Spring 2014

Wastewater and Water Quality Changes in Lower South San Francisco Bay, 1957-2013.

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WASTEWATER AND WATER QUALITY CHANGES IN LOWER SOUTH SAN FRANCISCO BAY, 1957-2013

A Thesis

Presented to

The Faculty of the Department of Environmental Studies

San José State University

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

by

Simret Kesete Yigzaw

May 2014

The Designated Thesis Committee Approves the Thesis Titled

WASTEWATER AND WATER QUALITY CHANGES IN
LOWER SOUTH SAN FRANCISCO BAY, 1957-2013

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APPROVED FOR THE DEPARTMENT OF ENVIRONMENTAL STUDIES

SAN JOSÉ STATE UNIVERSITY

May 2014

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ABSTRACT

WASTEWATER AND WATER QUALITY CHANGES IN LOWER SOUTH SAN FRANCISCO BAY, 1957-2013

by Simret Kesete Yigzaw

The San José-Santa Clara Regional Wastewater Facility (Facility), located in the Lower South Bay (LSB), is the largest advanced wastewater treatment plant in San Francisco Bay. From 1957 to 2013, the Facility added a series of expansions and upgrades that increased treatment capacity and improved the quality of treated effluent.

This study addressed the questions: (1) To what extent have expansions and upgrades in the treatment plant during the past six decades resulted in improvements to Facility effluent quality? (2) How and to what extent have the changes in the Facility effluent translated into changes in the water quality of the LSB? Five hypotheses were formulated to evaluate long-term trends and correlations regarding wastewater loads (BOD, TSS, NH_4^+, NO_3^-, and PO_4) in the LSB. R software was used to analyze the data.

All five hypotheses were confirmed by the data, with a number of qualifications that can be readily explained. The first major finding is that, in spite of substantial increases in population, both influent and effluent flow to the Bay decreased in the past decade. A second major finding is that the data show major load reduction in BOD, TSS, and nutrients corresponding to Facility improvements. Third, anoxia and hypoxia were virtually eliminated following the Facility's upgrade to nitrification, significantly improving DO concentrations in the LSB. Fourth, LSB nutrient concentrations showed significant decreases corresponding with capital improvements to the Facility.

ACKNOWLEDGMENTS

For believing in me and never giving up on me, I am ever grateful to my mother: Mom, I can't thank you enough. Amaha and Somit: Your love, support and daily inspiration make me feel truly blessed. I could not have done this without you. To my amazing family and friends: Thank you for all your encouragement and the help that has sustained me and kept me afloat throughout all the ups and downs of this thesis project.

To my thesis chair, Prof. Katherine Cushing: Thank you for sticking with me for all the years leading up to successful completion of this thesis. Your support, encouragement and guidance have been invaluable. Drs. Cloern and Crunk generously and graciously took time from their busy schedules to assist and guide me, for which I am also most appreciative.

This thesis project would not have been possible without the cooperation of the San José-Santa Clara Regional Wastewater Facility, which provided the data for this research. I am indebted to all my dedicated colleagues at the Facility for coming to my assistance when needed and for encouraging me throughout the process. Special thanks are due to my supervisors, Eric Dunlavey and James Ervin, for going well beyond the call of duty to support this research effort. Readers of this study will certainly appreciate Kent Brown's contribution with regard to producing the beautiful maps, for which I am also grateful.

David, Emily and Jin—of the San Francisco Estuary Institute (SFEI)—generously helped me with the R program, shared codes and responded to queries by email, which was all instrumental to the success of this research effort. Hongyue Song, Shruti Vaidyaand Nathan Grassman (three of Prof. Crunk's graduate students) took time from their busy schedules to conscientiously help me with the statistical analysis, a critical component of the project, in an unselfish demonstration of collegiality. Finally, I would like to thank Dr. Daniel Berman for his patient and expert editorial assistance, in the context of odd hours and frequent scheduling adjustments.

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Acronyms

BNR	Biological nutrient removal
BOD	Biochemical oxygen demand
C	Concentration
CBOD	Carbonaceous biochemical oxygen demand
CDO	Cease and Desist Order
CIP	Capital Improvement Program
CO_2	Carbon dioxide
COD	Chemical oxygen demand
CWA	Clean Water Act
DIN	Dissolved inorganic nitrogen
DIP	Dissolved inorganic phosphorus
DO	Dissolved oxygen
EPA	Environmental Protection Agency
Facility	San José-Santa Clara Regional Wastewater Facility
g	gram
HABs	Harmful algal blooms
kg/d	Kilogram per day
loess	Polynomial local regression
LSB	Lower South San Francisco Bay
MDL	Method of detection limit
mg/L	Milligram per liter

MGD	million gallons per day
N	Nitrogen
NBOD	Nitrogenous biochemical oxygen demand
NH_4^+	Ammonium
NNE	Nutrient numeric endpoints
NO_3^-	Nitrate
NO_X	Nitrogen Oxides
NPDES	National Pollutant Discharge Elimination Systems
NRC	National Research Council
O_2	Oxygen
P	Phosphorus
PO_4	Phosphorus
POTW	Publicly Owned Treatment Works
Q	discharge
RWF	San José-Santa Clara Regional Wastewater Facility
RWQCB	Regional Water Quality Control Board
SBAP	South Bay Action Plan
SBDA	South Bay Discharge Authority
SBWR	South Bay Water Recycling (SBWR)
SCVWD	Santa Clara Valley Water District
SFB	San Francisco Bay
SFRWQCB	San Francisco Bay Regional Water Quality Control Board

SJSC	San José/Santa Clara
SOD	Sediment oxygen demand
SPE	Streeter-Phelps equation
SPE	Streeter-Phelps equation
SWRCB	State Water Resources Control Board
TMDL	Total Maximum Daily Load
TSS	Total suspended solids
umol/L	Micromole per liter
WPCP	San José/Santa Clara Water Pollution Control Plant
wq	Water quality

Introduction

Wastewater that is not properly treated, over a certain length of time, has the potential to degrade receiving water quality. In recent history, substantial damage was done to the San Francisco Bay and connecting ecosystem as a direct result of improperly treated wastewater. Upgrades to the wastewater treatment system during the past several decades have resulted in significant improvements. Many other places, however, both in the United States and other countries, are experiencing environmental degradation because of the inadequate handling of wastewater. The issue is a priority because degradation of the environment over time not only diminishes water quality but actually threatens the continued existence of life in areas where the environment is being harmed. One rationale for this study is the application of what has been done by the San José-Santa Clara Regional Wastewater Facility (known as the Facility) to other places with secondary and tertiary treatment facilities, where similar methodology could be applied, to achieve similar results.

In spite of the tremendous progress made in the Bay Area, significant issues remain to be resolved with regard to the effects of wastewater released into the environment. There is concern that the Bay may be losing its resilience toward eutrophication (the acceleration of the delivery, in situ-production of organic matter, and accumulation of organic matter). This may be due to changes in water clarity resulting from less suspended sediment, increased seeding from ocean populations, declines in consumption by bivalves because of increases in predation by certain fish, and declines in phytoplankton consumption by consumers prompted by recent new invasive species introductions, resulting in algal growth being less light limited (Cloern et al., 1999; Cloern et al., 2005; Cloern et al., 2006).

Although the study has important implications for other areas, as mentioned above, its focus was an assessment of the progress made during the past six decades with regard to improvements of wastewater treatment processes by the Facility. The scope of the study also included changes in ambient water quality in the Lower South San Francisco Bay (LSB) during the corresponding period of time.

Estuaries are bodies of water, partially enclosed, where fresh river water meets and mixes with salty ocean water. They are highly dynamic systems that are subject to changes that occur over a broad spectrum of time scales, reflecting complex responses to numerous driving forces. An important habitat for a wide variety of marine life, estuaries are often threatened by pollution and habitat destruction that typically results from urban development, a major part of which is related to sewage from human and industrial waste.

The San Francisco Bay represents the largest bay on the California coast and the largest Pacific estuary in North America, whose system is comprised of the Delta, receiving the waters of the Sacramento and San Joaquin river systems, in addition to the Bay proper, into which waters from the Delta flow (Figure 1). The Bay is typically viewed as having two reaches, northern and southern. The northern reach passes south and westward from the Delta through Suisun and San Pablo Bays.

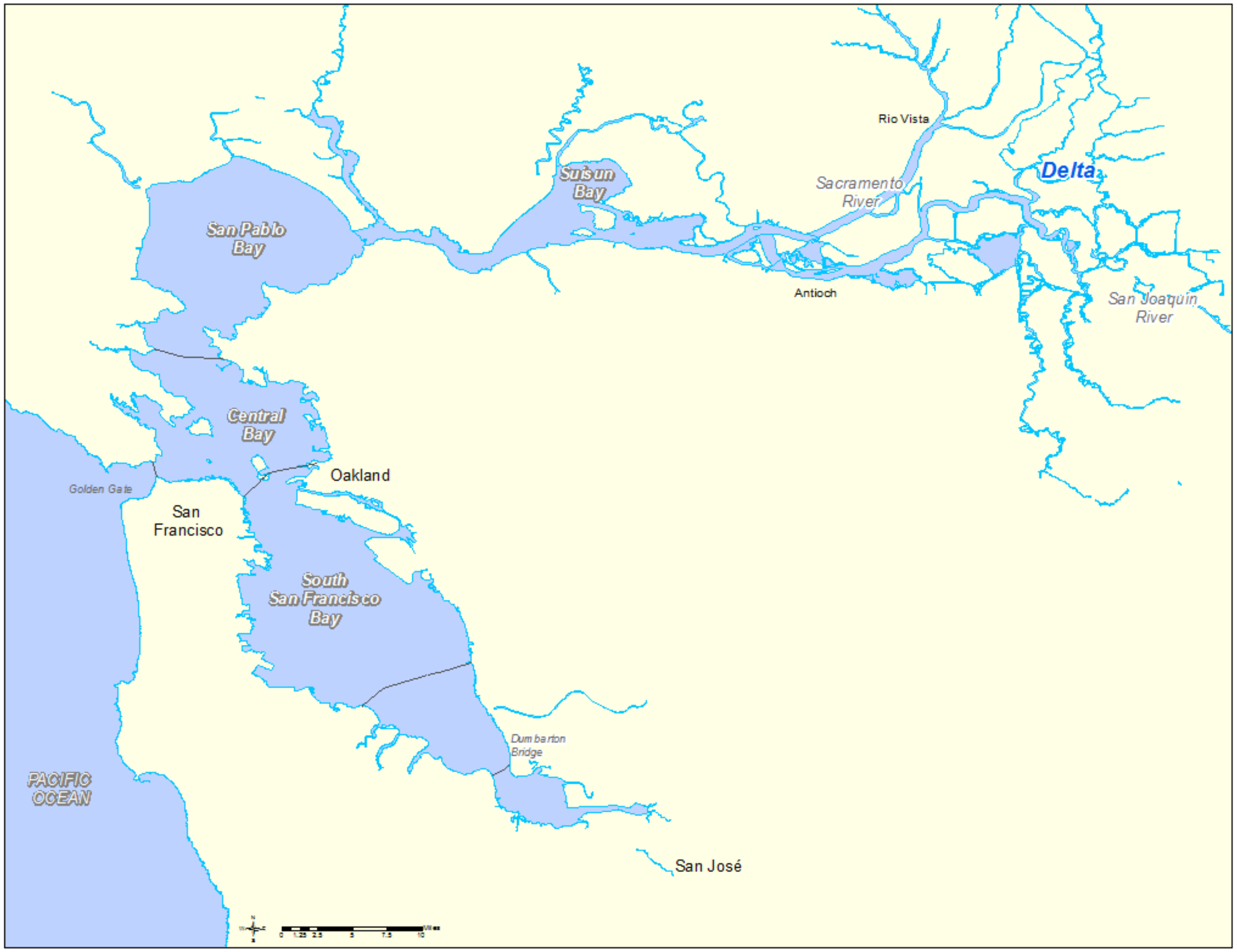

Figure 1. The San Francisco Bay, with two reaches, northern (encompassing the Delta, Suisun Bay, and San Pablo Bay) and southern (also known as the South Bay), joining in the Central Bay near the Golden Gate.

The southern reach, also called the South Bay—a marine lagoon situated in a densely populated urban setting—extends southeastward towards San José. The two reaches join in the Central Bay near the Golden Gate (Conomos, 1979).

Following World War II, San José, the largest city in the Bay Area, began to experience a rapid increase in population and development as it transitioned from an agricultural community, to industrial manufacturing, and then to high-tech. San José had been the world's largest canning and dried-fruit packing center until the 1950s, with the growth of the high-tech industry. Waste from the fruit canning industry caused gross seasonal pollution of the waters, resulting in hydrogen sulfide production and a widespread odor nuisance, exerting "a total BOD [biochemical oxygen demand] equal to that from the contributory population" (Norgaard et al., 1960, pp. 1088-1089). These seasonal canning wastes remained a major contributor of sewage to the Facility until the late 1980s.

Historical Chronology of San José's Sewage Disposal and the Facility

1867 Construction of first sewer: 3ft x 4ft redwood box

1870 Submission of plans for system designed to serve population of 100,000

1896 Completion of combined sewerage and drainage system: 60-inch brick line

1930 Extension of sewer line approximately 2.5 miles into Bay waters

1949 Engagement of consulting firm by City to design sewage facility

1955 Construction of sewage treatment plant begins

1956 36-MGD primary treatment facility completed and placed into operation

1959 Partnership created between cities of San José and Santa Clara

1960	Interim improvements added to expand design capacity to 51 MGD
1962	Construction of major expansion program begins
1964	Completion of secondary treatment plant, with upgrade to capacity of 94 MGD
1970	Addition of primary, secondary, and chlorination facilities of 160 MGD
1979	Addition of tertiary/advanced facilities, including nitrification and filtration
1980	First stage of new expansion project, to 167 MGD capacity
1986	Completion of expansion to currently rated capacity
1997	Implementation of biological nutrient removal (BNR) process
1997	Construction of South Bay Water Recycling (SBWR) pipeline system
2010	Agreement for Silicon Valley Advanced Water Purification Center
2012	Initiation of Plant Master Plan, for what is now known as the Facility

For more detailed information on this subject, please refer to Appendix I.

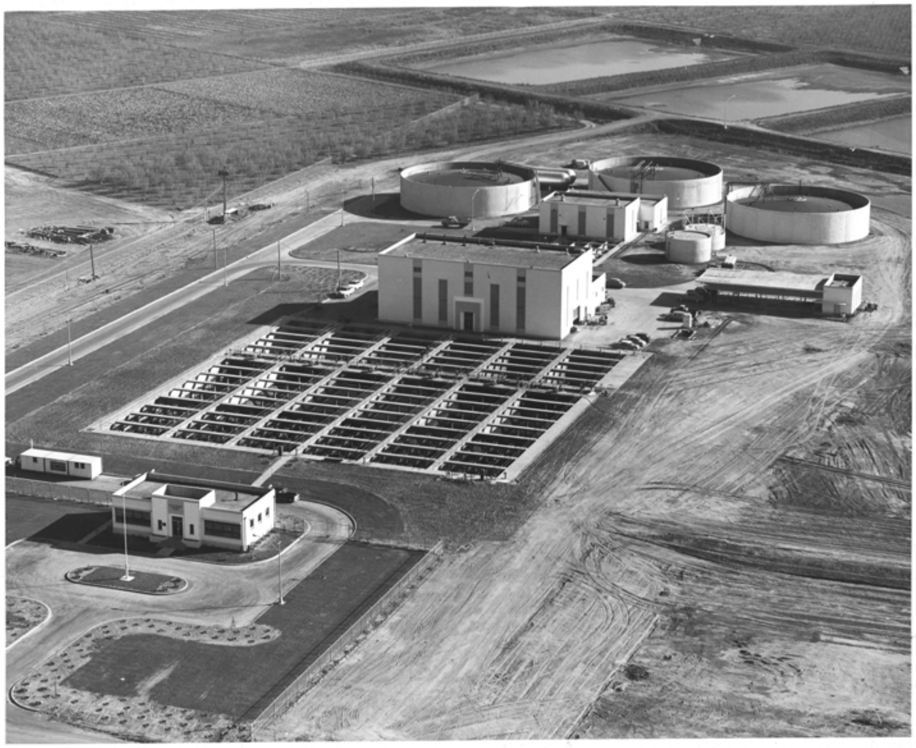

Figure 2. The Facility, as of 1956, which at the time only included the primary treatment processes. Reprinted with permission.

Figure 3. The Facility as of 2012, including primary, secondary, tertiary, and BNR treatment processes. Reprinted with permission.

Chronology of Sewage Treatment Issues and Policy Development

1946	Santa Clara County sewage disposal survey
1953	Survey of water conditions in the LSB
1958-59	Pilot pollution study of South San Francisco Bay
1960-64	Comprehensive study of San Francisco Bay
1969	SF Bay-Delta Water Quality Control Program final report
1971	Interim Water Quality Control Plan for SF Bay Basin
1972	Federal Clean Water Act (CWA)
1972	Water Quality Management Plan for SF Bay
1974	Water Quality Control Policy for enclosed bays and estuaries
1975	Water Quality Control Plan (Basin Plan)
1981-86	Water Quality Study under SBDA
1989	Cease and Desist Order (CDO)
1990	Order WQ 90-5 (action plan to protect salt marshes in the LSB)
1991	South Bay Action Plan (SBAP) to implement Order WQ 90-5

For more detailed information on this subject, please refer to Appendix II.

Problem Statement

The LSB—known also as the extreme South Bay or the dead end of the Bay—was highly polluted during the early 1940s as a result of untreated sewage released directly into the water. The area was known for "The Big Stench" resulting from high levels of organic materials released onto the water's surface. The problem was particularly severe during the summer months because of low water flow, high temperatures, and high levels of organic matter from the canneries in the area. The water had low levels of dissolved oxygen (DO), sometimes falling to zero, contributing to occasional fish kills (SFEI, 2007).

With increases in population, the sewage flow to the area increased substantially, prompting interest in constructing publicly owned treatment works (POTW) in the area. The Facility is the largest advanced wastewater treatment plant in San Francisco Bay. During the past 50 years, the Facility has added a series of expansions and upgrades that increased treatment capacity and improved the quality of treated effluent.

Recently, there has been substantial interest both regionally and at the state level with regard to the issue of nutrients in the San Francisco Bay. This concern has been prompted in part by the occurrence in recent years of algal blooms (rapid increases or accumulations in the population of algae, typically microscopic, in the aquatic system). The two dimensions of the LSB water quality problem stemming from the discharge of BOD and nutrients from the Facility relate to two different historical phases. In the earlier phase, from 1957 to 1964, BOD was released in substantial amounts by the Facility directly into the water. More recently, from 1964 to the present, BOD has been created in the Bay in the form of phytoplankton biomass as a result of nutrients, such as nitrogen and phosphorus, released by the Facility.

With all this in mind, this study's main research questions are as follows: (1) To what extent have expansions and upgrades in the treatment plant during the past six decades resulted in improvements with regard to Facility effluent load (BOD, TSS, NH_4^+, NO_3^-, and PO_4)? (2) How and to what extent have the changes in the Facility effluent load translated into changes in the water quality of the LSB?

Current nutrient loads to SFB and to some of its subembayments are comparable to or much greater than a number of other major estuaries (such as Chesapeake Bay), which are experiencing impairment as a result of nutrient over-enrichment (Cloern & Jassby, 2012). Also, nitrogen and phosphorus concentrations are highly elevated. There is growing evidence that the Bay may be losing its resilience to harmful effects of nutrient enrichment (eutrophication). One important step in addressing this critical problem is enhancing our understanding with regard to the details of this phenomenon.

The objective of this study is to present and analyze data on BOD, total suspended solids (TSS), nutrients, and DO, both with regard to the Facility effluent and the ambient water quality of the LSB. Specifically, the study reviewed the historical changes during the past six decades and how those changes correspond to effluent loading and water quality in the LSB.

The five hypotheses tested in this study are as follows:

1. Facility influent flow, BOD, TSS, and nutrient loading increased with increases in population over time.

2. Facility effluent BOD, TSS, and nutrient loading decreased with Facility treatment process upgrades.

3. There is an inverse correlation between ambient DO concentration in the LSB and both Facility BOD and nutrient loading.

4. Ambient DO and nutrient concentrations in the LSB follow a pattern relating to temporal, seasonal, and spatial aspects. (Temporal here refers to treatment era. Seasonal breaks down into dry and wet. Spatial refers to proximity to Facility discharge point.)

5. Ambient nutrient concentrations in the LSB are positively correlated with Facility nutrient loading.

These are all specific, testable predictions that have been evaluated on the basis of data collected by the researcher from the Facility. The open source software R was used to analyze the data, as described in more detail in the methods section below.

Literature Review

Conceptual Framework

The conceptual framework for this study is based on a modified version of what is known as the Streeter-Phelps equation (SPE) or model, also known as the DO sag equation or curve, a curve that shows the profile of DO content alongside the course of a stream, ensuing from deoxygenation (the removal of oxygen from a water column). The curve is used to measure concentrations of DO. The SPE determines the relationship between DO concentration and the BOD over time. This equation, which is used as a water quality modeling tool in the study of water pollution, was devised by Streeter and Phelps, based on field data from the Ohio River (Streeter & Phelps, 1958).

It was during the 1960s, when computers made it possible to include further contributions to the oxygen development in streams, that more complex versions of the Streeter-Phelps model were introduced. O'Connor (1960) and Thomann (1963) were at the head of this development. Contributions from photosynthesis, respiration, and sediment oxygen demand (SOD) were added by O'Connor. It was Thomann who expanded the Streeter-Phelps model to allow for multi-segment systems.

The original Streeter-Phelps model was based on the assumptions that (1) a single BOD input is distributed evenly at the cross section of a stream or river, and that (2) it moves as plug flow with no mixing in the river (Lin, 2001). Only one DO sink (carbonaceous BOD) and one DO source (reaeration), are considered in the original Streeter-Phelps model (Schnoor, 1996). These simplifications result in errors in the model.

In order to factor in the numerous additional processes in a stream affecting the DO, a more accurate, expanded, model was developed. In 1979, Bauer et al. developed a "one-dimensional steady-state stream water-quality model," based primarily on the Streeter-Phelps oxygen-sag equation. The program (written in FORTRAN) for the revised model included special options for the capability of handling nonpoint source waste inputs and conditions of anoxia (severe deficiency of oxygen).

In 1989, McCutcheon published a modified version of the Streeter-Phelps model. The data used to arrive at the modified version mainly defined the DO balance. Important components of the balance included reaeration, deoxygenation by decay of organic material, nitrification, and sediment oxygen demand (McCutcheon, 1989). Figure 4 below, developed by McCutcheon, depicts the "interrelationship of major kinetic processes for BOD, DO, and nutrient analyses as represented by water quality models" (EPA, 1997, pp. 2-14). The figure serves to explain the relationship and interplay between DO and its associated processes: (1) reaeration, (2) carbonaceous deoxygenation, (3) nitrogenous deoxygenation (nitrification), (4) photosynthesis and respiration, and (5) sediment oxygen demand (SOD).

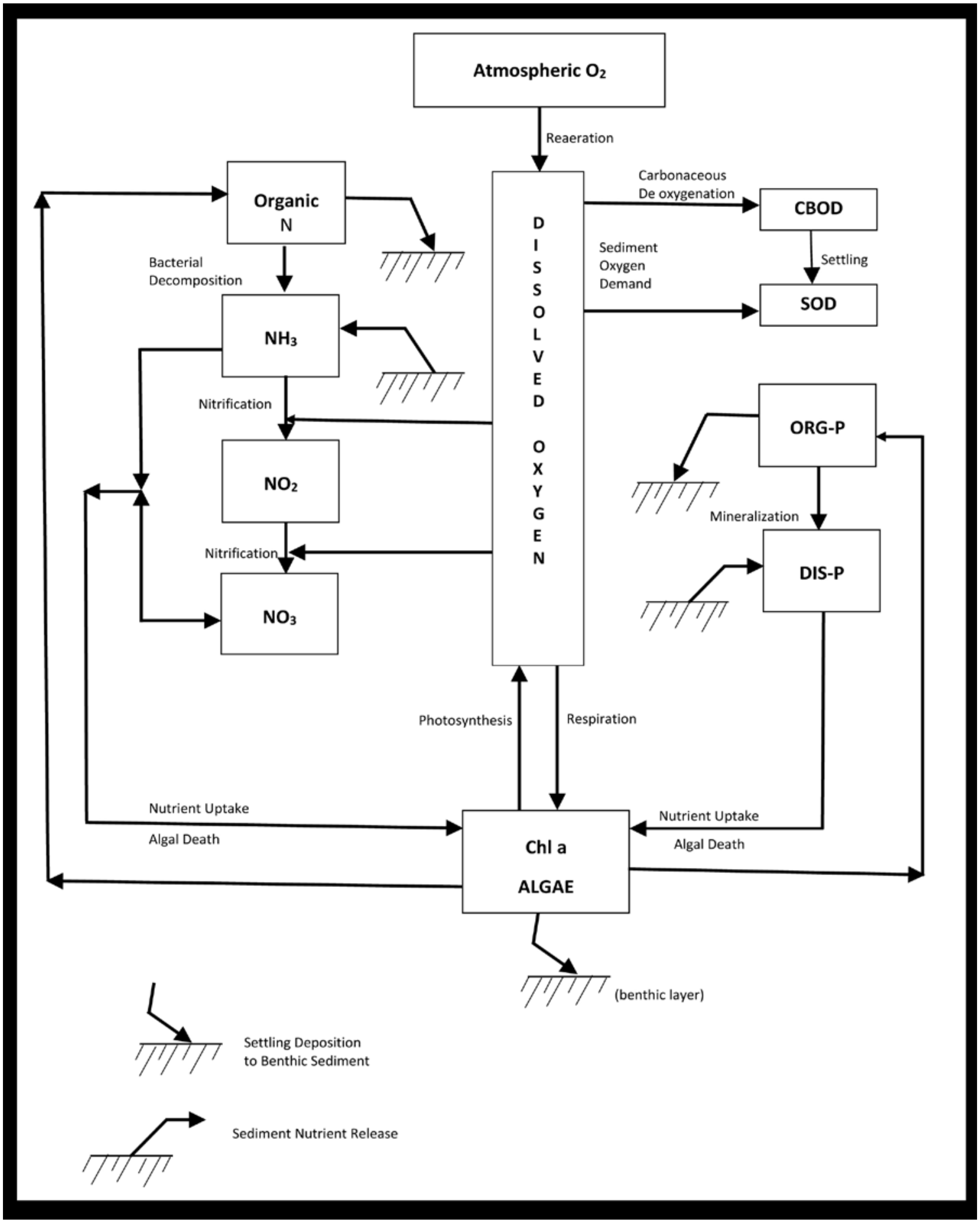

Figure 4. Schematic diagram of conceptual framework of the interrelationship between major processes involving BOD, DO, and nutrients. Adapted from "Technical Guidance: Manual for Developing Total Maximum Daily Loads: Streams and Rivers" (EPA, 1997, pp. 2-14), with permission from the copyright holder, Taylor and Francis Group LLC Books.

DO in the water column of estuaries and other bodies of water functions as an important determinant of water quality. As such, DO concentration can serve as an important indicator of nutrient-related impairment. One reason for this is that maintaining sufficient DO levels is critical for sustaining aquatic life. Another reason is that low levels of DO represent a typical response on the part of the ecosystem to high nutrient loads.

The estuary produces and consumes oxygen, which is transported into the water column through the air. Hypoxia (oxygen deficiency) or anoxia can develop in situations when the oxygen loss rate exceeds the oxygen production or input rate. In an extreme (persistent) state, hypoxia or anoxia can result in stress or death of aquatic organisms. This can also lead to sulfide gas production, potentially toxic to aquatic organisms, causing both odor and damage to infrastructure such as painted exteriors (discoloration and corrosion) (Senn & Novick, 2013).

Otherwise stated, the problem with DO begins with the input of oxygen-demanding wastes into the estuary or other body of water. The main inputs that affect the DO are municipal and industrial discharges of wastes, as well as combined sewer overflows and separate sewer discharges. These types of discharges contain matter that creates a chemical oxygen demand (COD), carbonaceous biochemical oxygen demand (CBOD), and oxidizable nitrogen. The latter is also represented by the nitrogenous biochemical oxygen demand (NBOD).

The sources of DO are (a) reaeration from the atmosphere, (b) photosynthetic oxygen production, and (c) DO in incoming effluents or tributaries. The internal sinks of

DO are: (a) oxidation of carbonaceous waste material, (b) oxidation of nitrogenous waste material, (c) oxygen demand of sediments of water body, and (d) use of oxygen for respiration by aquatic plants. Further information appears below in the description of associated processes of DO.

(1) *Reaeration.* A variety of physical, chemical, and biological reactions may result in oxygen being removed from or added to the water column. Reaeration occurs when oxygen is low in the water column. Conversely, when oxygen is high in the water column, saturation occurs, and DO transfer to the air. Oxygen transfer in natural waters depends on numerous factors, including temperature, wind mixing, surface films, and water column depth.

DO enters and leaves the LSB by three main mechanisms: fluvial transport (including the Delta, perennial ephemeral streams, stormwater inputs, and treated wastewater effluent), water exchange between subembayments, and mixing between zones within a subembayment (Senn & Novick, 2013).

(2) *Carbonaceous deoxygenation (CBOD).* BOD is a measure of the quantity of oxygen consumed by microorganisms during the decomposition of organic matter, the most commonly used parameter for determining the demand for oxygen on the part of the receiving water of a municipal or industrial discharge. As an indirect measure of biodegradable organic compounds in water, BOD also may be used to evaluate the efficiency of treatment processes. When elevated levels of BOD lower the concentration of DO in a body of water, this creates the potential for profound effects on the body of water and the aquatic life therein. When the DO concentration falls below 5 milligrams

per liter (mg/L), this results in stress on species intolerant of low oxygen levels. The lower the concentration of oxygen, the greater the stress will be. Carbonaceous oxygen demand and nitrogenous oxygen demand represent the two parts into which BOD is typically divided. CBOD results from of the breakdown of organic molecules such as cellulose and sugars into carbon dioxide and water.

(3) *Nitrogenous deoxygenation (nitrification).* This refers to the biological oxidation of ammonium with oxygen, first into ammonium and then into nitrite. This is followed by the oxidation of these nitrites into nitrates. Nitrification is an aerobic process, which is performed by small groups of autotrophic bacteria and archaea.

(4) *Photosynthesis and respiration*. It is through the processes of photosynthesis and respiration that phytoplankton and rooted aquatic plants, among other things, are able to significantly affect the water column's DO levels. Primary production by phytoplankton and benthic algae results in the production of oxygen during daylight hours. In most habitats of the SFB, the rate of oxygen production varies proportionately to the rate of primary production. For phytoplankton and MPB (microphytobenthos), this production rate is light-limited (Senn & Novick, 2013).

(5) *Sediment oxygen demand (SOD)*. This refers to the overall demand for DO from the water column, a demand that is exerted by a combination of biological, biochemical, and chemical processes at the interface of sediment-water. The main sources of SOD include anaerobic (low-oxygen) chemical compounds in the sediments collected at the bottom of the estuary, as well as particulate BOD (including algae and other sources of organic matter) which settle out of the water column.

SOD typically consists of biological respiration from benthic organisms and the biochemical decay processes in the uppermost layer of deposited sediments, together with the release of oxygen-demanding anaerobic chemicals, such as iron, manganese, sulfide, and ammonium. These soluble chemicals, when released into the water, exert a relatively rapid oxygen demand, with the oxidization of the reduced chemicals. Certain oxidation processes (nitrification of ammonium to nitrate, for example) require bacteria and may be considerably slower.

As mentioned in the problem statement, the two dimensions of the LSB water quality problem stemming from the discharge of BOD and nutrients (primarily nitrogen and phosphorus) from the Facility relate to two particular and distinct historical phases. In the earlier phase, BOD was released in substantial amounts by the treatment plant directly into the water. More recently, BOD has been created in the Bay in the form of phytoplankton biomass as a result of nutrients, such as nitrogen and phosphorus, released by the Facility.

The most common of the various factors that tend to increase the supply of organic matter to coastal systems is generally recognized to be nutrient enrichment (Nixon, 1995). The input of additional nutrients to coastal marine systems can in some cases have beneficial impacts such as an increase in fish production. Typically, however, the consequences of nutrient enrichment for coastal marine ecosystems are detrimental. Many of these detrimental consequences are associated with eutrophication, the increased productivity from which increases oxygen consumption in the system, and can result in water bodies becoming hypoxic or anoxic. This in turn can lead to both fish kills and more subtle changes in ecological structure and functioning. Eutrophication can also have negative ramifications for estuaries even in the absence of low-oxygen events. Harmful algal blooms (HABs) harm fish, shellfish, and marine mammals, presenting a direct threat to public health. Nutrient over-enrichment of coastal waters can lead to blooms of some organisms that occur more frequently and are longer in duration (NRC, 2000). Ammonium consumes oxygen from the water, which can lead to hypoxia. Ammonium is also preferred by phytoplankton over nitrate. It is phytoplankton dependence on NH_4^+ which leads to production. This in turn is cycled within the microbial loop. That based on NO_3^-, in contrast, typically leads to production that supports a food web prompting secondary production, and also export from the euphotic zone (Dugdale & Goering, 1967; Eppley & Peterson, 1979).

The question as to how anthropogenic nutrient enrichment causes change in the structure or function of nearshore coastal ecosystems has been a primary focus of coastal science during recent decades. Cloern's conceptualization of the evolving model of the coastal eutrophication problem involved three distinct phases (Cloern, 2001). Phase I emphasized changing nutrient input as a signal. The responses to that signal included phytoplankton biomass and primary production, decomposition of phytoplankton-derived organic matter, and enhanced depletion of oxygen from bottom waters. Phase II reflected important differences in the responses to nutrient enrichment of lakes and coastal-estuarine ecosystems. Phase III was organized around five specific questions intended to guide coastal science during the early 21st century. The first and perhaps most salient question, for purposes of this study, is "how do system-specific attributes constrain or amplify the responses of coastal ecosystems to nutrient enrichment?" (Cloern, 2001, p. 223).

Overview: Water Quality Problems for Estuaries

In "Overview of Hypoxia around the World," Diaz (2001) stated that DO has changed more dramatically than any other environmental variable of such ecological importance in such a short period of time. The occurrence of hypoxic and anoxic environments in shallow coastal and estuarine areas appears to be increasing, probably accelerated as a result of human activities. The author asserted that "many ecosystems that are now severely stressed by hypoxia may be near or at a threshold of change or collapse" (Diaz, 2001, p. 275).

Diaz, writing together with a coauthor, subsequently extended his warning about threats to marine ecosystems by reporting that "dead zones in the coastal oceans have spread exponentially since the 1960s," with "serious consequences for ecosystem functioning" (Diaz & Rosenberg, 2008, p. 926). The phenomenon has been aggravated by an increase in primary production and the resulting coastal eutrophication worldwide, prompted by riverine runoff of fertilizers and the burning of fossil fuels. Although the authors acknowledged that a return to preindustrial levels of nutrient input would be unrealistic, they proposed as an appropriate management goal the reduction of nutrient inputs to levels that occurred in the middle of the previous century, before the time that eutrophication began to spread dead zones globally.

Howarth et al. (2002) reported that approximately 60% of coastal rivers and bays in the U.S. have been moderately to severely degraded by nutrient pollution, with both nitrogen (N) and phosphorus (P) contributing to the problem. The flux of N and P from land to oceans has increased two-fold and three-fold, respectively, as a result of human activity, globally. Sewage treatment plants are the largest single input, in the case of some estuaries. Because of both improved point source treatment and control (especially for P), and of increases in the total magnitude of nonpoint sources (especially for N), nonpoint sources of nutrients are currently of greater importance for most systems.

An updated assessment of nutrient-related impacts in U.S. estuaries completed in 2007 was reported on by Bricker et al. (2008). The assessment evaluated three components for each estuary: the influencing factors (such as land use and nutrient loads), the overall eutrophic condition (such as chlorophyll a and the extent of DO problems), and the outlook for the future. With 65% of assessed systems demonstrating moderate to high level problems, eutrophication has been a widespread problem. Most of the assessed estuaries have been greatly affected by activities resulting from human behavior, contributing to land-based nutrient loads. The prediction for the future was that conditions would worsen in 65% of the cases, while improving in only 19% of the assessed estuaries. The symptoms described are more prevalent in systems with longer residence times such as coastal lagoons like the LSB. The findings from this article by Bricker et al. correspond very closely with those of the article by Howarth et al. (2002), described immediately above.

Related Research

Research relating to the Thames Estuary, Chesapeake Bay, Hudson River, and Delaware Estuary—as estuary systems in highly populated urban areas—may serve to shed light on other estuaries such as that of the LSB. These are the main estuary systems covered in the literature, as it relates to the impact of wastewater sewage treatment on aquatic environments. There are important similarities in terms of the progression of environmental degradation and responses on the part of local governing bodies, which can help to illuminate trends, patterns, and potential solutions.

Thames. Green wrote that the purpose of the 1998 collection to which he contributed was "to bring together certain aspects of the situation in the Thames estuary, and to update the changes that have occurred," with chapters covering a wide range of topics, "from physiochemical factors to algae and fish" (Green, 1998, p. 3). Chapter 2 traced the dramatic decline in the water quality of the Thames estuary during the 19th and 20th centuries (Tinsley in Attrill, 1998).

Improvements in water quality within the estuary were achieved by virtue of the combined effects of chemical precipitation of the sewage, reducing BOD and the addition of interceptor sewers (Tinsley, 1998). The book in its conclusion described the Thames as a "success story," benefitting from "the increasing awareness of environmental problems, from the enhanced monitoring techniques and from the stricter and more relevant regulations introduced," with the mechanism for protection and improvement firmly in place (Attrill, 1998, pp. 197-198).

Chesapeake Bay. Jaworski (1990) reported that the water quality of the upper Potomac Estuary near Washington, D.C. had changed dramatically during the preceding century. From the 1950s through the 1970s, major water quality problems in the upper Potomac Estuary resulting from inadequate treatment included high fecal coliform counts, low DO concentrations, and nuisance eutrophic conditions. It was during the 1970s that the wastewater treatment facilities for BOD, suspended solids and phosphorus removal were upgraded through a major construction program. The positive response from the estuary during the period of enhanced wastewater treatment (from 1970 to 1985) was significant. The author suggested that "benefit to both ecological resources and societal conveniences should be included in the cost of wastewater collection and treatment/benefit consideration," defining societal convenience benefits as "those gained when one utilizes water in the everyday course of living conveniently" (Jaworski, 1990, p. 12).

Hagy et al. (2004) described the long-term pattern of hypoxia and anoxia in Chesapeake Bay and its relationship to nitrate loading. During the half-century period under study, the volume of hypoxic water increased substantially and at an accelerating rate, during mid-summer. There was a positive correlation found between hypoxia and nitrate loading, though "more extensive hypoxia was observed in recent years than would be expected from the observed relationship" (Hagy et al., 2004, p. 634). According to the authors, the results suggested that the Bay may have become more susceptible to nitrate loading. Hagy et al. (2004) referenced an earlier study by Breitburg (1990), in which the focus was patterns and relationships among physical factors (e.g., salinity, wind forces, tidal currents, and temperature), rather than human impact, in the context of near-shore hypoxia in the Chesapeake Bay. The author stated that "severe hypoxia at the study site … appeared to result from intrusions of bottom water, which were most effectively driven by southerly winds" (Breitburg, 1990, p. 593). The final force that brought deep water close to shore was provided by tidal currents.

Hudson River. Brosnan and O'Shea (1996) used long-term trends in DO and total coliform bacteria concentrations to evaluate the impact of nearly 60 years of sewage abatement and treatment in the lower Hudson River. "Although some water pollution control plants have been in operation in the region since the 1930s, the most significant abatement of untreated sewage [there] has occurred since the late 1970s, when most of the existing plants were upgraded to secondary treatment, and additional plants were constructed" (Brosnan & O'Shea, 1996, p. 890). DO concentrations generally increased from the late 1970s through the 1980s and especially into the 1990s. This increase coincided with the upgrading of the North River plant in the spring of 1991 to secondary treatment. The significant improvements that were implemented during the early 1990s are reflected in the increases in surface and bottom DO concentrations, as well as in the reduction of the spatial and temporal extent of severe hypoxia.

A subsequent study on the Hudson several years later by Hetling et al. (2003) pointed to "a continued increase in wastewater flow and population over the past century but a decrease in contaminant loading during the last 25 years" (Hetling et al., 2003, p. 30). The authors stated that the decrease in effluent loads was a direct result of water quality management programs at both the state and federal levels, as well as a substantial public and private investment made with regard to upgrading the infrastructure of point source water pollution control.

Delaware Estuary. Sharp (2010) addressed the question as to what can be learned about hypoxia based on 40 years of consistent monitoring of data records of the Delaware Estuary from a multistate agency. The hypoxia and anoxia that occurred in the upper Delaware Estuary during much of the 20th century has diminished during the past few decades, according to Sharp. Reduced nitrogen and carbon appeared to be the primary cause of the DO decline. "In spite of extremely high nutrient concentrations, excess algal production did not influence DO anywhere along the tidal freshwater stretch or the saline portion of the well-mixed Delaware Estuary" (Sharp, 2010, p. 535). Although the nutrient loading to the Estuary is very high, Sharp stated, the typical signs of eutrophication were not obvious. The author cautioned that estuarine water conditions similar to those before nutrient enrichment will not necessarily be revived following nutrient removal. The deterioration of estuaries and complex coastal ecosystems is a condition that results from a combination of factors: nutrient enrichment, habitat alteration, depletion of higher tropic levels, and inhibition by contaminants other than nutrients.

San Francisco Bay. *San Francisco Bay: The Urbanized Estuary*, published in San Francisco in 1979, was further subtitled *Investigations into the Natural History of San Francisco Bay and Delta with Reference to the Influence of Man*. The volume represents a nearly 500-page collection of 20-some chapters, half of which were presented as during the 58th annual meeting of the Pacific Division of the American Association for the Advancement of Science at San Francisco State University in June 1977. The stated purpose was to summarize in individual chapters the state-of-the-art knowledge of the natural processes contributing to the maintenance of the estuary. The idea was "to be as comprehensive as possible, bringing together reports dealing with the many interrelated aspects of estuarine research ongoing in San Francisco Bay and Delta" (Conomos, 1979, p. 7). The four main sections into which chapters are divided are Physical Processes, Water Properties, and Quality, The Ecosystem and Fisheries Resources.

Conomos (1979) stated that "it was not until the last few decades that real progress has been made in our understanding of the processes and rates by which water, solutes, sediments and organisms interact." He further stated that, "The distribution of biologically reactive water properties such as plant nutrients, carbon, and DO are primarily related to seasonal variations in the supply of these components, to the intensity of water movement and mixing, and to a lesser extent to the amount of available light, which promotes biological activity" (Conomos, 1979, p.491). Despite the progress that had been made to date, Conomos wrote, "there is still much to learn before we can accurately describe the mechanisms that contribute to the maintenance of the estuary as we know it now, or before we can adequately predict what lies in the future" (Conomos, 1979, p.492).

Cloern and Nichols (1985) wrote that the purpose of their book was "to examine the temporal dynamics of [various] properties and processes in the San Francisco Bay estuary." They acknowledged that their "understanding in some areas is limited by the lack of comprehensive, long-term studies and/or the relative difficulty in achieving understanding of the intricate interrelations among components of the estuarine system" but that the compilation nonetheless "has provided the opportunity to demonstrate how several key driving forces affect individual components of the estuarine ecosystem" (Cloern & Nichols, 1985, p. V). In their concluding chapter, the editors indicated areas that represent the appropriate focus for future research. They mentioned that certain important estuarine properties for San Francisco Bay "remain almost completely unstudied." These included "the sources and fates of toxic contaminants (particularly organic compounds), nutrient budgets, and riverine inputs of organic material" (Cloern & Nichols, 1985, p. 236).

Nichols et al. (1986) have written that one potentially important result of waste discharge was "stimulation of plant growth through nutrient enrichment, with subsequent declines in oxygen content of the water as the plant material decomposes." "Waste-derived nutrients are more apparent in the South Bay, where storm drains and waste treatment plants are the principle sources of freshwater inflow" (Nichols et al., 1986, p. 572). In spite of their recognition that nutrient concentrations are sometimes high, the authors asserted that the Bay does not in fact exhibit symptoms of eutrophication. In fact, they argued, "the problems of San Francisco Bay appear less severe than those of other large urbanized estuaries," partly because much of the urban and industrial development occurred near the estuary mouth, and also because corrective actions taken since the 1960s have eliminated oxygen depletion, greatly reducing pathogenic bacteria (Nichols et al., 1986, p. 573).

From the 1980s through 2013, numerous articles in various journals have treated the specific subject of phytoplankton dynamics in the San Francisco Bay. Most of these articles were co-authored by Cloern. The significance of phytoplankton dynamics in the San Francisco Bay, simply stated, is that they reflect the effects of the disposal of waste into estuaries by humans on the natural environment. By extension, this condition is a measure of the resilience of the environment, providing critical warnings suggesting that limits have been exceeded.

Dimensions that have been treated in these publications include significance of biomass and light availability, and the role of nutrients such as ammonium and nitrate in spring bloom development, as well as the effects of turbidity, climate anomalies, tidal stirrings, and the introduction of new species of clams into the environment.

What is known with certainty is that phytoplankton biomass in the San Francisco Bay has increased significantly during the preceding decades. The ecological mystery that researchers have been trying to solve is the cause or causes of this phenomenon. Researchers have concluded that the phytoplankton increase "cannot be attributed to increases in nutrient concentration" (Cloern et al. in SFEI, 2006, p. 67). Researchers have asserted that what has been considered to be the inherent resistance on the part of the San Francisco Bay to the harmful consequences of nutrient enrichment might be changing. The reasons for these changes appear to be the result of a complicated variety of factors in the natural environment, including climate anomalies (Cloern et al., 2005; Cloern et al., 2007; Cloern & Jassby, 2012).

Most recently, Cloern and Jassby (2012) have stated that the resistance of the San Francisco Bay is weakening. Cloern and Jassby built upon observations from the San Francisco Bay, as a way of illustrating responses to six drivers regarded as common agents of change in places where land and sea meet: water consumption and diversion, human modification of sediment supply, introduction of nonnative species, sewage input, environmental policy, and climate shifts. Responses to these drivers included nutrient enrichment and elimination of hypoxia, as well as changes to the food web that decrease resistance to nutrientt pollution of the estuary. As a result of its urban location, "South San Francisco Bay is highly enriched with sewage-derived nitrogen and phosphorus" (Cloern & Jassby, 2012, p. 15). On the basis of high N and P concentrations, South San Francisco Bay exhibited the potential to produce phytoplankton biomass at levels that severely impair other nutrient-enriched estuaries such as Chesapeake Bay.

Current Efforts of Regional Nutrient Management Strategy

The United States Environmental Protection Agency (EPA) Office of Water report has described the national strategy as proposing to build on work accomplished, while at the same time establishing "an objective, scientifically sound basis for assessing nutrient overenrichment problems" (EPA, 1998, p. 5). In terms of specifics, the strategy proposed a two-phase process for the development of water quality standards for nutrients. The first step involved development of "nutrient criteria guidance" for nitrogen, phosphorus, and other nutrient parameters such as chlorophyll a, secchi depth, and algal biomass. For the second phase of the process, the EPA expected that both states and tribes would adopt nutrient water quality criteria to support designated uses of waters.

In order to implement EPA guidelines, The Planning and Standards Implementation Unit of the California State Water Resources Control Board (SWRCB) in

July 2006 published its *Technical Approach to Develop Nutrient Numeric Endpoints for California*. The approach described in the report provided a methodology for supporting several water quality program components. These components included setting numeric limits for National Pollutant Discharge Elimination System (NPDES) permits and development of Total Maximum Daily Load (TMDL) nutrient numeric endpoints. For those Regional Water Boards that opted to do so, an additional component would be the development of numeric nutrient criteria. The report was intended as a "starting point for a process that will lead to refinements in the classification framework, secondary indicators, and linkage analysis modeling tools through the development of site-specific endpoints" (SWRCB, 2006, p. 1-1).

Sutula et al. (2007) outlined "a conceptual framework for the development of nutrient numeric endpoints (NNE) for estuaries, and to highlight data gaps and research recommendations critical for their development" (Sutula et al., 2007, p. iv). Ultimately, the goal envisioned was to develop a set of tools for the purpose of supporting the water quality programs of the SWRCB, Regional Water Quality Control Boards (RWQCBs), and the regulated community. The proposed approach was described as having the advantage of "a more robust link to actual impairment of use, rather than an approach that relies on concentration data alone" (Sutula et al., p. iv).

McKee et al. (2011) reviewed literature and data relevant for an assessment of eutrophication in the San Francisco Bay, "with the goal of providing information to formulate a work plan to develop NNEs for this estuary" (McKee et al., 2011, p. iii). The three stated objectives of this review were to: (1) evaluate indicators for assessing eutrophication and other adverse effects of anthropogenic nutrient loading in the San Francisco Bay, (2) summarize the existing literature in the Bay using indicators, while identifying gaps in data, and (3) investigate what data and tools currently exist for evaluating nutrient loading trends. In its executive summary, the report stated that "evidence is building that the historic resilience of SF Bay to the harmful effects of nutrient enrichment is weakening." The report further identified that, although data with which to improve published load estimates from some sources exist, "Nutrient loads to SF Bay from external sources are poorly understood" (McKee et al., 2011, p. iv).

McKee and Gluchowski (2011) presented "new estimates of nitrogen loads for the South Bay, South of the Bay Bridge" (McKee & Gluchowski, 2011, p. 2). Treatment technology at each facility and weather conditions contributed to influencing nutrient concentration in wastewater discharges. The authors further reported that, "Loads of nitrogen from secondary treatment facilities [are] dominated by ammonium at an average ratio of 14:1 NH_4^+:NO_x, whereas for advanced treatment, the ratio is 15:1 NO_x: NH_4^+ favoring NO_x" (McKee & Gluchowski, 2011, p. 13). The authors concluded that, "On an annual average basis, wastewater loads appear to dominate for ammonium based on the calculations presented here," which would still be the case, "even if we assume stormwater ammonium loads are underestimated by a factor of 1.5 times" (McKee & Gluchowski, 2011, p. 22).

The San Francisco Bay Regional Water Quality Control Board (SFRWQCB) has presented "a draft strategy for developing the science needed to make informed decisions about assessing nutrient impacts on water quality, protecting beneficial uses, and managing nutrient loads to San Francisco Bay" (SFRWQCB, 2012, p. 1). In light of the compelling evidence of changing conditions in the San Francisco Bay, in combination with uncertainty about future monitoring programs and new nutrient policies in view, the report stated that "there is a strong need for a coherent nutrient science and management strategy for the Bay" (SFRWQCB, 2012, p. 3). The paper listed several primary anticipated management decisions, including: (1) establishment of Bay nutrient objectives, (2) evaluation of the need for revised objectives for DO (in sub-habitats) and ammonium, (3) development and implementation of a nutrient monitoring program, and (4) specification of nutrient limits in NPDES permits (e.g., municipal and industrial wastewater and municipal stormwater permits), as well as determining additional data collection needs.

The main findings of the May 2013 draft of *San Francisco Bay Nutrient Conceptual Model* by Senn and Novick included, first, the observation that the various changes in the SFB ecosystem during the past decade, in combination with the Bay's high nutrient loads and concentrations, justify growing concerns about elevated levels of nutrients. Second, the paper suggested uncertainty about the future trajectory for SFB. Although, it is plausible that SFB's resilience will be maintained and no additional degradation will occur, it is equally plausible that its resilience "will continue to decline until moderate to severe impairment occurs in some subembayments." Third, the authors acknowledged that widespread impairment was not currently occurring, in spite of evidence consistent with conditions in SFB moving toward a critical juncture (Novick & Senn, 2013, p. i). Changes in phytoplankton community composition and occurrences of harmful algal blooms possibly related to nutrients may represent an exception. "The degree to which impairment is occurring … needs to be a major and early focus of investigation and monitoring" (Novick & Senn, 2013, p. i).

Novick and Senn (2014) have listed the project's four main goals as: (1) use the best available current information to quantify external nutrient loads to SF Bay, (2) explore how current loads vary spatially at the subembayment scale and seasonally, (3) where data permit, assess long-term trends in nutrient loads, and (4) identify major data needs and important uncertainties. The focus of the analysis included loads from POTWs. The authors found that "most POTWs carry out only secondary treatment, which transforms nutrients from organic to inorganic forms, but generally does not remove much N or P" (Novick & Senn, 2014, p. 3), with "the five largest POTWs accounting for approximately 75% of NH_4^+ loads, 50% of NO_3^- loads, and 45% of PO_4 loads of total POTWs Bay-wide" (Novick & Senn, 2014, p. 3).

Method

Study Site

The study site includes the Facility (formerly known as the San José /Santa Clara Water Pollution Control Plant) and the receiving waters of the LSB. The Facility is located at 700 Los Esteros Road in San José, Santa Clara County. The Facility provides tertiary treatment of domestic, commercial, and industrial wastewater to a total service area population of approximately 1.4 million.

The Facility serves multiple cities and wastewater districts: the cities of San José, Santa Clara, and Milpitas; Santa Clara County Sanitation Districts No. 2 and No. 3; the West Valley Sanitation District, including Campbell, Los Gatos, Monte Sereno, and Saratoga; and the Cupertino, Burbank, and Sunol Sanitary Districts. Each associated satellite collection system is owned, operated, and maintained independently from the discharger, and collects wastewater from its respective service area (Figure 5).

The wastewater treatment process at the plant includes screening and grit removal, primary sedimentation, secondary treatment by activated sludge process, secondary clarification, filtration, disinfection, and dechlorination. The Facility has an average dry weather flow design capacity of 167 million gallons per day (MGD) and a 271 MGD peak hourly flow capacity for full tertiary treatment.

Primary treatment refers to the physical processes (settling or skimming) that remove a significant percentage of both the organic and inorganic solids from wastewater. Secondary treatment depends on a biological process known as activated sludge in which a mixture of wastewater and microorganisms is agitated and aerated to allow the microorganisms to break down organic material. This process removes fine suspended solids, dispersed solids, and dissolved organics through volatilization, biodegradation, and incorporation into sludge. Tertiary (advanced) treatment uses a variety of biological, physical, and chemical treatment approaches to reduce nutrients, organics, and pathogens. Tertiary treatment includes nitrification (the biological oxidation of ammonium with oxygen, then into ammonium, then into nitrite followed by the oxidation of these nitrites into nitrates) and filtration (the removal of minute solids to further improve the effluent quality before it is discharged to the receiving environment). BNR removes total nitrogen (TN) and total phosphorus (TP) from wastewater, through the use of microorganisms under different environmental conditions in the treatment process. Nitrification and denitrification are the biological processes responsible for removing nitrogen. During nitrification, ammonium is oxidized to nitrite, after which nitrite is then oxidized to nitrate under aerobic conditions. Denitrification involves the biological reduction of nitrate to nitric oxide, nitrous oxide, and nitrogen gas, under anaerobic conditions. Biological phosphorus removal relies on phosphorus uptake by aerobic heterotrophs capable of storing orthophosphate in excess of their biological growth requirements (Metcalf & Eddy, 2003).

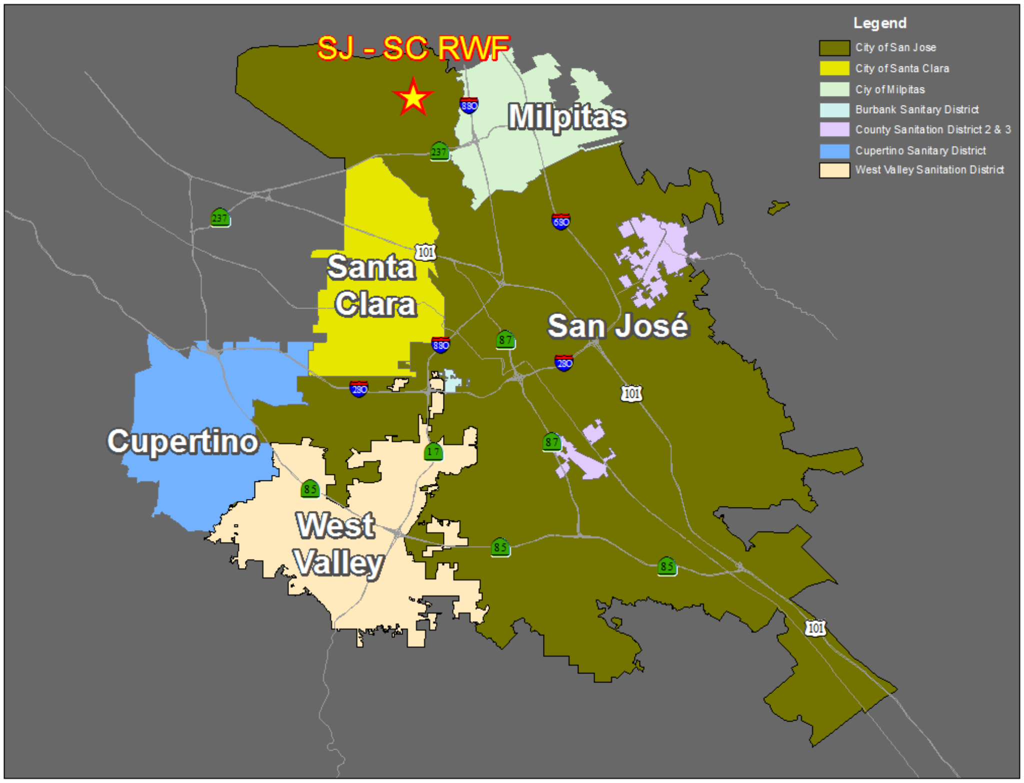

Figure 5. The tributary agencies served by the Facility.

The LSB area is generally considered to extend from the Dumbarton Bridge south to the Southern Pacific Railroad Bridge across Coyote Creek. The treated wastes from the Facility are discharged to Artesian Slough (also called Mallard Slough, 37° 26'23.38" Latitude and 121° 57' 29.18" Longitude), from where it flows by way of Coyote Creek, to the main body of South San Francisco Bay (Consoer, Townsend and Associates, 1968). The discharge point is situated approximately two miles from the Creek, 6.5 miles from South Bay proper (Larry Walker Associates, 1987).

The major factors influencing ambient water quality in the LSB include: location/physical characteristics, climatic conditions, delta outflows, tidal currents, local streamflows/urban runoff, other nonpoint sources, and existing point source discharges. South Bay lies in the Coastal Range between the Santa Cruz Mountains to the west and the Diablo Range to the east. The area's marine-type climate is characterized by mild and moderately wet winters, on the one hand, and cool, dry summers on the other. Approximately three-quarters of the total annual rainfall generally occurs during the winter months, from December to March (Harris et al, 1961).

The South Bay region is characterized by prevailing westerly or north-westerly winds in late spring, summer, and early fall, with more variable conditions in winter. As a result of inland solar heating, diurnal sea breezes reinforce the prevailing westerly summer wind. Winter storm tracks to the south that result in winds from the east or southeast influence the winter pattern (Cheng & Gartner, 1985).

The streams and sloughs that discharge to the South Bay include: Coyote Creek, Alviso Slough, Guadalupe Slough, Stevens Creek, Mountain View Slough, Mayfield/Charleston Slough, San Francisquito Creek, and Mowry Slough/Newark Slough/Plummer Creek. Each of these has its own tributary stream. All tributary streams are intermittent and of local drainage. Sewage water inflows to the southern reach during the summer exceed the natural stream inflows. The southern reach receives 10 % of the mean annual river runoff, and also 76 % of the total wastewater inflow to the Bay (Conomos, 1979).

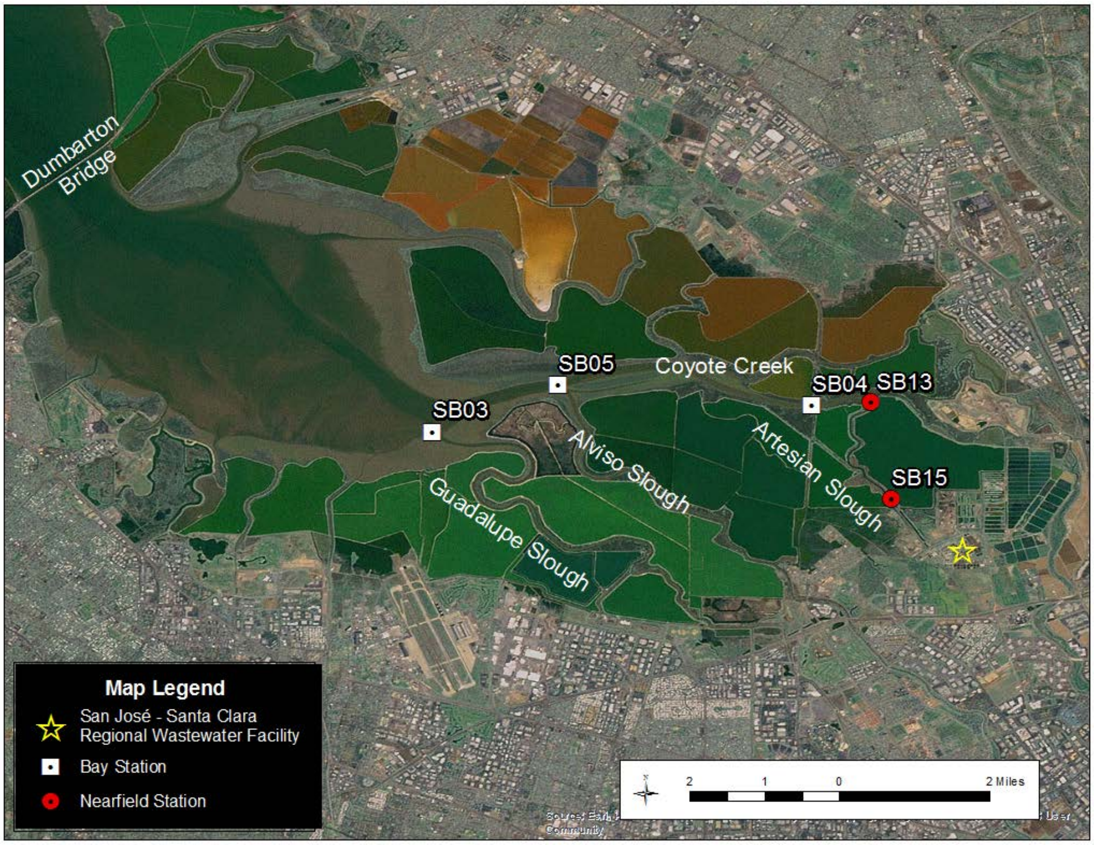

Figure 6. Map of the study area, showing the location of the Facility and the locations of the five stations in the LSB (SB15, SB13, SB04, SB05, and SB03).

Table 1. Location of stations, with approximate latitudes and longitudes (in decimal minutes).

Station	Lat	Long	Reference Locations
SB15	37^0 26.588'	121^0 57.640 '	Effluent discharge mixing point
SB13	37^0 27.683'	121^0 57.871'	Mouth of Artesian Slough
SB04	37^0 27.600'	121^0 58.540'	Coyote Creek Railroad Bridge
SB05	37^0 27.875'	122^0 1.406'	Mouth of Alviso Slough
SB03	37^0 27.437'	122^0 3.033'	Mouth of Guadalupe Slough

Tides throughout the Bay are both mixed and semidiurnal, with two cycles (two low and two high tides) occurring each tidal day. Both the highs and lows in each cycle are usually quite different in terms of height (Conomos, 1979). Due to the fact that the South Bay is an enclosed embayment, the wave reflections from the south end of the bay are superimposed upon the incoming tides from Central Bay, with the tides becoming nearly standing waves (Walters et al., in Cloern & Nichols, 1985). The southern reach, in contrast with the northern reach, is known for seasonally reversing but sluggish near-bottom and surface non-tidal currents. These "are generated by prevailing summer and episodic winter-storm winds, and by winter flows of Delta-derived low-salinity water from the northern reach" (Conomos, 1979, p. 47).

Hydrographic characteristics vary substantially between the landward and seaward reaches of the South Bay, which show different circulation characteristics. According to Schemel (1998), "The term, landward reach, refers to all of the bay landward of San Mateo Bridge, including Lower South Bay" (Schemel, 1998, p. 10).

South Bay salinity varies seasonally, controlled mainly by exchanges with the Pacific Ocean, as well as the northern reach. During wet years, some salinity stratification may be present in winter due to intrusion of low-salinity water from the northern reach into South Bay and local runoff from the south (Conomos, 1979).

Data Collection

Statistical data for effluent and receiving water were collected from the Facility and the Final Technical Report (December 1981-November 1986) of the South Bay Dischargers Authority Water Quality Monitoring Program. The U.S. Census Bureau's website was accessed for population data for the areas served by the Facility. A wide variety of publications—in addition to the *Final Technical Report* (December 1981-November 1986) of the South Bay Dischargers Authority Water Quality Monitoring Program—was utilized to frame the study, provide key background information and contribute to our understanding of the issues.

As authorized by the NPDES Permit, the Facility measured parameters, as described in Table 2 (periods during which no data were recorded are not reported).

Table 2. Facility influent and effluent parameters.

Parameters	Facility Load	Monitoring Period
BOD, TSS	Influent	1957 - 2013
	Effluent	1957 - 2013
NH_4^+	Influent	1965 - 1987 2012 - 2013
	Effluent	1965 - 2013
NO_3^-	Influent	2008-2013
	Effluent	1975 -2013
PO_4	Influent	1965 - 1977 2006 - 2013
	Effluent	1974 - 2013

The Facility conducted compliance monitoring in receiving waters (the LSB) at several stations since 1965, with differing frequency (Table 3). From December 1981 to November 1986, the Facility conducted monitoring of receiving water at 10 stations. This was done jointly with Sunnyvale and Palo Alto, under the South Bay Dischargers Authority. No receiving water quality data were collected from 1993 to 2003. The Facility resumed monitoring at the Bay stations in 2003 and near field stations in 2012. For this study, five stations—two near field stations (SB15 and SB13) and three Bay stations (SB04, SB05, and SB03)—were selected.

Table 3. Summary of LSB ambient water quality parameters.

Parameters	Stations	Monitoring period	Frequency
DO	SB03,SB04,SB05	1963-1993	Bi-weekly
		2003-2009	Monthly
		2009-2012	Quarterly
		2013	Monthly
	SB13,SB15	1963-1993	Bi-weekly
		2012-2013	Monthly
NH_4^+, NO_3^-, PO_4	SB03,SB04,SB05	1975-1993	Bi-weekly
		2003-2009	Monthly
		2009-2012	Quarterly
		2013	Monthly
	SB13,SB15	1975-1993	Bi-weekly
		2012-2013	Monthly

Data Analysis

Data were analyzed with R 3.0.2 (R Development Core Team, 2013) for all calculations and graphs, including use of ggplot2 (Wickham, 2009), and wq package (Jassby & Cloern, 2013). The sequence of data analysis leading to results involved reading, cleaning, deriving, generating wqData, reshaping, and analyzing and visualizing, as depicted in Figure 7, below:

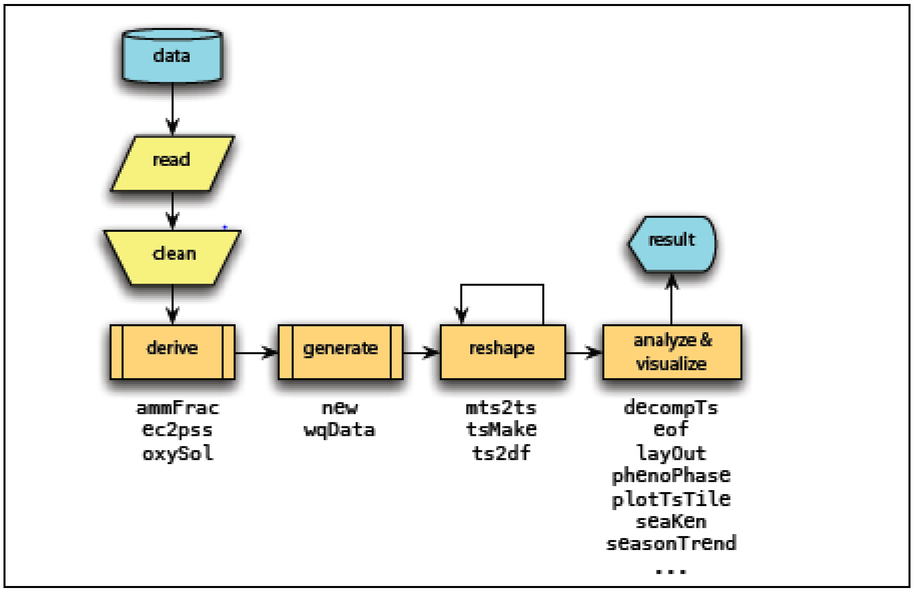

Figure 7. Schematic diagram of the data analysis sequence in the water quality (wq) package. From "wq: Exploring water quality monitoring data" by Jassby and Cloern, 2013. Adapted with permission from the creator, Alan Jassby.

Wastewater loadings, both influent (wastewater flowing into the Facility) and effluent (treated wastewater flowing out of the Facility), were calculated using monthly influent/effluent concentrations and monthly Facility discharge (influent/effluent flow), as reflected in the following formula:

$$\text{Loading (kg/d)} = \text{C (mg/L)} \times \text{Q (MGD)} \times 3.785$$

C = concentration of monthly influent and effluent pollutant discharge, mg/L

Q = monthly influent and effluent flow discharge, MGD

3.785 = conversion factor to convert (mg/L) x (MGD) into kg/day

For periods during which no data were collected, estimated per capita values were used to calculate the influent NH_4^+ and PO_4 loadings.

In the graphs, boxplots have lines within the box, representing the median, and boxes extend from the first through the third quartiles, with the vertical lines extending to all points within 1.5 times the interquartile distance (box height). Local polynomial regression (loess) is used for smoothing lines in graphs.

The approach used for hypothesis #1 (Facility influent flow, BOD, TSS, and nutrient loading increased with increases in population over time) was to track influent and effluent flow relative to population growth over time.

The statistical approach for hypothesis #2 (Facility effluent BOD, TSS, and nutrient loading decreased with Facility treatment process upgrades) was to complete a test of sample means with a two-tailed *t*-test. T-tests were therefore completed to compare the mean BOD, TSS, NH_4^+, NO_3^-, and PO_4 value during a period with the mean BOD, TSS, NH_4^+, NO_3^-, and PO_4 value during the subsequent treatment era (the different eras being: January 1957—primary treatment, February 1964—secondary treatment, February 1979—tertiary/advanced wastewater treatment, July 1997—BNR). The reason for using a two-tailed test was that while it was expected that there would be a decrease in means, the possibility also existed that there was an increase. A two-tailed approach allowed for this possibility.

In order to utilize this approach, it was necessary to first establish that the data were normal. Histograms of each data set were created, after which it was noted that the data were sufficiently normal. An F-test completed for each dataset comparison indicated that all variances were different. For this reason, a Welch two-tailed *t*-test was completed, after which the difference of the means was assessed for significance, based on a 5% significance level.

The statistical approach for hypothesis #3 (There is an inverse correlation between ambient DO concentration in the LSB and both Facility BOD and nutrient loading) was to calculate the correlation between the Facility BOD, NH_4^+, NO_3^- , and PO_4 load with the concentration of DO at the LSB stations. Monthly average effluent loading was paired with monthly average LSB nutrient concentration.

The effluent dataset includes one monthly average record from January 1957 to December 2013. However, the LSB dataset had a large gap, stretching out for a period of approximately ten years (1993-2003), in addition to missing data during other periods. Monthly averages for each variable and station were calculated. Nutrient concentration for the LSB dataset was converted from mg/L to umol/L. Nutrient loads from Facility effluent were compared with LSB DO concentration.

The full dataset was split into five separate datasets based on site location (ordered here based on distance from the facility, from closest to farthest): SB15, SB13, SB04, SB05, and SB03. The datasets were further broken down by DO within each site. Correlation values can range between 1 and -1. A value of 0 indicates no correlation, while a value of 1 indicates perfect correlation. Values in between provide an estimate of the level (degree) of correlation (Cohen, 1988):

- 0 to 0.1: no correlation
- 0.1 to 0.3: weak correlation
- 0.3 to 0.5: moderate correlation
- 0.5 to ~1: strong correlation
- 1: perfect correlation

The statistical approach for hypothesis #4 (Ambient DO and nutrient concentrations follow a pattern relating to temporal, seasonal, and spatial aspects) was to complete a test of sample median with two-sided nonparametric test.

A two-sided nonparametric test was completed to test the temporal (comparing ambient DO and nutrient concentration in the LSB across treatment eras), seasonal (comparing ambient DO and nutrient concentration in the LSB in terms of dry—from May to October—and wet—November to April—season), and spatial (comparing ambient DO and nutrient concentration in the LSB in terms of distance from the point of discharge). First, a histogram of each data set was created to check for normality, after which it was noted that the data were not sufficiently normal. A nonparametric Wilcoxon rank sum test alternative to the *t*-test was used. The difference of the medians was assessed for significance, based on a 5% significance level.

The statistical approach for hypothesis #5 (ambient nutrient concentration in the LSB is positively correlated with Facility effluent nutrient loading) was to analyze the correlation between the NH_4^+, NO_3^-, and PO_4 Facility effluent load and the LSB station concentration levels for those nutrients, utilizing the same approach as for hypothesis #3.

Study Limitations

Even though the Facility collected effluent data since 1957, with the completion of the primary treatment plant, ammonium was collected starting in 1965, and phosphorus beginning in 1975 (missing completely the primary treatment era). The Facility did not measure influent nitrate until 2008. There was a large data gap for influent ammonium (1989-2011) and phosphorus (1978-2006). Comparison for some of the treatment era was not possible.

The Facility started water quality monitoring in the LSB in 1963 (only capturing some data points for the primary treatment era) for some variables, with regular monitoring not implemented until 1975 at the earliest. At first, monitoring was done along Artesian Slough through the lower stretch of Coyote Creek. From 1981 to 1986, the Facility in collaboration with the cities of Sunnyvale and Palo Alto started LSB ambient water quality monitoring (South of Dumbarton Bridge) under the South Bay Discharger Authority, which was formed through a joint power agreement. However, the water monitoring program ceased in 1992, for unknown reasons. This resulted in a data gap of approximately ten years for the Bay station (the station farthest from the Facility), and a 20-year data gap for the near field stations (those closest to the Facility). The substantial and intermittent data gap limits the ability to measure annual seasonal changes of water quality in the LSB over the past five decades.

The Facility's high method of detection limit (MDL) to analyze ammonium LSB data also constrained the ability to do trend analysis and correlation. Data below the MDL level were removed from the data analysis, resulting in loss of data, especially for the farthest stations, where nutrient concentration decreases in proportion to distance from the facility.

Results

The results reported here are organized in the framework of the five hypotheses guiding this study.

1. *Facility influent flow, BOD, TSS, and nutrient loading increased with increases in population over time.* The Facility influent flow increased dramatically (approximately 244%, from an annual average of approximately 36 to 124 MGD) along with service area population increases (approximately 225%, from approximately 400,000 to 1.3 million) from the late 1950s through the late 1990s, with the exception of a period of drought that occurred between 1987 to 1992 (a less severe drought in 1976-1977 had relatively little effect in terms of influent flow) (Figure 8). In spite of continuing population increases (approximately 6% from 2000 to 2010), influent flow began to decline in the late 1990s, until the date for which most recent data are available (December 2013), with the level of influent flow at that time returning to the level of the early 1980s (107 MGD). The data for the Facility effluent flow increase are identical to that of the influent flow increase with the exception that by December 2013, the effluent flow returned to the level of the late 1970s (89 MGD) (Figure 9).

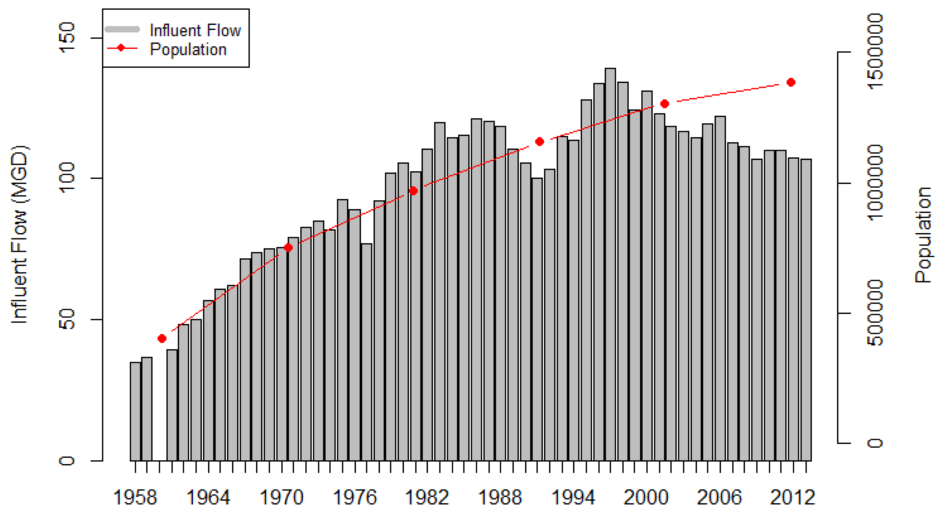

Figure 8. Service area population growth and Facility influent flow.

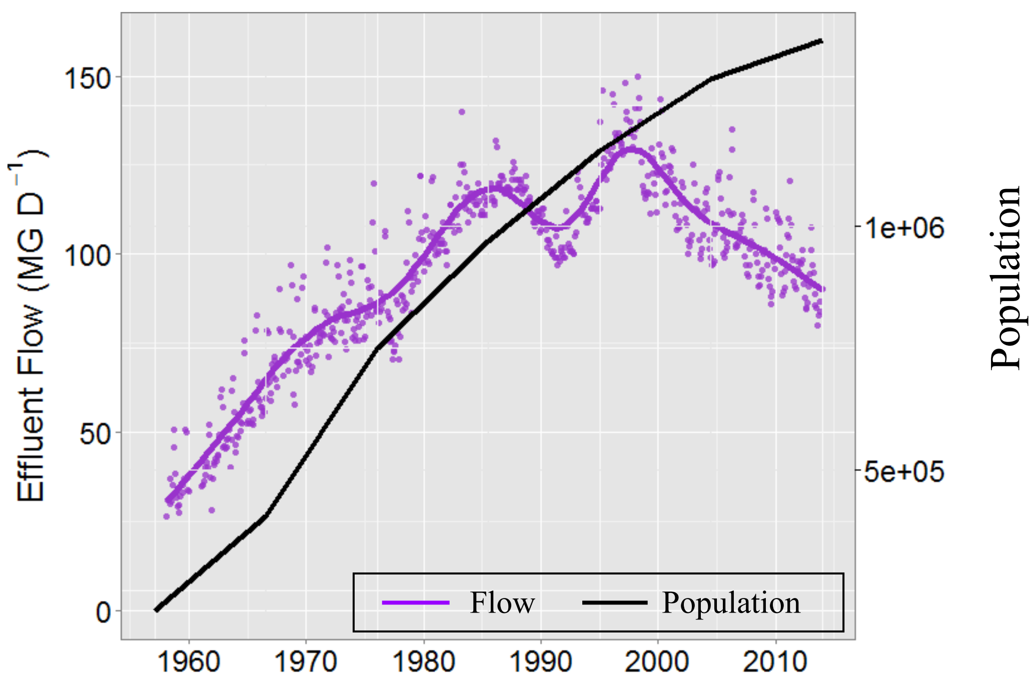

Figure 9. Facility influent flow over time, showing the association with changes in population. The curved line represents a loess smoother (span = 0.3), indicating a general pattern, not intended as a rigorous trend analysis.

Influent BOD loading showed a steady overall increase of approximately 215% (approximately 51,000 to 161,000 kg/d), from the late 1950s through 2010, with a dip during the period from the late 1980s to the early 1990s (roughly corresponding to the drought from 1987-1992), after which point it began to drop off, to an annual average of approximately 127,000 kg/d by December 2013, the level equivalent to approximately 1989 (Figure 10).

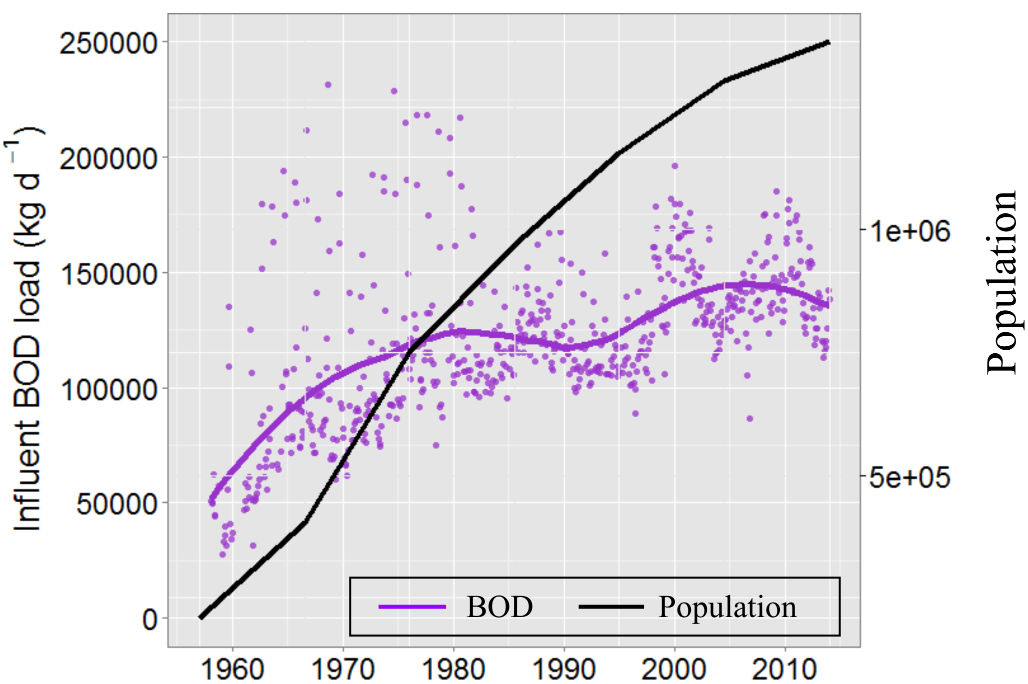

Figure 10. Facility influent BOD load over time, showing the association with changes in population. The curved line represents a loess smoother (span = 0.5), indicating a general pattern, not intended as a rigorous trend analysis.

Influent TSS loading showed an increase of approximately 160% (approximately 57,700 to nearly 150,000 kg/d), from the late 1950s through just after 2000, with a dip during the period from 1980-1995, after which point it declined steadily, to an annual average of approximately 113,000 kg/d by December 2013, the level equivalent to the mid-1960s (Figure 11). Figures 10 and 11 show that although BOD and TSS loads increased over time together with increasing population, BOD and TSS load increases did not "keep up" with increasing population. In other words, BOD and TSS loads did not increase proportionally with population, even though they followed the general rising trend (Figure 12). Possible reasons for this are covered in the discussion section below.

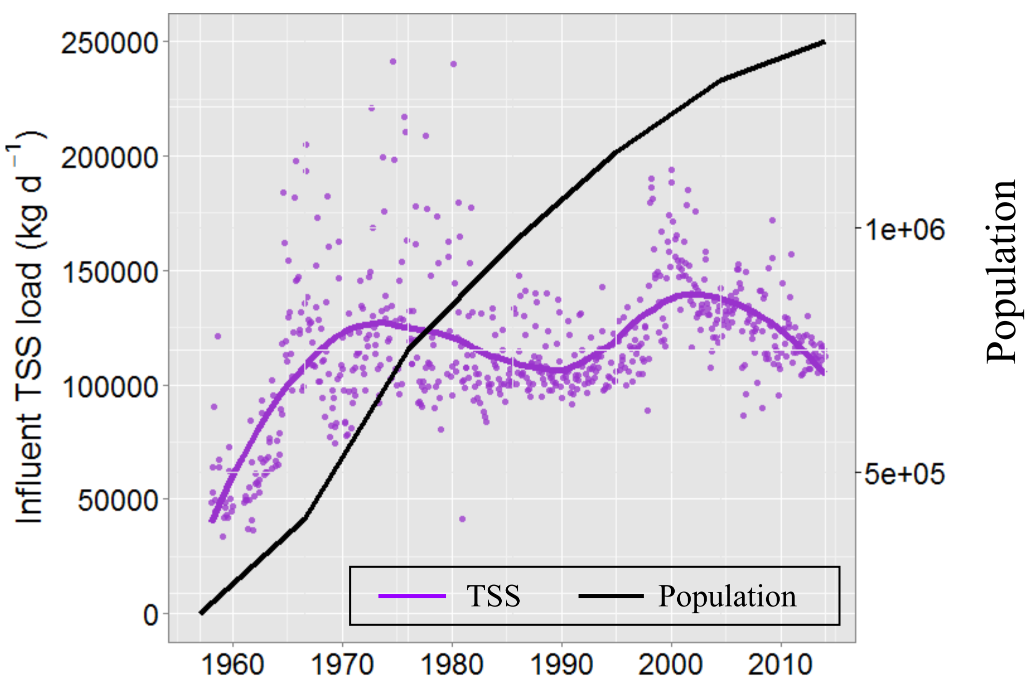

Figure 11. Facility influent TSS load over time, showing the association with changes in population. The curved line represents a loess smoother (span = 0.5), indicating a general pattern, not intended as a rigorous trend analysis.

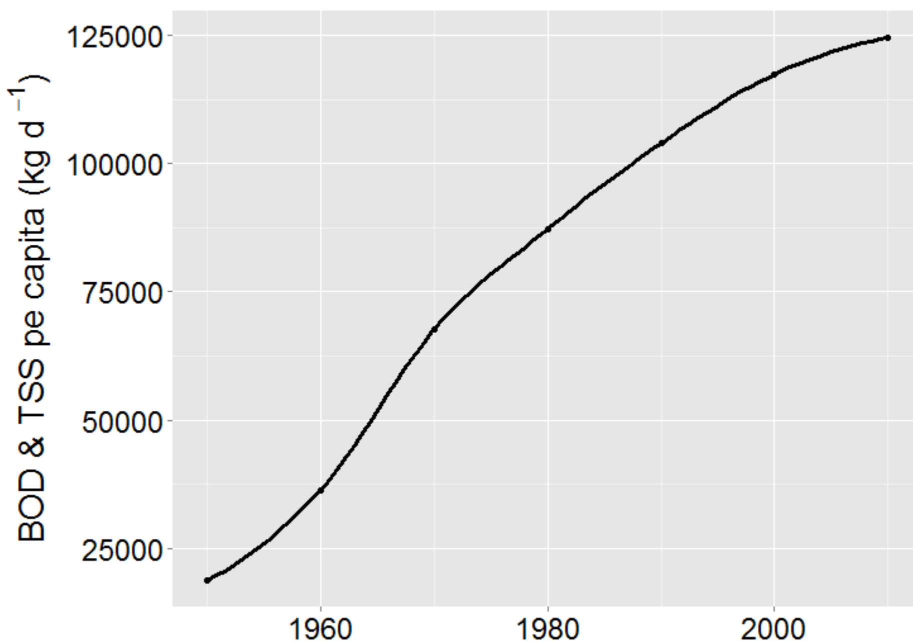

Figure 12. Facility influent BOD and TSS loads, calculated based on per-capita estimates, extrapolating data during the period from 1950-2010.

Influent ammonium loading showed an increase of approximately 66% (just under 6,000 to approximately 10,000 kg/d), from 1965 to 1989. From the period beginning in 2011 to December 2013, the level has been about 30% higher than it was in the late 1980s (Figure 13). To help fill in the data gap from 1977-2010, data per capita was used (Figure 14).

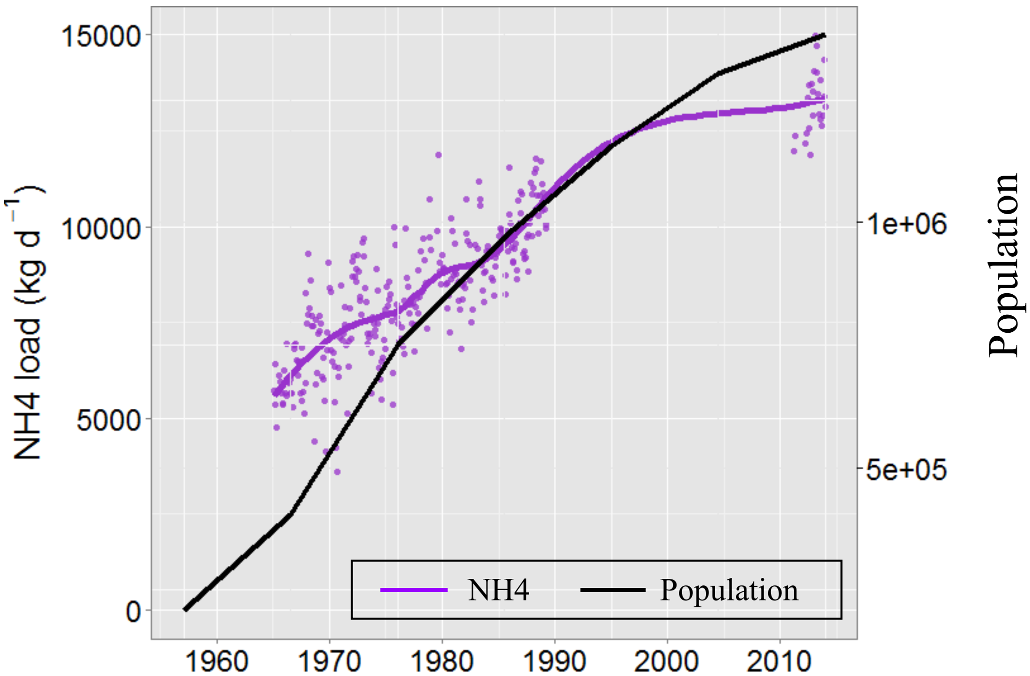

Figure 13. Facility influent NH_4^+ load, showing the association with changes in population, with a data gap for the period from 1989-2012. The curved line represents a loess smoother (span = 0.5) representing a general pattern, not intended as a rigorous trend analysis.

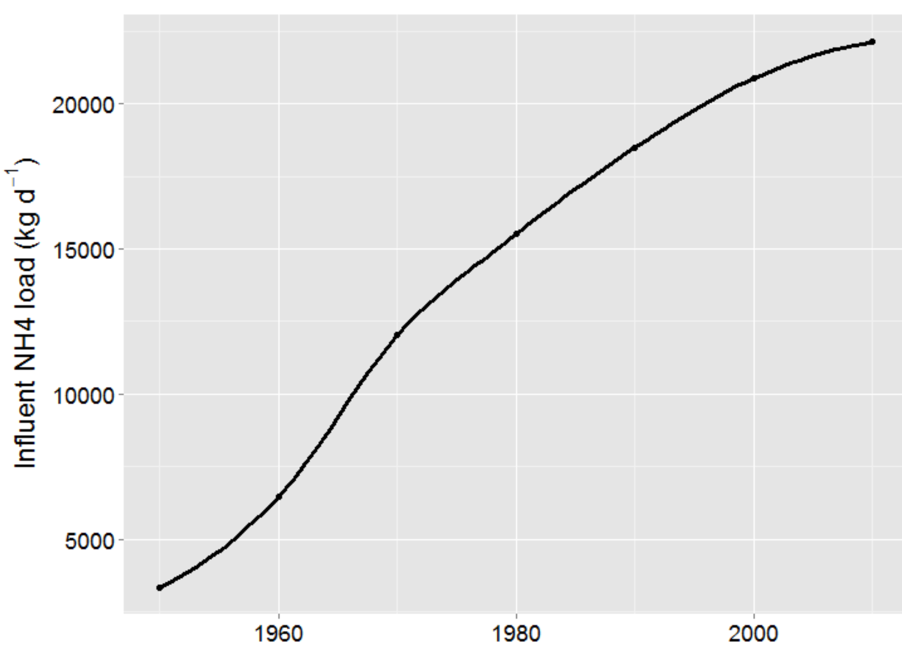

Figure 14. Facility influent NH_4^+ load, calculated based on per-capita estimates, extrapolating for a data gap during the period from 1977-2010.

Influent phosphate loading held steady from 1965 to 1977, at an average rate of 2,319 kg/d during that time period. There is a data gap from 1978 to 2005. From the period beginning in 2006 to December 2013, the average rate of 3629 kg/d was approximately 56% higher than during the period from 1965 to 1989 (Figure 15). To help fill in the data gap from 1978-2006, data per capita was used (Figure 16).

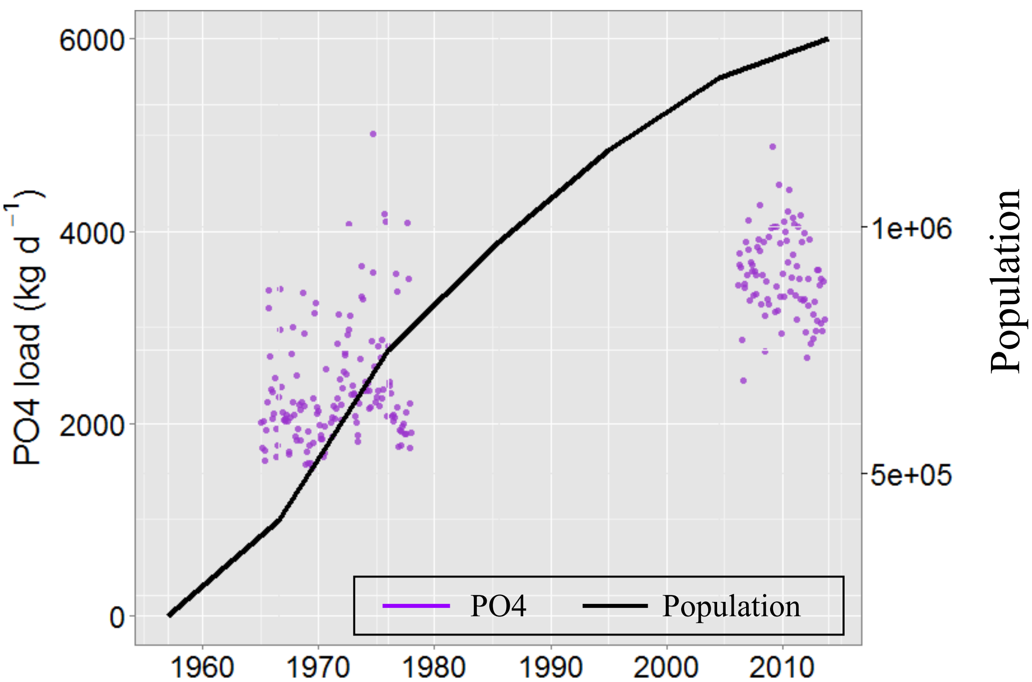

Figure 15. Facility influent PO_4 load, showing the association with changes in population, with a data gap for the period from 1978-2006.

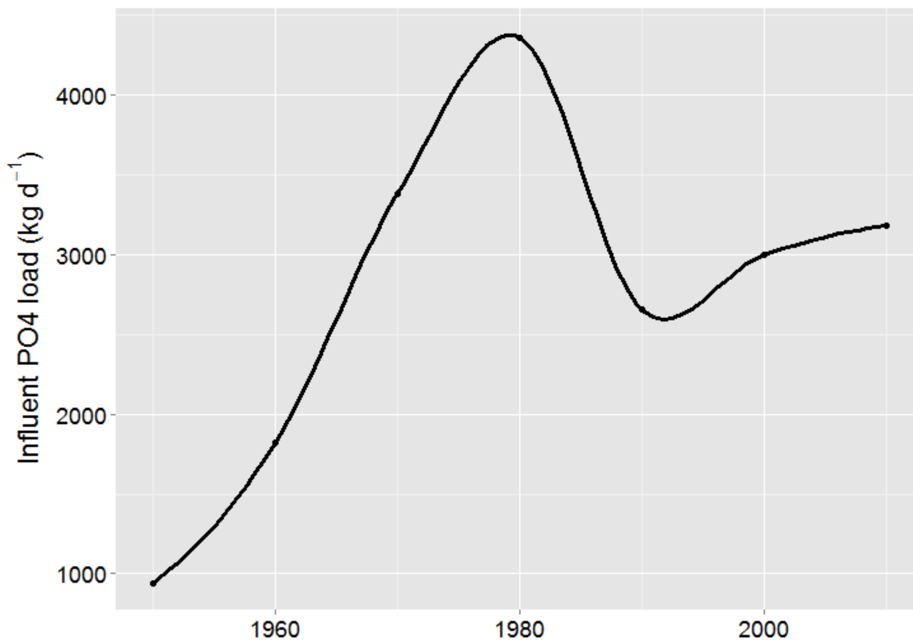

Figure 16. Facility influent PO_4 load, calculated based on per-capita estimates, extrapolating for a data gap during the period from 1978-2006.

2. *Facility effluent BOD, TSS, and nutrient loading decreased with Facility treatment process upgrades*. Based on monthly averages, effluent BOD load dropped from 47,138 kg/d (during the period from January 1957 to January 1964) to 10,318 kg/d (during the period from February 1964 to January 1979) to 1,667 kg/d (during the period from February 1979 to June 1997) to 1,439 (during the period from July 1997 to December 2013), corresponding to the wastewater improvement periods of primary treatment, secondary treatment, tertiary/advanced wastewater treatment, and BNR) (Figure 17 & 18). The percentage of BOD removal from one treatment era to the next was 31%, 91%, and 99%, respectively.

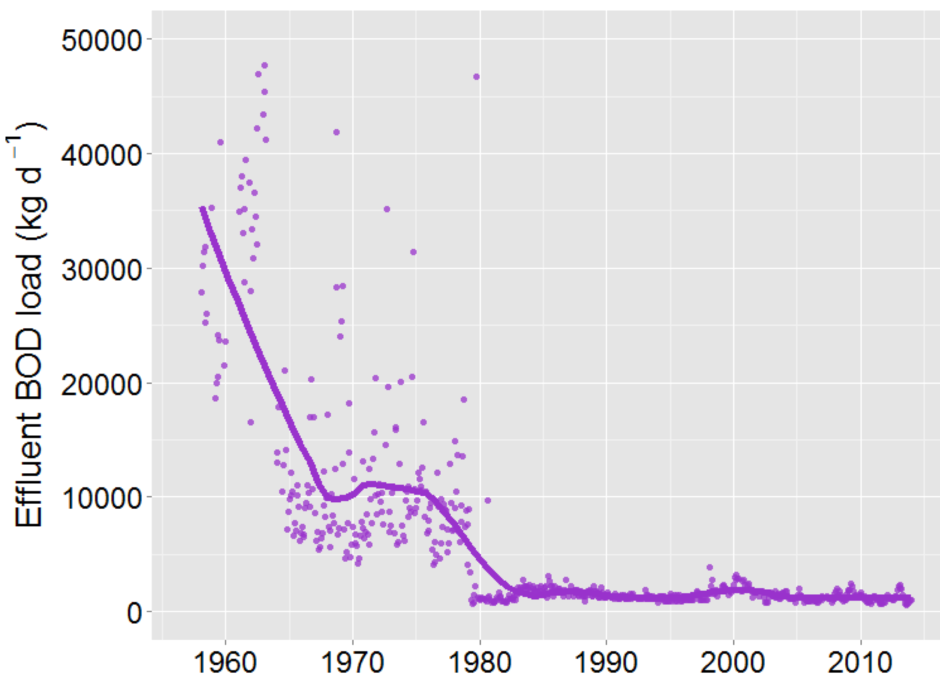

Figure 17. Facility effluent BOD load over time. The curved line represents a loess smoother (span = 0.3) representing a general pattern, not intended as a rigorous trend analysis.

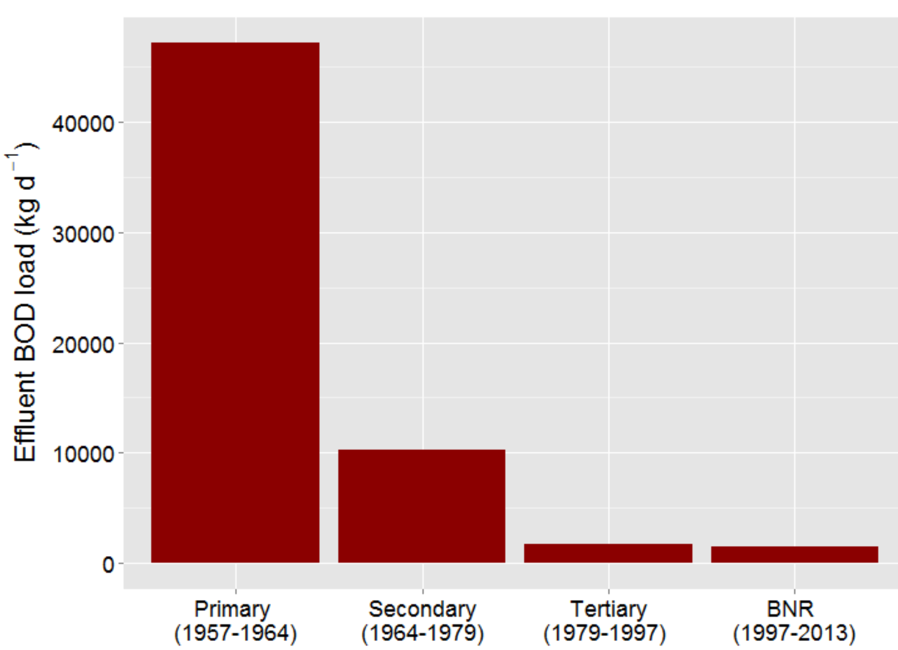

Figure 18. Facility effluent BOD load change between treatment eras: primary (1957-1964), secondary (1964-1979), tertiary (1979-1997), and BNR (1997-2013).

The p-values obtained from the tests showed statistically significant decline in BOD and TSS effluent load from one treatment era to another, with the exception of the p-value between February 1979 to June 1997 vs. July 1997 to December 2013 (Tables 4 to 6).

Table 4. Results of Welch's t-test for testing mean differences between two treatment eras (primary and secondary), for BOD and TSS effluent load. Significant factors ($p < 0.05$) are in bold text.

	Variance		Welch's two-sample t-test						
								95% Confidence Interval of the Difference	
	F	Sig.	t	df	sig.	Mean Difference	Std Error Difference	Lower	Upper
BOD	37.39	<2.2e-16	7.81	52.835	**2.30e-10**	36820	4715	27363	46278
TSS	0.62	0.03954	5.15	118.623	**1.047e-06**	8141	1581	5011	11271

Table 5. Results of Welch's t-test for testing mean differences between two treatment eras (secondary and tertiary), for BOD and TSS effluent load. Significant factors ($p < 0.05$) are in bold text.

	Variance		Welch's two-sample t-test						
								95% Confidence Interval of the Difference	
	F	Sig.	t	df	sig.	Mean Difference	Std Error Difference	Lower	Upper
BOD	3.20	6.66e-16	18.41	262.12	**2.20e-16**	8650	470	7725	9576
TSS	18.30	<2.2e-16	14.36	192.63	**<2.2e-16**	13528	942	11671	15386
NH_4^+	53.68	<2.2e-6	30.72	173.85	**<2.2e-16**	4847	158	4536	5158
NO_3^-	0.24	1.10e-07	-42.32	145.94	**<2.2e-16**	-6196	146	-6485	-5907
PO4	0.54	0.005279	3.87	131.06	**0.00017**	1495	386	171	528

Table 6. Results of Welch's t-test for testing mean differences between two treatment eras (tertiary and BNR), for BOD and TSS effluent load. Significant factors ($p < 0.05$) are in bold text.

	Variance		Welch's two-sample t-test						
								95% Confidence Interval of the Difference	
	F	Sig.	t	df	sig.	Mean Difference	Std Error Difference	Lower	Upper
BOD	32.77	<2.2e-16	1.07	235.02	0.2879	228	214	-194	650
TSS	47.50	<2.2e-16	0.77	230.37	0.4434	150	196	-235	536
NH_4^+	2.50	1.41E-10	11.69	376.72	**<2.2e-16**	264	23	219	308
NO_3^-	2.83	4.79E-13	29.28	365.00	**<2.2e-16**	3503	120	3268	3739
PO4	4.25	<2.2e-16	24.91	332.73	**<2.2e-16**	1495	60	1377	1613

Effluent TSS load dropped from 22,555 kg/d (during the period from January 1957 to January 1964) to 14,414 kg/d (during the period from February 1964 to January 1979) to 886 kg/d (during the period from February 1979 to June 1997) to 736 (during the period from July 1997 to December 2013), corresponding to the wastewater improvement periods of primary treatment, secondary treatment, tertiary/advanced wastewater treatment, and BNR (Figures 19 & 20). The percentage of TSS removal from one treatment era to the next was 63%, 88%, and 99%, respectively. The p-values obtained from the Welch's t-tests showed statistically significant decline from one treatment era to another, with the exception of the p-value for the comparison between February 1979 to June 1997 vs. July 1997 to December 2013 (Tables 4 to 6).

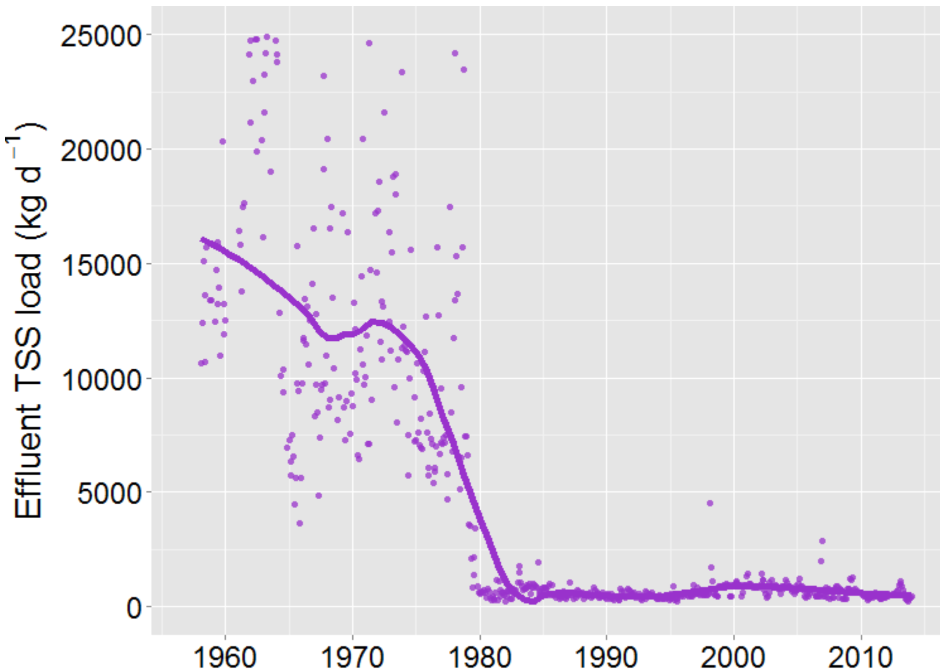

Figure 19. Facility effluent TSS load over time. The curved line represents a loess smoother (span = 0.3) representing a general pattern, not intended as a rigorous trend analysis.

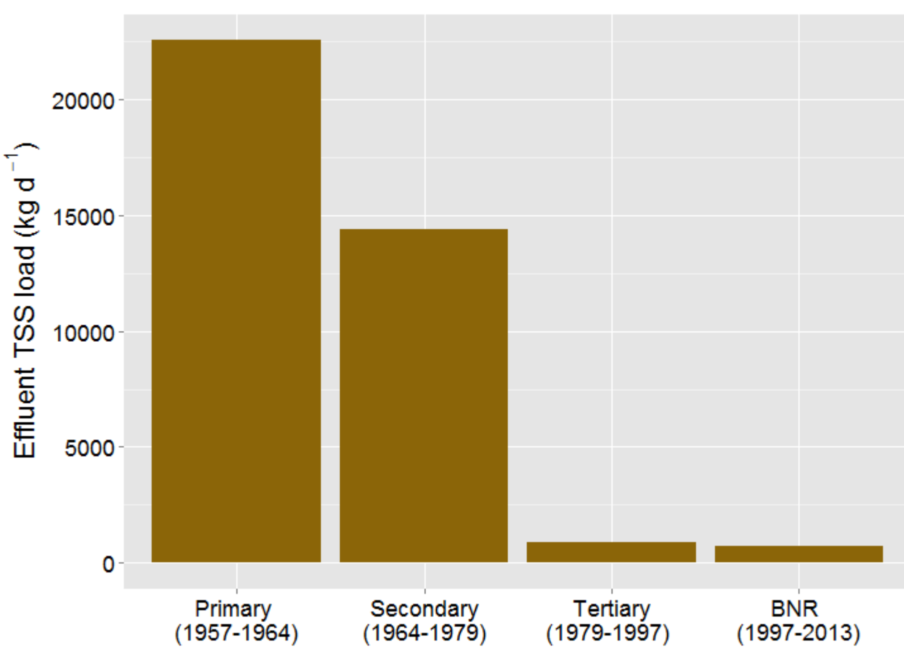

Figure 20. Facility effluent BOD load change between treatment eras: primary (1957-1964), secondary (1964-1979), tertiary (1979-1997), and BNR (1997-2013).

Effluent ammonium load dropped from 5,386 kg/d (during the period from February 1964 to January 1979) to 539 kg/d (during the period from February 1979 to June 1997) to 275 (during the period from July 1997 to December 2013), corresponding to the wastewater improvement periods of secondary treatment, tertiary/advanced wastewater treatment, and BNR (Figures 21 & 23). (The primary treatment era is not covered because no data exist for this period.) The percentage of ammonium removal from one treatment era to the next was 26% and 94%. The *p*-values obtained from the Welch's *t*-tests showed a statistically significant decline from one treatment era to another (Tables 4 to 6).

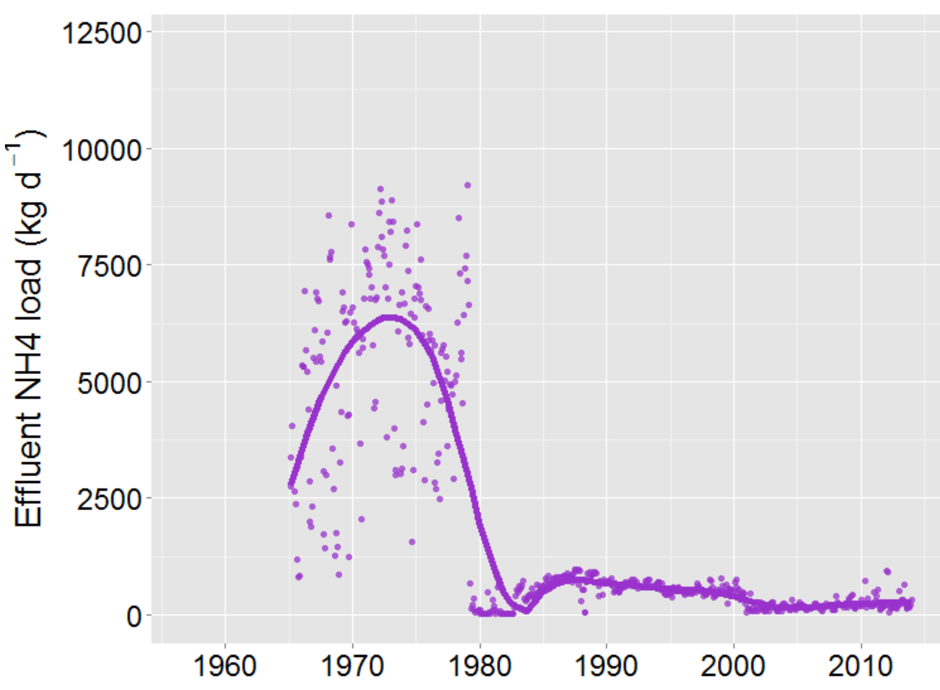

Figure 21. Facility effluent NH_4^+ load over time. The curved line represents a loess smoother (span = 0.3) representing a general pattern, not intended as a rigorous trend analysis.

Effluent nitrate load first jumped from 1,194 kg/d (during the period from February 1964 to January 1979) to 7,389 kg/d (during the period from February 1979 to June 1997), and then declined to 3,886 (during the period from July 1997 to December 2013), corresponding to the wastewater improvement periods of secondary treatment, tertiary/advanced wastewater treatment, and BNR (Figures 22 & 23). (The primary treatment era is not covered because no data exist for this period.) The percentage of nitrate removal from nitrification to BNR was 47%. The *p*-values obtained from the Welch's *t*-tests reflected this increase, followed by a decrease (Tables 4 to 6).

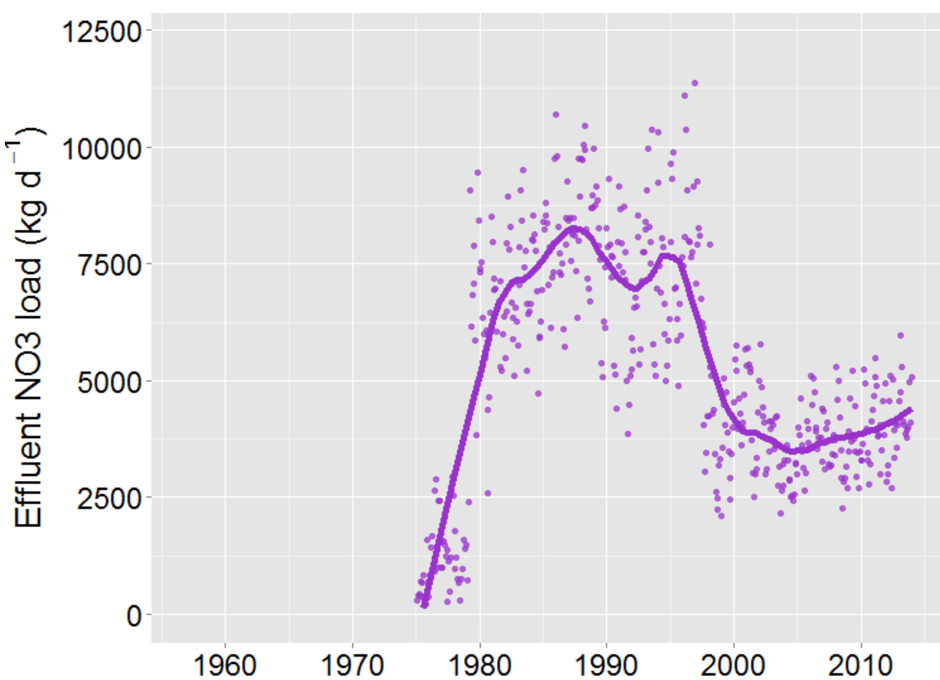

Figure 22. Facility effluent NO_3^- load over time. The curved line represents a loess smoother (span = 0.3) representing a general pattern, not intended as a rigorous trend analysis.

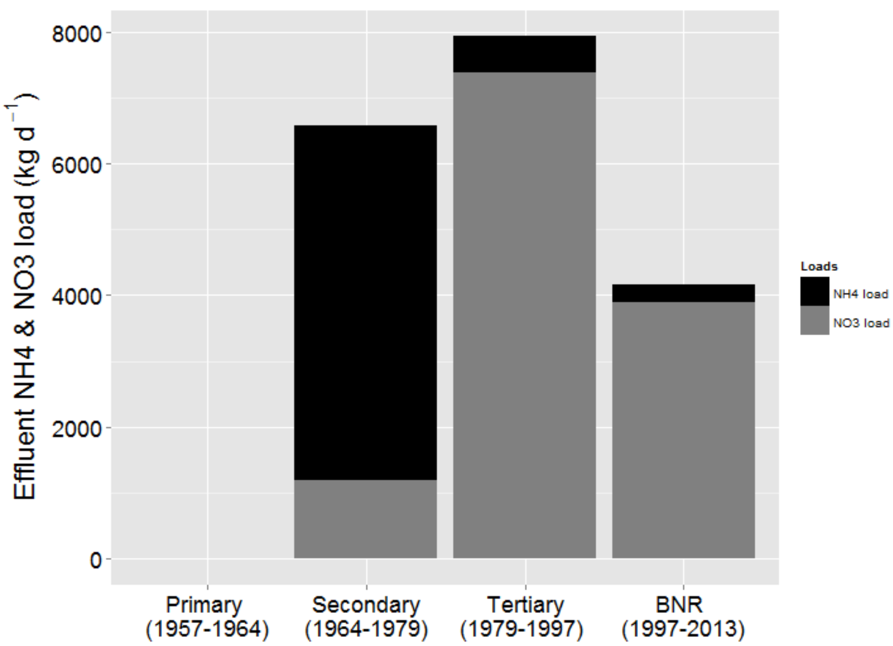

Figure 23. Facility effluent NH_4^+ and NO_3^- load change between three treatment eras: secondary (1964-1979), tertiary (1979-1997), and BNR (1997-2013).

Effluent phosphate load dropped from 2,261 kg/d (during the period from February 1964 to January 1979) to 1,912 kg/d (during the period from February 1979 to June 1997) to 417 (during the period from July 1997 to December 2013), corresponding to the wastewater improvement periods of secondary treatment, tertiary/advanced wastewater treatment, and BNR (Figures 24 & 25). (The primary treatment era is not covered because no data exist for this period.) The percentage of phosphate removal from one treatment era to the next was 56% and 89%. The *p*-values obtained from the Welch's *t*-tests showed a statistically significant decline from one treatment era to another (Tables 4 to 6).

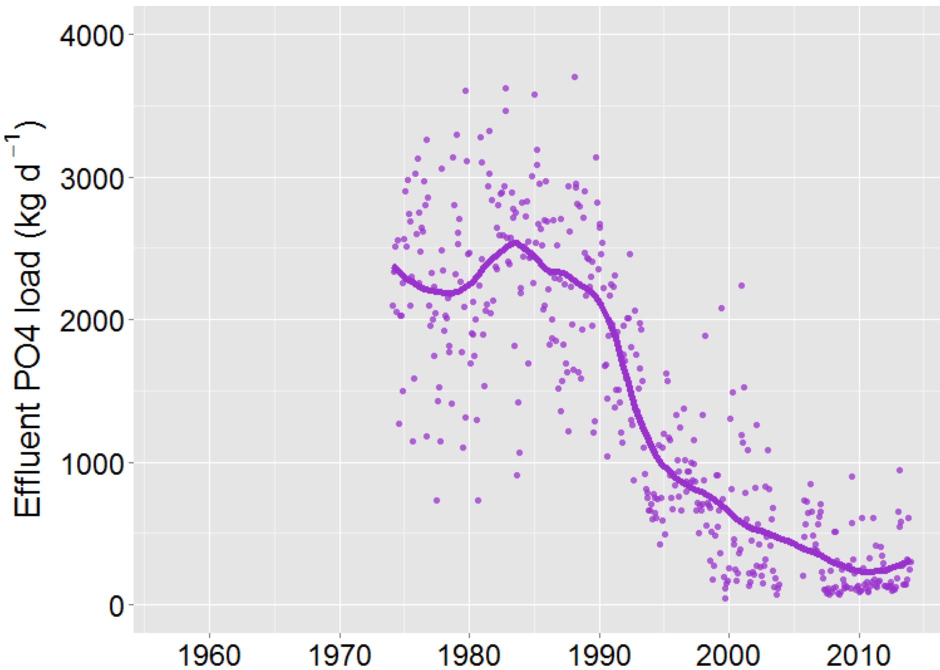

Figure 24. Facility effluent PO_4 load over time. The curved line represents a loess smoother (span = 0.3) representing a general pattern, not intended as a rigorous trend analysis.

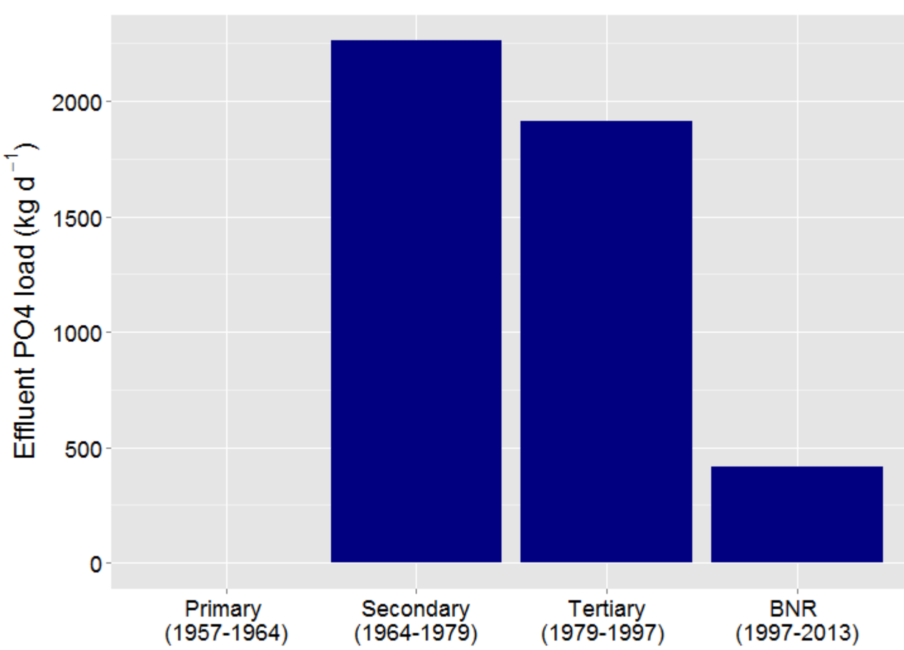

Figure 25. Facility effluent PO_4 load change between three treatment eras: secondary (1964-1979), tertiary (1979-1997), and BNR (1997-2013).

3. *There is an inverse correlation between ambient DO concentration in the LSB and both Facility BOD and nutrient loading*. Table 7 summarizes the information in Figures 26 through 33, showing that Facility effluent BOD and nutrient loading were negatively correlated with nutrient concentration in the LSB, with the exception of NO_3^-, for which the correlation was positive. Of the four nutrients tested, PO_4 showed the weakest correlation. (Only Stations SB15 and SB03 are represented in Figures 26 through 33, in order to show the closest station and the one that is farthest away from the Facility.)

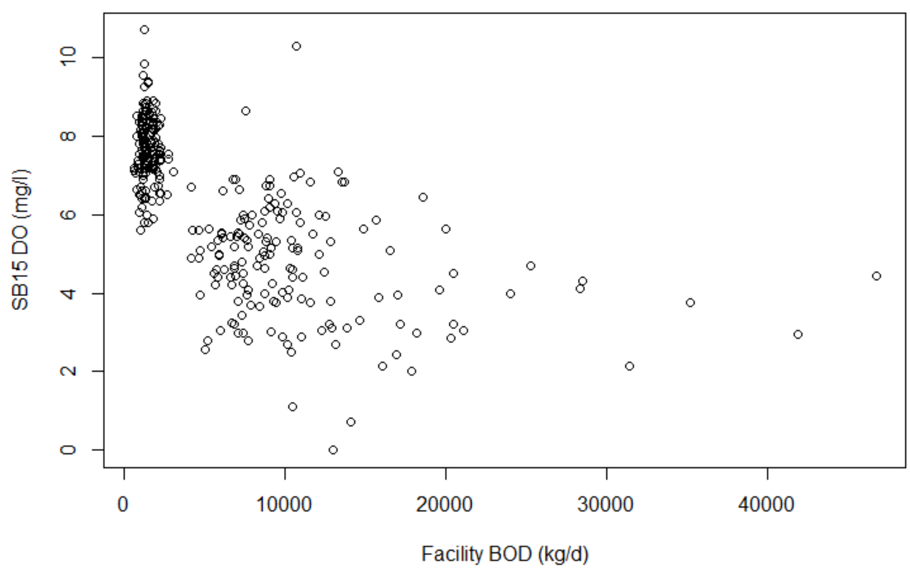

Figure 26. Scatter plot showing the relationship between DO concentration in the LSB and BOD load from the Facility (using monthly means) for SB15, showing a negative correlation.

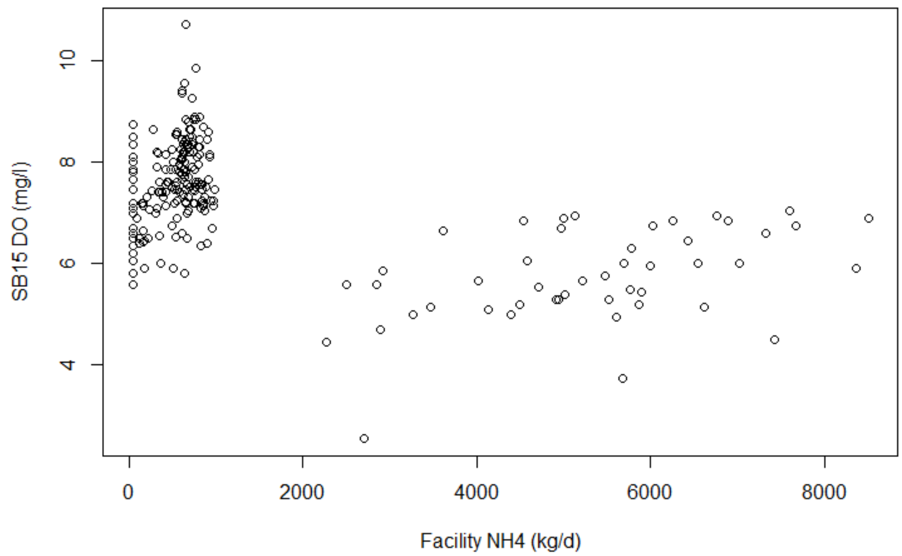

Figure 27. Scatter plot showing the relationship between DO concentration in the LSB and NH_4^+ load from the Facility (using monthly means) for SB15, showing a negative correlation .

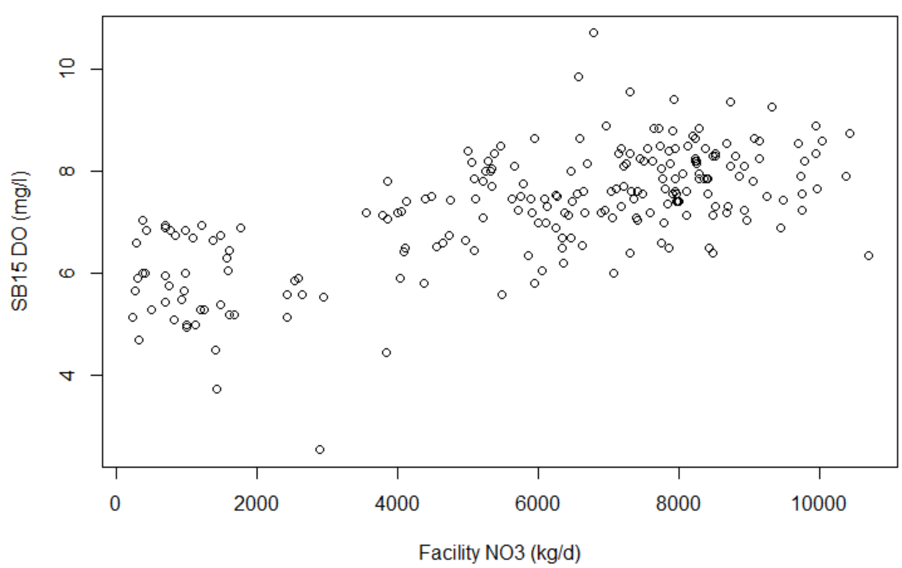

Figure 28. Scatter plot showing the relationship between DO concentration in the LSB and NO_3^- load from the Facility (using monthly means) for SB15, showing a positive correlation.

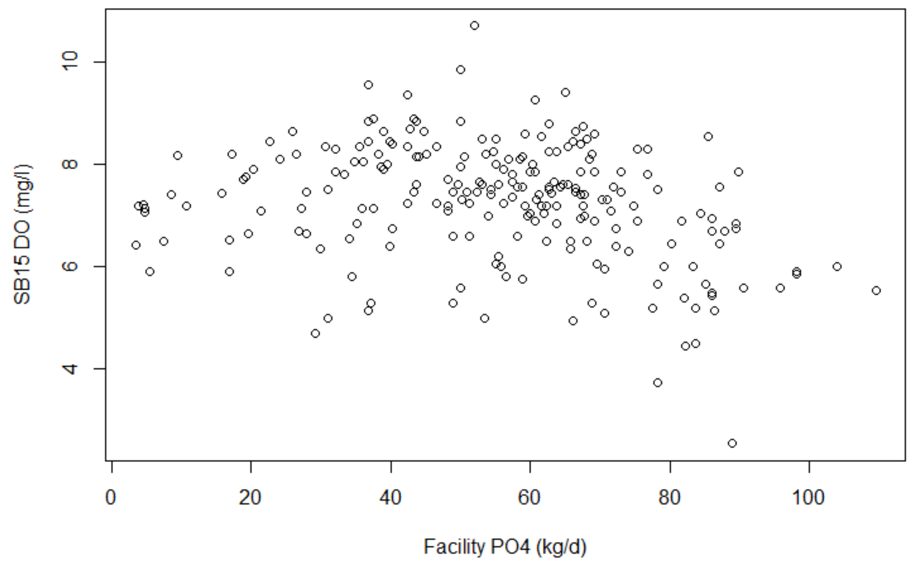

Figure 29. Scatter plot showing the relationship between DO concentration in the LSB and PO_4 load from the Facility (using monthly means) for SB15, showing a negative correlation.

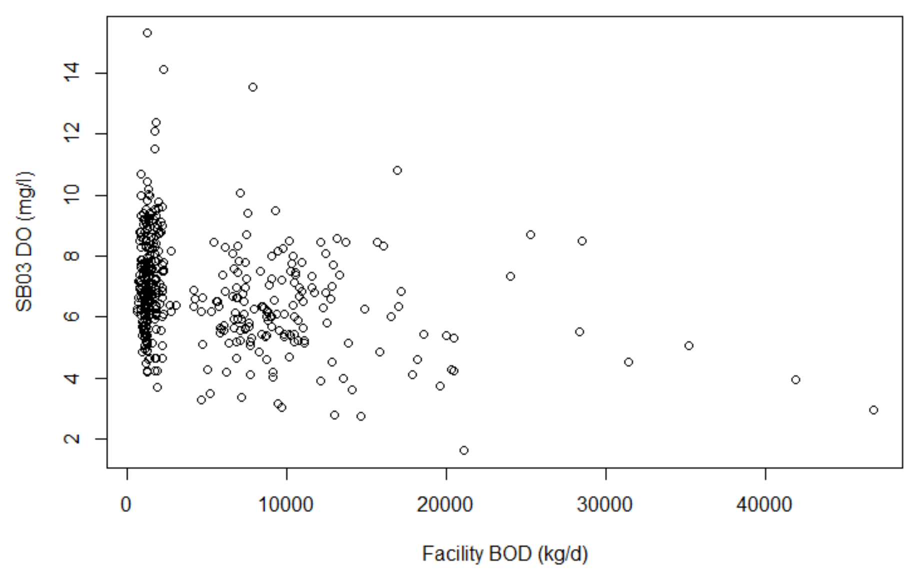

Figure 30. Scatter plot showing the relationship between DO concentration in the LSB and BOD load from the Facility (using monthly means) for SB03, showing a negative correlation.

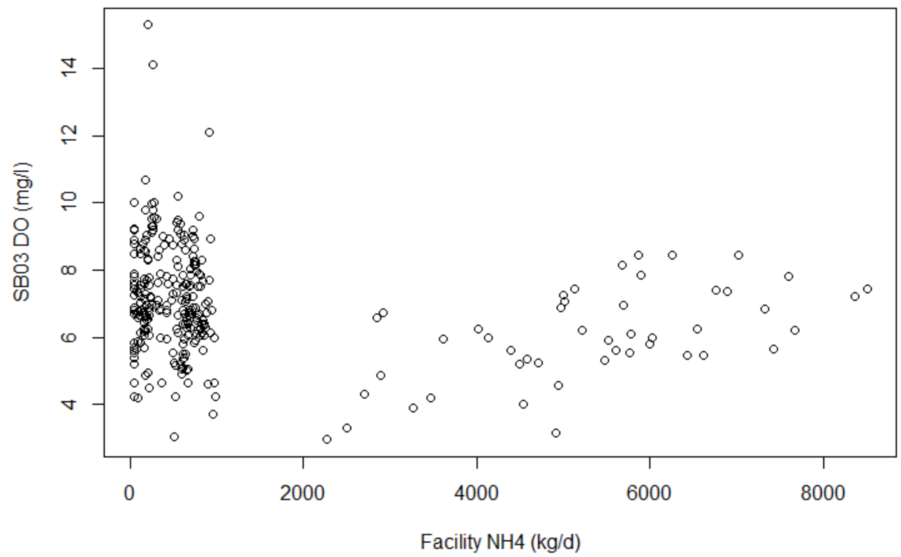

Figure 31. Scatter plot showing the relationship between DO concentration in the LSB and NH_4^+ load from the Facility (using monthly means) for SB03, showing a negative correlation.

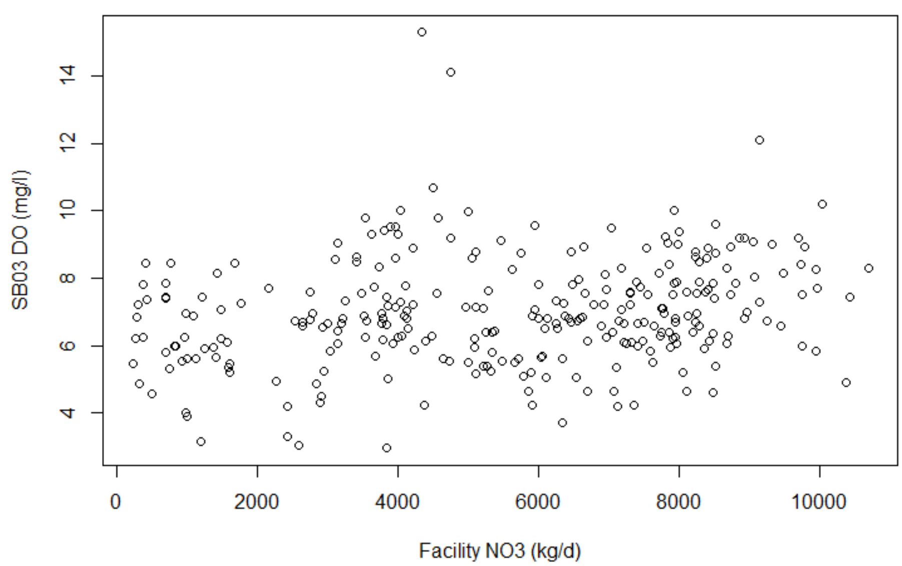

Figure 32. Scatter plot showing the relationship between DO concentration in the LSB and NH_4^+ load from the Facility (using monthly means) for SB03, showing a positive correlation.

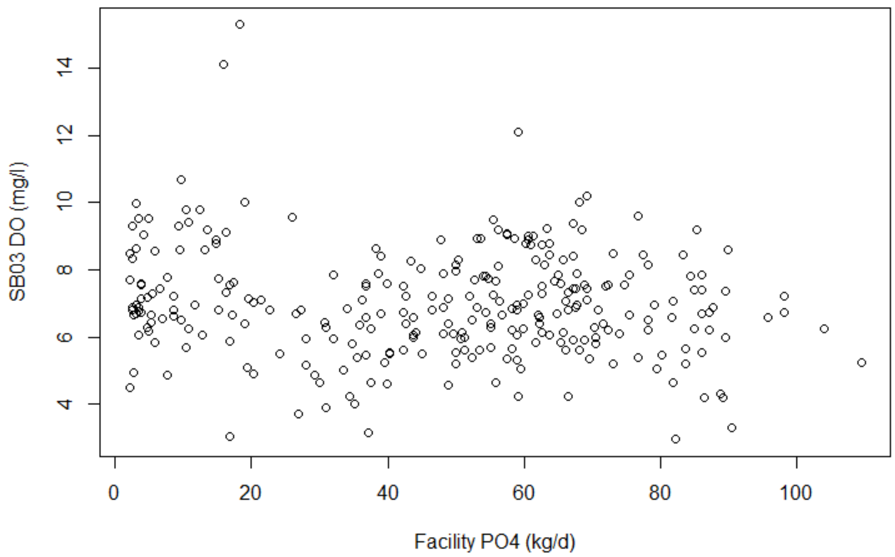

Figure 33. Scatter plot showing the relationship between DO concentration in the LSB and PO_4 load from the Facility (using monthly means) for SB03, showing a negative correlation.

Table 7. Results of Pearson Correlation, between DO concentration in the LSB and BOD, $NH4^+$, NO_3^-, and PO_4 load from the Facility (using monthly means), for all five stations.

	BOD - DO	NH_4^+ - DO	NO_3^- - DO	PO_4 - DO
SB15	-0.66	-0.55	0.68	-0.26
SB13	-0.45	-0.47	0.63	-0.18
SB04	-0.42	-0.43	0.49	-0.17
SB05	-0.37	-0.26	0.34	-0.12
SB03	-0.29	-0.19	0.22	-0.13

The strength of the correlation decreases in direct correspondence to the distance between the Facility and the site with the exception of PO_4 at the farthest station (Figures 34-37).

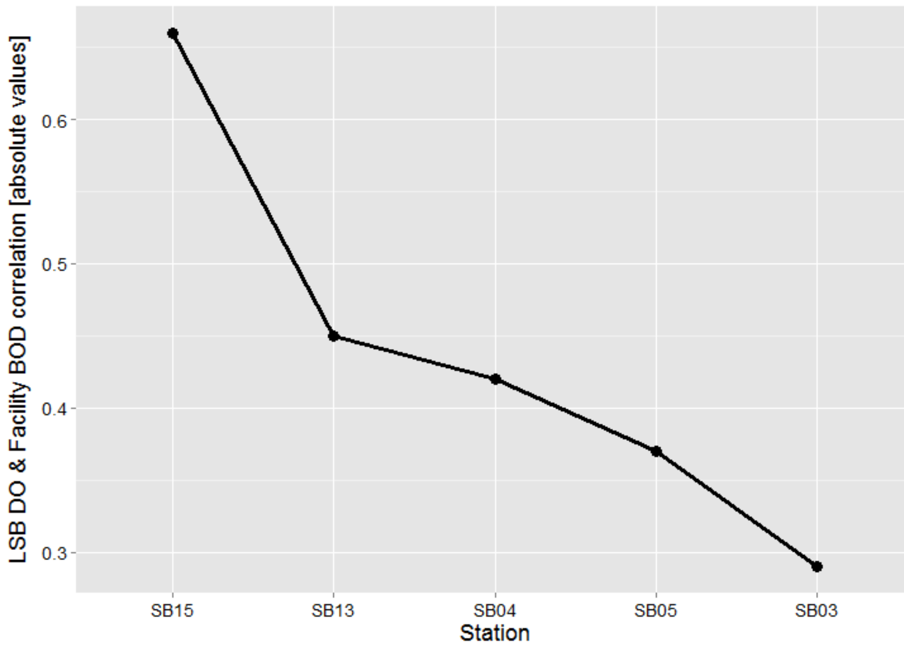

Figure 34. Plot showing how correlation change between LSB DO concentration and Facility BOD load with increasing distance (left to right) from the Facility (absolute values).

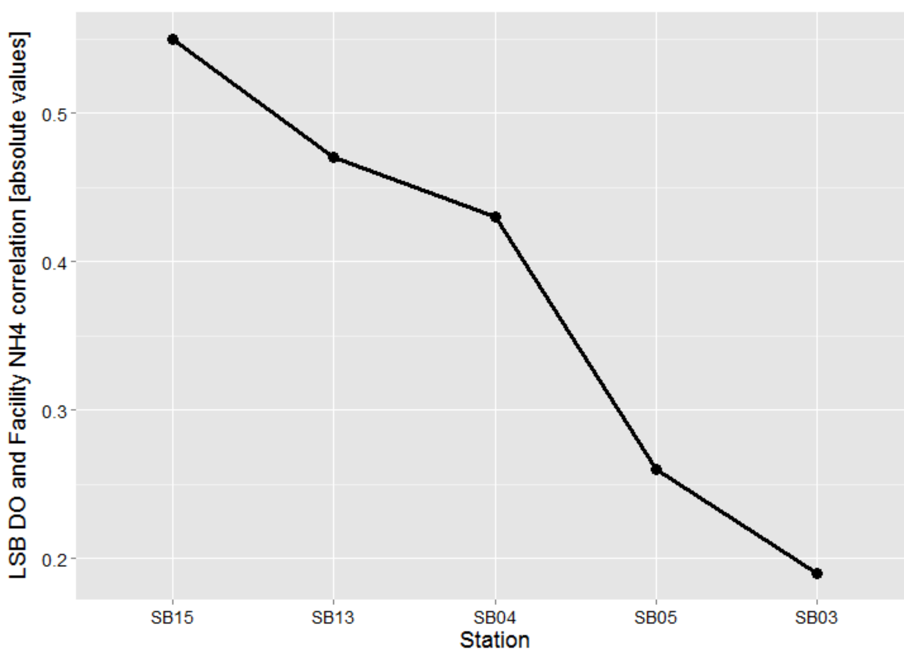

Figure 35. Plot showing how correlation change between LSB DO concentration and Facility NH_4^+ load with increasing distance (left to right) from the Facility (absolute values).

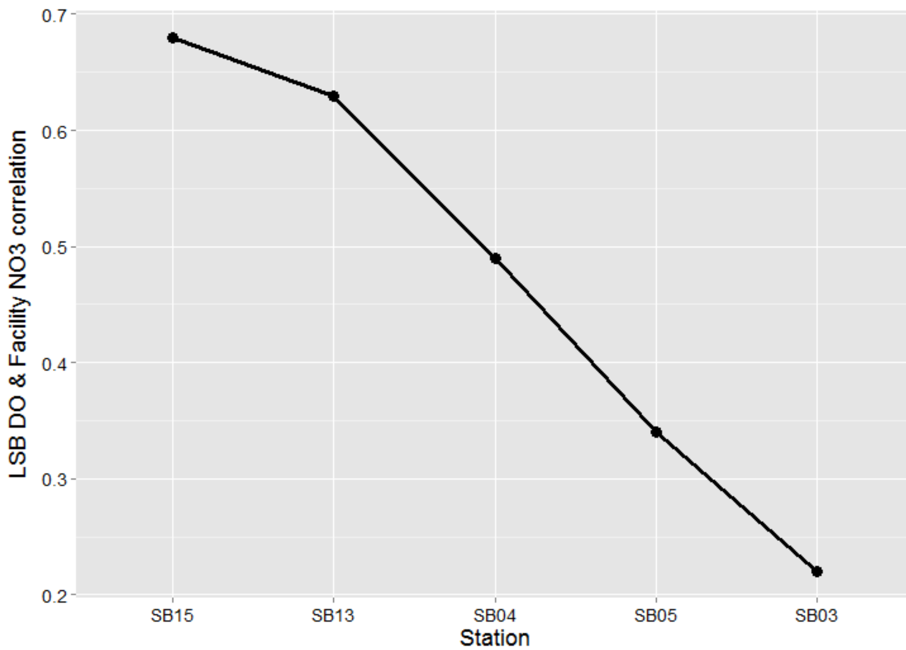

Figure 36. Plot showing how correlation change between LSB DO concentration and Facility NO_3^- load with increasing distance (left to right) from the Facility.

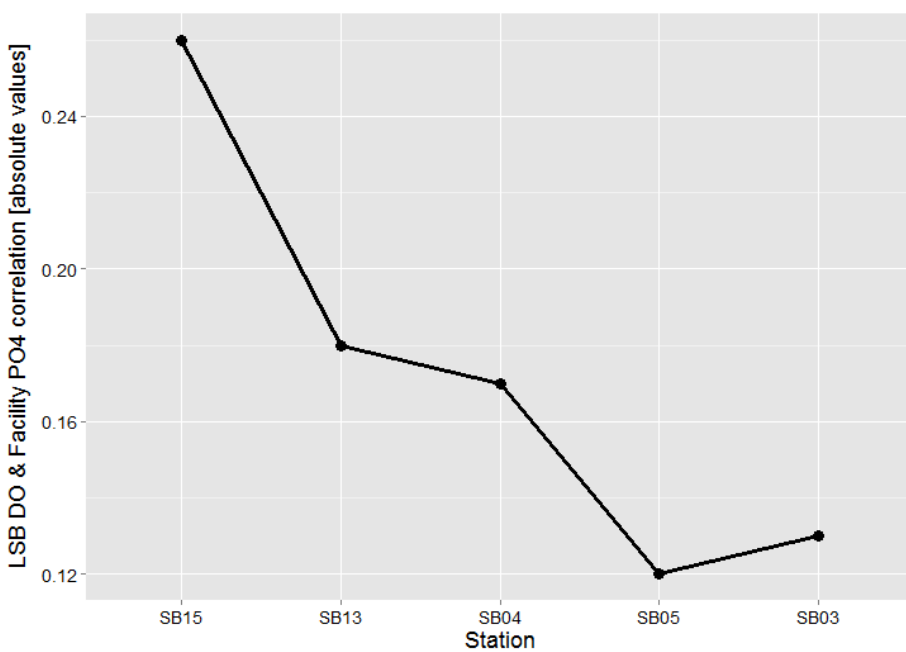

Figure 37. Plot showing how correlation change between LSB DO concentration and Facility PO_4 load with increasing distance (left to right) from the Facility.

4. *Ambient DO and nutrient concentration follow a pattern relating to temporal, seasonal, and spatial aspects*. The levels of DO in the LSB (the lower stretch of Coyote creek) over time, during the period from 1965 until December 2013, with a data gap from 1992 to 2002, are presented in Figures 38 and 39. After 1980 (nitrification was implemented in 1979), the level of DO shifted from a low of 0 mg/L (anoxia) to a low of 2.5 mg/L (outside the range of hypoxia [< 2 mg/L]). During the period from 1965 to 1980, the relative durations of dry season anoxia and hypoxia were 4% and 11%, respectively.

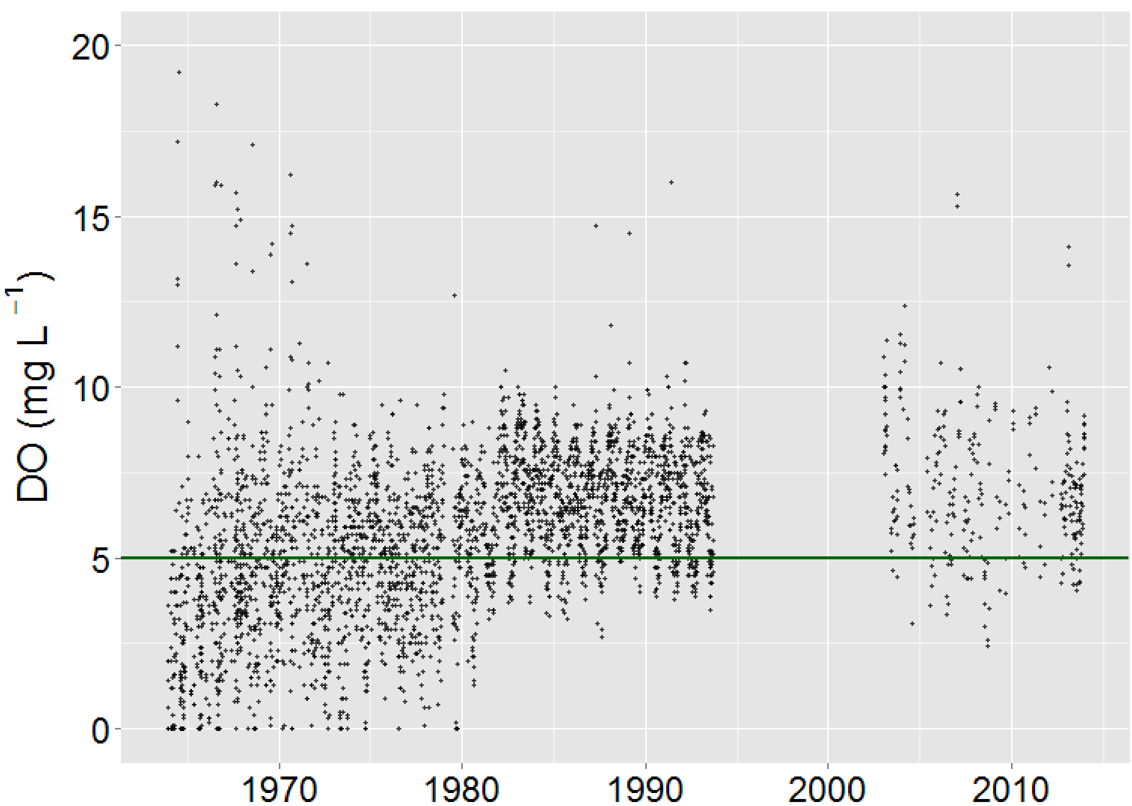

Figure 38. DO concentrations for the five stations in the lower stretch of Coyote Creek, for the years during which data were collected (with a data gap during the period from the early 1990s to the early 2000s). The green horizontal line represents the 5 mg/L water quality objective for DO for the San Francisco Basin.

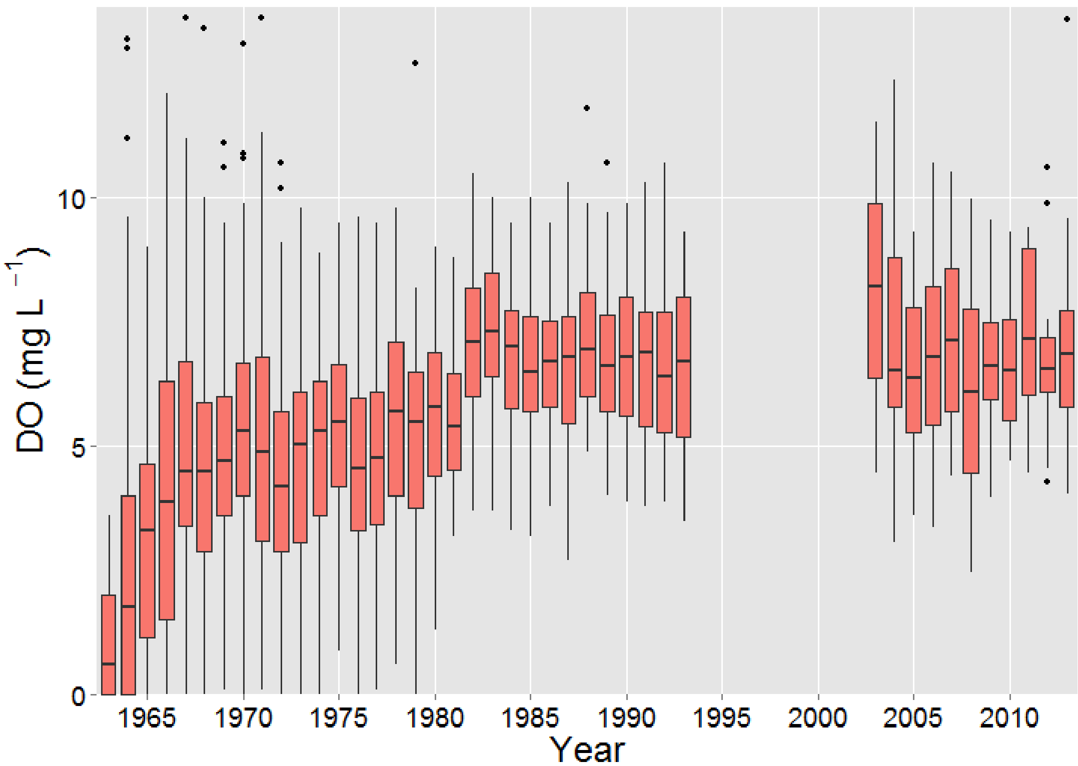

Figure 39. Long-term DO concentrations at the five stations in the lower stretch of Coyote Creek for the years during which data were collected (with a data gap during the period from the early 1990s to the early 2000s). Boxplots summarize the annual median.

The median value of DO concentration in the LSB significantly increased (47%) from the secondary period (from February 1964 to January 1979) to the tertiary period (from February 1979 to June 1997), and (3%) from the tertiary period to the BNR period (from July 1997 to December 2013), corresponding to the wastewater treatment eras (Figure 39). The *p*-values obtained from the Wilcoxon rank-sum tests reflected these increases (Table 8).

Ammonium concentration in the LSB declined steeply starting in 1979, corresponding with the beginning of the implementation of nitrification. Following a data gap from 1993 to 2003, there was another substantial decline, beginning in 2003, until 2012, with a slight uptick in 2012 and 2013 (Figure 40).

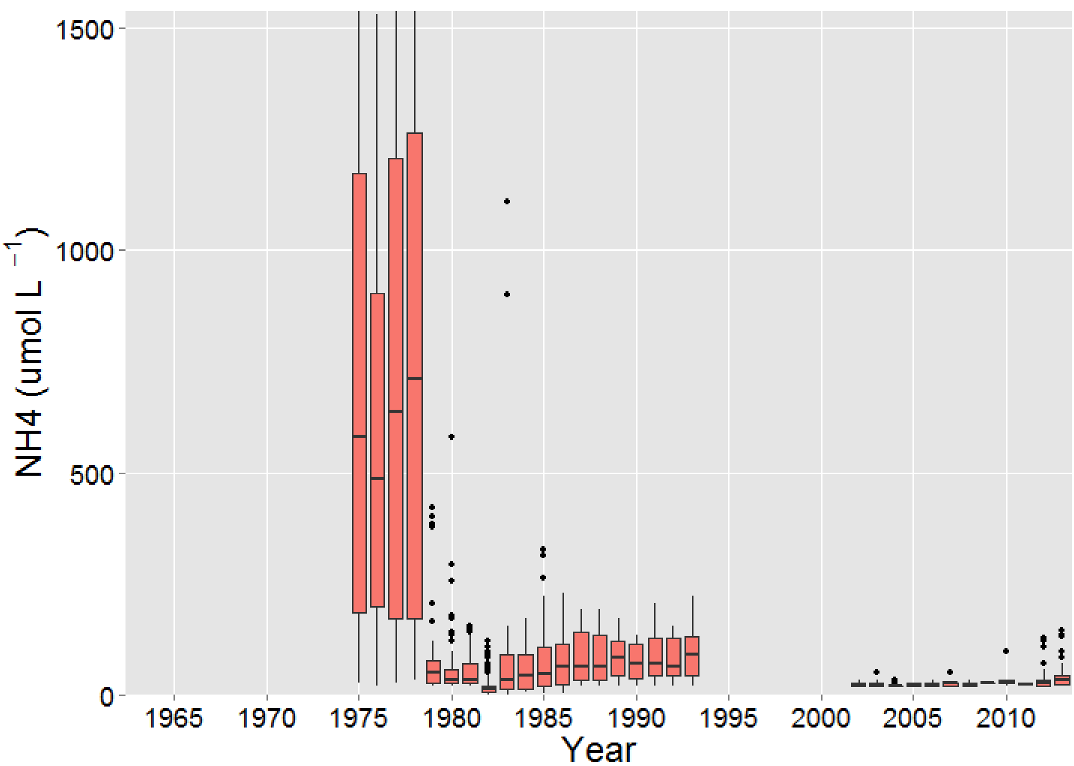

Figure 40. Long-term NH_4^+ concentrations at the five stations in the lower stretch of Coyote Creek for the years during which data were collected, 1975-2013 (with a data gap during the period from the early 1990s to the early 2000s). Boxplots summarize the annual median.

When ammonium declined, nitrate increased, in 1979, following the implementation of nitrification. Following a data gap from 1992 to 2002, there was a substantial decline, corresponding with BNR implementation (Figure 41).

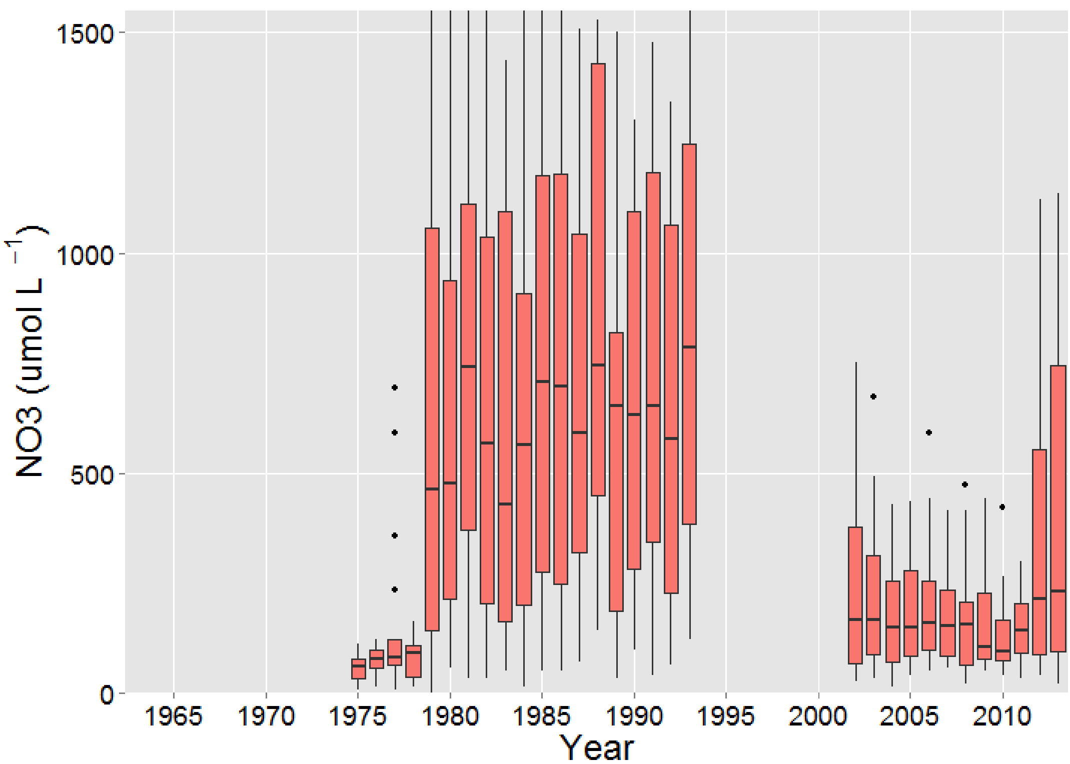

Figure 41. Long-term NO_3^- concentrations at the five stations in the lower stretch of Coyote Creek for the years during which data were collected, 1975-2013 (with a data gap during the period from the early 1990s to the early 2000s). Boxplots summarize the annual median.

The median value of NH_4^+ concentration in the LSB significantly decreased (91%) from the secondary period (from February 1964 to January 1979) to the tertiary period (from February 1979 to June 1997), and (42%) from the tertiary period to the BNR period (from July 1997 to December 2013), corresponding to the wastewater treatment eras. The *p*-values obtained from the Wilcoxon rank-sum tests reflected these decreases (Table 8).

The median value of $NO3^-$ concentration in the LSB significantly increased (627%) from the secondary period (from February 1964 to January 1979) to the tertiary period (from February 1979 to June 1997), while decreasing (70%) from the tertiary period to the BNR period (from July 1997 to December 2013), corresponding to the wastewater treatment eras. The *p*-values obtained from the Wilcoxon rank-rum tests reflected this increase, followed by a decrease (Table 8).

Phosphorus showed a decline in 1991, followed immediately by a data gap until 2003, after which phosphorus decreased to a fraction of what it was prior to 1991 (Figure 42).

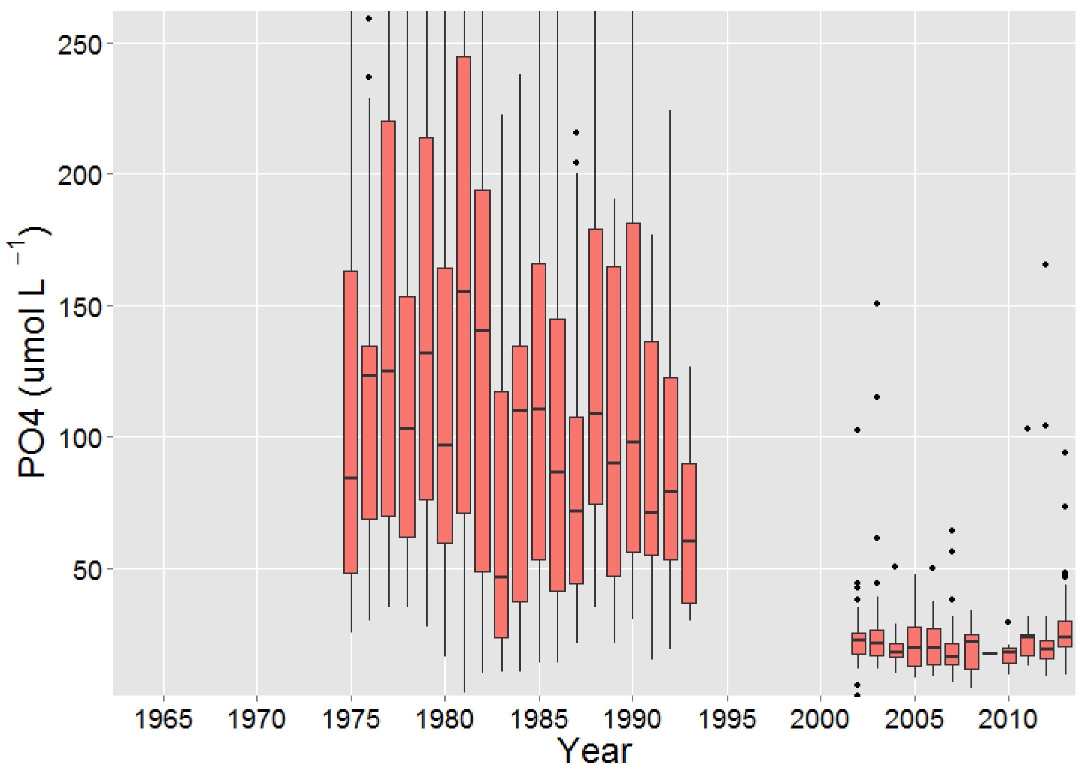

Figure 42. Long-term PO_4 concentrations at the five stations in the lower stretch of Coyote Creek for the years during which data were collected, 1975-2013 (with a data gap during the period from the early 1990s to the early 2000s). Boxplots summarize the annual median.

The median value of PO_4 concentration in the LSB showed no significant difference from the secondary period (from February 1964 to January 1979) to the tertiary period (from February 1979 to June 1997), while showing a significant decrease (170%) from the tertiary period to the BNR period (from July 1997 to December 2013), corresponding to the wastewater treatment eras. The *p*-value obtained from the Wilcoxon rank-sum test reflected the decline (Table 8).

Table 8. Results of a Wilcoxon rank-sum, non-parametric test for testing mean differences between treatment eras (secondary and tertiary) and (tertiary to BNR), for DO, NH_4^+, NO_3^-, and PO_4 concentration in the LSB. Significant factors ($p < 0.05$) are in bold.

	Secondary vs. Tertiary		Tertiary vs. BNR	
	W	sig.	W	sig.
DO	6.73e+05	**3.02e-10**	228093	**0.002537**
NH_4^+	575916.5	**< 2.2e-16**	192911.5	**< 2.2e-16**
NO_3^-	3551	**< 2.2e-16**	212726.5	**< 2.2e-16**
PO4	21741	**0.1688**	154269	**< 2.2e-16**

The *p*-value (2.2e $^{-16}$) obtained from the Wilcoxon rank-sum tests showed a statistically significant difference between the medians of DO concentration for the dry and wet seasons. The median value for DO dry seasons is less than the median DO for wet seasons. Figures 43 to 47 show the seasonal trends for the five stations.

The *p*-values for NH_4^+, NO_3^-, and PO_4 (0.06, 0.2277, and 0.52221, respectively) obtained from the Wilcoxon rank-sum tests show no statistically significant difference between the medians for the dry and wet seasons.

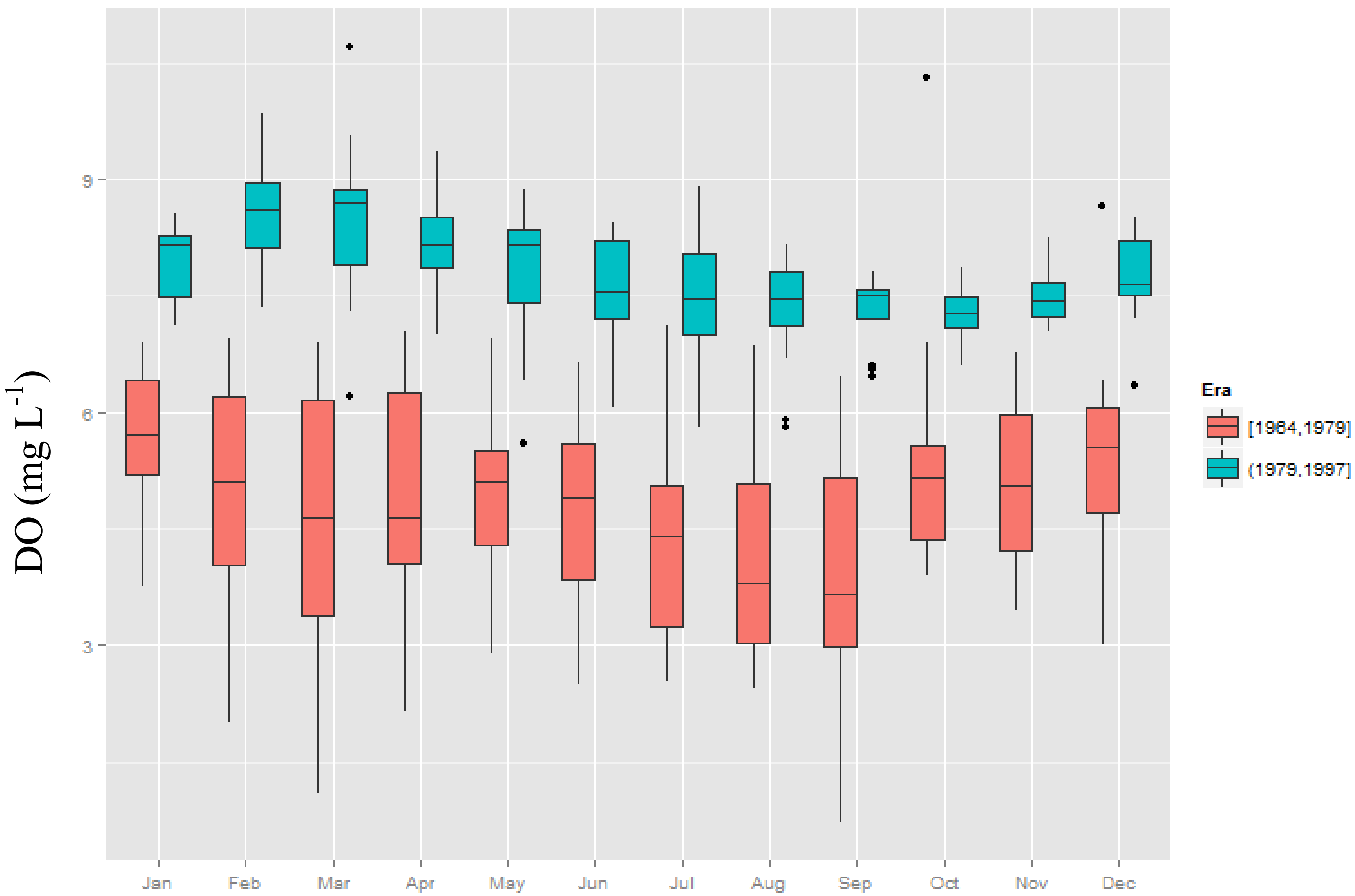

Figure 43. Seasonal patterns of DO concentration during two different eras: secondary (1964-1979), and tertiary (1979-1997), at Station SB15.

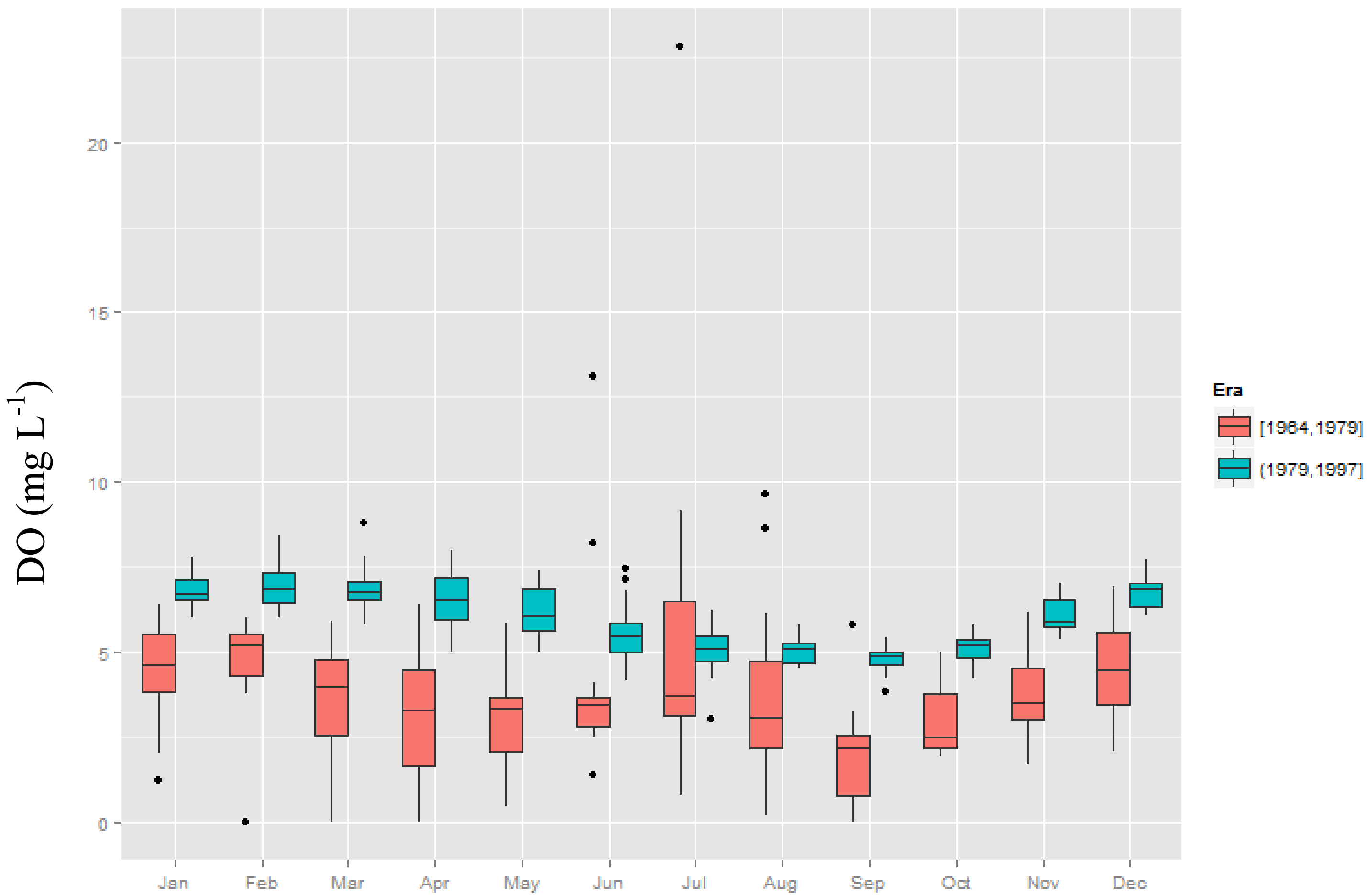

Figure 44. Seasonal patterns of DO concentration during two different eras: secondary (1964-1979), and tertiary (1979-1997), at Station SB13.

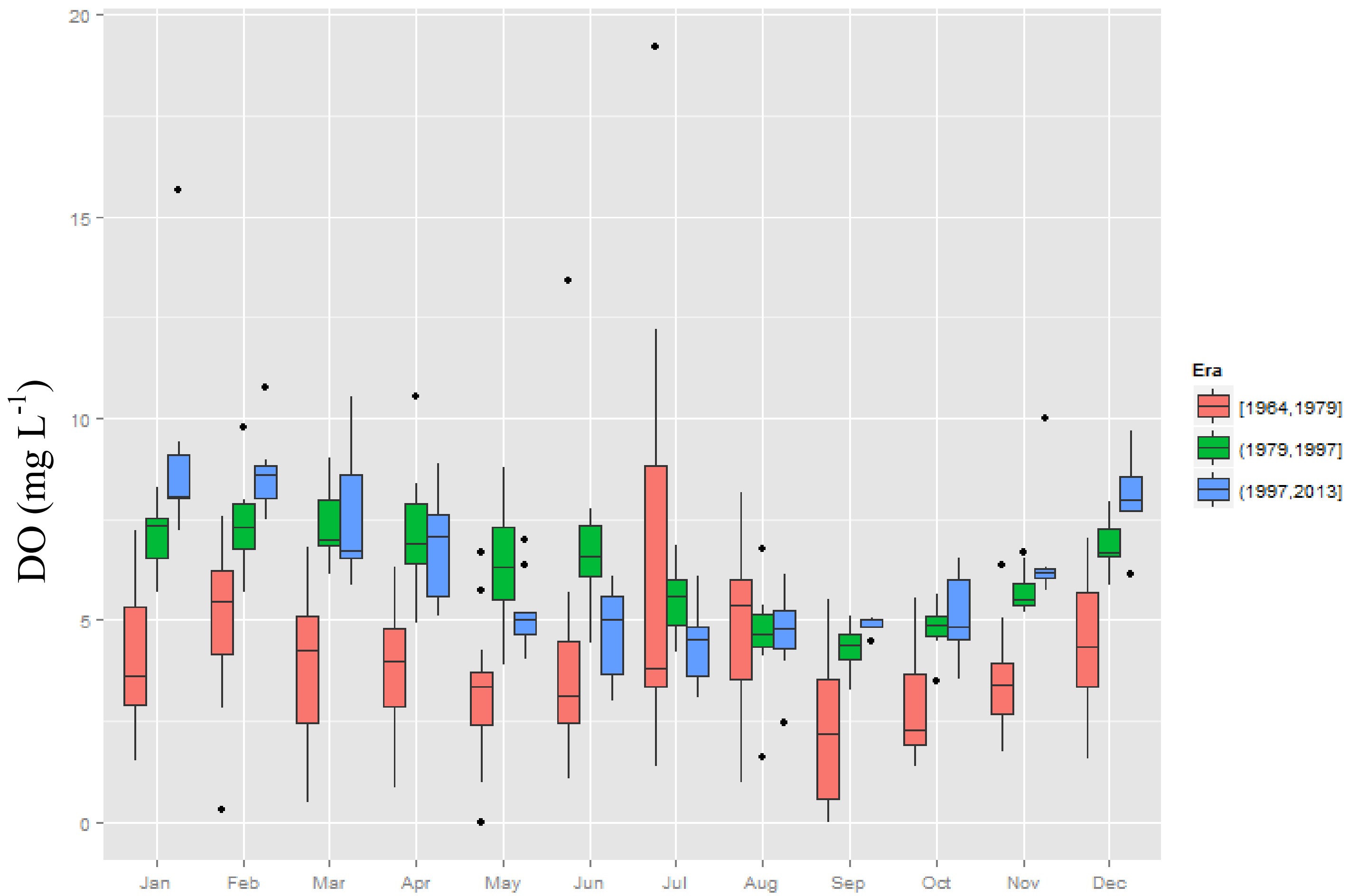

Figure 45. Seasonal patterns of DO concentration during three different eras: secondary (1964-1979), tertiary (1979-1997), and BNR (1997-2013), at Station SB04.

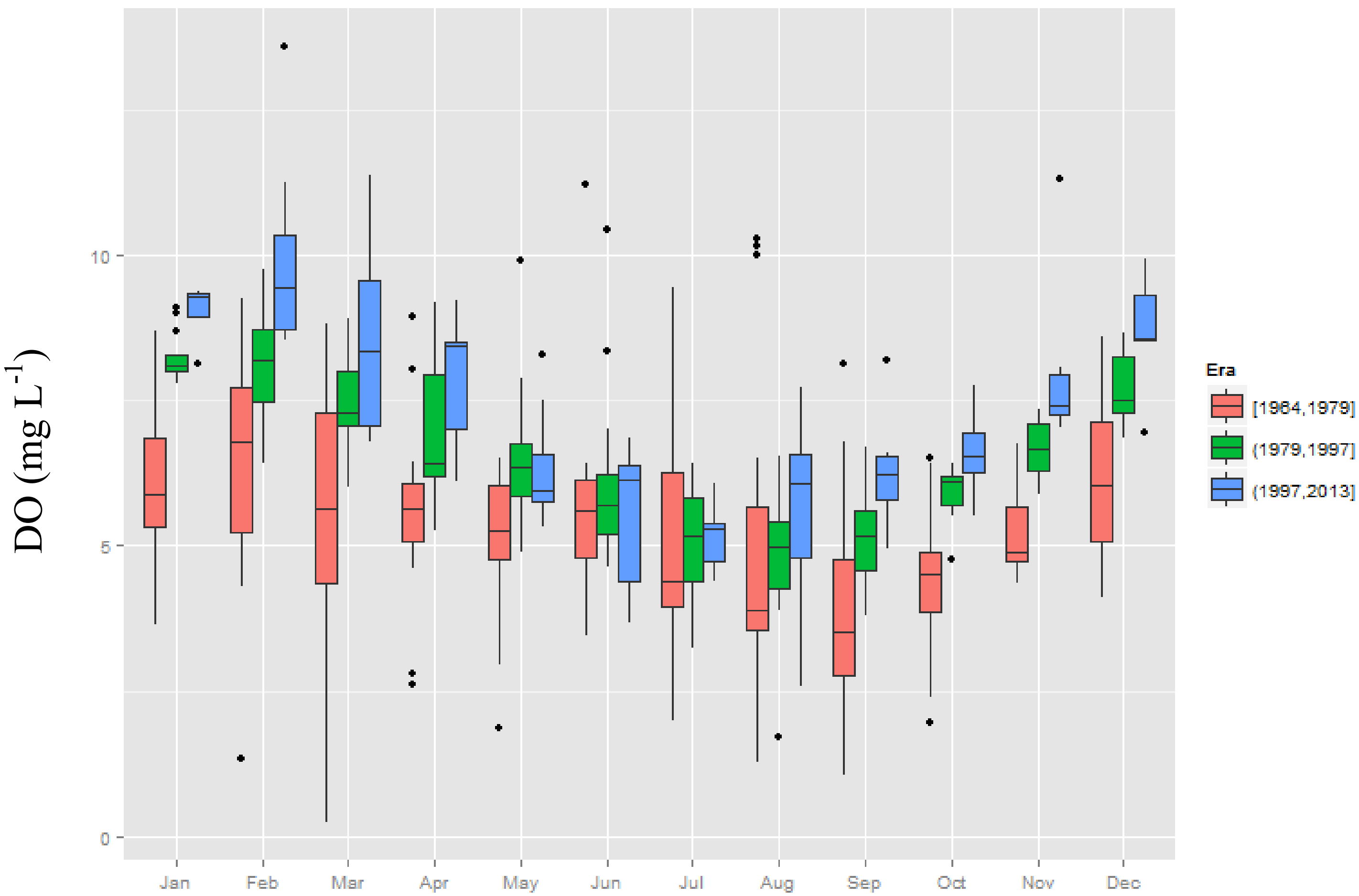

Figure 46. Seasonal patterns of DO concentration during three different eras: secondary (1964-1979), tertiary (1979-1997), and BNR (1997-2013), at Station SB05.

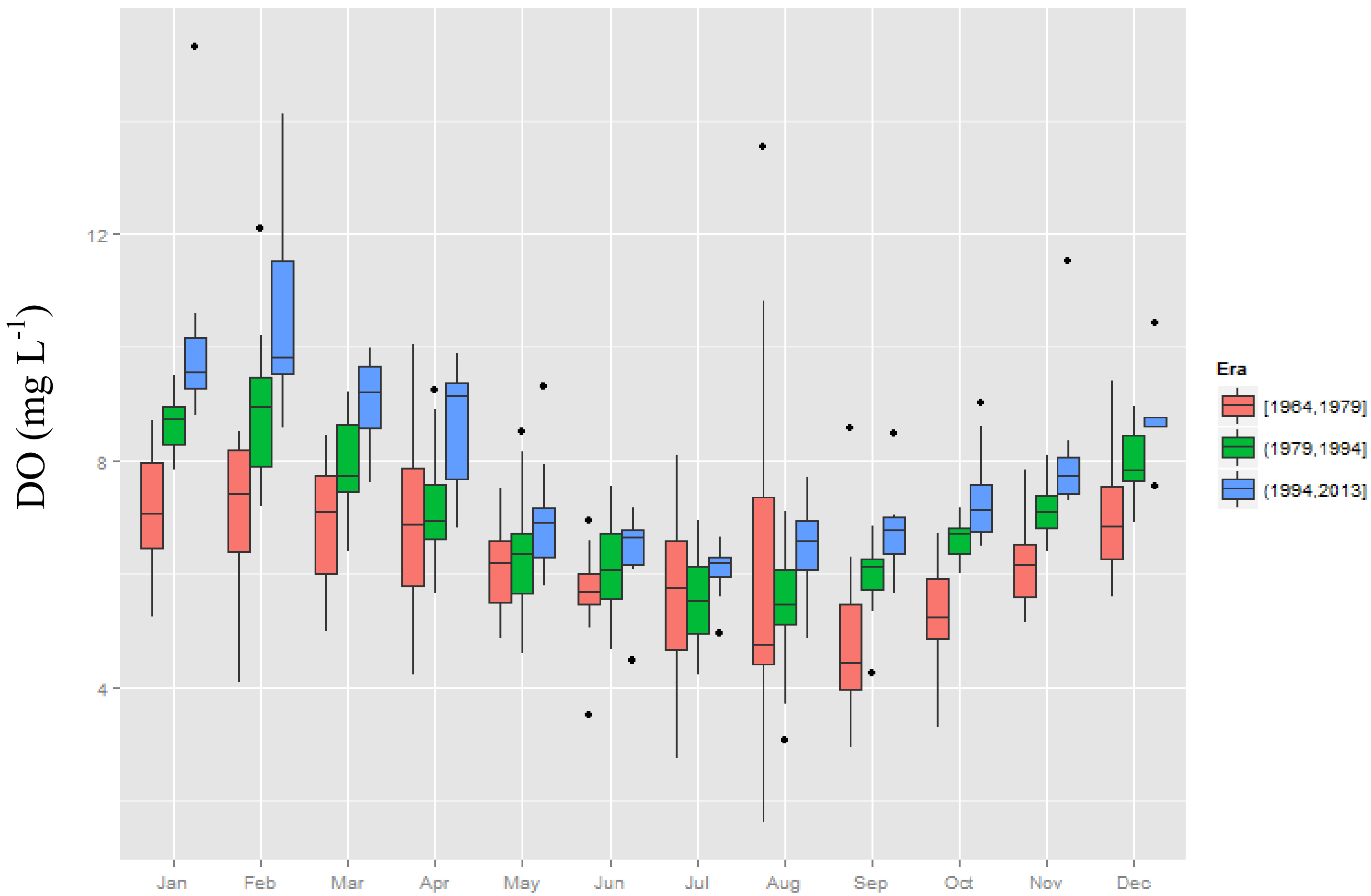

Figure 47. Seasonal patterns of DO concentration during three different eras: secondary (1964-1979), tertiary (1979-1997), and BNR (1997-2013), at Station SB03.

Spatially, there appears to be a pattern of DO concentration among the stations, as the concentration is highest for SB03 (the farthest station away from the Facility) and lowest (closer to the Facility) (Figure 48). The closest station to the Facility (SB15), however, reported a higher level of DO concentration than the ones that are further out (SB04 and SBO5).

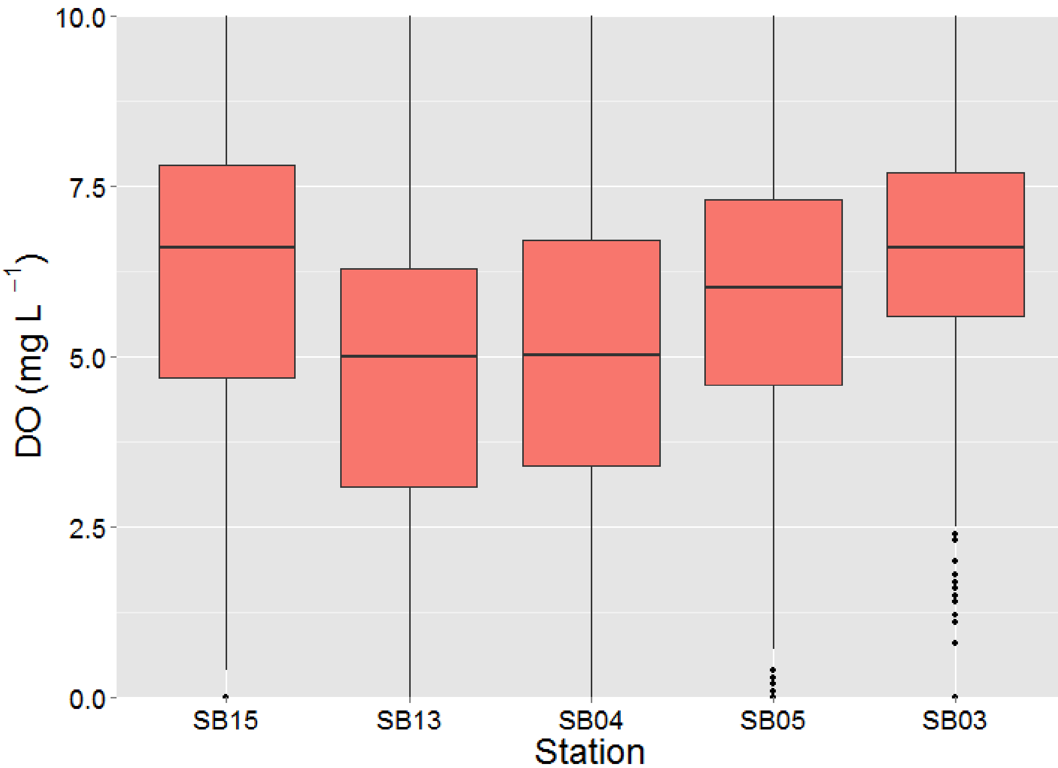

Figure 48. Spatial distributions of DO concentrations at Stations SB15, SB13, SB04, SB05, and SB03, from 1964 to 2013 (with a data gap, not showing, during the period from 1992-2003, for the three Bay stations, SB04, SB05, and SB03, as well as a different data gap, not showing, during the period 1992-2012, for the two near-field stations, SB15 and SB13), represented by boxplots.

Ammonium, nitrate, and phosphorus concentration in the five stations declined, in direct proportion with distance from the Facility (Figures 49 to 51).

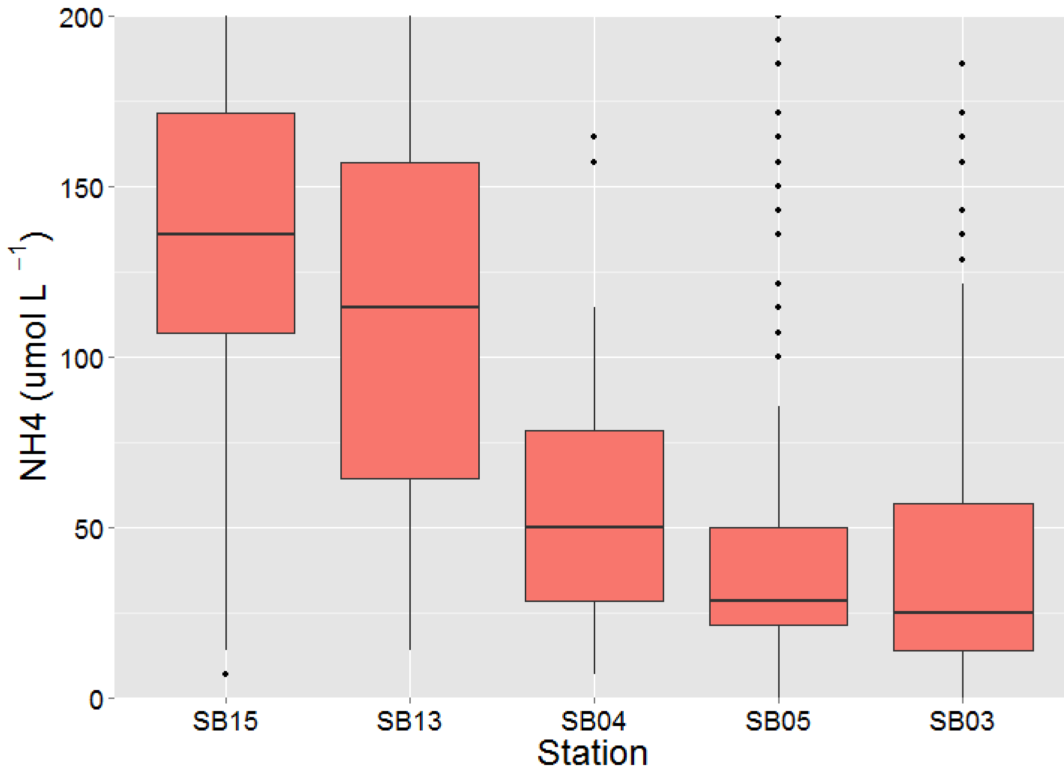

Figure 49. Spatial distributions of NH_4^+ concentrations at Stations SB15, SB13, SB04, SB05, and SB03, from 1975 to 2013 (with a data gap, not showing, during the period from 1992-2003, for the three Bay stations, SB04, SB05, and SB03, as well as a different data gap, not showing, during the period 1992-2012, for the two near-field stations, SB15 and SB13), represented by boxplots.

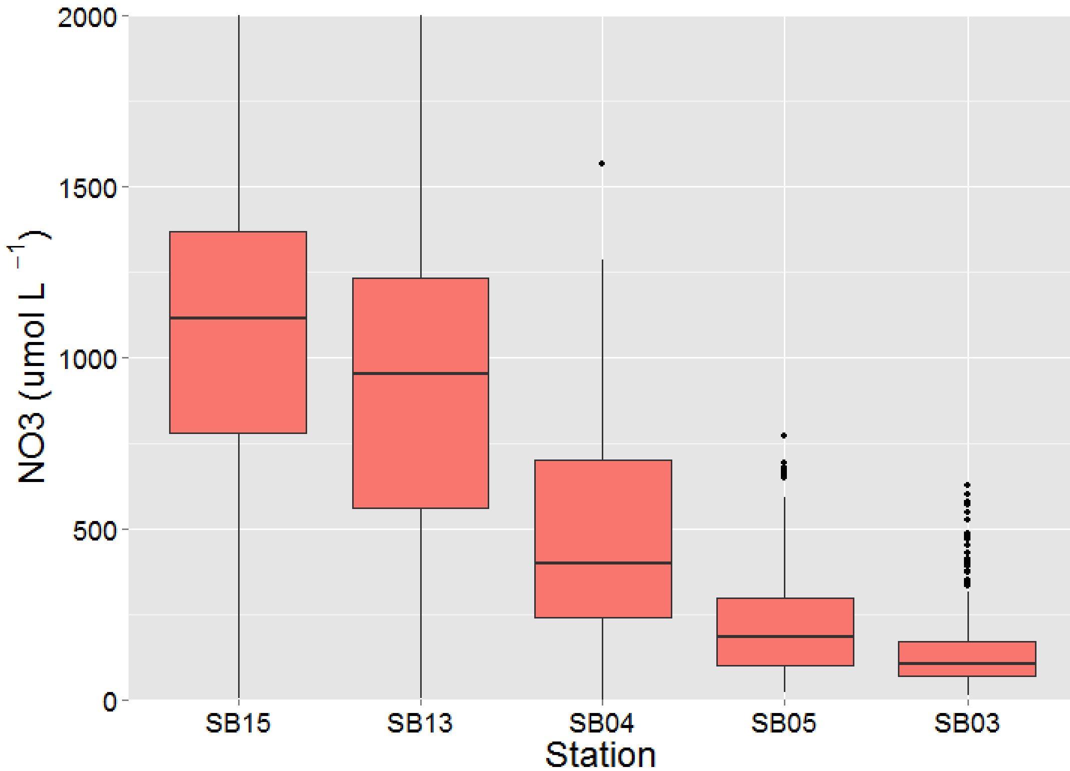

Figure 50. Spatial distributions of NO_3^- concentrations at Stations SB15, SB13, SB04, SB05, and SB03, from 1975 to 2013 (with a data gap, not showing, during the period from 1992-2003, for the three Bay stations, SB04, SB05, and SB03, as well as a different data gap, not showing, during the period 1992-2012, for the two near-field stations, SB15 and SB13), represented by boxplots.

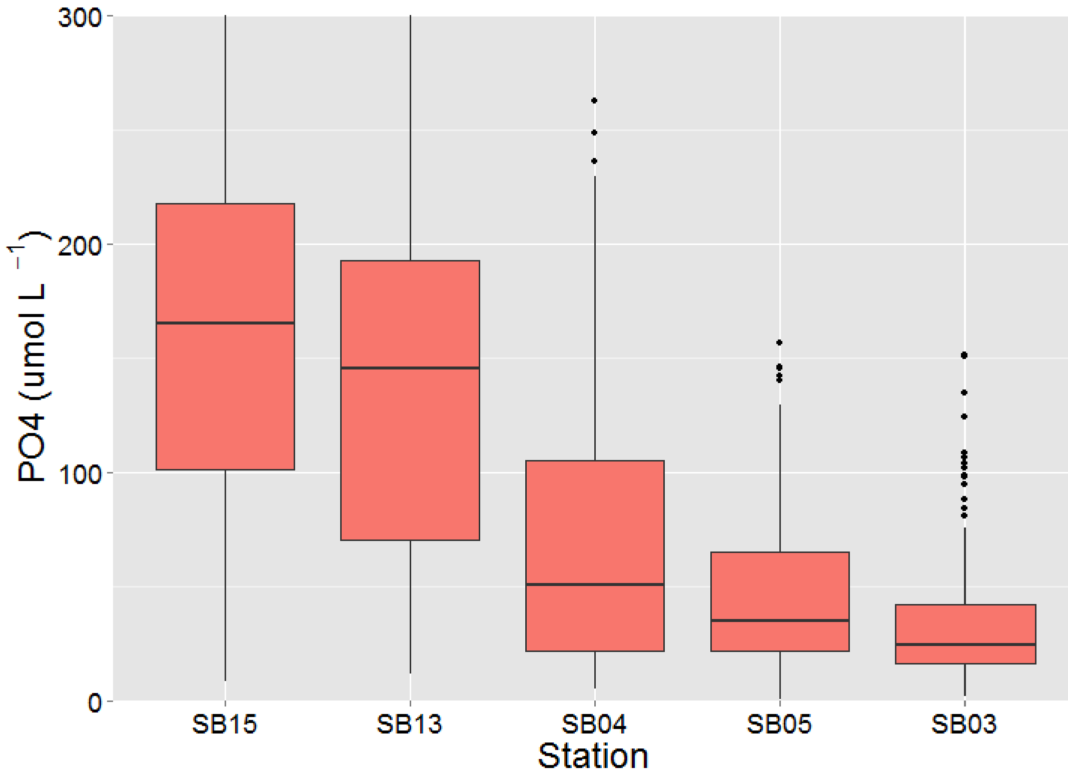

Figure 51. Spatial distributions of PO_4 concentrations at Stations SB15, SB13, SB04, SB05, and SB03, from 1975 to 2013 (with a data gap, not showing, during the period from 1992-2003, for the three Bay stations, SB04, SB05, and SB03, as well as a different data gap, not showing, during the period 1992-2012, for the two near-field stations, SB15 and SB13), represented by boxplots.

5. *Ambient nutrient concentrations in the LSB are positively correlated with Facility nutrient loading*. Table 9 summarizes the information in Figures 52 through 57, showing that Facility effluent nutrient loading was positively correlated with nutrient concentration in the LSB. The most highly correlated observations were for NH_4^+, with a high level of correlation at all sites. NO_3^- and PO_4 had similar levels (low to medium) of correlation. Correlation decreases in direct correspondence with distance between the Facility and the site. Based on the available data, there appears to be no correlation between PO_4 loading at the site farthest away from the Facility (SB03). (Only Stations

SB15 and SB03 are represented in Figures 51 through 57 in order to show the closest station and the one that is farthest away from the Facility.)

Table 9. Results of Pearson Correlation, between nutrient (NH_4^+, NO_3^-, and PO_4) concentration in the LSB and NH_4^+, NO_3^-, and PO_4 load from the Facility (using monthly means), for all five stations.

	SB15	SB13	SB04	SB05	SB03
NH_4^+	0.96	0.94	0.93	0.86	0.77
NO_3^-	0.52	0.58	0.46	0.32	0.21
PO_4	0.6	0.54	0.57	0.33	0.05

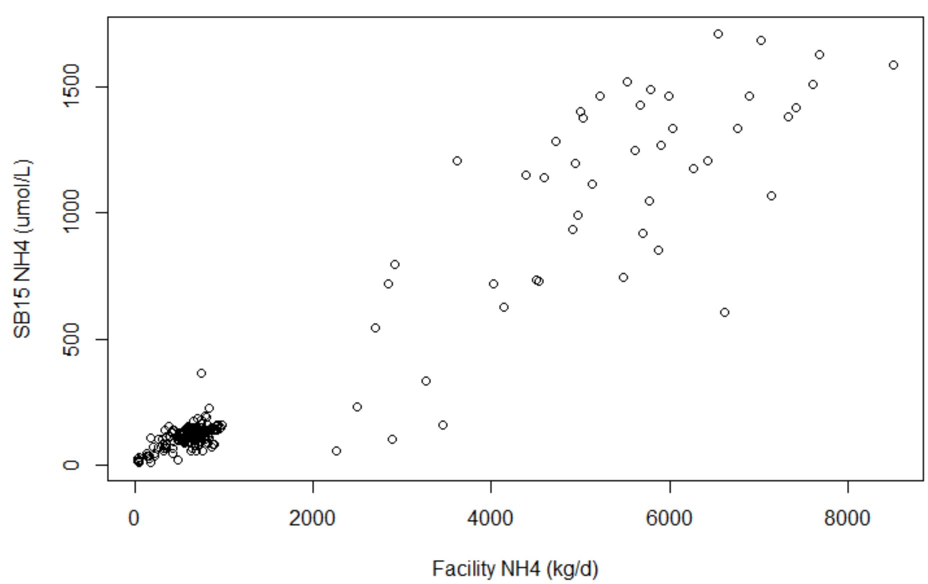

Figure 52. Scatter plot showing the relationship between NH_4^+ concentration in the LSB and NH_4^+ load from the Facility (using monthly means) for SB15.

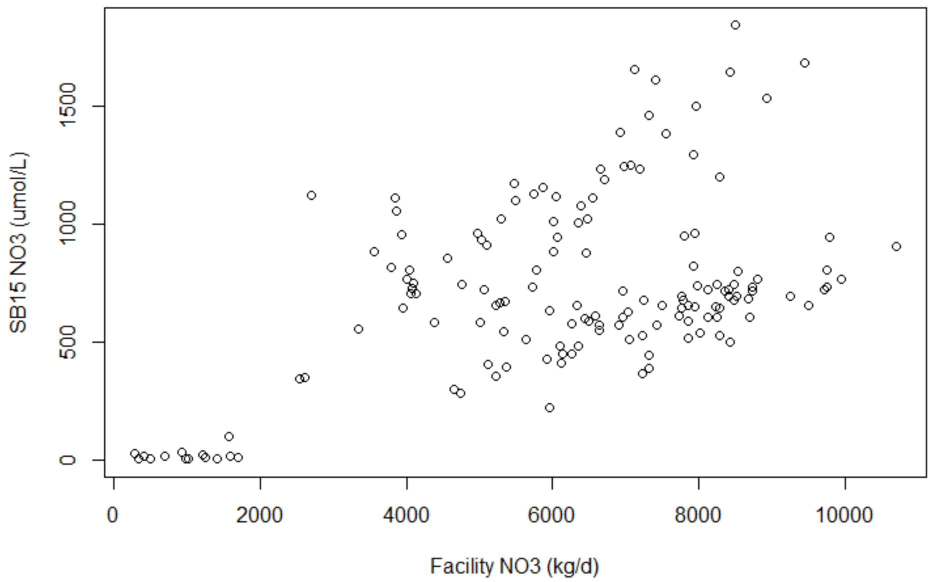

Figure 53. Scatter plot showing the relationship between NO_3^- concentration in the LSB and NO_3^- load from the Facility (using monthly means) for SB15.

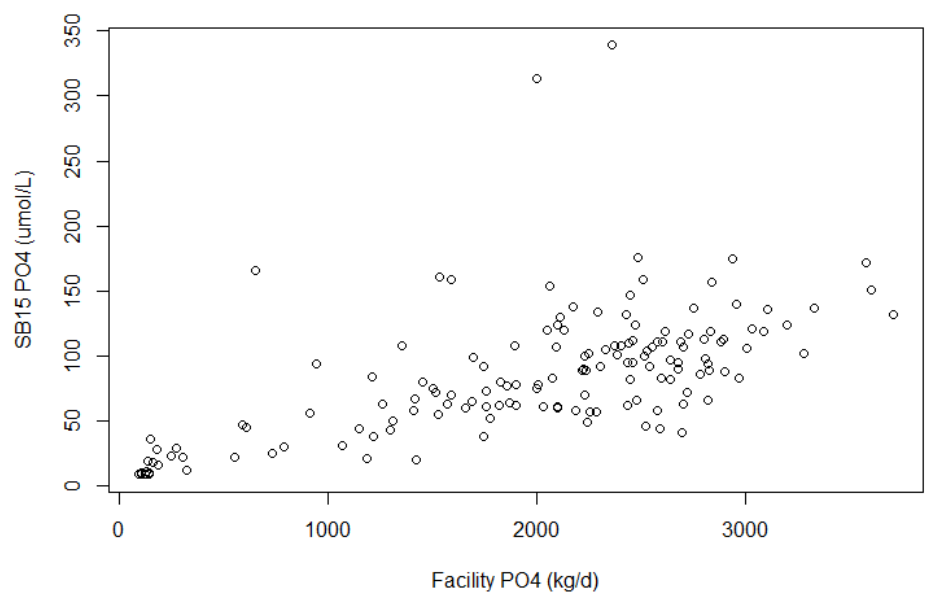

Figure 54. Scatter plot showing the relationship between PO_4 concentration in the LSB and PO_4 load from the Facility (using monthly means) for SB15.

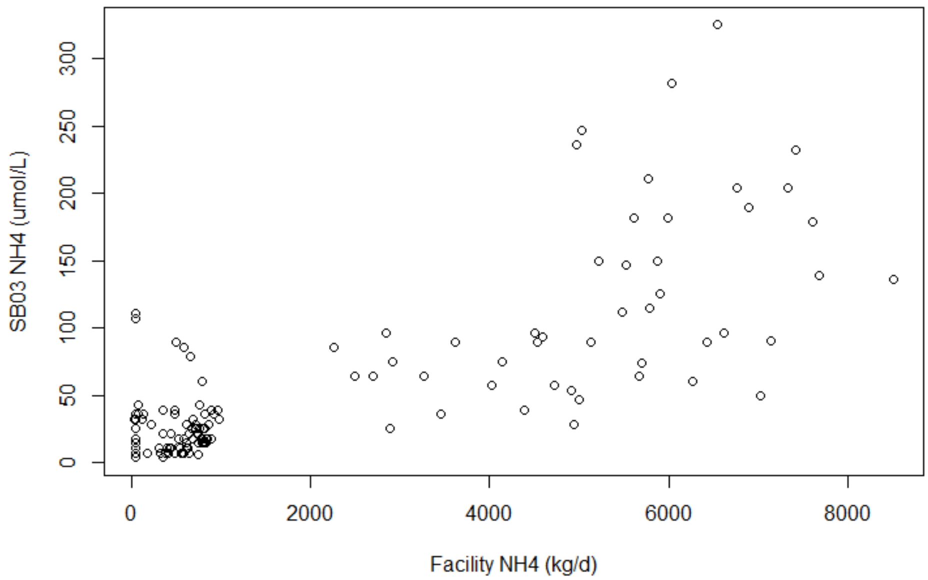

Figure 55. Scatter plot showing the relationship between NH_4^+ concentration in the LSB and NH_4^+ load from the Facility (using monthly means) for SB03..

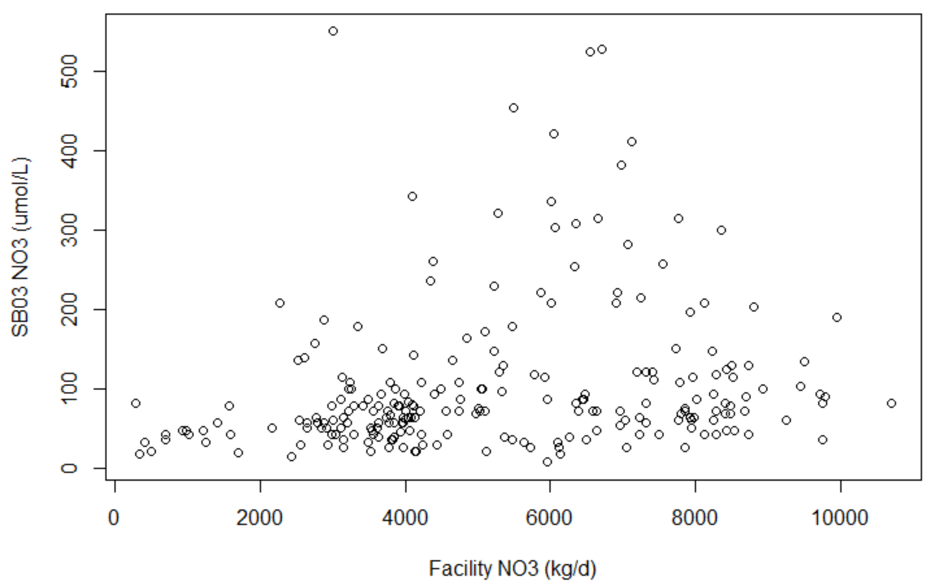

Figure 56. Scatter plot showing the relationship between NO_3^- concentration in the LSB and NO_3^- load from the Facility (using monthly means) for SB03.

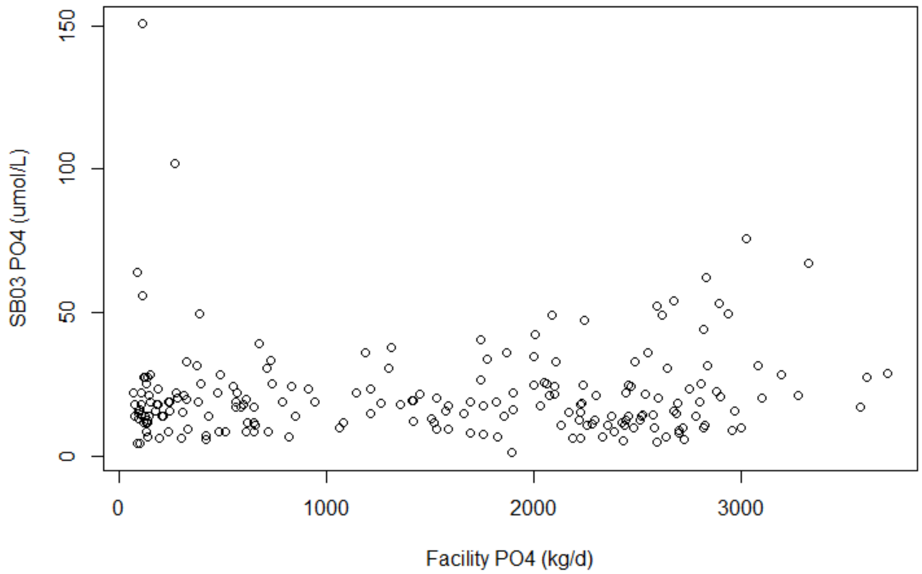

Figure 57. Scatter plot showing the relationship between PO_4concentration in the LSB and PO_4 load from the Facility (using monthly means) for SB03.

Discussion

For ease of comprehension, the discussion section adopts the framework utilized in the results chapter, organized in the framework of the five hypotheses guiding this study.

1. *Facility influent flow, BOD, TSS, and nutrient loading increased with increases in population over time*. The data reported under this hypothesis in the results section above show a clear trend over time with regard to the increase of Facility influent BOD, TSS, and nutrients. This trend can most likely be attributed to increases in population that occurred during the same period of time.

As mentioned in the results section, although BOD and TSS loads increased over time together with increasing population, BOD and TSS load increases did not "keep up" with increasing population. In other words, BOD and TSS loads did not increase proportionally with population, even though they followed the general rising trend. One possible explanation for the "gap" between the loess line representing BOD/TSS load over time and the line representing population over time is that the data for influent loading were thrown off, so to speak, by the disappearance in the late 1980s of the canneries, which had disproportionately been contributing to Facility influent. To show how BOD and TSS increased with population, data can be extrapolated by using an estimate from the literature (Metcalf & Eddy, 2003) of 90 g per capita BOD and TSS loads. Actual BOD and TSS loads have been higher than the per-capita-based estimates, with the closest point of correspondence during the drought from 1987-1992. The greatest gaps occurred around 1980 and 2000, with discrepancies of 34% and 28%, respectively. Since 2000, the gap for BOD has narrowed to approximately 23%, and for TSS narrowed to approximately 3.5%.

The steady reduction in flow (water usage) during the period from 2000 until December 2013 may be attributed to various water conservation efforts on the part of the SCVWD. Water conservation accounted for a 10% reduction in influent flow (SCVWD, 2014). Recycled water also played a major role in the reduction of effluent flow. In 2013, SBWR distributed nearly 5 billion gallons of recycled water through 142 miles of transmission and distribution pipeline, representing a 32% increase relative to the previous calendar year (SBWR, 2013). Because of contribution made by recycled water, the flow to the Bay was reduced by almost 16% in 2013.

The load coming into the Facility has been seasonal in nature. From the 1950s through the 1980s, the spikes implied by the data points appearing high on the scatter plot reflect canning industry activity, which operated at peak levels from July through September. Following the closure of the canneries, the spikes implied by the dots appearing high on the scatter plot reflect the effects of the wet season. The dip from the late 1980s to the early 1990s (see Figure 10) may be due to disappearance of the peak loads, in combination with the drought of 1987-1992.

The various data gaps that occurred and are reflected in the figures (20 years for ammonium and almost 30 years for phosphorus) were due to cessation of data gathering by the Facility for unknown reasons. To help fill in these data gaps, extrapolation can be done by using estimates from the literature of per capita ammonium and phosphorus loads. The range of ammonium per capita in the literature extends from 12-16 g. Based on historical data, as well as more recent data, the Facility's incoming total ammonium lies on the higher end of the range of estimation   Therefore, 16 g per capita was used, as per Sedlak (1991). The average for phosphorus from 1950 until 1990, when this ingredient was banned from detergents in California, was 4.5 g per capita. Subsequently, until December 2013, the average was 2.3 g per capita (Metcalf & Eddy, 1991). The considerable (approximately 1100 kg/d) difference of phosphorus load prior to 1990 between the actual Facility data (which was substantially lower than the estimate) and the figure based on per capita estimates may be understood as a reflection of a miscalculation or typographical error that was introduced into method I-2600/I-4600 and went unnoticed until 1989 (USGS, 1992). After the State of California ban on phosphorus in detergents in 1990, not surprisingly, the phosphorus load decreased dramatically, from a high of 4,362 in the late 1980s to 2,657 in 1990, a decrease of approximately 39%.

2. *Facility effluent BOD, TSS, and nutrient loading decreased with Facility treatment process upgrades*. The data reported in the results section above show a clear overall trend, with regard to BOD, TSS, NH_4^+, and PO_4 in terms of dramatically decreasing load levels over time, across treatment eras (primary, secondary, tertiary, and BNR), although the decreases were not consistently incremental from one era to the next. The same overall trend was true for nitrate (NO_3^-). Ammonium loadings drastically decreased 89% when treatment upgraded to tertiary (nitrification). Nitrification did not remove total nitrogen, but rather converted most of the ammonium to nitrate. While ammonium loads dropped following nitrification, nitrate loads increased approximately 520%. The upgrade to BNR in 1997 introduced denitrification to the treatment process,

with a subsequent nitrate reduction of 48% due to removal of total nitrogen via denitrification.

The major component of the nitrogen load to the LSB was nitrate (83.2%). Only 5.5% of the nitrogen load from the Facility was in the form of ammonium (RWF, 2013). There appears to be small upward trend in terms of an increase of ammonium in the past 10 years. The total nitrogen load coming into the Facility is comprised of 57.5% ammonium, 41% organic nitrogen, and 2% nitrate. The Facility removed almost all the ammonium (99%) from the influent but added 100 kg/d of ammonium for purposes of chloramination (ammonium is added to prolong the chlorine's effectiveness, as well as a cost-saving measure), to the existing 100 kg from the 1% of the ammonium remaining after treatment of the influent. With regard to phosphorus, the reduction in the early 1990s was due to the removal of this chemical element from detergents, as mandated by California law. Levels of phosphorus further declined with BNR, a reduction of 78%, relative to the previous treatment era.

For BOD, the major decrease occurred from the primary to the secondary treatment era, when a 74% removal rate was achieved. BOD load remained essentially the same between the tertiary to the BNR treatment era, as the results had already been maximized. The slight difference that did occur between these two treatment eras was due to a single event that occurred in September 1979, when more than four billion gallons of marginally treated sewage were released into the Bay, resulting in more than 20 times the normal concentration of BOD load from previous month.

As is the case with influent load, effluent load (containing BOD, TSS, NH_4^+, NO_3^-, and PO_4) shows seasonal activity. During the dry season, with BNR treatment operating efficiently, PO_4 removal from influent to effluent can exceed 90%. During the dry season, phosphorus in effluent was very low. In 2012 and 2013, for example, the wet season drop in phosphorus removal corresponded with rains, cooler temperatures, and changes to process operations in response to seasonal changes (RWF, 2013). Like phosphorus, both nitrification and denitrification are affected by temperature and therefore subject to seasonal effects (RWF, 2014).

A recent study found seasonality in terms of nutrient loads to the LSB. Though estimated stormwater loads varied seasonally, a portion of the overall variability was found to be due to seasonal differences in POTW loads. From the dry season to the wet season, NO_3^- loads increased by up to 50% at SJSC (Novick & Senn, 2014).

3. *There is an inverse correlation between ambient DO concentration in the LSB and both Facility BOD and nutrient loading.* The strong-to-medium negative/inverse correlation between BOD/ammonium load and DO concentration at LSB stations can be explained by the molecular breakdown of organic matter through the process of consuming oxygen, as stated in the conceptual framework. During the spill of 1979, for example, organic matter from sewage was oxidized into CO_2. The degree of oxidation was enough to create anoxia in the upper area of Coyote Creek. Similarly, ammonium was converted to nitrate, as DO increase in the area between Coyote Creek and the South Bay. Nitrification (oxidation of ammonium) in Coyote Creek's water column constituted a large DO sink. The DO returned to normal levels after two weeks. NO_3^- also declined

(Cloern & Oremland, 1983). Denitrification occurred in the absence of oxygen. The major source of nitrate for denitrification in most estuaries is nitrate produced in the sediments (Seitzinger, 1988).

The ostensible positive correlation between nitrate and DO is deceptive, as DO concentration increases when BOD and ammonium are converted to carbon dioxide and nitrate. The relationship between DO and nitrate can be explained by the fact that both nitrate and DO are inversely correlated with ammonium. So as ammonium decreases, nitrate and DO increase.

4. *Ambient DO and nutrient concentration follow a pattern relating to temporal, seasonal, and spatial aspects.* The data reported in the results section above show a clear overall trend, with regard to DO in terms of substantially increased concentration over time, across treatment eras (secondary, tertiary, and BNR), with a data gap from 1992 to 2002.

Both anoxia and hypoxia were virtually eliminated shortly after 1980, following the implementation of nitrification in 1979. DO concentrations below 5 mg/L (the water quality objective for San Francisco Basin)—as seen in the lower right-hand portion of the figure—still occurred during the summer months from 2003 to 20013. In most cases, hypoxia is associated with a semi-enclosed natural setting that results in restricted water exchange when combined with water-column stratification (Diaz & Rosenberg, 2008).

The data reported in the results section above also show a clear overall trend with regard to NH_4^+, NO_3^-, and PO_4 concentration in the LSB in terms of dramatically decreasing load levels over time, across treatment eras (secondary, tertiary, and BNR), although the decreases were not consistently incremental from one era to the next. The decrease in nutrient concentration corresponds to the decrease in Facility effluent load, reflecting the improvements of the treatment era, as explained under hypothesis #2 in the discussion section here.

In this study, no seasonality was observed for any of the three nutrients. Other studies such as that of the South San Francisco Bay by Smith & Hollibaugh (2006), did note significant uptake of phosphorus during summer, observing an uptake of nitrogen in both summer and winter.

The data reported in the results section show a clear spatial trend with regard to DO concentration relative to the distance of the Facility. DO levels at the stations tend to increase with distance from the Facility, with the exception of SB15. SB15 has higher levels of DO, because it is dominated by the Facility's highly oxygenated discharge. DO concentration in the LSB is significantly affected by the semi-diurnal and tidal cycle, as well as spring-neap tide. High DO concentration is associated with high tide, whereas low DO concentration is associated with ebb tide (Shellenbarger et al., 2007). However, the station closest to the Facility records low levels of DO during high tide, when Bay water dominates Artesian Slough (RWF, 2014).

The results show that DO concentration is higher during the wet season and lower during the dry season. During summer, residence time could be as long as ten weeks, while residence time at the northern end during winter could be perhaps two weeks (Smith & Hollibaugh, 2006). More abundant light, as a result of shallower depth, leads to high rates of primary production (including O_2 production) during daylight hours. Higher rates of respiration result from a greater amount of phytoplankton and MPB biomass, as well as higher loads of dead organic matter contributing to the sediments. In these systems, more influence on DO concentrations than pelagic respiration is exerted by SOD due to the comparatively high water-column-volume-to-sediment-area ratio. Net O_2 production is negative at night, which in turn leads to early morning DO minima (Senn & Novick, 2013).

The data reported under this hypothesis in the results section above show a clear spatial trend of ammonium, nitrate, and phosphorus concentration in the five stations declining in direct proportion to distance from the Facility. In the cases of each of the three nutrients, the boxplots show a clustering between the two nearest stations, on the one side, and the three farthest stations on the other, with nutrient levels clearly lower for the more distant stations. In estuaries, dissolved nutrients may either be assimilated or released as a result of biotic reactions of primary production, respiration, nitrogen fixation, and denitrification, as well as abiotic reactions such as sorption or desorption from sediment and coprecipitation (Smith & Hollibaugh, 2006).

5. *Ambient nutrient concentrations in the LSB are positively correlated with Facility nutrient loading*. The positive correlation between Facility effluent nutrient loading and nutrient concentration in the LSB—together with the fact that it decreases with distance from the Facility—might indicate that the near field station is dominated by Facility discharge.

The major source of DIN (dissolved inorganic nitrogen) and dissolved inorganic phosphorus loads to the LSB year-round were POTWs. SJSC comprised ~60% of POTW loads. In contrast with other embayments, DIN loads from POTWs to the LSB were mainly in the form of NO_3^- (90%), rather than NH_4^+. This is because the POTWs there tend to nitrify effluent before discharging it (Novick & Senn, 2014).

In comparison with other estuaries, LSB nutrient concentrations from wastewater are almost twice the total N input from all sources to Chesapeake Bay and its tributaries. The result is that N and P concentrations are much higher in the LSB than in Chesapeake Bay. That being said, the LSB has low phytoplankton biomass, relative to other enriched estuaries (Cloern & Jassby, 2012).

Efforts toward wetland and salt pond restoration around the Bay's margins have the potential to play an important role in an integrated nutrient management strategy. This is due to the potential for reducing N concentrations (and also P concentrations, to a lesser degree). Because denitrification converts NO_3^- to N_2 gas, it functions as a true N sink (and high denitrification rates can potentially occur in wetlands). Denitrification rates vary over a wide range, however. They are also highly dependent on temperature and other conditions. Although wetlands also retain P, unlike N, P has no true sink. The scale of planned wetland restoration efforts that are currently underway in the LSB and the South Bay is such that those sites could conceivably function as a major N sink (Senn & Novick, 2013).

Conclusion

All five hypotheses were confirmed by the data, with a number of qualifications that can be readily explained. Although BOD and TSS loads did not increase proportionally with population, they nonetheless followed the general rising trend. The "gap" between the loess line representing BOD/TSS load over time and the line representing population over time can be understood by the disappearance of the substantial loads from the canneries, which had disproportionately been contributing to Facility influent.

For hypothesis #4 (Ambient DO and nutrient concentration follow a pattern relating to temporal, seasonal and spatial aspects), for example, the seasonal aspect could not be confirmed, which may simply be due to data gaps. The data essentially lend support to what one would intuitively believe to be true on the basis of logic and is supported by findings from other studies (i.e., that Facility improvements have led to more effective wastewater treatment and that lower nutrient concentrations occur in direct proportion to distance from the Facility).

This is consistent with the general finding of related research for other estuaries. For the Chesapeake Bay, for example, a retrospective study stated that, "The improvements in water quality are a result of a massive wastewater management effort …." (Jaworski, 1990, p. 11). In the case of the Thames, major sewage treatment improvements implementing nitrification in the late 1970s led to significantly improved water quality, as was the case with the LSB at the same time (Attrill, 1998). A comparison relating LSB data with those from other estuaries (the Chesapeake Bay, the Delaware Estuary, and the Hudson River) follow below, by category.

BOD and TSS load reductions. Effluent BOD loadings in the LSB decreased 78% from 47,138 kg/d to 10,318 kg/d when the facility upgraded from primary to secondary treatment during the period from 1957 to 1964. Effluent BOD loadings in the Chesapeake Bay decreased 92%, from approximately 63,600 kg/d to 5,400 kg/d when the facility upgraded from primary to secondary treatment from 1970 to 1985 (Jaworski, 1990). In the case of the Hudson River, there was a 50% reduction for both BOD and TSS from primary to “subsequent upgrades to secondary treatment” (from 1920 to 1960) followed by a further 75% reduction when upgraded to “full secondary” in 1972 (Hetling et al., 2003). TSS loadings fell from 22,555 kg/d to 14,414 kg/d when the facility upgraded from primary to secondary treatment during the period from 1957 to 1964, a reduction of 36%. TSS loadings for the Chesapeake Bay fell from approximately 61,800 kg/d to 3,400 kg/d when the facility upgraded from primary to secondary treatment from 1970 to 1985, a reduction of 95%. It is difficult to ascertain the reason for the differences in reduction (36% versus 95%) between Chesapeake Bay and the LSB based on available published material. It bears mention, however, that the largest and most consistent TSS loadings reductions for the LSB Facility occurred when the facility upgraded to tertiary treatment in 1979 with the addition of a filtration facility, at which time TSS loadings fell from 14,674 kg/d to 899 kg/d, a reduction of 93%.

Nitrogen loads reduction. Ammonium loadings drastically decreased from 5,386 kg/d to 539 kg/d in the LSB, a reduction of almost 90%, when treatment was upgraded to tertiary (nitrification) during the period from 1964 to 1979. There was no reduction in total nitrogen during this period. The upgrade to BNR in 1997 introduced denitrification to the treatment process, and nitrogen loadings decreased from 7928 kg/d to 4161 kg/d, a reduction of 47%, due to removal of total nitrogen via denitrification. Although there are no equivalent data from Chesapeake Bay for the period during which the plant there upgraded from secondary to tertiary, "There was no change in nitrogen loading because of improved removal in the secondary treatment process" (Jaworski, 1990, p. 11). As Jaworski points out, however, "it should be noted that many of the wastewater plants are now nitrifying the wastewater and thus reducing the nitrogenous biochemical oxygen demand" (Jaworski, 1990, p. 17). In the case of the Hudson River, there was a 17% reduction of total nitrogen from primary to "subsequent upgrades to secondary treatment" (from 1920 to 1960), followed by a further 40% reduction when upgraded to "full secondary" in 1972. This was not true for all treatment plants in the area, however, with some achieving only a removal of 20% or less (Hetling et al., 2003).

Phosphorus loads reduction. There were no significant changes in effluent phosphorus loads in the LSB following the facility upgrade from secondary to tertiary in 1979. Phosphorus loadings started to decline in the early 1990s, when phosphorus was phased out from soaps and detergents, and further declined with BNR in 1997, from 1,912 kg/d to 417 kg/d, a reduction of 78% from previous levels. No comparable data are available for Chesapeake Bay but there was a reduction of 98% in terms of the amount of phosphorus discharged to the estuary, from 10,900 kg/d to 270 kg/d, during the period of upgrade from primary to secondary, from 1970 to 1985. In the case of the Hudson River, there was a 78% increase of phosphorus from primary to “subsequent upgrades to secondary treatment” (from 1920 to 1970), followed by a 63% reduction when upgraded to “full secondary” in 1972, in the wake of the state legislature’s ban on phosphorus-based detergents in 1973 (Hetling et al., 2003).

DO levels. After 1980 (nitrification was implemented in 1979), the level of DO shifted from a low of 0 mg/L (anoxia) to a low of 2.5 mg/L (outside the range of hypoxia (< 2 mg/L)). During the period from 1965 to 1980, the relative duration of dry season anoxia and hypoxia was 4% and 11%, respectively. For Chesapeake Bay, dissolved oxygen concentrations increased with the addition of nitrification to the wastewater treatment process during the early 1980s. In the years 1983, 1984, and 1985, “the average dissolved oxygen concentrations in the main channel below the Woodrow Wilson Bridge were usually above 5.0 mg/L” (Jaworski, 1990, p. 27). It bears mention that in the LSB, DO levels sometimes fall below 5 mg/L in summer. For the Hudson River, beginning in the late 1970s, DO concentrations generally increased through the 1980s and especially into the 1990s, corresponding with the upgrade to secondary treatment in the spring of 1991. In the years from the early 1970s to the 1990s, DO minima increased from less than 1.5 mg/L to more than 3.0 mg/L, with hypoxia during summer months greatly reduced (Brosnan & O’Shea, 1996). In the case of the Delaware Estuary, DO concentration from the late 19th century to the mid-20th century diminished from the saturation point to close to zero, and was close to saturation again as of 2010 . Before 1990, summer DO concentrations consistently fell below the Clean Water Act (CWA) standard of 3.5 mg/L. Since that time, summer DO concentrations are reported to have almost always been above the standard, indicating a successful recovery from chronic hypoxia (Sharp, 2010).

Nutrients in the Bay. Facility effluent BOD and nutrient loading were inversely correlated with nutrient concentration in the LSB, with the exception of NO_3^- for which the correlation was positive. Correlation decreases in direct correspondence to the distance between the Facility and the site. Ammonium concentration in the LSB declined after 1979. When ammonium declined, nitrate increased, in 1979, following the implementation of nitrification. There was a substantial decline, corresponding with BNR implementation. Phosphorus showed a decline in 1991, after which phosphorus decreased to a fraction of what it was previously, largely as a result of the ban on phosphorus detergents, as well as due to BNR. Ammonium, nitrate and phosphorus concentration declined, in direct proportion to the distance from the Facility.

In the case of the Delaware Estuary, the hypoxia in the mid-20th century has been attributed to a primary BOD (Sharp, 1994), the result of reduced carbon and nitrogen in sewage effluents (Sharp, 2010). The stations closest to the sewage treatment plant have been found to have high concentrations of nitrate and phosphorus. "All the nutrients showed decreases due to dilution going down the salinity gradient …." From the late 1960s through the 1980s, ammonium concentrations in the urban river showed a large decrease. The long-time increase in nitrate concentration was accompanied by a decrease in ammonium concentration. The result has been a decrease in nitrogenous oxygen demand corresponding with the DO increase. A change similarly occurred with phosphorus concentrations, although with a different pattern and for a different reason, partially the result of the ban on phosphate detergents. Probably also contributing to the decrease were improvements in sewage treatment combined with increased removal of phosphorus in sludge (Sharp, 2010, p. 544). Despite very high nutrient concentrations, DO was not influenced by excess algal production anywhere along the tidal freshwater stretch or the saline portion of the Delaware Estuary.

This research may be considered the most comprehensive study focusing specifically on evaluating how the long-term historical data (over a period of more than 50 years) demonstrate the effectiveness and performance of the Facility's wastewater treatment. The primary contribution that the study findings make to the existing literature in this area is to validate the overall benefits of improvements with regard to the LSB's physical environment that have been made to the Facility over time.

As population will almost inevitably continue to increase, so will influent flow and load, requiring higher capacity on the part of the facility that processes what is coming in, constituting cause for concern. In this case, however, the Facility at present is actually operating *under* the design capacity, probably because the Facility was originally intended to handle the effects of a large canning industry, which virtually disappeared during the late 1980s. The Facility therefore has the ability to continue to support the area's growing population, in coordination with the city's Master Plan for rehabilitating the Facility's aging infrastructure, consisting of process changes and long-range capital projects that will enable the Facility to meet future regulatory requirements and population demands using sustainable, energy-efficient, and cost-effective solutions.

The implications of the findings showing the benefits of improvements with regard to the LSB's ecosystem over time suggest that it would be highly advisable to continue along the lines of what has been done, especially in recent years.

Wahlin and Grimvall (2008) have pointed to "strong evidence that long-term trends in measured nutrient concentrations can be more extensively influenced by changes in sampling and laboratory practices than by actual changes in the state of the environment" (Wahlin & Grimvall, 2008, p. 115). This suggests the importance of exercising caution with regard to interpreting results, making sure to take into account any possible limitations in study design and data measurement that could possibly influence results and conclusions. In the context of this study, we may identify three specific issues: data gap, methods changes, and method of detection limit (MDL) problems.

With regard to the first issue, the substantial data gap—occurring between 1992 to 2002 for LSB stations SB03, SB04, and SB05 (the three Bay stations, farthest away from the Facility) and between 1992 to 2012 for SB13 and SB15 (the two near field stations, closest in proximity to the Facility)—constrained the ability to compare data between sites and limited trend analysis, such as seasonal trend analysis, utilizing the wq package. Related to this is the issue of missing data, within periods during which data were collected. Second, as laboratory methods changes are implemented over time, uncertainties are introduced that make long-term analysis more tenuous. Third, the high method of detection limit of ammonium for the LSB also constrained the ability to conduct trend analysis and derive correlation.

In spite of the tremendous progress made in the San Francisco Bay Area, however, significant issues remain to be resolved, with regard to the effects of wastewater released into the environment. There is concern that the Bay may be losing its resilience toward eutrophication, due to less suspended sediment in the water column, resulting in algal growth being less light limited. As suggested in San Francisco Bay Nutrient *Management Strategy* (Feger et al., 2012), there is a strong need for a coherent nutrient science and management strategy for the Bay. More specifically, what is needed is development and implementation of a nutrient monitoring program, to fill the data gap and answer uncertainties.

As important as it is to minimize the negative effects of nutrient enrichment, it is also important to acknowledge that estuarine water conditions similar to those before nutrient enrichment will not necessarily be revived following nutrient removal. The deterioration of estuaries and complex coastal ecosystems is a condition that results from a combination of factors: nutrient enrichment, habitat alteration, depletion of higher tropic levels, and inhibition by contaminants other than nutrients (Sharp, 2010).

It is apparent from the evidence that improvements in wastewater treatment are beneficial for the environment and necessary for the long-term sustainability of the ecosystem. In terms of public policy, this suggests that high priority should continue to be paid to sustaining the momentum that has been achieved with regard to ongoing improvements, in spite of budgetary challenges.

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Appendix I: History of San José's Sewage Disposal and the Facility

The City of San José began construction of its first sewers in 1867, starting with a 3-foot-by-4-foot redwood box. Plans for a system designed to serve a population of 100,000 were submitted in 1870, when the population at the time was only 10,000. The main element of this combined sewerage and drainage system was a 60-inch brick line, completed in 1896, extending from the city's downtown area to the south end of San Francisco Bay. An extension to this line was made in 1930, to convey raw sewage to a point approximately 2.5 miles into Bay waters from the plant site (Young, 1974).

The City of San José in 1949 engaged the services of a consulting company (Hyde and Sullivan) to design the necessary sewerage facilities for treating its wastes, initiating the restoration of South San Francisco Bay. The 36 MGD (million gallons per day) facility that was completed and became operational in 1956 at a cost of approximately $3.7 million included a number of unit processes: prechlorination, screening, grit removal, primary sedimentatic anaerobic digestion and sludge lagooning. At its time of completion in 1956, the San José facility served a population of nearly 200,000.

In 1959, the City of Santa Clara purchased an interest in the San José treatment facility and outfall to the Bay. By 1960, under the partnership, the capacity of the Facility was expanded to 54 MGD, with the capital investment project costing approximately $5 million. In 1964, for the purpose of removing BOD and TSS, a secondary treatment facility, utilizing the activated sludge process, was completed, with capacity increased to 94 MGD, at a cost of approximately $30 million. The subsequent major expansion occurred in 1970 at an approximate cost of $23 million, with the addition of primary, secondary and chlorination facilities and an enhanced nominal capacity of 160 MGD. In 1979, nitrification and filtration processes (considered "advanced waste treatment" or "tertiary") were completed for the facilities, at a cost of $116 million, removing nitrogen- and phosphorus-based nutrients from the secondary effluent (WPCP, *n.d.*).

Three major spills constitute a significant part of the history of the Facility. In 1979, more than four billion gallons of marginally treated sewage were released into the Bay, representing the worst spill in the history of the Bay (Cloern & Oremland, 1979). The next year, in 1980, two other spills occurred, both of them during the August-September canning season. The first one discharged approximately 1.8 billion gallons of inadequately treated sewage, while the second, on September 28, resulted in a discharge of approximately two million gallons of primary treated sewage. In all cases, mechanical/operational failure, triggered by biological upset of the secondary treatment stage, was determined to be at fault. The problem originated with discharge by canneries during their peak season, with the Facility failing to operate as designed at a capacity of 143 MGD (McEnery, 1981).

For the purpose of restoring the Facility to its rated capacity, emergency modifications were promptly implemented. At this time, in 1980, the first stage of the expansion project was begun. When completed six years later, the Facility was certified to have an operational capacity of 167 MGD, which remains its current capacity today (WPCP, 1997). In 1997, the Facility reconfigured its secondary and nitrification processes into a parallel Biological Nutrient Removal (BNR) process, which removed nitrogen and phosphorus. The total approximate cost of capital improvements to the Facility since 1979, in 2009 dollars, was $472 million (CH2M Hill, 2009).

In 1990, the SWRCB ordered San José to implement actions that would protect the salt marsh in the LSB from conversion caused by dry-weather flows exceeding 120 MGD (Order WQ 90-5). In October 1991, an Action Plan developed by the City was approved by the Regional Water Quality Control Board (RWQCB), which outlined water conservation programs totaling a 15-MGD reduction to be achieved by 1996. The measures to control discharge flows included public education, indoor water conservation and water reclamation projects (City of San José, 1992).

South Bay Water Recycling (SBWR) was formed specifically for the purpose of implementing water reclamation projects. Construction of the SBWR pipeline system was completed in 1997, with the $140 million project including sixty miles of pipeline, four pump stations, and a reservoir (SBWR, 2001). As of 2013, the system delivered up to 19 million gallons of recycled water daily to its approximately 740 customers (SBWR, 2014). This recycled water has been used for a wide variety of applications, including irrigation, golf courses, public parks, cemeteries, dust control, street cleaning, and car washes.

In 2010, an agreement was reached between the Santa Clara Valley Water District (SCVWD) and the City of San José to build the Silicon Valley Advanced Water Purification Center, a $68-million-dollar advanced water treatment facility (originally scheduled for completion in mid-2013) that will produce up to eight million gallons per day of highly purified recycled water. The project has received $8.25 million from the federal American Recovery and Re-investment Act and $5.25 million from the California Department of Water Resources. The resulting highly purified water will be blended into existing recycled water provided by the neighboring, which will improve overall recycled water quality so that the water can be used for a wider variety of irrigation and industrial purposes.

The state-of-the-art facility will take treated wastewater from the San José-Santa Clara Regional Wastewater Facility (RWF) and purify it by using microfiltration, reverse osmosis and ultraviolet light. The result will be eight million gallons per day of highly purified water that is expected to match California primary drinking water standards (SCVWD Website).

The Plant Master Plan was launched to prepare for the future of what is now known as the Facility. The Plan provides a roadmap for replacing the Facility's aging facilities and infrastructure, and consists of process changes and long- range capital projects that will enable the Facility to meet future regulatory requirements and population demands using sustainable, energy-efficient, and cost-effective solutions.

The Plant Master Plan proposes more than 100 projects as part of a 30-year Capital Improvement Program (CIP). The projects are divided into three separate phases. Phase 1 (2012-2021), totaling $450 million, involves repair and rehabilitation. Phase 2 (2013-2021), totaling $416 million, involves new biosolids dewatering and drying, as well as new energy generation. Phase 3 (2021-2040), totaling $1,124 million, involves projects related to possible regulatory changes and ongoing repair and rehabilitation (Carollo et al., 2012).

Appendix II: Sewage Treatment Issues and Policy Development

Hyde and Sullivan (1946) reported that San José and Sunnyvale discharge raw sewage into the waters of Lower San Francisco Bay. In their conclusion, they stated that, "The disposal of raw and inadequately treated sewage and industrial wastes into the waters of Lower San Francisco Bay and its annexa has destroyed their esthetic character and at times and places has created a noisome mass, evil to look upon and disagreeable to smell" (Hyde & Sullivan, 1946, p. 175). This has created conditions in which the existence of fish has been largely destroyed both in many of the sloughs and in the southern portions of the Lower Bay. The report recommended constructing a primary treatment plant, to be followed by secondary treatment in extensive oxidation ponds. These oxidation ponds, which were to have been located in the tidal marshlands south of Coyote Slough, were never implemented.

A 1953 *Survey of Water Conditions in Lower San Francisco Bay* prepared by Brown and Caldwell for the City of San José and County of Santa Clara reported on the effects of sewage discharges in the southeast bay. The results of the survey showed a change from moderate pollution of the Bay waters in July of 1953 to extreme pollution in August and September. The rapid recovery in October was followed by a return to the conditions of July by the first of December. The seasonal nature of the sewage flow was connected with the activities of the food processing plants (the canning industry) (Hyde & Sullivan, 1946).

A 1961 *Pilot Study of Physical, Chemical, and Biological Characteristics of Waters and Sediments of South San Francisco Bay* was prepared for the San Francisco

Bay Regional Water Pollution Control Board by Harris et al. because of the lack of data and potential adverse effects of existing water quality conditions. As a result, the Research Consulting Board offered general recommendations, outlining a minimum three-year investigation. The purpose of the investigation was to determine existing water quality and sediment characteristics, "as well as to develop a quantitative characterization and inventory of wastes discharges to the Bay" (Harris et al., 1961, p. 1). In its recommendations, the study suggested that the program be continued for at least two additional years and expanded to include sampling stations north of the San Mateo Bridge.

Final Report: A Comprehensive Study of San Francisco Bay (Volume VIII, Summary, Conclusions and Recommendations), published in July 1970, by the Sanitary Engineering Research Laboratory College of Engineering and School of Public Health of the University of California at Berkeley (SERL Report No. 67-5), covered the four study year periods from 1960-61, 1961-62, 1962-63 and 1963-64. The investigation was described as "probably the most extensive program ever undertaken in an estuary to characterize the water and sediment quality as well as the waste discharges having potentially adverse affects [sic] on the estuary" (Pearson et al., 1979, p. 62). In its assessment of major water quality problems in San Francisco Bay, the discussion section stated that "the study [did] not reveal gross or major water quality problems except, perhaps, in the southern most portions of the Bay." At the same time, however, it was acknowledged that "a number of very disturbing conditions" (Pearson et al., 1979, p. 67) were revealed through more extensive analysis of the data. Recommendations included establishment of a monitoring program, involving a minimum of 20 key sampling stations, for the most part near the main channels of the Bay.

The March 1969 *San Francisco Bay-Delta Water Quality Control Program* final report in its findings and recommendations section acknowledged the existence of significant water quality deterioration in the Bay-Delta, stating that this deterioration will worsen as a result of the accelerating growth of both population and industry. The recommended plan called for implementing a regional system involving the construction in three phases, the first of which comprised a network of interceptors to transport treated wastewaters from the San José area and Contra Costa County, as well as Marin and Sonoma Counties, to more central areas of San Francisco Bay, where the wastewaters could be flushed to the ocean through the Golden Gate. "The second and third phases of the recommended plan further transport treated wastewater effluents to the ocean and provide for progressively increasing wastewater reclamation as demands for reuse of wastewater and supplemental water supplies increase" (SWRCB, 1969, p. 2-2).

In June 1971, an *Interim Water Quality Control Plan for the San Francisco Bay Basin* was submitted by the California Regional Water Quality Control Board. The plan was prepared to satisfy the requirements on the part of the federal and state governments with regard to construction grant programs, as well as the Porter-Cologne Act requirements for water quality control plans (RWQCB, 1971). The Interim Plan's overall objective was "to set forth a definitive program of actions designed to preserve and enhance water quality and protect beneficial water uses in a manner [resulting] in maximum social and economic benefits of the people of the State" (RWQCB, 1971, p. II-

1). The water quality objectives covered the regulation of all controllable factors, for the purpose of protecting the quality of Basin waters from deterioration. "The most effective means of doing this," the plan stated, "appears to be by a combination of improved treatment and relocation of discharges to areas where the wastes would receive adequate dispersion and assimilation during the interim period" (RWQCB, 1971, p. VI-1).

The Federal Water Pollution Control Act (FWPCA), also known as the Clean Water Act (CWA) of 1972 established national goals for eliminating discharges of pollutants into navigable waters and of attaining fishable and swimmable waters. As part of the CWA, Congress created a major public works financing program for municipal sewage treatment. This involved a system of grants for construction of municipal sewage treatment plants. The initial permits issued in the 1970s and early 1980s by the National Pollution Discharge Elimination System (NPDES) permit program under the CWA focused on Public Owned Treatment Works (POTWs) and industrial wastewater.

The final report for the *Water Quality Management Plan for South San Francisco Bay* prepared by Consoer-Bechtel in March 1972 described a Bayside Dischargers Plan, the major features of which included, among other things, (1) consolidation of treatment plants, (2) general upgrading of the level of treatment to include filtration and substantial nitrification, and (3) export of wastewater from the South Bay.

The SWRCB in 1974 issued its *Water Quality Control Policy for the Enclosed Bays and Estuaries of California* for the purpose of providing water quality principles and guidelines, in order to prevent water quality degradation and to protect the beneficial uses of waters of enclosed bays and estuaries. The policy, still in effect, included a general prohibition against the discharge of municipal and industrial wastewater to enclosed bays and estuaries, including prohibition of discharge south of the Dumbarton Bridge. The policy allowed for a Regional Board to grant exceptions to this prohibition, in cases where “the Regional Board finds that the wastewater will be consistently treated and discharged in a manner that would enhance the quality of the receiving waters above that which would occur in the absence of the discharge” (SBWRQCB, 1990, p. 6).

The Water Quality Control Plan of July 1975 (major revisions of which were adopted in 1982, 1986, 1992, 1995, 2002, and 2004) developed water quality objectives from data reviewed during the planning process, as well as from both published and unpublished literature (Brown & Caldwell, 1975). The Plan listed various water quality objectives, for the protection of beneficial use, for waters inland from Golden Gate. An objective of 5.0 mg/L minimum with regard to DO was applied to all tidal waters in the Bay downstream of the Carquinez Bridge (Brown & Caldwell, 1975).

In “Chemistry and Microbiology of a Sewage Spill in South San Francisco Bay,” Cloern and Oremland (1983) reported on the breakdown of the San José-Santa Clara Waste Treatment Facility during three particular weeks in September 1979. This breakdown resulted in the discharge of a large volume of primary-treated sewage into South San Francisco Bay through its receiving water tributary, Coyote Creek. The article is perhaps most significant for its substantiation of two paradoxical key principles associated with the discharge of sewage into estuaries: (1) “the finite capacity of receiving waters to assimilate wastes,” and (2) the tremendous resilience of aquatic ecosystems, “even to extreme perturbations” (Cloern & Oremland, 1983, p. 404).

The *South Bay Dischargers Authority Water Quality Monitoring Program Final Technical Report* (a joint venture between Larry Walker Associates and Kinnetic Laboratories), published in August 1987, covered the period from December 1981 to November 1986. The South Bay Discharges Authority (SBDA) Water Quality Monitoring Program was a five-year study of the water quality and biological resources of the South Bay. With regard to major findings, it was stated that DO depressions in the South Bay have historically been a focus for concerns about water quality. Information from the study, however, showed that violations of Basin Plan objectives should not be attributed to the SBDA plants when operating at the observed treatment levels.