$_3^-$, NH$_4^+ $, pH, TC and E Coli. Conductivity has a mean of (221.61 μS/cm) which does not exceed the CA State Water Board mean objective (<336 μS/cm), but 6.66% of samples (i.e. 2 individual samples) did not meet the EPA range standards (150-500 μS/cm). These two individual conductivity samples were lower than the CA State Water Board objective and were recorded on dry sampling days. NO$_3^-$ has a mean of (1.13 mg/L) which does not exceed the San Bernardino Mountain Hooks Creek objectives of (0.8-2.5 mg/L) but has 33.33% of samples (i.e. 10 samples) which did not meet objectives. Nine low individual NO$_3^-$ samples were recorded during both the dry and wet season and one high individual sample was recorded during the wet season. NH$_4^+$ has a mean of (0.30 mg/L) which does not exceed the EPA Aquatic Life Criteria (0.02 – 0.4 mg/L), but 43.33% of samples (i.e. 13 individual samples) that did not meet the criteria. These individual samples were recorded on dry sampling days. pH has a mean (6.66) that was within the CA State Water Board objective (6.5-8.5), but has 40% of samples (i.e. 12 samples) that did not meet the objectives. The twelve individual pH samples were all lower than the CA State Water Board objectives and were recorded during the wet season which was November 2019 - April 2020. The bacteria Total Coliform has a mean
(276.82 cfu /1000 mL) which does not exceed the CA State Water Board objective, but 6.25% of the samples (i.e. 1 individual sample) that exceeded the objective. The individual Total coliform sample was recorded on a wet sampling day which was in March. E. coli has a mean of (39.31 cfu/100mL) which meets the EPA standards of (<126 cfu/100mL), but 6.25% of the samples (i.e. 1 sample) exceeded the EPA standards. The individual E. coli sample was recorded on the same wet sampling day as Total coliform. Total coliform (103478.70 cfu/1000mL) has the highest variance followed by E. coli (8398.30 cfu/1000mL) and conductivity (5182.83 μS/cm).
Table 6. Descriptive Statistics of Water Quality Data for Burnt Mill Creek (BMC) Samples.

<table>
<thead>
<tr>
<th></th>
<th>N</th>
<th>Min</th>
<th>Max</th>
<th>Mean</th>
<th>Std. Deviation</th>
<th>Variance</th>
<th>Criteria/ Standards</th>
<th># and % Exceeding</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow m/s</td>
<td>24</td>
<td>0.26</td>
<td>1.58</td>
<td>0.81</td>
<td>0.36</td>
<td>0.13</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>DO mg/L</td>
<td>30</td>
<td>8.24</td>
<td>11.98</td>
<td>10.02</td>
<td>0.79</td>
<td>0.63</td>
<td>&gt;4 mg/L</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>Temp. C</td>
<td>30</td>
<td>2.1</td>
<td>15.7</td>
<td>6.88</td>
<td>3.09</td>
<td>9.55</td>
<td>&lt;25 C</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>Conductivity μS/cm</td>
<td>30</td>
<td>101.6</td>
<td>385</td>
<td>221.61</td>
<td>71.99</td>
<td>5182.83</td>
<td>150-500 Range &lt;336 μS/cm (mean)</td>
<td>2 (6.66%)</td>
</tr>
<tr>
<td>NO₃ mg/L</td>
<td>30</td>
<td>0.1</td>
<td>3.2</td>
<td>1.13</td>
<td>0.73</td>
<td>0.54</td>
<td>0.8-2.5 mg/L</td>
<td>10 (33.33%)</td>
</tr>
<tr>
<td>NH₄ mg/L</td>
<td>30</td>
<td>0.0</td>
<td>1.1</td>
<td>0.30</td>
<td>0.29</td>
<td>0.08</td>
<td>0.02-0.4 mg/L</td>
<td>13 (43.33%)</td>
</tr>
<tr>
<td>Turbidity NTU</td>
<td>30</td>
<td>1.4</td>
<td>99.2</td>
<td>12.93</td>
<td>17.81</td>
<td>317.35</td>
<td>&lt;100 NTU</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>pH</td>
<td>30</td>
<td>5.87</td>
<td>7.50</td>
<td>6.66</td>
<td>0.42</td>
<td>0.17</td>
<td>6.5-8.5</td>
<td>12 (40%)</td>
</tr>
<tr>
<td>TC cfu/100 mL</td>
<td>16</td>
<td>36.9</td>
<td>920.8</td>
<td>276.82</td>
<td>321.68</td>
<td>103478.70</td>
<td>&lt;1,000 cfu/100mL</td>
<td>1 (6.25%)</td>
</tr>
<tr>
<td>E. coli cfu/100 mL</td>
<td>16</td>
<td>0</td>
<td>365.4</td>
<td>39.11</td>
<td>91.64</td>
<td>8398.30</td>
<td>&lt;126 cfu/100mL</td>
<td>1 (6.25%)</td>
</tr>
</tbody>
</table>

Source: Alford, 2021.
Table 7 shows that flow has a strong positive correlation with conductivity \((r = 0.653; p < 0.01)\) and a strong positive correlation with temperature \((r = 0.618; p < 0.01)\). This means when there is an increase in flow rate, conductivity, and temperature both increases, and when there is a decrease in flow, conductivity and temperature decrease as well. DO has a negative correlation with \(\text{NH}_4^+\) \((r = -0.427; p < 0.05)\) and a negative correlation with temperature \((r = -0.633; p < 0.01)\). This explains that as DO increases, ammonium and temperature decrease. DO has a positive correlation with \(E.\ coli\) \((r = 0.625; p < 0.05)\) which is a strong association, indicating that as DO increases, \(E.\ coli\) increases.

Table 7. Covariance Correlations Matrix for Burnt Mill Creek.

<table>
<thead>
<tr>
<th></th>
<th>Flow</th>
<th>DO</th>
<th>(\text{NH}_4^+)</th>
<th>(\text{NO}_3^-)</th>
<th>Cond.</th>
<th>Temp.</th>
<th>pH.</th>
<th>Turb.</th>
<th>TC</th>
<th>(E.\ coli)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>DO</td>
<td>.175</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(\text{NH}_4^+)</td>
<td>-.356</td>
<td>-.427</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(\text{NO}_3^-)</td>
<td>-.086</td>
<td>-.084</td>
<td>.018</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cond.</td>
<td>.653</td>
<td>.007</td>
<td>-.314</td>
<td>-.080</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Temp.</td>
<td>.618</td>
<td>-.633</td>
<td>.336</td>
<td>-.107</td>
<td>.192</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>pH.</td>
<td>-.143</td>
<td>-.177</td>
<td>.044</td>
<td>.025</td>
<td>-.146</td>
<td>-.107</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Turb.</td>
<td>-.303</td>
<td>-.415</td>
<td>.131</td>
<td>.215</td>
<td>-.129</td>
<td>.337</td>
<td>.057</td>
<td>.322</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>TC</td>
<td>-.336</td>
<td>.625</td>
<td>-.469</td>
<td>.287</td>
<td>.114</td>
<td>-.145</td>
<td>-.143</td>
<td>.279</td>
<td>.275</td>
<td>1</td>
</tr>
</tbody>
</table>

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

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

Orchard Creek (OC) (Table 8) does not have any water quality parameter means exceeding the criteria stated in Table 1. However, there are some parameters that have individual samples that failed to meet their criteria and objectives. These are Conductivity, NO$_3^-$, NH$_4^+$, pH, TC and E. coli. Conductivity has a mean of (208.08 μS/cm) which does not exceed the CA State Water Board mean objective (<336 μS/cm), but four individual samples did not meet the EPA range standards (150-500 μS/cm). These four individual conductivity samples were all lower than the EPA range standards and were taken on dry sampling days in the month of October. NO$_3^-$ has a mean of (1.61 mg/L) which does not exceed the San Bernardino Mountain Hooks Creek objectives of (0.8-2.5 mg/L) but has ten individual samples that did not meet the objectives. These ten individual samples were recorded during the wet season. NH$_4^+$ has a mean of (0.37 mg/L) which does not exceed the EPA Aquatic Life Criteria (0.02 – 0.4 mg/L), but four individual samples that did not meet the criteria. These four individual samples were recorded during dry sampling periods. pH has a mean (6.69) that was within the CA State Water Board objective (6.5-8.5) but has twelve individual samples that did not meet the objectives. The twelve individual pH samples were lower than the CA State Water Board objective and were recorded during the wet sampling period. The bacteria Total Coliform has a mean (456.55 cfu /1000 mL) which does not exceed the CA State Water Board objective of (<1,000 cfu/100mL), but two samples that exceeded the objectives.
The two samples were recorded during the wet sampling period. E. coli has a mean of (33.41 cfu/100mL) which meets the EPA standards of (<126 cfu/100mL), but one sample that did not meet the EPA standards which was recorded during the wet season. Total coliform (514993.33 cfu/100mL) has the greatest variance followed by conductivity (3678.63 μS/cm) and E. coli (3443.91 cfu/100mL).
Table 8. Descriptive Statistics of Water Quality Data for Orchard Creek (OC) Samples.

<table>
<thead>
<tr>
<th>Descriptive Statistics OC</th>
<th>N</th>
<th>Min</th>
<th>Max</th>
<th>Mean</th>
<th>Std. Deviation</th>
<th>Variance</th>
<th>Criteria/ Standards</th>
<th># and % Exceeding</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow m/s</td>
<td>29</td>
<td>0.10</td>
<td>1.34</td>
<td>0.46</td>
<td>0.29</td>
<td>0.08</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>DO mg/L</td>
<td>29</td>
<td>7.88</td>
<td>11.48</td>
<td>10.17</td>
<td>0.93</td>
<td>0.87</td>
<td>&gt;4 mg/L</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>Temp. C</td>
<td>29</td>
<td>4.3</td>
<td>14.9</td>
<td>7.55</td>
<td>2.57</td>
<td>6.64</td>
<td>&lt;25 C</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>Conductivity μS/cm</td>
<td>29</td>
<td>108</td>
<td>358</td>
<td>208.08</td>
<td>60.65</td>
<td>3678.63</td>
<td>150-500 Range &lt;336 μS/cm (mean)</td>
<td>4 (13.79%)</td>
</tr>
<tr>
<td>NO₃ mg/L</td>
<td>29</td>
<td>0.1</td>
<td>4</td>
<td>1.61</td>
<td>1.03</td>
<td>1.06</td>
<td>0.8-2.5 mg/L</td>
<td>10 (34.48%)</td>
</tr>
<tr>
<td>NH₄ mg/L</td>
<td>29</td>
<td>.00</td>
<td>1.6</td>
<td>0.37</td>
<td>0.40</td>
<td>0.16</td>
<td>0.02-0.4 mg/L</td>
<td>4 (13.79%)</td>
</tr>
<tr>
<td>Turbidity NTU</td>
<td>29</td>
<td>0.4</td>
<td>49.7</td>
<td>11.68</td>
<td>11.34</td>
<td>128.71</td>
<td>&lt;100 NTU</td>
<td>0 (0%)</td>
</tr>
<tr>
<td>pH</td>
<td>29</td>
<td>6.03</td>
<td>7.66</td>
<td>6.69</td>
<td>0.47</td>
<td>0.22</td>
<td>6.5-8.5</td>
<td>12 (41.37%)</td>
</tr>
<tr>
<td>TC cfu/100 mL</td>
<td>20</td>
<td>35</td>
<td>2419.6</td>
<td>456.55</td>
<td>717.63</td>
<td>514993.33</td>
<td>&lt;1,000 cfu/100mL</td>
<td>2 (12.15%)</td>
</tr>
<tr>
<td>E. coli cfu/100 mL</td>
<td>16</td>
<td>1</td>
<td>235.9</td>
<td>33.41</td>
<td>58.68</td>
<td>3443.91</td>
<td>&lt;126 cfu/100mL</td>
<td>1 (6.25%)</td>
</tr>
</tbody>
</table>

Source: Alford, 2021
Table 9 shows that flow has a strong positive correlation with conductivity ($r = 0.565; p < 0.01$), indicating that as flow increases, conductivity increases. DO has a negative correlation with temperature ($r = -0.800; p < 0.01$), explaining that as DO increases, temperature decreases. Ammonium has a positive correlation with nitrate ($r = 0.413; p < 0.05$), therefore as ammonium increases, nitrate increases. pH also has a negative correlation with turbidity ($r = -0.464; p < 0.05$). With an increase in pH, there is a decrease in turbidity.

### Table 9. Covariance Correlations Matrix for Orchard Creek.

<table>
<thead>
<tr>
<th></th>
<th>Flow</th>
<th>DO</th>
<th>NH$_4^+$</th>
<th>NO$_3^-$</th>
<th>Cond.</th>
<th>Temp.</th>
<th>pH</th>
<th>Turb.</th>
<th>TC</th>
<th>E. coli</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flow</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>DO</td>
<td>0.246</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NH$_4^+$</td>
<td>-0.362</td>
<td>0.290</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>NO$_3^-$</td>
<td>-0.031</td>
<td>-0.003</td>
<td>-0.413</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cond.</td>
<td>0.565**</td>
<td>-0.085</td>
<td>-0.310</td>
<td>-0.132</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Temp.</td>
<td>-0.201</td>
<td>-0.800*</td>
<td>0.095</td>
<td>-0.201</td>
<td>0.228</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>pH</td>
<td>0.192</td>
<td>-0.079</td>
<td>-0.087</td>
<td>0.072</td>
<td>0.006</td>
<td>-0.175</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Turb.</td>
<td>-0.266</td>
<td>0.023</td>
<td>0.112</td>
<td>0.146</td>
<td>-0.354</td>
<td>0.130</td>
<td>-0.464*</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>TC</td>
<td>-0.467</td>
<td>-0.433</td>
<td>0.140</td>
<td>0.185</td>
<td>-0.299</td>
<td>0.339</td>
<td>-0.078</td>
<td>0.180</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>E. coli</td>
<td>0.075</td>
<td>-0.234</td>
<td>-0.315</td>
<td>0.032</td>
<td>0.176</td>
<td>0.170</td>
<td>-0.374</td>
<td>0.082</td>
<td>0.351</td>
<td>1</td>
</tr>
</tbody>
</table>

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

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

### Seasonal Trends in Water Quality Parameters

When considering seasonal variations (i.e. wet vs. dry), it was observed that in the dry season, the mean NH$_4^+$, conductivity, pH and TC concentrations...
were greater compared to the wet season. NO$_3^-$ and E. coli concentrations were greater in the wet season compared to the dry season. DO and temperature concentrations stayed consistent during both the wet and dry season across all watersheds. Pearson’s correlation showed that DO was negatively correlated with temperature in SC, BMC and OC. This means that as DO increases, temperature reduces. DO was also positively correlated with NO$_3^-$ in SC and E. coli in BMC. With an increase in DO, there was an increase in NO$_3^-$ and E. coli. In the wet season when temperatures were lower there was a higher concentration of DO in the stream and in the dry season when temperatures were higher there was a lower concentration of DO. Data also showed that TC, E. coli, NH$_4^+$, NO$_3^-$, pH and turbidity were all positively correlated to each other.
Figure 6. Dry Season Percentage Exceedance for SC, LBC, BMC and OC: DO, Temperature, Conductivity, \( \text{NO}_3^- \), \( \text{NH}_4^+ \), Turbidity, pH, TC and E. coli.
Environmental Protection Agency Watershed Tool Scenario Settings

The application of the US Environmental Protection Agencies (EPA) Modeling My Watershed Tool was applied to determine watershed landscape and precipitation influences on surface water quality for Total Suspended Solids (TSS), Total Nitrogen (TN) and Total Phosphorus (TP). The model also displayed results for runoff, infiltration, and evapotranspiration. Different scenarios were developed by combining the different land use covers and climate periods. In using the model, artificial scenarios were created that focus on changes to the LULC that are the primary drainage areas that create the headwaters of the
watershed. The daily flows were simulated by changing the land use under specific climate conditions, and vice versa. To create scenarios, total size of each watershed was obtained and 25% was calculated with the headwaters as a starting point with possible changes that could occur. Four contrasting land use scenarios were constructed based on current land-use in the San Bernardino Forest area. The scenarios are as follows:

Scenario 1: Conversion of current conditions to developed low land cover.
Scenario 2: Developed medium intensity land cover.
Scenario 3: Developed high intensity land cover.
Scenario 4: Forested land cover.

The scenarios were created using the lowest and highest points of precipitation events that occurred between April 2019 through April 2020 in the watershed with four different land use/land cover (LULC). Higher frequencies of precipitation were noted to be 0.81 inches of precipitation for LBC, OC, and BMC sites and 1.81 inches for SC. Lower frequencies of precipitation were noted to be (0.25 – 0.28 inches) across all four sites. The major changes in land use across all watersheds were seen in developed high intensity and forested land cover. Low precipitation scenarios showed more percentage change than high precipitation scenarios. The results show the effects of climate change and also indicate that land use is the main agent of hydrological variations in the catchment. The results also reveal the irregularity in rainfall and landcover changes as they relate
to the topography, land cover and soil type in the watershed, describes the hydrology of a headwater watershed.

**Scenario 1**

Scenario 1 estimates the impacts from converting 25% of current land condition covers to developed low intensity land cover in the areas draining to create the headwaters across all watersheds.

MMW responses for modeled parameters showed OC with the highest percentage runoff by 116% followed by LBC with 110% increase from current conditions in the season of low precipitation (0.28 inches). OC also had the highest percentage runoff with an increase of 157% followed by LBC with 133% increase from current conditions in the season of high precipitation (0.81 - 1.81 inches). Infiltration was seen to have increased by 20% from current conditions in OC followed by SC with an increase of 19.6% in the season of low precipitation. Infiltration decreased from current conditions across all watersheds with SC taking a lead by 4.8% followed by LBC with 2.7% decrease in the season of high precipitation. LBC had the highest decrease in evapotranspiration from current conditions by 15% followed by OC with a decrease of 14% in both seasons (Figures 12a - 15b). OC was the site with the highest percentage increase (196%) in TSS followed by SC with 139% in the season of low precipitation. OC also had the highest percentage increase (147%) in TSS in the season of high precipitation followed by LBC with 124% (Figure 8a). TN loadings were seen to have spiked by 200% from current conditions in OC watershed in the dry season.
and 133% in the wet season. SC had the second highest TN increase with 140% in the dry season. LBC had the second highest TN increase with 104% in the wet season (Figure 8b). All four watersheds had a 100% increase from current conditions in TP loadings in the dry season and OC had the highest increase by 133% in the wet season (Figure 8c).

Figure 8a. Developed low land cover for BMC, OC, LBC, SC: Percent increase in TSS.
Figure 8b. Developed low land cover for BMC, OC, LBC, SC: Percent increase in TN.

Figure 8c. Developed low land cover for BMC, OC, LBC, SC: Percent increase in TP.
Scenario 2

Scenario 2 estimates watershed response that results from increased developed land covers from developed low intensity to developed medium intensity in the areas of the watershed draining and forming headwaters. MMW responses for modeled parameters showed OC with the highest runoff increase of 206% percent from current conditions in the season of low precipitation (0.28 inches) followed by LBC with 201%. OC also had the highest runoff increase of 294% from current conditions in the season of high precipitation (0.81 - 1.81 inches) followed by SC with 265% and LBC with 254%. In the season of low precipitation, SC had the highest increase in infiltration by 21% followed by LBC and OC with 18% from current conditions. SC also had the highest percentage decrease in infiltration in the season of wet precipitation by 10% followed by LBC with 7.5% decrease. Evapotranspiration was seen to have decreased by 21% in SC in both the season of low and wet precipitation followed by LBC with 20% in both seasons (Figures 12a - 15b). OC had the highest TSS loadings in the dry season with 391% followed by SC with 361%. OC also had the highest TSS loadings in the wet season with 276% followed by LBC with 241% (Figure 9a). OC had the highest TN loadings in the dry season with 525% followed by SC with 460% and 338% in the wet season followed by LBC with 275% (Figures 9b). In the dry season, all four watersheds had a 200% increase in TP loadings. OC was the watershed with the highest TP loadings of 233% in the wet season.
followed by LBC with 175%. OC was also the only site that experienced an increase in TP loadings in the wet season from the dry season (Figure 9c).

Figure 9a. Developed medium land cover for BMC, OC, LBC, SC: Percent increase in TSS.
Figure 9b. Developed medium land cover for BMC, OC, LBC, SC: Percent increase in TN.

Figure 9c. Developed medium land cover for BMC, OC, LBC, SC: Percent increase in TP.
Scenario 3

Scenario 3 estimates watershed response that results from increased developed land covers of high intensity in the areas of the watershed draining and forming the headwaters. MMW responses for modeled parameters showed that OC had the highest percentage increase in runoff with 468% in the season of high precipitation (0.81 - 1.81 inches) followed by SC with 434%. OC had the highest percentage increase in runoff in the season of low precipitation (0.28 inches) with 312% followed by LBC with 297% from current conditions. SC had the highest percentage decrease in infiltration by 17% followed by LBC with 13% decrease in the season of high precipitation. SC also had the highest percentage increase in infiltration in the season of low precipitation by 12% from current conditions. Evapotranspiration was seen to have decreased by an average of 22% across all watersheds in both the wet and dry seasons (Figures 12a - 15b). MMW tool recorded that OC watershed had the highest percentage of TSS and TN increase in both the wet and dry season with 750% increase in the dry season and 512% in the wet season. SC had the second highest percentage increase of TSS and TN with 712% in the dry season and the lowest with 242% in the wet season. LBC had the second highest increase in TSS and TN in the wet season with 432%. (Figures 10a and 10b). Figure 10c explains that OC was the watershed with the highest percentage of TP loadings in the wet season with 333% increase and was the only site that had higher TP loadings from the dry season. BMC, LBC and SC had a reduced percentage of TP loadings in the wet
season compared to the dry season. All four watersheds maintained the same level of TP loadings in the dry season.

Figure 10a. Developed high land cover for BMC, OC, LBC, SC: Percent increase in TSS.
Figure 10b. Developed high land cover for BMC, OC, LBC, SC: Percent increase in TN.

Figure 10c. Developed high land cover for BMC, OC, LBC, SC: Percent increase in TP.
Scenario 4

Scenario 4 estimates watershed response that results from an added forested cover to the current conditions. Model My Watershed responses for modeled parameters showed SC with the highest runoff decrease of 30% in the wet season followed by OC with 21% from current land conditions. OC had the highest decrease in the dry season with 20% followed by LBC with 15%. There were no changes in infiltration levels from current conditions in LBC and OC but there was an increase by an average of 1.95% from current conditions in BMC and SC in the season of low precipitation. In the season of high precipitation, infiltration increased by 1.51% across all watersheds. Evapotranspiration was seen to have increased by an average of 0.6% from current conditions in LBC and SC during both wet and dry seasons. There were no recorded changes from current conditions in OC and BMC (Figures 12a -15b). SC was the site with the highest percentage decrease in TSS loadings in both the dry and wet seasons with 25.95% and 20.13% respectively. OC had the second highest decrease in TSS in both the dry and wet seasons with 17.85% and 19.21% (Figure 11a). SC also had the highest percentage decrease in TN loadings in both the dry and wet seasons with 20% and 19.68% respectively. OC had the second highest decrease in TN in the wet season with 19% and no decrease in the dry season. BMC and LBC had the same percentage decrease in TN in the dry season with 14 (Figure 11b). All four watersheds had no changes in TP loadings from current
conditions in the dry season and OC was the site with the highest reduction in TP loading in the wet season with 33% followed by LBC with 25% (Figure 11c).

Figure 11a. Forested land cover for BMC, OC, LBC, SC: Percent decrease in TSS.
Figure 11b. Forested land cover for BMC, OC, LBC, SC: Percent decrease in TN.

Figure 11c. Forested land cover for BMC, OC, LBC, SC: Percent decrease in TP.
Seasonal Variations

The MMW tool shows that the general trend across all watersheds is that in the dry season, runoff increased, infiltration rates increased, evapotranspiration decreased as impervious land use covers were modeled. In the wet season, runoff rates increased greatly by over 50% compared to the dry season, infiltration rates decreased, and evapotranspiration decreased as impervious land use covers were modeled. Increase in evapotranspiration in the dry season can be linked to warmer temperatures in the dry season. Evapotranspiration is the loss of soil through both evaporation and transpiration. Forested land cover was different with both seasons having a reduced amount of runoff across all watersheds and maintaining the same level of evapotranspiration and infiltration with current conditions.
Figure 12a. Dry Season for SC: Runoff, Evapotranspiration, Infiltration.

Figure 12b. Wet Season for SC: Runoff, Evapotranspiration, Infiltration.
Figure 13a. Dry Season for LBC: Runoff, Evapotranspiration, Infiltration.

Figure 13b. Wet Season for LBC: Runoff, Evapotranspiration, Infiltration.
Figure 14a. Dry Season for BMC: Runoff, Evapotranspiration, Infiltration.

Figure 14b. Wet Season for BMC: Runoff, Evapotranspiration, Infiltration.
Figure 15a. Dry Season for OC: Runoff, Evapotranspiration, Infiltration.

Figure 15b. Wet Season for OC: Runoff, Evapotranspiration, Infiltration.
CHAPTER FOUR
DISCUSSION

Seasonal Trends
While comparing the trends observed using both the in-situ data and the modeling tool, it is observed that there is an adverse impact of precipitation events on water quality in the study area. The observation from the model showed a higher percentage change of water quality parameters observed in the season of low precipitation than in the season of high precipitation events across all watersheds. The in-situ data showed significant changes in water quality parameters in the wet versus the dry season across all watersheds. The results show the adverse impact of climate change (changes in precipitation and runoff, drought, and warming temperatures) and indicate that land use and soil types are also primary drivers of hydrological variations in the catchment.

Seasonal trends during the study period were relatively variable. NO$_3^-$, NH$_4^+$, accumulations did not meet the San Bernardino Mountain Hooks Creek objectives (0.8-2.5 mg/L) and EPA Aquatic Life criteria (0.02 - 0.4 mg/L) during both wet and dry periods across all sites. Individual Conductivity tests during both the dry and wet seasons in Seeley Creek, BMC and OC did not meet regulatory standards. In LBC, 15 out of 28 individual conductivity tests had the highest number of individual exceedances that did not meet the EPA range standards (150-500 $\mu$S/cm) with one test exceeding the range with a value of (846 $\mu$S/cm)
on 12/10/2019. There was no precipitation on that sampling day but precipitation was recorded 48 hours before the sampling event. Although turbidity remained consistent across all sites, it met a spike of (190 NTU) at Seeley Creek on 08/19/2019 which significantly exceeds EPA criteria (<100 NTU) and a spike of (99.2 NTU) at BMC on 9/9/2019. This still falls within standard but is relatively high compared to values recorded on other sampling days. There was no known precipitation event 24 hours before the sampling day. The pH levels were low across all sites in both the wet and dry periods.

**Water Quality Parameters**

In this study, four sites were investigated to assess in situ conditions and to determine the changes that would occur if the landscape is altered. To understand the physicochemical components of the streams under observation, in situ data were analyzed to determine if the stream was within the federal criteria and state objectives. Various parameters were observed from a database and compared to the U.S. Environmental Protection Agency Recreational Water Quality and Aquatic Life Criteria, California State Water Resources Control Board, South Lahontan Region Objectives, and San Bernardino Mountains Hooks Creek Objectives. This assists with understanding how the model can be an effective tool for watershed analysis related to how changes in land types may influence water quality. Various parameters had a mean that exceeded the criteria and objectives (such as, NO₃, NH₄, TC, and others) while other parameters had individual samples that exceeded the criteria and objectives.
(such as, Conductivity, pH, E. Coli, and others). It was also observed that across all sites, there were high concentrations of NO$_3^-$ This is a matter of serious concern because excessive NO$_3^-$ inputs in water ways leads to eutrophication which causes an exhaustion of DO thereby posing as a risk to human and aquatic health (Mallin and Cahoon, 2003; Fink and Mitsch, 2004; Smith et al., 2013).

Modelling

**Total Suspended Solids**

One of the main concerns of this study is erosion which manifests as the process of sedimentation in the form of Total Suspended Solids. The modeling suggests that potential impairment in water quality is caused by sedimentation which is as a result of transportation and adsorption of particles such as heavy metals, phosphorus, pesticide, nutrients and hydrocarbons onto particulate surfaces. Land use/land cover scenarios were created using the Model My Watershed tool to determine how changes in landscape affect water quality and erosion. The scenarios created were 25% at the headwaters of each watershed. The number and extent of impervious surfaces in an area is one of the most important factors in determining the negative effects of development on water quality (Moglen, 2009). However, studies reveal that water quality decline starts when there is 10% to 20% impervious surface in the watershed area (Holland et al., 2004; Schueler, 2009). It was observed that there was a relatively high percentage increase in stormwater pollutants and runoff across all watersheds.
using 25% of impervious surfaces (developed low intensity, developed medium intensity and developed high intensity). With 25% of forested land cover, there was a significant decrease in stormwater pollutants and runoff across all watersheds.

The Model My Watershed tool revealed an increase in TSS across all watersheds during both low rainfall and high rainfall. Although the percentage increase in TSS was higher in the season of low rainfall compared to the season of high precipitation. This is a result of the first flush phenomenon. The season of low rainfall was recorded during the periods (April – October 2019) and the season of high rainfall was recorded during the period (November – April 2020). First flush phenomenon is the initial volume of runoff in urban catchments during rainfall events containing the highest pollutants levels. Vegetative cover also plays an important role in this trend. In the season of low precipitation events, soil particles can break up causing them to move to waterways (Bach et al., 2020). In this case, the amount of soil exposure due to vegetative density plays an important role. The soil type of the area and steep slopes can cause erosion. As a result of lack of precipitation, the soils become dry causing them to be more erosive when water first comes in contact with it. The physical raindrop can break up the soil particles causing more to become loose and thus exposed to erosion. In higher precipitation events the rainfall may initially cause a spike in TSS but over the rainfall event it will flush those sediments out (Mamun et al., 2020).
Scenario 1 with a possibility of developed low land cover shows an increase in runoff, infiltration and TSS across all sites. This explains that when rainfall exceeds a soil's infiltration capacity, there is an increase in saturation therefore causing an increase in runoff which in turn produces high TSS rates. With the possibility of developed medium intensity in scenario 2, there is an increase in runoff and TSS and a decrease in infiltration in the period of low rainfall across all sites. This explains a higher percentage of impervious surface in the watershed therefore causing a decrease in infiltration and an increase in runoff and TSS which can initiate erosion in the area, with losses of nutrients. Scenario 3 with the possibility of developed high land cover, there is an increase in runoff and TSS and a decrease in infiltration across all sites. Scenario 4 being a possible forested land cover, there is a decrease in both runoff and TSS and an increase in infiltration.

This tells us that impervious surfaces are directly related to stormwater nutrient loadings. The model records OC with the highest percentage increase in runoff and TSS loadings followed by SC with the change in land covers. SC was recorded as the site with the highest decrease in infiltration when changing from developed low to developed high intensity. Recall SC has 27.25% developed open space, drains a residential and highly compact area and is characterized by sewer systems. This means that SC has a large amount of impervious surfaces compared to the other sites. This greatly explains the increase in all water quality parameters as indicated by the model.
Developed areas create surfaces that increase stormwater runoff rates that collect and move pollution inputs to nearby water features, while reducing soil infiltration opportunities that support groundwater recharge (Frazer, 2005). Sedimentation in the form of the form of total suspended solids (TSS) causes siltation which decreases the stream capacity of the river; it causes changes in coastline and erosion; it also affects the life span functionality and fertility of the river over time. Accumulation of sediments in rivers are as a result of erosion caused by rainfall. Past literature (Bilotta and Brazier, 2008; Ibrahim and Eko, 2018) show that precipitation events have a high impact on sedimentation increase which negatively affects water quality. TSS are solid particles that are bigger than 2 microns suspended in water which contains biotic and abiotic constituents. In addition, excessive amount of sediments gives room for development of turbidity in water bodies which prevents sunlight from reaching into the waters. When turbidity levels surpass 100 mg/l, they cause death to aquatic species and also affect the water productivity.

The results from the MMW tool shows that water quality parameters displayed remarkable disparities between the forest-dominated and urban-dominated sites LULC scenarios. The percentage of urban land had a strong positive correlation with total suspended solids and total nitrogen concentrations primarily driven by the presence and amount of impervious surfaces across the watershed.

BMP selection in the successful elimination of sediments from urban stormwater runoff is dependent on characteristics of the site and sediments in
urban runoff, particle size and the unit treatment processes present (flow velocity and detention times in the BMP) (Jones et al., 2012). Consideration of BMPs for removal of suspended solids include sedimentation and filtration. Porous pavements, wetland basins, infiltration trenches, detention basins, media filters, and retention ponds are all sedimentation and filtration BMPs that aid in the removal of TSS in urban runoff. Porous pavement or permeable paving system is an infiltration system in which stormwater runoff infiltrates into the ground through a stabilized pervious surface or a permeable layer of pavement. It is known to have a pollutant removal efficacy of 91% (USEPA, 1999; CDT 2004). This infiltration system redirects runoff via a permeable layer of asphalt into a stone reservoir underground and then gently enters into the subsoil (Field and Sullivan, 2003). Wetland basins capture and temporarily store high capacity of runoff for a long period of time while maximizing elimination of pollutants through wetland vegetation uptake, retention and settling (Guerrero et al., 2020). It is known to have a pollutant removal efficacy of 80% for TSS when used with other BMPs (Shamma and Zhu 2001; FHWA, 2003). Sand filters are a recommended BMP to manage first flush phenomenon and remove TSS as well with an efficacy of 80%. Grass swales are often used to manage erosion in open channels. Land surface is formed to direct stormwater through a stabilized grassed area (Sayre et al., 2006). For filtration- and infiltration-oriented BMPs, maintenance is imperative in order to avert sediment clogging (USEPA, 2004). BMPs have to be monitored at
the inlet and outlet to ensure they are being maintained properly and are remaining effective in mitigating pollution inputs to waterways.

**Nutrients**

Nutrients are needed for plant/species growth, reproduction and to survive. They are either an element or compound consumed by an organism to grow, repair itself or create energy. Phosphorous (P) and Nitrogen (N) are the primary nutrients that in excess impair water quality. Excessive amounts of nutrients lead to eutrophication in the ecosystem which leads to extensive growth of algae (Carpenter, 1998; Peters and Maybeck, 2009; Chislock et al, 2013). When these algae die, bacteria is needed for decomposition which in turn requires dissolved oxygen (DO), hence a high amount of DO is used and there is a reduction in DO. The presence of Phosphorus affects the salinity level in water which also decreases DO because DO levels are higher in freshwater than in salt water.

MMW tool revealed an increase in Total Nitrogen (TN) across all watersheds while moving from developed low land cover to developed high land cover during both low rainfall and high rainfall. Although the percentage increase in TN was higher in the season of low precipitation compared to the season of high precipitation. With the amount of impervious surfaces present, this increase could be as a result of human activities such as a waste water treatment plans, leakage from fertilized soil, landfills, animal feedlots, septic systems, or urban drainage. In situ data revealed that nitrate was the parameter with the highest
number of individual samples that did not meet regulatory standards across all watersheds. Nitrate is an organic form of nitrogen that is found as a result of human activities (such as urban drainage and refuse dumps) and also naturally occurring in the environment (such as pet slops, plant and animal decomposition). The average concentrations of nitrate in the wet season was higher than in the dry season. This is comparable to the research conclusions by Freguso-Lopez et al. (2020) in the El Fuerte river, southern Gulf of California, Mexico. Past literature reveals that during high precipitation events, urban areas with high impervious surfaces contribute the most to nonpoint nitrate contamination in waterways. This contamination is mostly as a result of presence of animal feedlots, septic systems, fertilizer runoff and wastewater treatments (Carpenter, 1998; Tong and Chen, 2002; Barakat et al., 2016).

MMW tool revealed an increase in TP loadings in the dry season and a decrease in TP loadings in the wet season in three watersheds except for Orchard creek (OC). The percentage accumulation of TP was found to be higher in the season of wet precipitation compared to the dry season. OC is characterized by 70% evergreen forest and 26% developed area with housing and a marina. There are also a ton of old houses in the area with old septic systems. This explains that the main sources of phosphorus in the waterways are the human wastes, leakage from old septic systems, phosphorus containing household detergents and some industrial wastes. Only little precipitation runoff contributes to P-loads in the water ways if combined sewer systems are applied.
Applying the forested land cover scenario in the MMW tool, there was a reduction in both TN and TP across all watersheds in the season of high precipitation. This means that when vegetated landscape is added to the watershed landscape, there is a decrease in nutrient loadings but when impervious surfaces are increased in the watershed, there is an increase in nutrient loading which in turn affects the salinity and DO of the watersheds.

When considering BMPs to solve the nutrient pollution problems in the watersheds, EPA selected over 14,000 water bodies across the country that were impaired with high levels of organic enrichment, nutrients and algal growth in 2010. The BMP database focus was on phosphorus and nitrogen (Strecker et al., 2005). It was discovered that sedimentation and filtration BMP processes are a good removal for total phosphorus as well as vegetated BMPs such as bioretention, swales and filter strips. Studies reveal that the infiltration basin has a removal efficacy of 65% for nitrogen and 69% for phosphorus (Sayre et al., 2006). Filtration BMPs should be designed with adequate protection and the use of chemical fertilizers should be avoided within BMPs (Hunt et al., 2006). BMPs with permanent pools such as wetlands and retention ponds appear to be effective for reducing nitrate loadings. Bioretention designs with pore storage above and below the underdrain, harvesting of vegetation and removal of algal mats and captured sediment may also be key maintenance practices for reliable removal of nitrogen.
It is also essential to regularly maintain and monitor BMPs for continued excellent performance.

**Total Coliform and E. Coli**

Total Coliform is the presence of coliform bacteria (E. coli). E. coli found in water is a powerful evidence that there is pollution from animal or human waste. It is generally associated with feces from humans or animals, polluted stormwater runoff, broken sewage systems and agricultural runoff (EPA, 2019; Cahoon 2006). The existence of E. coli in water along with TC signifies that pathogenic organisms exist in the water which is potentially harmful to human health. TC found in water alone with the absence of E. coli could be as a result of environmental pollution that happened during a plumbing construction (Cahoon, 2006; LQ2, 2018). The mean concentrations for E. coli were within the EPA’s criteria (<126 CFU/100mL) but all four watersheds had individual samples that did not meet the criteria. SC had the most individual samples that exceeded the criteria followed by LBC. This could be as a result of the mixed land use types characterized by increases in variable impervious surface types. Seasonal variations were observed in this study with TC and E. coli due to the higher mean concentrations of TC in the dry season compared to the wet season. Studies reveal similar trends in the past and it was proposed that higher concentrations of TC in the dry season could be as a result of less storm flows and warmer temperatures (Heaney et al., 2015; Wilson et al., 2007). To mitigate this, certain BMPs should be considered such as stormwater wetlands, wet ponds,
bioretention and dry retention basins. These methods have proven effective as tested and applied by Hathaway et al., 2010, with an above 50% efficiency for removing TC and only bioretention was found with an efficacy of above 50% for removing E. coli. Vegetated buffers/filter strips/swales were found with a 75% efficacy for both bacteria as observed by Tilman et al., 2011. Infiltration basin has been proven to have an efficacy of 100% and wetlands have an efficacy of 97% for the coliform bacteria (Sayre et al., 2006). Facucette et al. (2009) found that having runoff flow through compost filter socks with porous rolls of compost placed on the ground over removed bacteria by 75% and adding flocculant to the compost stocks increased the removal rate to 99%.
CHAPTER FIVE
CONCLUSION

The primary goals of this research were: to determine the type of physical changes to soil and water resources that will occur if the watershed landscape is altered; to model aspects of these changes especially as they relate to soil erosion and changes in surface runoff related to precipitation events; and to use findings to identify and recommend appropriate stormwater and watershed best management practices.

Results indicate that multiple parameters had mean concentrations and individual samples that exceeded the criteria and objectives set by federal and state regulations when observing in situ and field data from observed watersheds. There were notable differences in seasonal patterns as some parameters (e.g. Nitrate, and E. coli) had higher mean concentrations in the wet season compared to the dry season, while (Ammonium, Conductivity, pH and TC) had higher mean concentrations in the dry season compared to the wet season.

When applying the EPA My Watershed Modeling tool, the extreme changes in the physicochemical characteristics of surface water quality, use across all watersheds, were seen in developed high intensity (i.e. higher runoff, lower infiltration, TSS, TN) and forested land cover (i.e higher infiltration, lower runoff, low percentages of TSS, TN and TP loadings). Seasonally, the water quality parameters (TSS, TN, TP) had higher percentage loadings in the dry
season compared to the wet season. Runoff was seen to have increased
dramatically in the wet season with a reduction in infiltration as land cover
scenarios were modeled to reduce forest cover and increase impervious surfaces
associated with development of the landscape. The results show the impact of
climate change (i.e. changes in precipitation events and runoff, drought, and
warming temperatures). The results also indicate that land use changes in the
landscape draining to and forming headwater streams is a primary factor of
hydrological variations in the catchments. Although the model is useful in
determining potential impacts to water resources related to landscape alterations,
it is limited in determining frequent changes to water resources over time. This
emphasizes the importance of water quality testing year round in the headwater
watershed streams of the San Bernardino National Forest since they are the
source of connection to surrounding landscape and downstream regions in the
hydrologic unit including the two largest river basins in Southern California (the
Santa Ana and Mojave). Additionally, the study site is of importance because it
drains into Silverwood Lake as well as Lake Arrowhead where significant
recreational activities occur, and it provides drinking water to most of the San
Bernardino and High Desert residents.

To mitigate both short- and longer-term impacts to water resources both in
situ data collection and the EPA My Watershed Modeling tools are essential in
identifying appropriate BMPs as well as determining their effectiveness in
mitigating pollution inputs from entering downstream waterways. Sediment and
filtration best management practices such as grass swales, wetland basins, infiltration trenches, were recommended for erosion control, sediment removal, and elimination of nutrients and bacteria. With the unpredictable weather conditions, Southern California as an arid environment will keep having prolonged periods of droughts and periods of high precipitation. Management practices need to be enforced by decision makers to protect the headwater watershed streams and water resources to enable a safe and resilient community.
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