Lake Sammamish Water Quality
Response to Land Use Change

December 2014

King County
Department of Natural Resources and Parks
Water and Land Resources Division
Science and Technical Support Section
King Street Center, KSC-NR-0600
201 South Jackson Street, Suite 600
Seattle, WA 98104
206-477-4800  TTY Relay: 711
www.kingcounty.gov/EnvironmentalScience

Alternate Formats Available
Cover photo:

Top right corner: 2013 Pictometry. The dataset consists of tiled orthogonal imagery produced from nadir images captured by Pictometry International. This ortho-mosaic is a visualization product and is not intended or valid for authoritative or definitive use.

Bottom left corner: 1936 Aerial Photos for western King County panchromatic imagery. The source is the original hardboard-mounted aerial prints. The original 0.5 foot pixel resolution was resampled to 1 foot pixel. The purpose of the imagery is historical comparison and archiving original documents.
Lake Sammamish Water Quality Response to Land Use Change

Submitted by:
Eugene B. Welch
Emeritus Professor, University of Washington
Consulting Limnologist

Debra Bouchard
King County Science and Technical Support Section
Water and Land Resources Division
Department of Natural Resources and Parks
Seattle, WA 98104
Acknowledgements

Water quality sampling and analysis was conducted by the King County Environmental Laboratory. We would like to thank Jeff Droker and Jean Power for their help sifting through boxes of historical data records and for describing detailed sampling protocols. We are also grateful to Katherine Bourbonais for her assistance coordinating our data needs and making sure our questions were answered.

We would like to thank staff in the King County Science and Technical Support Section for their help. Rachael Gravon and Andrew Miller provided help with data analysis. Sally Abella, Curtis DeGasperi, Jim Simmonds, and Kate O’Laughlin provided significant contributions with their technical review. A special thank-you goes to Jonathan Frodge with Save Lake Sammamish for his technical review. Warren Perkins authored the Lake Sammamish Total Phosphorus Model, frequently cited in this report as King County 1995.

Project funding was through the Major Lakes Monitoring Program, supported by the Science and Technical Support Section of King County Department of Natural Resources and Parks' Water and Land Resources Division.
Citation

King County. 2014. Lake Sammamish Water Quality Response to Land Use Change. Prepared by Dr. Eugene B. Welch, Emeritus Professor, University of Washington, and Consulting Limnologist, and Debra Bouchard, Science and Technical Support Section, King County Water and Land Resources Division, Seattle, WA.
Table of Contents

EXECUTIVE SUMMARY .................................................................................................................. viii
1.0 INTRODUCTION .......................................................................................................................... 1
  1.1 Study Purpose ............................................................................................................................. 1
  1.2 Lake Sammamish Characteristics ............................................................................................ 2
  1.3 Background .................................................................................................................................. 4
    1.3.1 Past Studies ............................................................................................................................. 5
    1.3.2 Watershed Changes .................................................................................................................. 8
    1.3.3 Increased and Ongoing Development .................................................................................. 10
    1.3.4 Lake Sammamish Initiative and Management Plan .................................................................. 17
2.0 METHODS ...................................................................................................................................... 18
  2.1 Lake Sampling ............................................................................................................................. 18
  2.2 Stream Sampling ......................................................................................................................... 21
  2.3 Laboratory Analysis ..................................................................................................................... 23
  2.4 Statistical Analysis ...................................................................................................................... 23
  2.5 Water Quality Parameters ......................................................................................................... 24
    2.5.1 Temperature ........................................................................................................................... 24
    2.5.2 Dissolved Oxygen .................................................................................................................. 25
    2.5.3 Phosphorus ............................................................................................................................ 26
    2.5.4 Sediment Phosphorus Recycling and DO Demand ................................................................. 27
    2.5.5 Nitrogen ................................................................................................................................ 28
    2.5.6 Conductivity ........................................................................................................................... 28
    2.5.7 Alkalinity ............................................................................................................................... 29
    2.5.8 pH ........................................................................................................................................ 29
    2.5.9 Transparency .......................................................................................................................... 30
    2.5.10 Phytoplankton and Chlorophyll a .................................................................................... 30
    2.5.11 Total Suspended Solids ....................................................................................................... 30
3.0 RESULTS ....................................................................................................................................... 32
  3.1 Water Inflows ............................................................................................................................. 32
    3.1.1 Precipitation .......................................................................................................................... 32
    3.1.2 Issaquah Creek Flows ........................................................................................................... 34
3.1.3 Tributary Water Quality ........................................................................................................ 35
3.1.4 Tributary Phosphorus Loading ................................................................................................. 38
3.1.5 Residence Time and TP Load .................................................................................................. 39
3.2 Temperature .................................................................................................................................. 39
3.3 Alkalinity, pH, and Conductivity .................................................................................................... 42
3.4 Trophic State Indicators .................................................................................................................. 45
  3.4.1 Phosphorus .............................................................................................................................. 45
  3.4.2 Nitrogen .................................................................................................................................... 51
  3.4.3 Transparency and Chlorophyll a (chl a) .................................................................................... 54
  3.4.4 Areal Hypolimnetic Oxygen Demand ...................................................................................... 58
4.0 DISCUSSION ................................................................................................................................. 62
  4.1 Ongoing Recovery from Wastewater Diversion .......................................................................... 62
  4.2 Phosphorus and Development ................................................................................................... 62
  4.3 Trophic State ............................................................................................................................... 66
  4.3.1 TP and Chl a ............................................................................................................................ 66
  4.3.2 DO Demand ............................................................................................................................. 68
5.0 CONCLUSIONS AND RECOMMENDATIONS .......................................................................... 70
6.0 REFERENCES ................................................................................................................................. 73

Figures

Figure 1.1 Lake Sammamish location in Washington State, WRIA 8, and with watershed boundary. ........................................................................................................................................ 3
Figure 1.2 Single-family residential land use year built by decade and current multi-family, commercial, and forest land use (2011) ....................................................................................................................................... 13
Figure 1.3 Permitted land use and urban growth boundary in the Sammamish Watershed. (King County. 2014. Urban Growth Boundary. King County Department of Permitting and Environmental Review. Seattle, WA.) ........................................................................................................ 14
Figure 1.4 Impervious area in the west, east and Issaquah basins in 1991 and 2011 ........................................................................................................................................................................ 16
Figure 2.1 Depth contours and water quality sampling stations on Lake Sammamish ............................................................................................................................................. 20
Figure 2.2 Tributary Sampling Locations ........................................................................................... 22
Figure 3.1 Monthly accumulated precipitation at the National Climatic Data Centers Sea-Tac International Airport Station (# 24233) calculated for each study year 1990 to 2011 ........................................................................................................ 33
Figure 3.2  Annual total precipitation from 1963 to 2013 and the 50 year historical average (1948–2014) measured at the National Climatic Data Centers Sea-Tac International Airport station (#24233). ................................................................. 34

Figure 3.3  Annual average Issaquah Creek flow (water year) and wet season (November – March) from 1964 through 2012. ................................................................. 35

Figure 3.4  Volume-weighted temperatures in Lake Sammamish and Lake Washington during January and August. .................................................................................. 41

Figure 3.5  Onset, duration, end date, and maximum stability of thermal stratification in Lake Sammamish from 1993 through 2012. ................................................................. 42

Figure 3.6  Annual mean whole-lake alkalinity (as CaCO3) at 0612 from 1981 through 2011. (Means +/- 1 SD). P < 0.01. Note: 1980, 1981, 1984 and 1989 not included due to insufficient data. .................................................................................. 43

Figure 3.7  Average annual epilimnetic (0 – 10 m), metalimnetic (10 – 15 m), and hypolimnetic (>15 m) pH at 0612 from 1982-2011. Note: 1980 and 1981 not included due to insufficient data. .................................................................................. 44

Figure 3.8  Annual mean whole-lake conductivity (μmhos/cm) at 0612 from 1982 through 2011. (Means +/- 1 SD). Note: 1980 and 1981 not included due to insufficient data. .................................................................................. 44

Figure 3.9  Mean annual whole lake volume-weighted total phosphorus (TP) at 0612, 1964–2011, compared with lake management goal of 22 µg/L and projected full build-out (King County, 1995). Arrow indicates wastewater diversion in 1968. .................................................................................. 45

Figure 3.10 Volume-weighted mean summer hypolimnion (June – turnover) total phosphorus (TP) at 0612, 1964–2011. Arrow indicates wastewater diversion in 1968. .................................................................................. 46

Figure 3.11 Mean summer (June – September) epilimnetic total phosphorus (TP) concentrations at Station 0612, 1964–2011. Arrow indicates wastewater diversion in 1968. .................................................................................. 46

Figure 3.12 Volume-weighted whole lake winter total phosphorus (TP) at 0612, 1964–2011. .................................................................................. 47

Figure 3.13 Interval mean whole lake, summer hypolimnion (June – turnover), and summer epilimnion (June – September) total phosphorus (TP). All values volume-weighted. .................................................................................. 49

Figure 3.14 Relation between volume-weighted mean summer (June – turnover) hypolimnion total phosphorus (TP) (x) and mean annual total phosphorus (TP), 1964–2011. .................................................................................. 50

Figure 3.15 Mean volume-weighted annual whole-lake, summer (June – turnover) epilimnetic, and hypolimnetic soluble reactive phosphorus (SRP) concentrations at 0612, 1982–2011. .................................................................................. 51
Figure 3.16  Mean volume-weighted total N in the whole lake annually, and the summer (June to turnover) in the epilimnion and hypolimnion at 0612, 1993–2011. ..............................................................52

Figure 3.17  Mean volume-weighted nitrate+nitrite nitrogen in the whole lake annually, and the summer (June to turnover) in the epilimnion and hypolimnion at 0612, 1985–2011..................................................................................................................52

Figure 3.18  Annual and summer (June – September) volume-weighted epilimnion ratios for TN:TP, compared with the Redfield ratio of 7.2 (by weight) at 0612, 1994–2011..................................................................................................................53

Figure 3.19  Annual and summer (June – September) volume-weighted epilimnion ratios for DIN:SRP at 0612, 1994–2011. ..................................................................................................................53

Figure 3.20  Annual summer (June – September) Secchi transparency at 0612, 1964–2011, compared with the lake management goal of 4.0 meters. .................................55

Figure 3.21  Annual summer epilimnetic (June – September) chlorophyll a at 0612, 1964–2011, compared with the lake management goal of 2.8 meters.................................55

Figure 3.22  Chlorophyll and transparency summer means in compared to a typical Carlson relationship. ......................................................................................................................56

Figure 3.23  Biovolume of phytoplankton from 2003 to 2013. Cyanophyta refers blue-green algae. Chrysophyta are mostly diatoms. Other includes chlorophyta and Euglena. Note: data not recorded for 2008...............................................................57

Figure 3.24  Maximum spring bloom chlorophyll a and timing of spring bloom maximum chlorophyll a (March – May) since the 1960s......................................................58

Tables

Table 1.1  Physical characteristics........................................................................................................ 4

Table 1.2  Total developed land in the Sammamish watershed from 1900 to 2011 based on total “As Built” area (acres/hectares) for the specified time period .................................................................11

Table 1.3  Impervious area in the east, west, and Issaquah basins in 1991 and 2011. (acre/hectare)..........................................................................................................................15

Table 2.1  Period of record for tributaries entering into Lake Sammamish.................................21

Table 2.2  Laboratory methods and detection limits for conventional water samples........23

Table 3.1  Summary water quality parameter trend analysis, means and number of samples (n) for Lake Sammamish tributaries.................................................................37

Table 3.2  Summary of Historical Annual TP Loading Estimates for Issaquah Creek........38
Table 3.3  Interval mean (±SD) TP in µg/L in different time periods and portions of the lake. All values volume-weighted. Pre-1981 data from King County (1995). Wastewater was diverted in 1968. ...........................................................................................................48

Table 3.4  Interval mean (±SD) sediment P release rates in mg/m² per day for the time periods shown. Wastewater diversion was in 1968.................................................................49

Table 3.5  Interval mean (±SD) summer transparency (meters) and epilimnetic chl a (µg/L) in different time periods. Pre-1995 data from King County (1995). Wastewater was diverted in 1968. ...........................................................................................................54

Table 3.6  Response of areal hypolimnetic deficit rate (AHOD) to changes in TP input during pre- and post-diversion of wastewater, except the change in TP input to Lake Tenkiller was an increase from agricultural nonpoint runoff. .....59

Table 3.7  Summer (from May through when DO reaches ≤1 mg/L) mean volume-weighted DO concentration, (± 1 SD), in the hypolimnion below 15 m, and AHOD, 1981-2011. A hypolimnietic mean depth of 8.17 m and volume of 98 x 10⁶ m³ was used..............................................................61

Appendices

Appendix A  Volume Weighting

Appendix B  Water Quality Data – Means and SD

Appendix C  Stream WQ Time Series Plots

NOTE: The above appendices are bound separately from this document.
EXECUTIVE SUMMARY

As the sixth largest lake in Washington and the second largest in King County, Lake Sammamish is designated a water of statewide significance and is an important and valued natural resource. It is one of the major recreational lakes in King County and in 2014 was selected as one of eight national urban wildlife refuge programs by the United States Fish and Wildlife Service.

Water quality monitoring of Lake Sammamish began in the early 1960s when wastewater was being discharged into both Lake Washington and Lake Sammamish. Since then, the continued monitoring of the lake has provided data to evaluate changes in water quality, examine effects of land-use changes and different management actions in the watershed on the lake’s quality, and help agencies maintain awareness of water quality conditions and identify problems in order to inform management decisions. This study evaluates lake data collected from mid-1960s through 2011. Specifically, water quality data were analyzed with the following objectives:

- Describe the current lake water quality, specifically for the parameters total phosphorus, chlorophyll a, water clarity (Secchi transparency), rate of dissolved oxygen depletion, and temperature.
- Compare lake water quality to established trophic state indicators for lakes, and goals specific to Lake Sammamish.
- Determine any long-term trend(s) in water quality and assess them in relation to watershed changes such as deforestation, sewage diversion, and ongoing development before and after the adoption of the Lake Management Plan in 1996.
- Summarize data to inform future management decisions in the Lake Sammamish Watershed that affect the lake’s water quality.

In the mid-1960s, the Municipality of Metropolitan Seattle (METRO; now merged with King County Department of Natural Resources and Parks) conducted a study of Lake Sammamish water quality to determine if sewage discharged from the City of Issaquah’s wastewater treatment plant and a large dairy facility (Darigold) were having an adverse effect on the lake. (The population of Issaquah at the time was approximately 4,000 people). Wastewater diversion was proposed for Lake Sammamish even though its water quality was not degraded to the same degree as that in Lake Washington before diversion began there in 1963.

Wastewater diversion has been implemented on many lakes in the United States and has been shown to successfully reverse the process of eutrophication (nutrient enrichment) by reducing a lake’s total phosphorus content. High total phosphorus levels in lakes contribute to increased phytoplankton abundance as measured by high levels of chlorophyll a in the upper, warmer layers of a lake (the epilimnion) in the summer, and to low water clarity (transparency). Decomposition of settled phytoplankton is the principal cause for dissolved oxygen depletion in the hypolimnion (the deep layer in the lake). High total phosphorus
levels also contribute to greater frequency and magnitude of cyanobacteria blooms (also known as blue-green algae) that are often toxic and capable of producing adverse effects to the nervous system and liver. Therefore, an increase in total phosphorus, chlorophyll $a$, and dissolved oxygen depletion rate, and a decrease in clarity point to water quality degradation.

In 1968, METRO diverted wastewater from Lake Sammamish into a regional wastewater collection system and to a treatment facility in Renton that currently discharges to Puget Sound. The diversion reduced the external total phosphorus load to the lake by about 35 percent. After wastewater diversion, additional studies were conducted by the University of Washington to track improvements in lake water quality. Recovery of lake quality was slower than anticipated, but by the late 1970s, mean annual total phosphorus (volume-weighted, whole-lake) had decreased substantially from 32 µg/L (parts per billion) before diversion to below 20 µg/L. Summer epilimnetic (upper water layer) chlorophyll $a$ also declined and water clarity increased, providing evidence that wastewater and dairy waste diversion was effective in improving lake water quality.

In the mid-1970s, research on the effects from land development raised concerns about future water quality degradation in Lake Sammamish due to stormwater runoff from increasing urbanization. An intensive study in the mid-1970s of stormwater entering the lake’s west side indicated that future development could represent a threat to the lake’s water quality (Welch et al., 1980). At that time, only 14 percent of the west side of the lake was developed. That analysis prompted the establishment of METRO’s routine water quality monitoring program for Lake Sammamish in 1979. Ongoing monthly or twice monthly monitoring has provided over three decades of data that, combined with the prior data, were used here to determine further changes in water quality in response to watershed land-use changes. Results from this monitoring program showed that lake annual total phosphorus content gradually increased from an apparent equilibrium of around 17 µg/L in the 1980s to over 20 µg/L by the mid-1990s, likely related to the increased urbanization surrounding the lake (King County, 1995).

Given the probable effect of land conversion and development on lake water quality, King County Water and Land Resources Division and the University of Washington Department of Civil and Environmental Engineering cooperated in a modeling assessment of long-term land-use change and lake water quality in the mid-1990s (King County, 1995). The conclusion of that effort was that mean annual total phosphorus would increase from a post-diversion equilibrium varying around 17 µg/L to a predicted 28 µg/L with complete build-out in the watershed (based on zoning codes at the time) if no controls on runoff were instituted and only 30 percent of the forest were retained. (Under these conditions, total phosphorus was predicted to be nearly as high as before wastewater diversion.) For a scenario of 65 percent forest retention and 50 percent total phosphorus removal in runoff from new development, a smaller rise in total phosphorus to 24 µg/L was predicted. These projections prompted the creation of the 1995 inter-jurisdictional Lake Sammamish Initiative, and a citizens’ task force, Partners for a Clean Lake Sammamish, which worked to complete the 1996 Lake Sammamish Water Quality Management Plan. This plan set limits of 22 µg/L annual mean total phosphorus, and summer means for epilimnetic chlorophyll $a$
of 2.8 µg/L and water clarity of 4.0 meters. The plan also called for long-term watershed protections involving forest retention and runoff treatment to retain total phosphorus from newly developed land, as well as short-term actions to reduce phosphorus loading to the lake.

The assessment of lake conditions provided in this report shows that mean annual total phosphorus has not changed significantly since 1980. There has been substantial population growth in the basin and the amount of impervious area has greatly increased since the mid-1990s. However, annual total phosphorus has remained quite stable, ranging from 19.5 µg/L in the 1990s to 17.5 µg/L in more recent years. Also, mean summer chlorophyll a (about 3.5 µg/L) and mean summer total phosphorus (about 11.5 µg/L) have remained stable since the late 1990s and summer transparency has averaged over 5.0 meters since 1998. That is in marked contrast to the 1994 model-predicted summer total phosphorus of 24 µg/L, transparency of about 3.8 meters, and chlorophyll a of about 4.0 µg/L at build-out with 65 percent forest retention in rural areas and 50 percent total phosphorus retention from new development. These model predicted levels would have exceeded the set limits of the Lake Sammamish Management Plan, but actual levels did not.

This report describes several likely contributing factors that explain the lake’s stable annual and summer total phosphorus and chlorophyll a concentrations and the improvement in transparency since the mid-1990s. First, the concentration of total phosphorus in the lake’s hypolimnion during the summer when the lake is thermally stratified decreased by one-half by the 1970s due to a decline in the release of phosphorus from sediments. This decrease in internal phosphorus loading (over three fourths since pre-diversion, and 60 percent since the 1970s) is most probably a long-term response to wastewater diversion. Before wastewater diversion, total phosphorus in the hypolimnion reached maximum concentrations around 300 µg/L, but more recently those maximums were rarely over 60 µg/L. Decreased hypolimnetic total phosphorus was associated with a marked decrease in soluble reactive phosphorus – the phosphorus fraction released from sediment. Had hypolimnetic total phosphorus not decreased, mean annual whole-lake total phosphorus would be more like 23 instead of 17.5 µg/L.

Second, watershed protections instituted in the mid-1990s may have reduced total phosphorus in runoff that otherwise would have resulted from land-use change. These stormwater controls were in support of the Sensitive Lake Standard requiring 50 percent removal of total phosphorus in runoff from all new urban development and were implemented as part of King County’s 1998 Surface Water Design Manual and Ecology’s 2001 Stormwater Management Manual. Runoff controls (i.e., retention basins) installed with new development would have retained phosphorus bound to particulate matter and, therefore, may be partly responsible for the lake’s stable total phosphorus level. As evidence, total phosphorus concentration in Issaquah Creek, which supplies 70 percent of the lake’s inflow, has remained stable at a mean annual concentration between 40 and 50 µg/L over the past three decades. Other tributaries to the lake also show no significant increase in total phosphorus (and a decline in soluble reactive phosphorus) since the 1980s. That was despite a marked increase in developed land and impervious area, especially in the east side of the lake. While inputs of total phosphorus from the east and
west side streams did not increase as predicted with increased development, changes in inputs from other non-stream stormwater sources (e.g., nearshore pipes and ditches) may have increased, allowing mean annual total phosphorus to remain relatively stable rather than decline in conjunction with reduced loads from internal lake sediments.

Research has shown that stormwater retention basins are effective at removing particles, but not dissolved substances. It is therefore not surprising that this study found that conductivity, which is a measure of conservative (dissolved) substances in solution that do not settle out in retention basins, has increased in the lake’s tributaries. The conductivity increase was significant in two eastside creeks (Ebright and Eden) and two west side creeks (Idylwood and Lewis). The lake’s conductivity, while lower in magnitude than its tributaries, has increased more in the last three decades than the increase in Issaquah Creek, indicating that runoff from the near built-out east and west sides may have had a more substantial effect on the lake than Issaquah Creek.

Third, specific activities were undertaken to reduce or prevent increased discharge of total phosphorus to the lake. Multiple specific actions recommended by the Lake Sammamish Initiative were implemented that reduced total phosphorus discharges. Contrary to earlier projections, increased development and urbanization did not occur in rural areas following establishment of the urban growth boundaries (UGB), and the refinement of the urban growth area as part of the 1994 King County Comprehensive Plan. The Comprehensive Plan included special designations for the protection of farming areas and provisions for housing on larger lots in rural areas, forest production districts, and open space areas for protection of the natural environment. Forest land was acquired in the upper Issaquah Creek basin while forest cover declined inside the UGB. Also implemented were actions recommended in the 1994 Issaquah Creek Basin and Nonpoint Action Plan, and the 1994 East Lake Sammamish Basin and Nonpoint Action Plan to control pollution from future development.

Fourth, other regulatory changes have resulted in less total phosphorus loading to tributaries to Lake Sammamish. These changes include bans on phosphorus in dish soap, dishwasher detergent, and laundry detergent, which would otherwise be released in rural areas from on-site septic systems. These changes also include the implementation of the King County Critical Areas Ordinance, which intends to limit development in riparian areas along tributaries to the lake and on steep slopes that might erode and result in sediment transport and problems with increased total phosphorus in receiving waters.

Finally, there has been a long-term reduction in Issaquah Creek flow and the corresponding increased water retention time in the lake since the 1970s. Decreased flows probably reflect the lower rainfall that is related to the shift in the Pacific Decadal Oscillation (PDO) from a cool/wet phase to a warm/dry phase in 1976–1977. The increased residence time allows for greater settling of phosphorus containing particles to the lake bottom, contributing to improved water clarity.

Summarizing the likely factors contributing to the stable annual total phosphorus of 17–18 µg/L, despite an increase in developed land over the past two decades:
1. A declining sediment phosphorus release rate that resulted in a reduction in hypolimnetic total phosphorus by nearly 50 percent since the 1970s – a legacy of the diversion of wastewater and dairy waste away from the lake. Without that reduction, internal loading of phosphorus would have been greater and resulted in a higher annual total phosphorus of approximately 23 µg/L.

2. Instituted stormwater controls on new development (e.g., the Sensitive Lake Standard requiring 50 percent total phosphorus removal for all new urban development) may have reduced the amount of particulate phosphorus entering the lake via stormwater runoff. Issaquah Creek and other tributaries show no significant increase in total phosphorus, and soluble reactive phosphorus declined in regional streams, despite a marked increase in developed land and impervious area, especially in the east side basin, and despite increased conductivity in the lake and some streams.

3. Specific actions identified in the Lake Management Plan and the two basin plans intended to reduce non-point pollution may have been effective in reducing erosion and particulate phosphorus inputs to tributaries and the lake. In addition, recent salmon recovery efforts to protect riparian habitat may have reduced erosion and particulate phosphorus inputs.

4. Other regulatory changes banning the use of phosphorus in household products may have resulted in less total phosphorus loading to tributaries to Lake Sammamish.

5. A long-term reduction in Issaquah Creek flow and increased water retention time in the lake since the 1970s allows for greater settling of phosphorus containing particles to the lake bottom, contributing to improved water clarity.

In addition, the oxygen depletion rate in Lake Sammamish’s hypolimnion has not changed since before wastewater diversion, although annual average temperature has increased at the same rate as in Lake Washington – a quarter of a degree centigrade per decade. Increased warming has led to an earlier onset of stratification in Lake Washington and is predicted to occur in Lake Sammamish in the future. The spring algae (diatom) bloom, which is timed with increased stratification, appears to be occurring earlier in Lake Sammamish when comparing maximum chlorophyll a values since the 1960s. Studies suggest that a longer period of stratification and a continued relatively high rate of dissolved oxygen depletion in the hypolimnion will further shrink the habitat in the middle layer of the lake with adequate oxygen supply and cool temperatures to be acceptable for growth of salmonid fishes. Thus, climate change may already be having an adverse effect on the lake’s water quality.

Based on this investigation, the following are recommended to ensure that Lake Sammamish water quality is maintained for the next 25 years:
Enforce Management Strategies

- Continue to enforce strategies to control phosphorus input and instituted land-use zoning, which were put into place by multi-jurisdictional efforts that resulted in the Lake Sammamish Initiative in 1995.

Monitoring

- Continue to collect and evaluate lake water quality data. As this report demonstrates, a consistent long-term data set to evaluate trends over time is essential to evaluate the effectiveness of management strategies in response to the lake's condition, and make adjustments as needed.

- Continue to monitor tributary water quality. A five-year hiatus (2009-2013) in water quality data in some of the Lake Sammamish tributaries occurred due to budgetary constraints. A consistent long-term data set to evaluate trends over time in loading of pollutants to the lake is essential to evaluate the effectiveness of management strategies.

- Continue to collect routine lake phytoplankton (algae) and zooplankton data. King County began collecting phytoplankton data in 2003 and has intermittently collected zooplankton data. Having a consistent long-term data set on the plankton community structure will enable future analyses of food web (includes impacts to kokanee and other species) interactions and their response to regional climate changes, which may have already affected the lake.

- Maintain and expand the flow-monitoring gaging network for tributaries to Lake Sammamish. Flow is currently measured in Issaquah Creek, Lewis Creek, and Laughing Jacobs Creek. Installing gages at other tributary mouths where water quality data are collected (e.g., Idylwood Creek, Eden Creek, Ebright Creek, Pine Lake Creek, and Tibbetts Creek) would allow comparison of flow and phosphorus (and other constituent) loading in the future.

Outreach

- Conduct outreach to jurisdictions and stakeholder groups within the Lake Sammamish watershed to maintain efforts to protect of the lake and ensure that water quality improvements from diversion and other planning efforts are not overshadowed by future development.
This page intentionally left blank.
1.0 INTRODUCTION

1.1 Study Purpose

This study evaluates lake data collected from 1960 through 2011 to describe how Lake Sammamish has responded to watershed development and associated nutrient inputs. Long-term data sets are often necessary to detect changes in water quality due to changes in watershed development over and above the normal short-term variability due to seasonal and year-to-year patterns in climate and other natural factors.

Specifically, water quality data were analyzed with the following objectives:

- Describe the current status of the lake’s quality relative to established trophic state indicator boundaries for transparency (water clarity), rate of dissolved oxygen (DO) depletion, total phosphorus (TP), and chlorophyll a (chl a), as well as goals specific to Lake Sammamish.
- Determine any long-term trend(s) in water quality and their relation to watershed changes such as deforestation, sewage diversion, and ongoing development.
- Provide data to inform future management decisions in the Lake Sammamish Watershed that impact the lake’s water quality.

This study is part of the ongoing King County Major Lakes Monitoring Program that assesses water quality in Lake Washington, Lake Sammamish, and Lake Union. The King County Major Lake and Stream Monitoring Program is designed to protect the significant investment in water quality improvement and protection made by the people of King County beginning in the 1950s. With the formation of METRO in 1958, construction began on two of the county’s regional treatment plants, West Point in Seattle’s Magnolia neighborhood and South Treatment Plant in Renton. By the late 1960s, regional water quality began improving dramatically. The improving water quality was an important milestone and was tracked by this program. The long-term environmental impacts of all types of pollution that affect the quality of lakes and streams can only be evaluated by sampling at multiple sites throughout the watershed. In 1994, King County assumed authority of METRO and its legal obligation to treat wastewater for 34 local jurisdictions and local sewer agencies that contract with King County. The County also assumed responsibility for continuing the monitoring of these surface waters.

As the sixth largest lake in Washington and the second largest in King County, Lake Sammamish is an important and valued natural resource. It is one of the major recreational lakes in King County, and receives high use by fishermen, boaters, water skiers, swimmers, and picnickers. There are both State and County parks along the shore, and the lake has been designated a water of statewide significance and is one of eight national urban wildlife refuge programs designated by the United States Fish and Wildlife Service (USFWS, 2013).
1.2 Lake Sammamish Characteristics

The lake is located within the Cedar-Sammamish Watershed, or Water Resource Inventory Area (WRIA) 8 (Figure 1.1). The major tributary to the lake is Issaquah Creek, which enters at the south end of the lake and contributes about 70 percent of the surface water flow and 60 to 70 percent of the TP loading (Moon, 1973; King County, 1995). Tibbetts Creek to the south and Pine Lake drainage to the east contribute about 6 percent and 3 percent respectively of the surface water external TP load (Moon, 1973). The other tributaries are smaller and contribute less to the total load. The outlet level of the lake is controlled by a weir in Marymoor Park at the north end of the lake.

The basin of the lake was formed by glaciation 14,000 years ago into a long, uniform trough with steeply sloping sides. The lake has a maximum depth of 32 m (105 ft.). The physical characteristics of Lake Sammamish and its drainage basin are summarized in Table 1.1.

Lake Sammamish is monomictic having one mixing and one stratification period per year. December through February the lake is isothermal and completely mixed from the surface to bottom. Typically by June, stratification due to density difference is relatively strong. The lake generally turns over in late autumn–early winter when the density difference between the surface waters (epilimnion) and deeper waters (hypolimnion) is insufficient to resist wind mixing. The thermocline, or metalimnion, gradually sinks and the epilimnetic zone of mixing increases. That entrainment process transports nutrients into the photic zone upwards for use by algae.
Figure 1.1  Lake Sammamish location in Washington State, WRIA 8, and with watershed boundary.
Table 1.1  Physical characteristics.

<table>
<thead>
<tr>
<th>Characteristic</th>
<th>English Units</th>
<th>Metric Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Watershed Area (including the lake)</td>
<td>63,012 acres</td>
<td>255 km²</td>
</tr>
<tr>
<td>Lake Area</td>
<td>4,893 acres</td>
<td>19.8 km²</td>
</tr>
<tr>
<td>Lake Volume</td>
<td>265,103 acre-ft</td>
<td>3.27 x10³ m³</td>
</tr>
<tr>
<td>Mean Depth</td>
<td>54 ft</td>
<td>16.5 m</td>
</tr>
<tr>
<td>Maximum Depth</td>
<td>105 ft</td>
<td>32 m</td>
</tr>
<tr>
<td>Flushing Rate</td>
<td>0.56 per year³</td>
<td></td>
</tr>
<tr>
<td>Depth of the Epilimnion</td>
<td>33-39 ft</td>
<td>10-12 m</td>
</tr>
<tr>
<td>Length</td>
<td>8.1 miles</td>
<td>13 km</td>
</tr>
<tr>
<td>Main Inflows</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• Issaquah Creek (70 percent of surface</td>
<td></td>
<td></td>
</tr>
<tr>
<td>water flow and TP loading)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• Tibbetts Creek (6 percent TP loading)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>• Pine Lake Creek (3 percent TP loading)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Outlet</td>
<td>Sammamish River</td>
<td></td>
</tr>
<tr>
<td>Trophic State</td>
<td>Mesotrophic</td>
<td></td>
</tr>
</tbody>
</table>

a  Volume from Birch 1976
b  Water renewal rate, or flushing rate, is the fraction of the lake's volume replaced per year.

1.3  Background

Historically, the lowlands areas of the basin were cleared of forest and converted to agricultural uses. Urban and suburban uses were concentrated in a few small communities such as Issaquah, Redmond, and east Bellevue. Over time agricultural uses have been replaced with more commercial and residential development. Beginning in the 1960s, large areas of the basin have experienced rapid urban and suburban development as part of the overall growth in King County (Figure 1.2), subsequently transforming the lake’s watersheds, particularly the east and west basins into urban and suburban land uses.

In 1968, the Municipality of Metropolitan Seattle (METRO; now King County Department of Natural Resources and Parks) diverted the discharge from the City of Issaquah’s wastewater treatment plant and a large dairy facility (Darigold) from Lake Sammamish to a regional wastewater collection and treatment facility in Renton that discharges to Puget Sound. (The population of Issaquah at the time was approximately 4,000 people.) The diversion reduced the external TP load to the lake by about 35 percent. Numerous studies have been conducted to evaluate the changes in water quality. Monitoring has continued
over the years to help county agencies maintain awareness of water quality conditions and to identify problems in order to inform management decisions and ensure that water quality improvements from diversion are not overshadowed by continued development in the watershed.

### 1.3.1 Past Studies

Numerous reports and publications have been written on Lake Sammamish quality and inputs from its watershed. The earliest record of water quality in the lake was an 1895 anonymous article in Scientific American, which stated, “one can see down into the glassy mirror like depths for 30 feet or more” (Birch, 1976). In 1897, Evermann and Meek recorded the first water quality complaint in the watershed: “The water in Issaquah Creek is said to be very clear in the summer and fall months, or was, previous to the coal company using the creek as a dumping ground for coal screenings, since which time the water has become muddy. There are a number of coal mines in this region, and the people complain of the screenings from them being dumped into the creeks, thereby doing considerable injury to the fishing grounds.” Evermann and Meek went on to report that the lake was “considerably discolored by vegetable matter and sediment washed from the surrounding hills and brought down by the swollen creeks” (Rock, 1974). A one-day survey in 1913 reported a Secchi depth of 3.3 meters in August (Kemmerer et al., 1923).

The following list summarizes studies conducted on Lake Sammamish by agency and/or institutions that sponsored the efforts. However, this list is not meant to be completely comprehensive. Discussion of relevant information from past studies will follow.

**METRO Study, 1964–1965**

The newly created METRO investigated the trophic state of the lake during an 18-month period from July 1964 to December 1965 (Isaac et al., 1966). The purpose of the study was to assess the impact of projected growth in the east side communities, as well as to anticipate the effect of the proposed sewering of the Lake Sammamish basin. This report recommended the treatment plant effluent and the processing waste from the Darigold plant be diverted out of the basin.

**University of Washington, 1969–1975**

The University of Washington monitored the lake and Issaquah Creek inflow for six years from late 1969 to 1975 following the diversion in 1968 of wastewater that discharged directly to the lake. Results were recorded in several theses and publications. Initially it appeared there was little change in trophic state in the years directly following diversion (Emery et al., 1973). Further data gathering through 1975 indicated there had been a positive response to the reduction in phosphorus (P) loading as evidenced by a reduction in hypolimnic P concentrations (Welch et al., 1980).

Annual water and nutrient budgets for the lake were developed by Moon (1973) as part of a master’s thesis, University of Washington. Moon estimated that 70 percent of the total annual hydraulic and external TP loading to Lake Sammamish came from Issaquah Creek...
and most of the remainder from 11 minor tributaries. That estimate was used in several subsequent modeling efforts.

**University of Washington Stormwater Studies 1977–1978**

Three major stormwater discharges on the west side of Lake Sammamish were intensively monitored to determine the extent of non-point source TP loading (Welch and Perkins, 1980). Based on these data, it was concluded that TP loading to the lake would increase substantially with future urbanization (Welch et al., 1980).

**METRO Study 1979–1980**

METRO monitored Lake Sammamish from May 1979 to May 1980 as part of a study of Seattle/King County regional lakes (Davis and Swartz, 1981). The water quality of the lake appeared to be improving based on the 1979-80 mean Secchi disk depth of 4.5 m compared with the 1971-75 mean of 3.3 m. Due to the infrequent sampling, conclusions were difficult to draw from the data.

**METRO Study 1981–present**

METRO began monitoring Lake Sammamish on a routine basis as part of the Major Lakes monitoring project beginning in April 1981. King County Department of Natural Resources and Parks (DNRP) continues to collect samples at multiple depths and locations.

**University of Washington modeling efforts 1985–1987**

A whole-lake TP model was developed to estimate the effects of increased urbanization within the Lake Sammamish watershed (Schuster, 1985; Welch et al., 1986). Effects of bioavailable P loading to the lake were examined (Butkus, 1987; Horner et al., 1987).

**Lake Sammamish Water Quality Management Project 1989**

A cooperative effort in watershed planning between the Washington State Department of Ecology, METRO, King County, and the Cities of Bellevue, Redmond, and Issaquah resulted in the Lake Sammamish Water Quality Management Project Report (Entranco, 1989). This report combined the work from the mid-1980s to project future water quality of the lake with and without P controls.

**Lake Sammamish Total Phosphorus Model 1995**

In 1995, the County produced an updated TP model. This model incorporated approximately 10 years of data and land-use information that was unavailable during previous modeling efforts in the mid-1980s. The new model predicted TP in surface and bottom waters and was used as a management tool to estimate future water quality based upon various land-use alternatives in the watershed and P control strategies (King County, 1995; Perkins et al., 1997).

**Lake Sammamish Initiative 1995**

The Lake Sammamish Initiative began in August 1995 in response to current and future impacts to water quality due to urbanization within the basin. The Initiative was an inter-
jurisdictional effort begun by County Executive Gary Locke and supported by the mayors of
the cities of Bellevue, Issaquah, and Redmond. Executive Locke appointed eight citizens
within the watershed to a task force called Partners for a Clean Lake Sammamish. The
Partners were asked to recommend future water quality goals for the lake, to develop a set
of management actions to achieve these goals, and a financial plan to pay for the actions
Sammamish,” was an important stimulus to this effort in the early 1990s.

Development of a Three-Dimensional Hydrodynamic Model of Lake Sammamish 2008

A 3-dimensional hydrodynamic model of Lake Sammamish was developed to support
analyses of the impacts of possible future land use and climate change scenarios on
hydrology and water quality, and the resulting impacts on aquatic biota (King County,
2008). An existing coupled 3-D hydrodynamic and water quality modeling framework
originally developed for the Chesapeake Bay Program (CH3D-Z and CE-QUAL-ICM) was
selected for application to Lake Sammamish and Lake Washington. The CH3D-Z model code
used in the ACOE-ERDC Lake Washington application was used with minor modifications to
simulate Lake Sammamish hydrodynamics. The current version of the model is capable of
reliably reproducing the seasonal and spatial thermal dynamics of the lake based on
comparison to routine temperature profile data collected between 1995 and 2002. Small
systematic, seasonal, inter-annual, and spatial (primarily vertical) thermal errors remain
that will require further data collection, testing, and refinement to reduce. Nonetheless, the
model represents one of several potentially useful tools to evaluate the effects of land use
and climate change on the aquatic resources of King County.

Regional stream studies, 2005–2006

Several studies were conducted by the University of Washington on non-point source
impacts from urban development in regional streams.

- A 10-year record of stream nutrient and sediment concentrations was evaluated for
  17 streams in the greater Seattle region to determine the impact of urban non-point
  source pollutants on stream water quality. Issaquah and Tibbetts creeks were part
  of the assessment (Brett et al., 2005a).
- Over a 1-year period, daily total phosphorus and weekly SRP were measured in four
  Seattle area streams, including Issaquah Creek, spanning the gradient of forested to
  urban dominated land cover (Brett et al., 2005b).
- Phosphorus bioavailability as a function of stream flow in forested, urban,
  agricultural (i.e., dairy farm) and mixed land cover streams was evaluated at 16
  stream sites, including Tibbetts and Issaquah creeks (Ellison and Brett, 2006).

Effects of a Temperature-Oxygen Squeeze on Distribution, Feeding, Growth, and Survival of
Kokanee (Oncorhynchus nerka) in Lake Sammamish 2009

The seasonal and diel distribution of salmonids in response to changing dissolved oxygen
and temperature profiles in Lake Sammamish, Washington were examined by combining
concurrent limnological measurements with gill netting and hydroacoustic surveys (Berge,
2009). Thermal stratification intensified through summer and fall with increasing temperatures in a deepening epilimnion. As the summer progressed, low hypolimnetic DO encroached into the metalimnion, creating a temperature-DO squeeze that reduced the amount of favorable (<17°C and > 4 mg/L) habitat available for salmonids by as much as 90 percent. Kokanee *Oncorhynchus nerka* and cutthroat trout *O. clarki*, responded to these limnological changes by moving to the metalimnion during peak stratification. The consequent overlap among zooplankton, juvenile kokanee, and piscivorous cutthroat trout influenced growth efficiency, condition, and spatial-temporal patterns of predation risk for kokanee. Climate change model projections suggest more protracted thermal stratification in future years, creating an uncertain future for kokanee in Lake Sammamish.

**Predicting Climate Change Effects on Kokanee Habitat Suitability in Lake Sammamish 2013**

Two lake temperature models were used to evaluate impacts of climate change on Lake Sammamish. This work utilized 2- and 3-dimensional lake temperature models developed as part of an earlier study to simulate observed lake temperatures over an 8 year period (1994-2002). Both models were generally consistent in their predictions of warming throughout the lake in response to future warming of the local climate, with a disproportionate amount of warming predicted to occur in the surface layer in the summer. Overall, the available habitat for kokanee is predicted to decline in response to warming. The decline in habitat volume also results from earlier onset of stratification and delay of destratification, which results in an extension of the period that the lake is stratified during the summer. The earlier onset of stratification results in warmer surface waters in the spring than would have occurred historically. The lake is also predicted to become more thermally stable under future warmer conditions.

**1.3.2 Watershed Changes**

There have been three important changes in the watershed during the past two centuries that have affected the water quality of Lake Sammamish. The first change was the deforestation of the watershed beginning in 1880. Second was the diversion of wastewater in 1968. And thirdly, the rapid increase in urban development in the 1990s leading to the development of the Lake Sammamish Management Plan and the Lake Sammamish Initiative.

**1.3.2.1 Forest Conversion**

The watershed was deforested during 1880 to 1930 with the greatest rate of removal occurring during the last 20 years of that period. The majority of the old growth forest was cut by 1930 leaving most of the watershed restocking as second growth forest. Coal mining also started in the 1800s, declining after its peak in 1914. Residential development was also occurring during the same time period, particularly along the lake front (Rock, 1974; Birch, 1976).

Lead $^{210}$ dating of sediment cores showed that the highest sediment accumulation rate in Lake Sammamish (0.67 cm/year) occurred from 1932–1944, the period following the greatest rate of deforestation and with probably the most erosion (Birch, 1976). The
sedimentation rate over the subsequent 30 years (0.32 cm/year) decreased by about one
half, but was still higher (nearly 50 percent) than the pre-deforestation rate (0.21
cm/year). While P loading increased fourfold following deforestation, the diatom
assemblage showed a consistent pattern of mesotrophy from 1880 onward (Birch, 1976;
Rock, 1974). A one day survey in August 1913 indicated that even then the lake was
probably mesotrophic and the hypolimnion likely anoxic (Kemmerer et al., 1923).
However, oxygen depletion rate may have been much less (see Section 3.5.1).

1.3.2.2 Wastewater Diversion

The eutrophication of Lake Washington, caused by direct discharge of treated sewage
effluent, prompted METRO to investigate the condition of Lake Sammamish in the 1960s.
Results of that work indicated that the lake was in an early stage of eutrophication that
could lead to a condition similar to that occurring in Lake Washington (Isaac et al., 1966).
The lake at that time was receiving secondary treated sewage effluent (568 m³/day) from a
trickling filter facility built in Issaquah in 1940. The population at the time of diversion was
about 4,000 people. Wastewater from this facility was discharged directly into Issaquah
Creek just upstream from where it enters the lake. In addition to treated sewage, the
wastewater contained treated waste from the Darigold plant (284 m³/day), and wastes
from the salmon hatchery at Issaquah. METRO diverted the wastewater at a cost of $3 (16
million in 2002 dollars) (Gibbs et al., 1972).

With funding from then Federal Water Pollution Control Act (now EPA), Washington Water
Research Center and National Science Foundation (NSF), the University of Washington,
Department of Civil Engineering began monitoring the lake in late 1969. Complete water
and nutrient budgets were constructed for water year 1971, and the contribution of TP
from Issaquah Creek, which yielded 70 percent of the lake’s water and TP, was determined
through 1975. Results of the TP budget, and analyses of the Issaquah and dairy wastewater,
showed that the best estimate of the TP diverted in 1968 was about 7,000 kg or 35 percent,
a decrease from 1,020 to 670 mg/m² of lake surface per year (Welch, 1977).

Contrary to early expectations, Lake Sammamish did not respond to wastewater diversion
as quickly as Lake Washington. Trophic state indicators (epilimnetic TP, chl a, and
transparency) during 1970–1974 had changed little from pre-diversion values (Welch,
1977; Emery et al., 1973). In contrast to Lake Washington, recovery to equilibrium TP level
was delayed in Lake Sammamish by about seven years. Also, in contrast to Lake
Washington, whole-lake TP increased in the 1980s, reaching 23.7 µg/L in 1987 (See
Section 3.5).

Expectations for wastewater diversion were that Lake Sammamish would show rapid
recovery to near pre-wastewater conditions as had been observed in Lake Washington.
However, there were significant differences between the two lakes that affected such
expectations. Lake Washington was considered eutrophic at the time of diversion. Lake
Sammamish was mesotrophic, as indicated by the sediment diatom index (pinnate:
centrate ratio 1.2) and summer epilimnetic means of TP (~ 20 µg/L), chl a (~ 6 µg/L) and
transparency (3.3 m) during 1970–1974 (Welch, 1977). The pre-diversion values for these variables in Lake Washington were, respectively, 65 μg TP/L, 36 μg chl a/L and 1.0 m Secchi transparency, indicating a near hyper-eutrophic state (Edmondson, 1970). Cyanobacteria dominated the summer phytoplankton in both lakes, but their biomass, comprised mostly of Oscillatoria, was much greater in Lake Washington (Isaac et al., 1966; Edmondson and Litt, 1982).

Another important difference was the inflow concentration of TP. The Cedar River, representing 66 percent of the water input to Lake Washington, had an average inflow concentration of only 17 μg/L (King County DNRP, 2003). In contrast, Issaquah Creek, with 70 percent of water input into Lake Sammamish, had a post-diversion inflow concentration of 65 μg/L (Welch et al., 1980). Therefore, a lower post-diversion lake TP due to external loading would have been expected in Lake Washington, given similar water residence times and sedimentation rates.

The most important difference between the two lakes was DO in the hypolimnion. By mid to late summer in Lake Sammamish, hypolimnetic waters were anoxic. In contrast, DO has never been less than 2 mg/L in the hypolimnion of Lake Washington (King County DNRP, 2003). This process of internal loading in Lake Sammamish accounted for average hypolimnetic TP reaching maxima of 250 μg/L prior to diversion (1964 and 1965; Welch, 1977) and a maxima of 394 μg/L in 1987.

### 1.3.3 Increased and Ongoing Development

The third important watershed change has been the increased and ongoing rate of development since the 1980s, converting second-growth forested land to residential use. The Pacific Northwest is dominated by moderately well drained outwash soils, a humid climate, and low-intensity rainfall (Brett et al., 2005b). Prior to development, these conditions generated runoff predominantly via subsurface flow processes because of the absorbent coniferous forest soil, which provided interception storage and evapotranspiration (Booth, 1991; Schueler, 1994). Land use changes in the watershed alter the quantity, quality, and timing of rainfall runoff. As forests are cleared and replaced by impervious surfaces (paved surfaces and roofs of structures), the rate of water percolation decreases, affecting the amount of water stored in the soils, and the rate of surface runoff increases. These changes increase stream flows and even small rainfall events are capable of washing accumulated pollutants into surface waters (Brett et al., 2005b). Unmitigated flows resulting from development cause additional erosion and instability in the stream channels and carry more sediment and P in particulate form that is sorbed to soil particles (Booth and Henshaw, 2000). Urbanization can also increase storm nutrient transport by supplying anthropogenic P sources from lawn fertilizers, septic drainfields, pet wastes, and construction sites.

From 1900 to 1970, about 15 percent of the total watershed area (not including the lake) had been developed, of which 70 percent was residential properties (Table 1.2,
Figure 1.2). The developed fraction of the watershed more than doubled by 1990, primarily due to the increase of SFR on the east side and Issaquah basins (Figure 1.2). Concern was expressed that P loading from increasing urbanized area runoff, which was shown to carry an average TP concentration of 109 μg/L in three west side storm flows, would counteract the effort expended for, and success of, wastewater diversion (Welch et al., 1980). Development was projected to increase further when completely built-out (King County, 1995; Perkins et al., 1997). (Using runoff coefficients for designated land-use categories TP loads were estimated to increase markedly by full build-out based on 1992 zoning (King County, 1995).)

### Table 1.2 Total developed land in the Sammamish watershed from 1900 to 2011 based on total “As Built” area (acres/hectares) for the specified time period.

<table>
<thead>
<tr>
<th>Time Period</th>
<th>SFR</th>
<th>MFR</th>
<th>Commercial</th>
<th>Total Development Activity</th>
<th>% Developed over time</th>
</tr>
</thead>
<tbody>
<tr>
<td>1900-1920</td>
<td>482 / 195</td>
<td></td>
<td>69 / 28</td>
<td>551 / 223</td>
<td>1</td>
</tr>
<tr>
<td>1921-1940</td>
<td>684 / 277</td>
<td></td>
<td>667 / 270</td>
<td>1,351 / 547</td>
<td>3</td>
</tr>
<tr>
<td>1941-1960</td>
<td>1,865 / 755</td>
<td>46 / 19</td>
<td>1,350 / 546</td>
<td>3,215 / 1,301</td>
<td>9</td>
</tr>
<tr>
<td>1961-1970</td>
<td>2,455 / 994</td>
<td></td>
<td>1,037 / 420</td>
<td>3,538 / 1,432</td>
<td>15</td>
</tr>
<tr>
<td>1971-1980</td>
<td>4,152 / 1,680</td>
<td>69 / 28</td>
<td>2,403 / 972</td>
<td>6,624 / 2,680</td>
<td>26</td>
</tr>
<tr>
<td>1981-1990</td>
<td>4,079 / 1,651</td>
<td>213 / 86</td>
<td>1,259 / 509</td>
<td>5,551 / 2,247</td>
<td>36</td>
</tr>
<tr>
<td>1991-2000</td>
<td>3,150 / 1,275</td>
<td>401 / 162</td>
<td>1,627 / 658</td>
<td>5,178 / 2,095</td>
<td>45</td>
</tr>
<tr>
<td>2000-2011</td>
<td>1,949 / 789</td>
<td>234 / 95</td>
<td>818 / 331</td>
<td>3,001 / 1,214</td>
<td>50</td>
</tr>
<tr>
<td>total acres / hectares</td>
<td>18,816 / 7,615</td>
<td>963 / 390</td>
<td>9,230 / 3,735</td>
<td>29,009 / 11,739</td>
<td></td>
</tr>
<tr>
<td>percent developed</td>
<td>32%</td>
<td>2%</td>
<td>16%</td>
<td>50%</td>
<td></td>
</tr>
</tbody>
</table>

Predictions based on a two-layer (epilimnion and hypolimnion) mass balance TP model in 1994 indicated that mean whole-lake TP concentrations would increase from the post-diversion level of 15 μg/L in the mid-1980s to 24–28 μg/L at build-out depending upon management strategies (King County, 1995). The increase would be due to runoff from the increased developed portion of the watershed that yields P at a rate six times greater than from forested land if unmitigated (King County, 1995; Perkins et al., 1997). The estimated TP load from single family residences was estimated to make up 28 percent of the total load in 1992 and was projected to increase to 57 percent at build-out with proposed controls discussed below. Therefore the 1995 Lake Sammamish Initiative set management goals to protect the lake from projected impacts.

---

1 Single family residential parcels were selected from the countywide parcel dataset using the Property Type attribute and the Residential Buildings table. The selected parcels were symbolized using the Year Built attribute, grouping them into 8 categories. The Multi-family and Commercial parcels were selected using relevant tables. Forest parcels were parcels zoned for Forestry. Note: early years did not have MFR designations.
In response to rapid growth in the region the state adopted the 1990 Growth Management Act requiring state and local governments to manage growth by identifying and protecting critical areas and natural resource lands, and designate urban growth areas. In response to these requirements, King County and cities within its boundaries established Urban Growth Boundaries (UGB) within the western one-third of the County where most growth and development were projected to occur. As growth continued, the County refined the Urban Growth Area (UGA) as part of its 1994 Comprehensive Plan and included special designations for the protection of farming areas and provisions for housing on larger lots in rural area; forest production districts; and open space areas for protection of the natural environment (Figure 1.3).

Land use change was a key factor for predicting annual lake TP concentrations in 1994, though the UGB was not in place when the TP model was developed. Countywide parcel data show that from 1995–1998, most residential building permits issued after the implementation of the 1994 Comprehensive Plan were for parcels inside the UGB, indicating that building density increased within the UGB (Robinson et al., 2005). In contrast to where permits were issued, the total land area newly devoted to housing was greater outside the UGA. This trend continued from 1995 through 2001. Over all countywide total land area newly devoted to residential housing has been greater outside the UGB (14,343 ha) compared with inside (9,084 ha) since 1994.
Figure 1.2 Single-family residential land use year built by decade and current multi-family, commercial, and forest land use (2011).
Figure 1.3  Permitted land use and urban growth boundary in the Sammamish Watershed. (King County. 2014. Urban Growth Boundary. King County Department of Permitting and Environmental Review. Seattle, WA.)
Impervious area was used to evaluate changes in land use between 1991 and 2011. This analysis was conducted using the 1991 and 2011 NOAA Coastal Change Analysis Program (C-CAP) data with the following land classes – “developed open space” and “high, medium and low intensity developed.” Impervious area has increased dramatically in the last 20 years, particularly in the east side and Issaquah Creek basins, by 98 percent and 88 percent, respectively (Table 1.3, Figure 1.4). Though the Issaquah Creek basin remains roughly 70 percent forested, primarily in the upper watershed (Brett et al., 2005a), the impervious area has substantially increased from 535 hectares in 1991 to 1,005 hectares in 2011. Only 2 percent of the east side and 14 percent of the west were not classified as impervious (Table 1.3).

Table 1.3  Impervious area in the east, west, and Issaquah basins in 1991 and 2011. (acre/hectare).

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>East basin</td>
<td>1,001 / 405</td>
<td>1,984 / 803</td>
<td>98%</td>
</tr>
<tr>
<td>West basin</td>
<td>2,072 / 838</td>
<td>2,557 / 1,035</td>
<td>23%</td>
</tr>
<tr>
<td>Issaquah Creek basin</td>
<td>1,322 / 535</td>
<td>2,484 / 1,005</td>
<td>88%</td>
</tr>
<tr>
<td>Total land in watershed</td>
<td>58,103 / 23,514</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Percent Impervious area</td>
<td>8%</td>
<td>12%</td>
<td></td>
</tr>
</tbody>
</table>

A recent evaluation found that between 1991 and 2006 there had been a decline in forest cover in 42 of the 47 Cedar/Sammamish watershed subbasins (Vanderhoof et al., 2011). In particular, the percent decline in the Issaquah Creek North Fork basin was 30.1 percent. The east and west basins of Lake Sammamish both had over 10 percent declines in forest cover from 1991–2006. In aggregate, the majority of the forest cover loss between 1991 and 1996 occurred within the UGB and the rate of loss decreased for the most part in later years. The Issaquah East Fork basin was one exception where the rate of forest decline increased between 2001 and 2006.
Figure 1.4 Impervious area in the west, east and Issaquah basins in 1991 and 2011
1.3.4 Lake Sammamish Initiative and Management Plan

The goals developed for Lake Sammamish as part of the 1995 Lake Sammamish Initiative and Lake Sammamish Management Plan were a mean summer (June–September) chlorophyll $a$ concentration less than or equal to 2.8 $\mu$g/L, and mean Secchi transparency of greater than or equal to 4.0 meters, and an annual mean volume-weighted TP concentration of less than or equal to 22 $\mu$g/L.

The Lake Sammamish Management Plan had two long-term and multiple specific short-term action recommendations to meet the goals, the most costly being 50 percent P removal from new development. Multiple specific short and long-term recommendations identified in the Lake Sammamish Management Plan have been implemented to some extent (Joanna Richey, pers. com., June 26, 2012; Joanna Buehler, pers. com., April 3, 2014). In particular, the two long-term recommendations were:

- 65% forest retention in rural areas.
- 50% retention of TP on future non-rural development. 1998 King County Surface Water Design Manual followed by the 2001 State Stormwater Management Manual for Western Washington. Both of these documents have been updated over the years.

In addition, two other multi-jurisdictional basin planning efforts were completed in 1994 in recognition of potential impacts to surface waters from increasing urbanization in the Sammamish watershed. These basin plans were funded in part by the Washington Department of Ecology Centennial Clean Water Fund – The Issaquah Creek Basin and Nonpoint Action Plan (King County, September 1994), and The East Lake Sammamish Basin and Nonpoint Action Plan (King County, December 1994). Both of these plans had multiple recommended actions to protect residents from flooding, control flows and erosion, prevent environmental degradation of aquatic habitat, and reduce non-point pollutant loads to surface waters. Recommendations included regulatory controls, multiple specific Capital Improvement Projects (CIPs), and education, enforcement, and monitoring.
2.0 METHODS

Monitoring of Lake Sammamish began in the early 1960s when wastewater was being discharged into both Lake Washington and Lake Sammamish. METRO/King County began a routine monitoring program in the three major lakes (Union, Washington, and Sammamish and their tributary streams in the late 1979 to track long-term changes in water quality. The continued long-term monitoring of lake water quality has provided data to evaluate changes in water quality, examine effects of land-use changes and different management actions in the watershed on the lake’s quality, and help agencies maintain awareness of water quality conditions and identify problems in order to inform management decisions.

This study evaluates lake data collected from mid-1960s through 2011. In general, lake measurements are made on a monthly basis December through February, with bimonthly measurements from March through November. Streams are monitored monthly.

2.1 Lake Sampling

Water samples have been collected routinely by King County at seven lake stations (Figure 2.1). The five nearshore stations distributed along the shoreline near the mouths of influent streams were sampled by King County from the mid-1980s through 2008. These stations were eliminated for budgetary reasons. The two deep, or pelagic, stations have been monitored continuously since 1994 (station 0611) and 1979 (station 0612). These deep water stations have maximum sampling depths ranging from 20 to 25 m. Changes in water quality observed over time at the pelagic stations reflect the effects of long-term, landscape changes in the watershed. Data from station 0612 was used in this report to compare with earlier data collected by the University of Washington researchers that sampled the same location beginning in 1970 (Moon, 1973) and by METRO in the 1960s (Isaac et al., 1966).

Grab (instantaneous) samples for total alkalinity, and nutrients, were collected at various depths in the water column using Van Dorn bottles at the shallow stations and Niskin bottles at the deeper, open water station. Some parameters were determined in the field (pH, temperature, DO, and conductivity) using a calibrated Hydrolab unit equipped with electronic sensors that was lowered to selected depths. Transparency was measured at each station using a 28-cm-diameter black-and-white Secchi disk equipped with a graduated line.

Sampling methods for chl a samples varied over the time period discussed in this report. Prior to 1994, chl a samples were collected at 1 m depth. Discrete compositing was implemented in March 1994 and consisted of compositing a sample collected from 1 m below the surface with a sample collected at Secchi depth. Beginning in 2004, sample compositing changed to a vertically integrated sample collected using a weighted length of 1.6-cm Tygon® tubing let down vertically through the water column. The tube is plugged at the surface and at the submerged end by a check valve and retrieved. Although observed differences between the two methods were often significant (King County, 2004), it was determined that these small differences would not significantly affect the ability to detect
long-term trends in seasonally averaged chl a concentrations. Phytoplankton samples were collected using the same methods as chl a and were preserved with Lugol’s solution.

To simplify the characterization of the lake through the water column over time, all samples collected from depths above 10 meters at station 0612 were designated epilimnetic samples and samples below 15 meters were designated as hypolimnetic samples. See Appendix A for tables summarizing volume-weighting methodology. Annual means and standard deviations for all lake water quality data are in Appendix B. Some data from earlier work done by the University of Washington was used in historical comparisons as discussed below.
Figure 2.1  Depth contours and water quality sampling stations on Lake Sammamish
2.2 Stream Sampling

Monthly grab samples were collected by King County at the mouth of the seven tributaries to Lake Sammamish for nutrients and total suspended solids (Table 2.3, Figure 2.2). Conductivity, temperature, and DO were measured using a calibrated Hydrolab unit. Continuous gages measured flow for several of the creeks (Table 2.1).

Period of record varies for each creek with Issaquah Creek having the longest consistent record beginning in 1980. Budget reductions in 2009 eliminated five of the routine water quality monitoring stations. Some monitoring of Issaquah Creek had occurred in the 1970s but data collection was not consistent until 1979. Issaquah Creek is the major tributary to the lake and there are several studies conducted through the University of Washington Department of Civil and Environmental Engineering that are referenced throughout this report.

Initially, METRO/King County’s long-term ambient monitoring program was designed to collect one random sample each month. Beginning in 1993, additional sampling of wet weather events was initiated. However, the wet weather events were sometimes substituted for the routine random sample; i.e., the single routine monthly sample may have been designated a non-random wet weather event sample. The storm sample was used in this analysis of long-term data for months when no regular routine monthly sample was collected—assuming that the designated routine sampling event was reclassified as a wet-weather event.

<table>
<thead>
<tr>
<th>Table 2.1</th>
<th>Period of record for tributaries entering into Lake Sammamish.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tributary</td>
<td>Water Chemistry Period of record - water year (Oct–Sept)</td>
</tr>
<tr>
<td>West side Tributaries</td>
<td>Idylwood Creek (A620)</td>
</tr>
<tr>
<td>East side Tributaries</td>
<td>Eden Creek (A690)</td>
</tr>
<tr>
<td></td>
<td>Ebright Creek (A685)</td>
</tr>
<tr>
<td></td>
<td>Pine Lake Creek (A680)</td>
</tr>
<tr>
<td>South Tributaries</td>
<td>Issaquah Creek Mouth (0631)</td>
</tr>
<tr>
<td></td>
<td>Lewis Creek (A617)</td>
</tr>
</tbody>
</table>
Figure 2.2 Tributary Sampling Locations
2.3 Laboratory Analysis

Grab samples from the water column were analyzed at the King County Environmental Laboratory. Laboratory methods and detection limits for conventional parameters are provided in Table 2.2. Additional information about the King County Environmental Lab can be obtained at the laboratory’s website: http://www.kingcounty.gov/environment/wlr/sections-programs/environmental-lab.aspx

Quality assurance/quality control procedures included the use of blanks, duplicates, and spikes where appropriate. All data were reviewed by King County Environmental Lab staff before entry into the Laboratory Information Management System (LIMS) database.

Table 2.2 Laboratory methods and detection limits for conventional water samples.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Standard Methods1</th>
<th>MDL* (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alkalinity</td>
<td>SM 2320-B</td>
<td>0.2</td>
</tr>
<tr>
<td>Chlorophyll a</td>
<td>SM 10200-H</td>
<td>0.0005</td>
</tr>
<tr>
<td>Ammonia-Nitrogen</td>
<td>SM 4500-NH3-H</td>
<td>0.02</td>
</tr>
<tr>
<td>Total Nitrogen</td>
<td>SM4500-N-D +</td>
<td>0.05</td>
</tr>
<tr>
<td></td>
<td>SM4500-NO3-F</td>
<td></td>
</tr>
<tr>
<td>Nitrate/Nitrite</td>
<td>SM4500-NO3-F</td>
<td>0.05</td>
</tr>
<tr>
<td>Soluble Reactive Phosphorus</td>
<td>SM 4500-P-F</td>
<td>0.002</td>
</tr>
<tr>
<td>Total Phosphorus</td>
<td>SM 4500-P,B,E</td>
<td>0.005</td>
</tr>
<tr>
<td>Total Suspended Solids</td>
<td>SM2540-D (03-01-009-001)</td>
<td>0.5</td>
</tr>
</tbody>
</table>

* Method Detection Limit
1 – Codes listed are from Standard Methods for the Examination of Water and Wastewater.

2.4 Statistical Analysis

A non-parametric Seasonal Kendall test was used to determine if an observed trend was unlikely to be due to random variation (Helsel and Frans, 2006). A standard $\rho$-value of < 0.05 was used to determine if the null hypothesis should be rejected – meaning the trend was weakly significant if the $\rho$-value was < 0.05 and not due to random variation. The $\rho$-values reported throughout the report are as follows:

- $\rho < 0.01$ - Very strong statistical significance
- $0.01 < \rho < 0.05$ - Strong statistical significance
- $0.05 < \rho < 0.10$ - Weak statistical significance
- $\rho > 0.10$ - No statistical significance
2.5 Water Quality Parameters

General lake water quality indicators temperature, dissolved oxygen (DO), conductivity, alkalinity, total phosphorus (P), soluble reactive phosphorus (SRP), total nitrogen (N), nitrate-nitrite nitrogen (NO23), ammonia (NH4), were measured at multiple depths. In addition, Secchi disc transparency (SD) was measured. Chlorophyll a (chl a) was measured as a composite sample. Some parameters have been added over time and do not have continuous measurements spanning from the 1980s. Beginning in 2003 quantitative phytoplankton enumeration was determined from samples collected at the pelagic station. Below is a detailed discussion of these water quality constituents.

All stream and lake data reported is time-weighted by calculating the monthly mean when more than one sample is collected in a given month. Many of the lake constituents were volume-weighted using a total lake volume of 327,000,000 m³ based on the hypsographic curve from Birch, (1976). A depth starting at 15 meters was used for the hypolimnion, which represented a volume of 98 x 10⁶ m³ and a mean hypolimnetic depth of 8.17 m. Using a shallower depth to calculate hypolimnetic volume would encompass some of the metalimnion. Hypolimnetic and epilimnetic concentrations are usually weighted from the thermocline, which declines gradually during the summer. However, depth and volume of the hypolimnion used here were kept constant to determine volume-weighted concentrations and subsequent sediment release and oxygen depletion rates. Volume-weighted concentrations are indicated as “whole lake,” “epilimnetic,” “metalimnetic,” and/or “hypolimnetic” means throughout the report.


2.5.1 Temperature

Water temperature is an important water quality variable because it (1) directly affects biological and chemical activity, (2) affects water density, which determines water column stability (i.e., thermal stratification), and (3) defines available habitat for a variety of aquatic species.

The seasonal pattern of temperature throughout the water column is determined largely by climatic factors. During winter, temperature throughout the water column is relatively constant, as in other temperate, monomictic lakes, because the lake is at a uniform temperature and density and therefore well mixed. During the summer the water column stratifies into a warm, less dense surface layer (or epilimnion), an intermediate metalimnion, and a colder, denser hypolimnion. This stratified condition develops because increased solar radiation in the spring heats the surface water which becomes less dense. The depth of mixing defines the bottom of the epilimnion and occurs where wind energy exerted to mix the water column equals the energy of resistance due to the higher density. Because the epilimnion and hypolimnion do not mix with each other during the summer-
stratified period, chemical and biological characteristics in the two layers usually diverge. In the fall, as the surface water cools and becomes denser, mixing by wind becomes easier, and thermal stratification moves downward until complete mixing returns.

Sammamish temperature profiles collected in the 1970s (from the University of Washington Department of Civil and Environmental Engineering) and 1980s through 1993 had too many anomalies and inconsistent sampling frequencies to directly compare stratification changes over time. Therefore for long-term temperature comparisons, surface temperature at one meter was used.

2.5.2 Dissolved Oxygen

DO is an important constituent that directly affects, and is affected by, abundance and diversity of aquatic organisms. Vertebrate and invertebrate taxa have specific tolerances for low DO for metabolic needs. Water quality criteria for DO are often established to protect the reproduction and growth of sensitive species. The amount of oxygen that can be held in water varies with water temperature (saturation). Water bodies with DO near saturation levels (e.g., 9 mg/L at 20°C) at all depths are capable of sustaining a diverse assemblage of aquatic organisms while fewer organisms are capable of tolerating low DO environments.

In lakes, DO is produced as a byproduct of photosynthesis and consumed through respiration by organisms as well as diffusion from the atmosphere into the water, generally maintaining high concentrations in the epilimnion. However, consumption can easily exceed supply in the hypolimnion, where photosynthesis and atmospheric diffusion are absent, while settled organic matter in the water and sediment create a DO demand. During summer stratification, DO concentrations may change dramatically with depth below the thermocline to the point of total depletion near the bottom sediments in the hypolimnion. As oxygen levels in the hypolimnion decline as the result of sediment oxygen demand and the decomposition of detritus settling from the epilimnion, species of fish and invertebrates not tolerant of low DO are replaced by less tolerant species.

Some regional lakes may lose dissolved oxygen entirely in deep water during extended periods of thermal stability, other do not. The magnitude of the loss of DO in the hypolimnion is correlated with shallow water algal production (Edmondson, 1966; Lehman, 1988). Thus, the level of DO and the rate of its loss are used as a type of trophic state index – the aerial hypolimnetic oxygen depletion rate, or AHOD, as DO loss per unit area. The AHOD rate was calculated by determining the slope of a regression line that defines the relation between volume-weighted hypolimnetic DO and time from May until just before DO drops below 1 mg/L. The determined slope, or g DO/m³ per day, was multiplied by the hypolimnetic mean depth (below 15 m) of 8.17 m to yield the areal rate in g/m² per day.
2.5.3 **Phosphorus**

Phosphorus is an essential naturally occurring element for the metabolic processes and growth of both plants and animals. It occurs naturally in soil and rock and can be found in plant and animal tissue as well as on particles in the atmosphere. Total phosphorus (TP) represents both organic and inorganic P in particulate and dissolved forms. Soluble reactive phosphorus (SRP) generally represents that portion of P (largely phosphate) that is dissolved in water and is readily available for biological uptake.

Phosphorus is important to algal growth and has historically been the nutrient most closely linked to the long-term change in algal production in Lake Washington (see Section 3). Lake Sammamish is also P limited (see Section 3), so increased P may lead to increased algal blooms. Human activities within the watershed, through land-use change that increases the rate and magnitude of stormwater runoff and direct discharge of treated sewage effluent, increases the input of P and is usually the most important cause of eutrophication and serious water quality degradation at moderate latitudes in the northern hemisphere (Carpenter, 2008). Anthropogenic sources of phosphorus include residential and agricultural application of fertilizer, septic systems, leaky wastewater conveyance systems, animal waste, and accelerated erosion of sediment bound phosphorus due to alterations in the physical and hydrologic features of the watershed.

Phosphorus data cited in King County, 1995, was used for data prior to 1995 and included historical TP data from 1964–1966 (METRO), from 1970–1975 (University of Washington Department of Environmental and Civil Engineering), from 1979–1981 (METRO/King County). From 1964–1966, METRO analyzed for total soluble phosphorus (TSP) and not TP. Comparative analyses by METRO in the 1980s indicated that the ratio of TP/TSP was variable, but averaged about 1.2. Thus, pre-diversion TSP data were converted to TP using that ratio. Why the converted pre-diversion epilimnetic TP values were not higher, consistent with higher whole-lake TP may be due to the conversion ratio being too low. Since most of biologically available P in spring-summer is in algal cells (Butkus, 1987; Butkus et al., 1988), that particulate P would not have been measurable as part of the TSP procedure (the unheated acid digestion process does not liberate cellular P) and high algal concentrations may have been missed in samples used to determine the conversion ratio. Determining TP (hot acid digestion) accounts for available P as well as the P within algal cells. TP data from 1981–1994 cited in King County (1995) was used but volume weighting was modified as described below. Any additional changes in data processing from that reported by King County (1995) are noted in Appendix A.

Phosphorus concentrations were volume-weighted using a total lake volume of 327,000,000 m$^3$ based on the hypsographic curve in Birch (1976). A depth starting at 15 meters was used for the hypolimnion which represented a volume of $98 \times 10^6$ m$^3$ and a mean hypolimnetic depth of 8.17 m. King County (1995) used a depth of 11 meters from the surface to define the hypolimnion which includes much of the metalimnion and results in a nominal hypolimnetic mean depth of 11 m and a volume of about $152 \times 10^6$ m$^3$. This difference in hypolimnetic depth was shown to have a rather small effect on volume-weighted hypolimnetic TP from what King County (1995) reported. Means of the
differences between using hypolimnetic mean depths of 11 m (King County, 1995) and 8.17 m were only $1.5 \pm 1.9 \, \mu g/L$ for the 1981-1994 data. That small effect is apparently due to most of the seasonal change in concentration occurring below 15 m, the nominal top of the hypolimnion.

Note: The King County Environmental Lab changed its TP analysis method in July 1998 and again in January 2007. These changes resulted in a bias toward lower reported TP values. For comparison with historical data, in particular values reported in King County (1995), values reported after July 1, 1998 were increased by a factor of 1.262 ($R^2 = 0.989$).

### 2.5.4 Sediment Phosphorus Recycling and DO Demand

Sediment P release was calculated based on the rate of buildup of TP in the hypolimnion. This method was found to produce results similar to results from Lake Sammamish sediment cores brought into the laboratory and incubated under anoxic conditions at hypolimnetic temperatures (Nürnberg, 1987). Rates were calculated from regression lines relating volume-weighted hypolimnetic TP versus time – in general August to turnover – for data from 1964–2011.

Most, but not all, of sediment P release is due to anoxia at the sediment-water interface. Under anoxia, iron is reduced and P that was sorbed to the ferric hydroxy complex is solubilized into the sediment pore water and may diffuse into the overlying water. The deepest determination of TP and DO was 25 m, which is not the bottom at station 0612, so the onset of anoxia occurring at the sediment-water interface was not measured. TP fluctuates at 25 m until late July or August and then starts to increase more consistently. Therefore, the beginning of TP increase (typically the first sampling date in August) was chosen to determine rates. The same procedure was used by King County (1995). As phosphorus is released from the sediment during stratification, the concentration builds in the hypolimnion. When the lake turns over in late autumn – early winter, the phosphorus in the hypolimnion is distributed throughout the water column where it available for utilization by phytoplankton.

Areal hypolimnetic oxygen deficit rate (AHOD) was determined by regressing volume-weighted DO in the hypolimnion below 15 m from the first sampling date in May to the last sampling date at which DO at 25 m reached $\leq 1 \, mg/L$. King County (1995) used a constant hypolimnetic mean depth of 11 m in volume-weighting DO concentrations and determined the oxygen deficit by difference rather than by regression. There turned out to be little difference in AHODs using the two mean depths. Similar to TP, most of the seasonal DO concentration change occurred below 15 m, the point below which DO addition through photosynthesis and atmospheric aeration was minimal or nonexistent. The mean of the differences in AHOD between using 11 m mean depth (King County, 1995) and 8.17 m was only -0.041 g/m² per day, or 7 percent less for the 1981–1994 data.

There is considerable variation in calculated AHODs because:

1. The seasonal decrease in DO is not always constant, so there is variation about best-fit regression lines.
2. Thermal stratification depth, intensity and duration are often variable from year-to-year.

3. Stratification can be disrupted by changing climatic conditions, resulting in altered hypolimnetic depths and DO additions to the hypolimnion.

These factors result in normal year-to-year variations of ± 25 percent. Thus, trends would need to be substantial to have statistical significance (see Section 3.5).

2.5.5 Nitrogen

Nitrogen exists in several forms in natural water, including nitrite-nitrogen, nitrate-nitrogen, ammonium-nitrogen, organic nitrogen, and elemental nitrogen. Aquatic organisms commonly use the dissolved forms of nitrogen; ammonium-nitrogen and nitrate-nitrogen. Total nitrogen (TN), nitrate plus nitrite, and ammonium-nitrogen are the forms historically sampled in Lake Sammamish. King County did not begin to measure total nitrogen until 1993. Nitrate and nitrite nitrogen are often reported as one value due to the method of analysis, as nitrate+nitrite nitrogen, which is the case in this report. Nitrite nitrogen in ambient water is typically negligible and so reported nitrate+nitrite values are almost always predominantly nitrate. An increase in only nitrogen input might have little effect on productivity because Lake Sammamish is generally P-limited (Section 3.5). However, long-term changes in nitrogen to phosphorus ratios may cause changes in phytoplankton community composition (Downing et al. 2001) that could have negative impacts. Also, long-term change in nitrogen concentrations may indicate changes in watershed activity. Increased inputs of N to Lake Sammamish could also affect water quality downstream in Puget Sound because marine waters tend to be N-limited, even if changes in Lake Sammamish trophic structure are not detectable.

A study of the regional long-term stream data found that seasonally high nitrate concentrations during the winter were not the result of anthropogenic impacts on the watershed but rather the presence of alders, which are an important source of inorganic N due to symbiotic N$_2$ fixation (Brett et al., 2005a).

2.5.6 Conductivity

Specific conductance (conductivity) is a measure of the capacity of water to conduct an electric current standardized at 25 °C, allowing comparisons among waters of different temperatures. Temperature and the concentration of major dissolved ions in water determine the conductivity. Waters in the Puget Sound region generally have low levels of dissolved minerals and relatively low conductivity due to slow weathering igneous bedrock. Conductivity is generally higher during base flows in King County streams and decreases during storm events due to dilution from rainwater. Changes in water flow pathways with less infiltration to groundwater due to direct stormwater conveyance to streams may result in longer groundwater residence times with greater leaching of major ions so that groundwater discharge to streams during baseflow conditions has higher conductance. Thus, annual average stream conductance is increased overall. In addition,
increased conductivity in urban streams has been attributed to several sources including leaky sewer and septic systems and concrete weathering (EPA, 2012).

Active land use and land-use conversion from open space to developed areas tend to increase conductivity, and such increases may indicate the presence other pollutants. For example, chloride from drain field leachate, nitrite+nitrate-N and soluble P from fertilizer, and/or major dissolved ions in stormwater from disturbance land.

2.5.7 Alkalinity

Total alkalinity, also called acid neutralizing capacity, of water is based on the concentrations of bicarbonates, carbonates, and hydroxides, and is expressed in mg CaCO$_3$/L (Wetzel, 1983). It relates to the ability of water to resist pH change, also known as buffering capacity.

Total alkalinitis of surface waters in western Washington are generally low due to the lack of sedimentary bedrock containing carbonates (Carroll and Pelletier, 1991) and therefore have low resistance to pH changes. In local water bodies, pH often increases to high levels (>10) during large algal blooms when photosynthetic removal of CO$_2$ by algae is faster than its replenishment from the atmosphere.

2.5.8 pH

Hydrogen ion activity in water is measured as the negative log of the hydrogen ion (H$^+$) concentration (pH), which indicates the degree of acidity of a lake. The pH is inversely related to hydrogen ion activity, so waters with pH above neutral 7.0 are termed “alkaline” and those below 7.0 are “acidic.” As discussed above, photosynthesis removes carbon dioxide (in the form of carbonic acid and bicarbonate) from water, reducing the concentration of hydrogen ions, increasing pH. For this reason, pH is often higher at the surface during daylight hours in the summer, especially in low-buffered waters. Dense, rooted aquatic macrophyte communities can also increase pH during intense photosynthesis. Frodge et al. (1991) observed pHs greater than 10 in dense beds of milfoil in Lake Washington. The addition of CO$_2$ by diffusion from the atmosphere and microbial respiration, lower pH. Organic matter that settles onto the lake bottom and is decomposed contributes to lower pH with depth. Water near the bottom and in surficial sediments usually has a pH around 6 due to bacterial decomposition of settled organic matter. However, most surface waters have a pH between 7.0 and 8.5, which is slightly alkaline.

High-elevation lakes in the Cascade Mountains and open water wetlands with bog-like characteristics in Puget Sound lowlands often have a pH below 7.0 due to very low alkalinity and water inflows directly from rainfall, snow, and from basins with little soil build-up. Thus they have poor buffering capacity, and are therefore highly sensitive to acid precipitation.
2.5.9 Transparency

Water transparency, or clarity, is measured with a standard black-and-white metal Secchi disk that is 28 cm in diameter. The depth at which the disk disappears from sight is determined by attenuation of light penetrating the water column and reflected back from the white portion of the disk to the observer. Light attenuation through the water column is influenced by several factors, including small sized particles made up of living planktonic algae, suspended sediment and organic detritus, and water color generally derived from incompletely decomposed organic molecules. Therefore, the depth that the disk is visible decreases as the concentration of small particles and dissolved colored substances increases.

Transparency of Lake Sammamish is dependent largely on the concentration of algal particles during periods of dry weather when inflows are not bringing in large amounts of erosive materials into the lake. This is especially true in summer, which is usually the season used to indicate the state of lake quality and trophic state. Chlorophyll a, as an index of algal biomass, is inversely and non-linearly related to Secchi transparency (Carlson, 1977). The impact to water clarity from phytoplankton in Lake Sammamish is influenced by dominance shifts from unicellular to multicellular phytoplankton.

2.5.10 Phytoplankton and Chlorophyll a

Chlorophyll a (chl a) is a photosynthetic pigment present in all algae and cyanobacteria. Chl a is used by these organisms in the process of photosynthesis, which converts light energy, carbon dioxide, and water to chemical energy that can be stored for future use. The ratio of algal biomass, or carbon, to chl a varies with species, nutrient availability, light and other environmental conditions. Thus, chl a is not an exact measurement of algal biomass. Nevertheless, it is used universally as an indicator of algal biomass and lake trophic state and this ratio is conventionally assumed to be about 50:1.

King County Environmental Lab changed analytical methods for chl a in July 1996. This method change resulted in a 14 percent increase in chlorophyll values due to improved extraction. Values reported prior to July 1996 were adjusted by increasing them by a factor of 0.14 for comparison with more recent data.

2.5.11 Total Suspended Solids

Total suspended solids (TSS) was measured in the streams and is used to estimate the amount of suspended material in the water, whether it is mineral (e.g., soil particles) or organic (e.g., algae). Particulate matter provides attachment places for pollutants such as nutrients to enter the receiving water. High concentrations of particulate matter can result in increased sedimentation that can impair important habitat for fish and invertebrates. In general, it is human activities within the watershed that usually results in higher TSS measurements (e.g., development results in loss of vegetation, increased erosion, and runoff).
TSS is a measure of the actual weight of material per volume of water and is reported in milligrams per liter. This measurement becomes important when trying to calculate total quantities of material in a stream, or when trying to determine the loading of particulate matter into receiving waters. There is no state water quality standard for TSS.
3.0 RESULTS

This section describes the current status and long-term changes in water quality conditions in Lake Sammamish. The relationship between changes in water quality and changes in the watershed including wastewater diversion, and ongoing development following the implementation of the Lake Sammamish Management Plan in 1994 is examined. Specifically, the data were analyzed with the following objectives:

- Characterize the water quality status of the lake relative to accepted indicators, such as transparency, DO, TP, and chl a.
- Identify any trends in water quality, with reference to historical conditions where applicable.
- Provide information to be used in making future environmental management decisions that may impact the lake.

3.1 Water Inflows

Water enters Lake Sammamish through direct precipitation to the lake surface, flows from tributaries, groundwater, and lateral flow along the eastern shoreline (King County 2008). The quantity of these flows is affected by localized land use (Section 1.3.3) as well as regional climate variability.

Climate variability in this region is strongly influenced by variation in Pacific Ocean circulation. Two measures of this variability that differ in the time-scales of their influence are the El Nino Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO). ENSO varies from warm to cool phases on the scale of years, while PDO varies on a decadal scale. The PDO shifted from a cool/wet phase to a warm/dry phase in 1976–1977.

3.1.1 Precipitation

Timing of precipitation can affect lake water quality. Heavy rains can result in high nutrient loading to the lake. If these inputs occur during the growing season, then algae can utilize the nutrients and algal production can increase. If the loading takes place during the fall and winter months, it is less likely to have an immediate effect on algal abundance, but a portion (less the fraction buried in bottom sediments) will remain in the lake until the next growing season. The seasonal distributional pattern of precipitation in the Pacific Northwest is one of wet winters and springs, and relatively dry summer. Figure 3.1 shows monthly accumulated rainfall for each year and the variation from year to year from the 52-year average. For example, 1996 and 1997 were exceptionally wet years. 2001 was relatively dry with cumulative precipitation well below the 50-year average until the last few months of the year, at which time the rainfall accumulations caught up with the historical average.
Accumulated precipitation generally increases from October through April, but the curves are relatively flat with little additional precipitation between May and September (Figure 3.1). This means that external loading of nutrients is usually low in summer and high from late fall to early spring. Therefore, any new supply of nutrients to the epilimnion during summer must come from internal recycling from sediments and the residual low stream inflows.

Epilimnetic nutrient concentrations decline during summer as they are assimilated by algae, which gradually settle through the water column, and the depleted nutrients are not fully replaced. On the other hand, any late spring to early fall rain events (during stratification) that cause runoff from urbanized areas will probably enter the epilimnion, because runoff from impervious surfaces is usually warmer than the lake surface water, and thus, contribute new nutrients to the epilimnion.

Annual precipitation averages 39 inches per year at Sea-Tac International Airport and was highly variable in the last 50 years (Figure 3.2). During the last two decades annual total precipitation was above the 50-year average 45 percent of the time and ranged from a high of 51 inches (130 cm) in 1996 to 28 inches (71 cm) in 1993, and 2000 (Figure 3.2). The six years from 2000–2005 averaged 6 and 7 inches (15–18 cm) less than the periods 2006–2011 and 1994–1999, respectively. Although variable, there was no consistent trend in precipitation.
Precipitation in the Lake Sammamish basin is substantially higher than at Sea-Tac International Airport. Previous analyses estimated a 15 percent higher annual rainfall total onto the surface of Lake Sammamish relative to Sea-Tac International Airport (King County 2008). Analysis of rainfall totals at King County’s Tibbetts Creek rain gage (available online at http://green.kingcounty.gov/WLR/Waterres/hydrology/default.aspx) show an annual average of 55 inches per year between 1988 and 2013. This is about 16 inches per year higher than the 39 inch annual average rainfall at Sea-Tac International Airport over the past 50 years. At higher elevations, even more rainfall occurs, with an annual average of 60 inches per year from 1995 to 2013 at King County’s Hobart rain gage.

### 3.1.2 Issaquah Creek Flows

Average annual flows in Issaquah Creek have dropped by roughly 20 percent over the past nearly 50 years (Figure 3.3) (King County, 2010). The lower flows probably reflect the lower rainfall that is related to the shift in the PDO from a cool/wet phase to a warm/dry phase in 1976–1977. The horizontal lines in Figure 3.3 represent mean flows for the two periods 151.8 cfs (4.3 m³/s) during 1964–1976, and 123.6 (3.5 m³/s) during 1977–2012. The highs and lows have been about 30 cfs less in the past three decades with an average of 123.6 cfs (3.5 m³/s) (about 30 cfs less). The winter pattern was similar with 1976–1977, 1987–1988, 1993–1994, and 2000–2001 being exceptionally low flows.
Figure 3.3  Annual average Issaquah Creek flow (water year) and wet season (November – March) from 1964 through 2012.

### 3.1.3 Tributary Water Quality

Time series plots for nutrients, TSS and conductivity were generated for each of the six tributaries based on the data available (Appendix C). Seasonal Mann Kendall tests were used to determine if trends were significant for any of the constituents, which are summarized in Table 3.1 below.

Periods of recorded data for tributaries have varied with some only a short duration. The longest records are for Pine Lake and Issaquah creeks. Issaquah Creek SRP, nitrate+nitrite-N and TN significantly decreased, and conductivity weakly increased from 1972 to 2011. Conductivity increased slightly and SRP and TSS decreased significantly, while TN decreased slightly in Pine Lake Creek from 1987 to 2011. Significant increases in TN and
conductivity, and decreased TSS were observed during 1996 to 2008 in Ebright Creek. Nitrate+nitrite-N, TN and conductivity significantly increased in Eden Creek, with a slight increase in SRP from 1987 to 2008. Nitrate+nitrite-N, SRP and TN significantly decreased in Idylwood Creek, while conductivity increased significantly from 1995 to 2008. During the same period, nitrate+nitrite-N, TN and TSS decreased significantly in Lewis Creek, while conductivity weakly increased. Finally, TP increased significantly while nitrate+nitrite-N weakly decreased in Tibbetts Creek.

In general, SRP concentrations have decreased in the Sammamish tributaries while there has been no significant change in TP (Section 3.5.1). Brett et al. (2005b) determined TP daily and SRP weekly in Seattle urban streams and found that SRP follows a smooth seasonal cycle with the higher concentrations occurring during low flow periods when ground water dominated stream flows. However, TP concentrations were highly variable and influenced by short-term flow fluctuations. Issaquah Creek was the exception showing very little annual variation in SRP; both TP and SRP were higher in the fall than would be expected, perhaps due to salmon spawning.
Table 3.1  Summary water quality parameter trend analysis, means and number of samples (n) for Lake Sammamish tributaries.

<table>
<thead>
<tr>
<th>Site</th>
<th>Parameter</th>
<th>routine monthly samples*</th>
<th>Tau</th>
<th>p</th>
<th>Mean</th>
<th>n</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pine Lake Creek</td>
<td>SRP (μg L⁻¹)</td>
<td>-0.34</td>
<td>&lt;0.01</td>
<td>44.4</td>
<td>222</td>
<td></td>
</tr>
<tr>
<td>A680</td>
<td>TP (μg L⁻¹)</td>
<td>-0.10</td>
<td>0.19</td>
<td>88.6</td>
<td>222</td>
<td></td>
</tr>
<tr>
<td>1987 - 2011</td>
<td>Nitrite + Nitrate Nitrogen (μg L⁻¹)</td>
<td>-0.17</td>
<td>0.12</td>
<td>376.3</td>
<td>222</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TN (μg L⁻¹)</td>
<td>-0.13</td>
<td>0.10</td>
<td>754.3</td>
<td>222</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TP:TN</td>
<td>-0.31</td>
<td>0.01</td>
<td>0.5</td>
<td>167</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TSS (mg L⁻¹)</td>
<td>-0.22</td>
<td>&lt;0.01</td>
<td>7.0</td>
<td>222</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Conductivity (umhos cm⁻¹)</td>
<td>0.15</td>
<td>0.08</td>
<td>135.1</td>
<td>220</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ebright Creek</td>
<td>SRP (μg L⁻¹)</td>
<td>-0.16</td>
<td>0.19</td>
<td>32.4</td>
<td>151</td>
<td></td>
</tr>
<tr>
<td>A685</td>
<td>TP (μg L⁻¹)</td>
<td>0.03</td>
<td>0.75</td>
<td>58.9</td>
<td>151</td>
<td></td>
</tr>
<tr>
<td>1996 - 2008</td>
<td>Nitrite + Nitrate Nitrogen (μg L⁻¹)</td>
<td>0.07</td>
<td>0.41</td>
<td>1313.9</td>
<td>151</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TN (μg L⁻¹)</td>
<td>0.20</td>
<td>0.03</td>
<td>1505.9</td>
<td>151</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TP:TN</td>
<td>-0.17</td>
<td>0.18</td>
<td>0.6</td>
<td>152</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TSS (mg L⁻¹)</td>
<td>-0.16</td>
<td>&lt;0.01</td>
<td>7.1</td>
<td>151</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Conductivity (umhos cm⁻¹)</td>
<td>0.49</td>
<td>&lt;0.01</td>
<td>137.6</td>
<td>151</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Eden Creek</td>
<td>SRP (μg L⁻¹)</td>
<td>0.22</td>
<td>0.06</td>
<td>20.7</td>
<td>161</td>
<td></td>
</tr>
<tr>
<td>A690</td>
<td>TP (μg L⁻¹)</td>
<td>0.14</td>
<td>0.20</td>
<td>43.0</td>
<td>162</td>
<td></td>
</tr>
<tr>
<td>1987 - 2008</td>
<td>Nitrite + Nitrate Nitrogen (μg L⁻¹)</td>
<td>0.24</td>
<td>0.02</td>
<td>1961.7</td>
<td>161</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TN (μg L⁻¹)</td>
<td>0.35</td>
<td>&lt;0.01</td>
<td>2069.5</td>
<td>162</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TP:TN</td>
<td>0.04</td>
<td>0.69</td>
<td>0.54</td>
<td>164</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TSS (mg L⁻¹)</td>
<td>-0.08</td>
<td>0.30</td>
<td>11.2</td>
<td>162</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Conductivity (umhos cm⁻¹)</td>
<td>0.52</td>
<td>&lt;0.01</td>
<td>119.5</td>
<td>162</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Issaquah Creek</td>
<td>SRP (μg L⁻¹)</td>
<td>-0.35</td>
<td>&lt;0.01</td>
<td>15.2</td>
<td>394</td>
<td></td>
</tr>
<tr>
<td>0631</td>
<td>TP (μg L⁻¹)</td>
<td>-0.04</td>
<td>0.45</td>
<td>41.8</td>
<td>398</td>
<td></td>
</tr>
<tr>
<td>1972 - 2011</td>
<td>Nitrite + Nitrate Nitrogen (μg L⁻¹)</td>
<td>-0.34</td>
<td>&lt;0.01</td>
<td>923.3</td>
<td>395</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TN (μg L⁻¹)</td>
<td>-0.47</td>
<td>&lt;0.01</td>
<td>1075.3</td>
<td>242</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TP:TN</td>
<td>0.23</td>
<td>0.48 /0.34 /0.39</td>
<td>168 /403</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>TSS (mg L⁻¹)</td>
<td>&lt;0.01</td>
<td>0.93</td>
<td>12.8</td>
<td>397</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Conductivity (umhos cm⁻¹)</td>
<td>0.10</td>
<td>0.07</td>
<td>111.1</td>
<td>396</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lewis Creek</td>
<td>SRP (μg L⁻¹)</td>
<td>0.03</td>
<td>0.76</td>
<td>38.4</td>
<td>167</td>
<td></td>
</tr>
<tr>
<td>A617</td>
<td>TP (μg L⁻¹)</td>
<td>0.04</td>
<td>0.63</td>
<td>168.0</td>
<td>168</td>
<td></td>
</tr>
<tr>
<td>1995 - 2008</td>
<td>Nitrite + Nitrate Nitrogen (μg L⁻¹)</td>
<td>-0.33</td>
<td>0.01</td>
<td>877.2</td>
<td>167</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TN (μg L⁻¹)</td>
<td>-0.25</td>
<td>0.01</td>
<td>1095.0</td>
<td>167</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TP:TN</td>
<td>-0.07</td>
<td>0.26</td>
<td>0.54</td>
<td>168</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TSS (mg L⁻¹)</td>
<td>-0.14</td>
<td>0.02</td>
<td>19.8</td>
<td>167</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Conductivity (umhos cm⁻¹)</td>
<td>0.17</td>
<td>0.05</td>
<td>156.0</td>
<td>168</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Idylwood Creek</td>
<td>SRP (μg L⁻¹)</td>
<td>-0.25</td>
<td>0.02</td>
<td>22.2</td>
<td>159</td>
<td></td>
</tr>
<tr>
<td>A620</td>
<td>TP (μg L⁻¹)</td>
<td>0.10</td>
<td>0.25</td>
<td>57.3</td>
<td>159</td>
<td></td>
</tr>
<tr>
<td>1995 - 2008</td>
<td>Nitrite + Nitrate Nitrogen (μg L⁻¹)</td>
<td>-0.36</td>
<td>&lt;0.01</td>
<td>581.2</td>
<td>159</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TN (μg L⁻¹)</td>
<td>-0.24</td>
<td>0.02</td>
<td>808.6</td>
<td>159</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TP:TN</td>
<td>-0.3</td>
<td>&lt;0.01</td>
<td>0.3</td>
<td>159</td>
<td></td>
</tr>
<tr>
<td></td>
<td>TSS (mg L⁻¹)</td>
<td>-0.14</td>
<td>0.05</td>
<td>17.7</td>
<td>158</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Conductivity (umhos cm⁻¹)</td>
<td>0.18</td>
<td>0.03</td>
<td>212.2</td>
<td>159</td>
<td></td>
</tr>
</tbody>
</table>
Table 3.1 (Continued)

<table>
<thead>
<tr>
<th>Tibbetts Creek</th>
<th>SRP (μg L⁻¹)</th>
<th>0.09</th>
<th>0.40</th>
<th>16.8</th>
<th>139</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>TP (μg L⁻¹)</td>
<td>0.25</td>
<td>&lt;0.01</td>
<td>61.4</td>
<td>140</td>
</tr>
<tr>
<td>1997-2008</td>
<td>Nitrile + Nitrate Nitrogen (μg L⁻¹)</td>
<td>-0.19</td>
<td>0.06</td>
<td>1092.2</td>
<td>139</td>
</tr>
<tr>
<td></td>
<td>TN (μg L⁻¹)</td>
<td>-0.08</td>
<td>0.22</td>
<td>1407.3</td>
<td>138</td>
</tr>
<tr>
<td></td>
<td>TP:TN</td>
<td>-0.15</td>
<td>0.18</td>
<td>0.4</td>
<td>140</td>
</tr>
<tr>
<td></td>
<td>TSS (mg L⁻¹)</td>
<td>0.01</td>
<td>0.84</td>
<td>12.8</td>
<td>140</td>
</tr>
<tr>
<td></td>
<td>Conductivity (umhos cm⁻¹)</td>
<td>0.05</td>
<td>0.51</td>
<td>219.5</td>
<td>140</td>
</tr>
</tbody>
</table>

* Regular monthly samples; storm samples included when no routine samples taken

3.1.4 Tributary Phosphorus Loading

Since the 1970s there have been several studies to determine TP loading for Issaquah Creek. Rock (1974) demonstrated the effect of various numerical methods and sampling frequencies on estimating annual loads. In an effort to improve the accuracy of loading estimates there have been several daily sampling efforts to aid in this effort (Rock, 1974, Birch, 1976, Butkus, 1987, and Brett et al., 2005b). Other historical daily observations compiled along with historical USGS flow measurements of Issaquah Creek daily flow rates were compiled to estimate TP loading from the creek (DeGasperi, 2004). In addition, tributary load estimates were part of the inputs for the Lake Sammamish CE-QUAL_ICM water quality modeling effort (King County, 2008). Table 3.2 summarizes these TP loading estimates. Estimated TP loads – both measured and modeled, with the exception of the August 2000 to August 2001 study – range from 7,126 to 8,617 kg TP/year. Note that Brett et al. (2005b) study was conducted during a particularly dry year as noted by the relatively low annual average flow rate.

Table 3.2 Summary of Historical Annual TP Loading Estimates for Issaquah Creek.

<table>
<thead>
<tr>
<th>Water Year</th>
<th>TP Load kg/yr</th>
<th>Annual Mean TP μg/L</th>
<th>Annual Mean Flow cfs</th>
</tr>
</thead>
<tbody>
<tr>
<td>1973</td>
<td>Rock (1974)</td>
<td>7,126</td>
<td>65</td>
</tr>
<tr>
<td>Jun 74 – Jun 75</td>
<td>Birch (1976)</td>
<td>7177</td>
<td>57</td>
</tr>
<tr>
<td>Apr 86 – Mar 87</td>
<td>Butkus (1987)</td>
<td>8,617</td>
<td>45</td>
</tr>
<tr>
<td>Aug 00 – Aug 01</td>
<td>Brett et.al. (2005b)</td>
<td>2,699</td>
<td>39</td>
</tr>
</tbody>
</table>

1. Un-biased regression model loading estimates.

The loading of TP from Issaquah Creek, which is 70 percent of the input to the lake, has apparently not changed since the 1970s based on loading estimates – both measured and modeled – and are directly influenced, as would be expected, by flows (Table 3.2).
Ellison and Brett (2006) found in a study of 16 streams of various land-use types in the region that the dominant P fraction during storm events was particulate forms (TP). Total dissolved phosphorus was the dominant fraction during baseflows – with the exception of agricultural (dairy) catchments where particulate P was highest in baseflows. They found that particulate phosphorus and total phosphorus increased during storm events in all stream types (forested, urban, agricultural dairy farms and mixed land cover) by 614 and 200 percent respectively. However, they also found that on average only 20 percent of the particulate phosphorus transported during storm events was biologically available.

### 3.1.5 Residence Time and TP Load

The inflow rate of Issaquah Creek, plus other sources of minor streams, ground water and precipitation directly on the lake surface, determine the lake's water residence time. Issaquah Creek represents 70 percent of the total annual inflow to the lake, so for example, in the 1964–1976 time period; mean total lake inflow rate was 217 cfs (6.2 m³/s) or 196 x 10⁶ m³/year. Water residence time during that period was about 1.79 years (350 x10⁶ m³ lake/196 x10⁶ m³). During the past three decades total average inflow was 177 cfs (5.1 m³/s), and residence time averaged about 2.17 years, or 21 percent longer.

Water residence time is an important determinant of lake TP concentration. The simplest and well verified equation for lake TP is:

\[
TP_l = TP_i / (1+\sqrt{\tau})
\]

Where \( \tau \) is water residence time, \( TP_i \) is annual average, flow weighted inflow concentration and \( TP_l \) is annual average, volume-weighted concentration in the lake (Welch and Jacoby, 2004).

This simple model did not fit the Lake Sammamish data well during the 1994 modeling exercise. Therefore, the model was adjusted to the annual inflow TP concentration of 80 µg/L, based on loading determined in 1985 (King County, 1995), a lake concentration of 26 µg/L in 1971–1975 and a residence time of 1.79 years during that period. That produced an exponent of 1.25 \((\tau^{1.25})\). Increasing the residence time to 2.17 years, with the same inflow concentration, gave a lake concentration of 22 µg/L. Thus, a lower inflow volume and subsequent slightly longer residence time could theoretically result in a greater sedimentation of TP. Low flow years would reduce the annual load to the lake as well as potentially increase the sedimentation rate.

### 3.2 Temperature

Annual average temperatures in Lake Sammamish have varied from year to year depending on changes in weather, especially regional air temperature. Temperature varied similarly from year-to-year in both Lake Sammamish and Lake Washington where annual averages have trended toward higher values (Figure 3.4) – approximately 0.25°C (0.45°F) per decade (King County, 2013). The lack of statistical significance in the trend is primarily
due to the large inter-annual variability in average lake temperature and the length of the record available. Temperature data for Lake Washington from 1963 to 2012, provided by the University of Washington, indicates a similar long-term increase in annual average lake temperature, which is statistically significant. Temperatures in Lake Sammamish were lower in the winter and higher in summer, indicating more susceptibility to regional air temperature than Lake Washington, probably due to the volume differences (Figure 3.4).

Water temperature of these two large lakes is affected by regional climate, which in turn is influenced by global climate variability and change. Studies of long-term temperature changes of large lakes throughout the world have detected the influence of human-caused warming of the atmosphere superimposed on regional scale variability. As mentioned above, climate variability in this region is strongly influenced by variation in Pacific Ocean circulation.

Some of the observed long-term variability in warming of Lake Washington and Lake Sammamish is likely due to PDO, which shifted from a cool to a warm phase in 1976–1977 and may be returning to a cool phase (King County, 2014). Research has also shown that the effect of climate variability and change is not limited to lake temperature, but includes ecological changes that result from shifts in the timing of the onset of lake thermal stratification. The processes that lead to warmer lake water generally also lead to earlier thermal stratification in these lakes.

Temperature profile data, collected in Lake Sammamish since the early 1990s, were used to calculate the timing of thermal stratification onset from the Schmidt’s stability index (Hutchinson, 1957; Idso, 1973). Earlier data were not collected at enough depths and were not of sufficient quality to be used in the calculation. The onset date of stratification over the past two decades is not statistically significant given random variation ($\rho = 0.14$); Figure 3.5). Also, there were no trends in length of stratification over that time period, as shown in Figure 3.5. Lake Washington has a longer and more consistent dataset beginning in the 1960s and has shown statistically significant long-term increases in annual lake temperatures of approximately 0.14°C per decade (0.26°F per decade) (King County, 2014) and a shift to an earlier onset of stratification (Winder and Schindler, 2004). There is a significant amount of synchrony in regional lake temperatures, so it is reasonable to assume that Lake Sammamish has experienced similar warming and shift in the timing of the onset of stratification as Lake Washington since the 1960s. However, neither lake has continued to warm significantly since 1993, likely due at least in part to a shift to a wet/wet/cool phase of the PDO. The lack of warming trend since 1993 explains why there are no significant trends in the onset or duration of stratification in either lake over this time period.
Figure 3.4  Volume-weighted temperatures in Lake Sammamish and Lake Washington during January and August.

Except for limited diffusion, significant transport of P from the hypolimnion to the epilimnion usually does not occur until the thermocline mixes down in the fall, well past the main growing season (King County, 1995). Also, the Issaquah Creek inflow, being colder than the epilimnion, usually disperses through the lake’s metalimnion, and its contained nutrients are unavailable to phytoplankton in the surface waters.
3.3 Alkalinity, pH, and Conductivity

Lake Sammamish is of relatively low ionic strength and has limited buffering to pH changes, as are most other western Washington lakes. Total alkalinity, also known as acid neutralizing capacity, in low calcium water is due mostly to bicarbonate. Concentration of dissolved ions in natural waters is due to weathering of watershed soil and bedrock. The dissolution of base chemicals from rocks, such as calcium, is part of the weathering process.

Annual mean, whole-lake alkalinity ranged from 32.5 to 42.5 mg/L expressed as CaCO$_3$, over the past 30 years (Figure 3.6). While alkalinity ranged from 26 to 42 mg/L during the early 1970s, the annual average was 33.3 mg/L (Welch, 1977). Only two values have been as low as the 1970s average (1982 and 1983; Figure 3.6). There has been a net increase of about 10 mg/L over this period. The year-to-year variation probably reflects shifts in the relative amount of stormwater runoff and base flow in the watershed as a result of differences in the intensity and frequency of precipitation events (Figure 3.1).
The significantly increasing trend in total alkalinity may be due to the continued land disturbance due to development. That was suggested to have caused total alkalinity in Lake Washington to increase from about 27 mg/L in 1956 to 40 mg/L by the early 1990s (Edmondson, 1994; King County, 2003). Also, greater dissolution of calcium and magnesium (weathering) may occur from increasing amount of pavement once development is in place, as opposed to undisturbed/unpaved land and contribute to greater alkalinity. Weathering is enhanced by the acidity of precipitation, which has a pH of around 5 locally (Tim Larson, pers. com., April 7, 2014).

Contrary to the trend shown by alkalinity data, pH in the epilimnion and metalimnion showed no trend, while there was a significant decreasing trend in pH in the hypolimnion ($\rho<0.05$) (Figure 3.7). While equilibrium pH increases with alkalinity over a wide range, the increase of alkalinity by 10 mg/L is not enough to show a substantial correlated difference in pH, especially given the day-to-day and diurnal variability caused by photosynthesis and respiration. A difference of 10 mg/L alkalinity would be equivalent to a pH difference of only 0.1 pH units.

The effects of photosynthesis and respiration are shown by consistently higher pH levels in the epilimnion due to photosynthesis (CO$_2$ removal) versus consistently lower levels in the hypolimnion due to respiration (CO$_2$ addition) without photosynthesis. The decreasing trend in the hypolimnion could indicate an increasing predominance of respiration over photosynthesis. Intermediate pH levels in the metalimnion reflect a combination of other processes along with the variation in mixing conditions.

Conductivity is a measure of the total dissolved solids content in water. The major ions, which largely produce conductivity, tend to be conservative. That is, they are not lost as water moves through the lake processes, unlike non-conservative substances that are subject to removal processes; e.g., phosphorus uptake by phytoplankton. Thus, conductivity
may tend to be higher during years of low rainfall and runoff, and diluted during high runoff years.

Figure 3.7  Average annual epilimnetic (0 – 10 m), metalimnetic (10 – 15 m), and hypolimnetic (>15 m) pH at 0612 from 1982-2011. Note: 1980 and 1981 not included due to insufficient data.

Conductivity has significantly increased over the past 30 years ($\rho < 0.01$)(Figure 3.8). The increase was about 23 percent, compared to the alkalinity increase over the same period of about 16 percent. The increase in conductivity is probably a reflection of increased development in the watershed, as was suggested for alkalinity. Impervious area in the watershed increased dramatically between 1992 and 2011 (Figure 1.3) with only 8 percent of the total watershed categorized as impervious in 1992, compared to 12 percent in 2011.

Figure 3.8  Annual mean whole-lake conductivity ($\mu$hos/cm) at 0612 from 1982 through 2011. (Means +/- 1 SD). Note: 1980 and 1981 not included due to insufficient data.
3.4 Trophic State Indicators

The common indicators used to determine trophic state in lakes are summer means for TP, chl a, and Secchi disc transparency. The accepted boundary levels for these indicators between oligotrophy and mesotrophy are 10 μg/L, 3 μg /L and 4 m for TP, chl a and transparency, respectively. Between mesotrophy and eutrophy the levels are: 30 μg /L TP, 9 μg /L chl a, and 2 m transparency (Nürenberg, 1996). These trophic state indicator levels also represent a gradient of lake water quality that relate to the value of a lake for recreation and aquatic life.

3.4.1 Phosphorus

Phosphorus is the primary nutrient of concern in Lake Sammamish, as well as most other lakes, and, as such, has been the focus of study for many years (Section 1.3.1). Annual, whole-lake, volume-weighted TP (v-w TP) concentration represents the mass of TP in the lake and takes into account its unequal distribution, mainly due to summer thermal stratification that leads to DO depletion and P accumulation in the hypolimnion from bottom sediment. Distribution is usually uniform in winter due to complete mixing. The hypolimnetic period runs from the onset of stratification in May until lake turnover in early November. Annual whole-lake v-w TP, summer hypolimnetic v-w TP (June-turnover), summer epilimnetic v-w TP, and winter whole-lake v-w TP are shown in Figures 3.10, 3.11, 3.12, and 3.13.
Figure 3.10 Volume-weighted mean summer hypolimnetic (June – turnover) total phosphorus (TP) at 0612, 1964–2011. Arrow indicates wastewater diversion in 1968.
Note: data for 1964–1966 from METRO; 1970–1975 from UW-Dept. of Civil Engineering; 1979–1981 from METRO/King County (King County 1995), 1981–2011 from King County. TSP (total soluble P) data from 1964–1966 corrected upward to TP by TP/TSP ratio of 1.2.

Figure 3.11 Mean summer (June – September) epilimnetic total phosphorus (TP) concentrations at Station 0612, 1964–2011. Arrow indicates wastewater diversion in 1968.
Note: data for 1964–1966 from METRO; 1970–1975 from UW-Dept. of Civil Engineering; 1979–1981 from METRO/King County (King County 1995), 1981–2011 from King County. TSP (total soluble P) data from 1964–1966 corrected upward to TP by TP/TSP ratio of 1.2.
Lake Sammamish Water Quality Response to Land Use Change

E.B. Welch and D. Bouchard

December 2014

Figure 3.12 Volume-weighted whole lake winter total phosphorus (TP) at 0612, 1964–2011. Note: summer data only in 1981 so not included. Data for 1964–1966 from METRO; 1970–1975 from UW-Dept. of Civil Engineering; 1979–1981 from METRO/King County (King County 1995), 1981–2011 from King County. TSP (total soluble P) data from 1964–1966 corrected upward to TP by TP/TSP ratio of 1.2.

3.4.1.1 TP Response to Diversion

As mentioned earlier, Lake Sammamish annual v-w TP did not respond to wastewater diversion as quickly as expected based on the Lake Washington experience (Section 1.3.2.2). This is illustrated by mean annual whole-lake TP concentrations, which averaged 32 µg/L before versus 26 µg/L during the first four years after diversion (Figure 3.9). However, whole-lake TP had begun to decline by 1975, reaching an apparent equilibrium of less than 20 µg/L through 1986.

Hypolimnetic TP, on the other hand, has shown a progressive and significant trend downward since wastewater diversion (p<0.01) and since 1981 (p<0.05), decreasing by about two-thirds since 1964-1966 (Figure 3.10; Table 3.3). That decrease is due to a gradual, long-term decline in sediment P release rate – 77 percent since before and 61 percent since the 1970s (Table 3.4). If a high rate in 2006 were omitted from the most recent 7 years, the resulting mean is 1.87 mg/m² per day, an 80 percent decrease and 70 percent decrease since before and after diversion. Gradually declining sediment P release following wastewater diversion has been observed in other lakes and may be due in part to burial of the high P sediment. Given the sedimentation rate determined in the mid-1970s of 0.32 cm/year (Birch, 1976), almost 14 cm of sediment would have accumulated since wastewater diversion in 1968.

Perkins (King County, 1995) estimated that a significant percentage of the entire TP loading to the lake (about half the total annual load) occurs in a few weeks of high flow, but not necessarily during flood events, based on data from the 1970s and 1980s (King County, 1995). However, annual and winter lake TPs have not increased significantly since the
1980s. Winter TP did appear to be increasing in the late 1980s and early 1990s, but by the late 1990s had declined again, resulting in no significant trend overall (Figure 3.12).

Summer epilimnetic TP content does not show a clear pattern of recovery from an enriched condition (Figure 3.11). Summer epilimnetic TP is usually the determining nutrient for the amount of algae during the growing season, and in turn, water clarity or transparency. Pre-diversion epilimnetic TP content was in the range of 5–15 µg/L, which was similar to post-diversion years from the late 1970s onward, probably due to spring bloom of phytoplankton removing phosphorus from the water column during April–June. Moreover, 50 percent of the immediate post-diversion observations from 1970–1975 were well above 15 µg/L, higher than pre-diversion years. Some of the differences between pre- and post-diversion epilimnetic TP data may be due to different methods of analysis discussed earlier (Section 2.5.3).

Table 3.3  Interval mean (±SD) TP in µg/L in different time periods and portions of the lake. All values volume-weighted. Pre-1981 data from King County (1995). Wastewater was diverted in 1968.

<table>
<thead>
<tr>
<th>Year Interval</th>
<th>Annual Whole lake</th>
<th>Summer Epilimnion (Jun-Sept)</th>
<th>Summer Hypolimnion (June-turnover)</th>
<th>Winter Whole lake (Dec-Mar)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1964–1966</td>
<td>31.6 ± 6.8</td>
<td>13.3 ± 2.11</td>
<td>58.3 ± 13.6</td>
<td>26.4 ± 0.3</td>
</tr>
<tr>
<td>1970–1975</td>
<td>26.4 ± 4.9</td>
<td>18.6 ± 7.5</td>
<td>37.5 ± 13.7</td>
<td>29.6 ± 2.33</td>
</tr>
<tr>
<td>1981–1986</td>
<td>16.7 ± 1.7</td>
<td>11.3 ± 0.9</td>
<td>26.9 ± 6.5</td>
<td>19.5 ± 1.3</td>
</tr>
<tr>
<td>1987–1992</td>
<td>19.1 ± 2.6</td>
<td>11.6 ± 2.4</td>
<td>30.2 ± 14.5</td>
<td>26.1 ± 2.8</td>
</tr>
<tr>
<td>1993–1998</td>
<td>19.5 ± 3.4</td>
<td>13.4 ± 4.2</td>
<td>25.0 ± 7.2</td>
<td>24.6 ± 5.9</td>
</tr>
<tr>
<td>1999–2004</td>
<td>17.1 ± 1.4</td>
<td>11.2 ± 2.5</td>
<td>24.1 ± 2.9</td>
<td>22.8 ± 3.3</td>
</tr>
<tr>
<td>2005–2011</td>
<td>17.9 ± 2.2</td>
<td>11.5 ± 1.4</td>
<td>19.4 ± 5.1</td>
<td>22.9 ± 2.8</td>
</tr>
</tbody>
</table>

1. 1964 not included - no June data available.
2. 1970 data set not complete. Only summer data available.
3. 1975 not included in Dec–Mar calculation as no data were collected in 1976.

Summer epilimnetic TP has also shown no real trend since 1981 (Figure 3.11). Although variable, ranging from 7.6 to 20.1 µg/L, the long-term mean has remained at about 12 µg/L subsequent to the 1970s when the mean was considerably higher with maximums of 25–30 µg/L (Table 3.3; Figure 3.11).

Some of the year-to-year variability in sediment P release (±50-75percent) might be related to sparse data that missed the onset of anoxia at the sediment-water interface (Table 3.4). Choice of time interval and hypolimnetic volume can also lead to differences in calculated rates. A consistent start time in August (with the exception of 1966 and 1995), continuing to turnover, and data from 15 m to the lake bottom, were used to calculate the reported rates. The start times differ slightly from those by King County (1995), but the trend is similar. Nevertheless, hypolimnetic buildup is a reliable method to determine sediment P release (Nürnberg, 1987). Rates determined by both hypolimnetic buildup at
20-27 m and from *in vitro* cores (8 experiments) in 1974 showed close agreement with average rates of 5 and 3.4 mg/m² per day, respectively (Welch, 1977). Those rates are considerably less than calculated here for 1974 (7.5 mg/m² per day). Nevertheless, rates have gradually declined since the mid-1970s, indicating a slow but steady recovery from legacy sediment P.

Table 3.4 Interval mean (±SD) sediment P release rates in mg/m² per day for the time periods shown. Wastewater diversion was in 1968.

<table>
<thead>
<tr>
<th>Time Interval</th>
<th>Sediment P release rates ±SD</th>
</tr>
</thead>
<tbody>
<tr>
<td>1964–1965</td>
<td>10.65 ± 6.3</td>
</tr>
<tr>
<td>1971–1975</td>
<td>6.26 ± 2.6</td>
</tr>
<tr>
<td>1981–1986</td>
<td>4.68 ± 2.6</td>
</tr>
<tr>
<td>1993–1998</td>
<td>2.40 ± 2.0</td>
</tr>
<tr>
<td>1999–2004</td>
<td>3.90 ± 1.6</td>
</tr>
<tr>
<td>2005–2011</td>
<td>2.45 ± 1.7</td>
</tr>
</tbody>
</table>

3.4.1.2 TP Response to Development

Annual whole-lake v-w TP ranged from 13.4 to 23.5 during 1982 to 2011 (Figure 3.9). A goal of 22 µg/L was set in 1994 to protect the quality of Lake Sammamish, and so far that goal has been exceeded only once, in 1996. In the longer term, annual v-w TP has shown a decline since 1964 but no trend subsequent to 1982 (Figure 3.9). Lumping the data in groups of roughly six-year intervals to reduce year-to-year variability shows slightly higher means in the 1990s during the period of rapid development, but overall the six-year means ranged between 16.7 and 19.5 µg/L and are not significantly different since the 1980s (Table 3.3, Figure 3.13).

![Figure 3.13](image-url) Interval mean whole lake, summer hypolimnion (June – turnover), and summer epilimnion (June – September) total phosphorus (TP). All values volume-weighted.
Annual mean TP has been affected by the magnitude of hypolimnetic TP. Highs and lows in the six-year TP means and in individual years show rather close association with hypolimnetic TP (Table 3.3; Figure 3.13). That association is not surprising, because maximum hypolimnetic TP had reached over 300 µg/L before wastewater diversion, 100 µg/L in the 1970s after diversion, but much less since then. In fact, there is a strong relationship between annual and hypolimnetic TP (Figure 3.14). Also, the decrease in hypolimnetic TP of 18.1 µg/L since the 1970s amounts to 1,774 kg overall less TP from internal loading. That represents 64 percent of the 8.5 µg/L (2,780 kg) decrease in annual whole-lake TP since the 1970s (Table 3.3). Had hypolimnetic TP not decreased, annual whole-lake concentration would have been closer to 23 instead of 17.9 µg/L.

Summer epilimnetic TP has shown less effect from the high hypolimnetic TPs because the thermocline minimizes transfer. In contrast, the high TP hypolimnetic water is mixed throughout the lake at turnover and, thus, has a large effect on the annual TP, in particular, winter concentrations as the high post-turnover concentrations tend to linger in the water column before being taken up by phytoplankton in early spring. However, unlike summer hypolimnetic TP, winter whole-lake TP has not declined significantly since the 1970s (Figures 3.11 and 3.13).

Periodic high winter TP inputs from storm events may have been offset by the decreased internal loading, although Tibbetts Creek is the only tributary to Lake Sammamish to show a significant increase in TP concentrations between 1997 through 2008 (Table 3.1).

![Graph](image)
from sediment (Table 3.4), the decrease in hypolimnetic SRP is consistent with decreased sediment release rate. The less pronounced decreasing tendency in epilimnetic SRP is understandable because the strong stratification during summer minimizes the transfer of SRP from high hypolimnetic concentrations to the epilimnion. Also, SRP is quickly utilized by phytoplankton and tends to remain at a normally low, growth-limiting level.

SRP concentrations have decreased in the Issaquah, Pine Lake, and Idylwood creeks while TP had not changed significantly (Section 3.2.3). In a regional study of 16 streams, SRP had declined significantly in many streams within the urbanizing region of the county (Ellison and Brett, 2006). That was most likely due to connecting former septic systems to the regional wastewater system with discharge to Puget Sound, conversion of agricultural (dairy farms) to low density single family residential and regrowth of landscape and forest vegetation. During both storm and baseflow events, TP and total dissolved P concentrations were highest in the agricultural streams due to the leaching of dissolved P from the catchments soil, which may have accumulated manure P from dairy operations. The fluctuation in TP in the tributaries was determined by baseflow dissolved P, which was the portion that contributed most of the TP during baseflow.

### 3.4.2 Nitrogen

Total nitrogen ranged from around 300 to 600 µg/L during the past 20 years (Figure 3.16). Annual, whole-lake TN approximated the average of summer hypolimnetic concentrations and all three varied proportionately. Epilimnetic TN was lower than whole-lake due to settling of particulate N through the stable water column in summer. There was a downward trend in TN regardless of the lake fraction or season, though the trend was only weakly significant.
Comparison of TN with TP earlier than 1993 is not possible because TN was not determined. A large fraction of TN is nitrate+nitrite-nitrogen, which has not changed significantly since 1993 (Figure 3.17). Thus, increased retention of runoff TN is not likely to be as effective as for TP, which usually has a larger particulate fraction. However, like TP, TN is not conservative while conductivity and alkalinity are conservative, which have no in-lake loss.

![Figure 3.16 Mean volume-weighted total N in the whole lake annually, and the summer (June to turnover) in the epilimnion and hypolimnion at 0612, 1993–2011.](image)

![Figure 3.17 Mean volume-weighted nitrate+nitrite nitrogen in the whole lake annually, and the summer (June to turnover) in the epilimnion and hypolimnion at 0612, 1985–2011.](image)

The ratio of TN:TP and DIN:SRP has been well above the Redfield ratio, which is considered to be the ratio at which algae take up and assimilate these nutrients (Figures 3.18 and
3.19). Thus, nitrogen is in excess and if growth were limited by nutrients, phosphorus would disappear first and limit growth. For this and other reasons, management of lake quality has been focused on phosphorus.

There was no consistent trend in the TN:TP ratios, despite lower TN concentrations and rather stable TPs in recent years. Ratios were slightly higher in the late 1990s in the epilimnion when TNs were high and TPs low during some of those years (Figures 3.18 and 3.19). In spite of the variability, there was a significant decrease in the summer epilimnentic DIN:SRP ratio ($\rho<0.05$).

![Annual and summer (June – September) volume-weighted epilimnionic ratios for TN:TP, compared with the Redfield ratio of 7.2 (by weight) at 0612, 1994–2011.](image1)

![Annual and summer (June – September) volume-weighted epilimnionic ratios for DIN:SRP at 0612, 1994–2011.](image2)
3.4.3 **Transparency and Chlorophyll \(a\) (chl \(a\))**

Overall, chl \(a\) and transparency, as with whole-lake TP, showed a delayed recovery following diversion of wastewater. Trophic state, as indicated by summer mean epilimnetic chl \(a\) and transparency, increased only slightly from before diversion (4.5 µg/L, 3.2 m) to after diversion during the early 1970s (4.3 µg/L, 3.6 m). Further recovery occurred subsequently with the lower epilimnetic TP in the early 1980s (Figure 3.11), associated with higher transparency and lower chl \(a\) (Figure 3.21).

Though variable, transparency has, in some years, been up to 2 m greater than prior to wastewater diversion (Table 3.5, Figure 3.20). The range in summer transparency since 1994 was 4.1 – 6.8 meters. Although there has been no significant trend since 1980 (\(\rho > 0.1\)), transparency has averaged over 5 m since 1999 (Table 3.5) and the 4.0 m goal has been reached every year since the inception of the Lake Management Plan, except for 1999 when the mean summer transparency was 3.9 m (Figure 3.20). That goal is also the boundary between mesotrophy and oligotrophy (Nürnberg, 1996). Thus, the lake’s trophic state has been oligotrophic for the past 30 years, based on transparency (Table 3.5). This is consistent with summer epilimnetic TP, which also has changed little since the 1980s (Figure 3.11).

From 1985 through 1994 mean summer chl \(a\) remained below the 1994 management goal of 2.8 µg/L (Table 3.5, Figure 3.21). However, since 1995 chl \(a\) has exceeded that goal more often than not. Nevertheless, there has not been a significant change in chl \(a\) since the 1960s (\(\rho > 0.1\)). Based on chl \(a\), trophic state has remained slightly above the mesotrophic threshold of 3.0 µg/L since 1994.

**Table 3.5** Interval mean (±SD) summer transparency (meters) and epilimnetic chl \(a\) (µg/L) in different time periods. Pre-1995 data from King County (1995). Wastewater was diverted in 1968.

<table>
<thead>
<tr>
<th>Year Range</th>
<th>Transparency (m)</th>
<th>Chlorophyll (a) (µg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1964–1966</td>
<td>3.2 ± 0.4</td>
<td>4.5 ± 2.7</td>
</tr>
<tr>
<td>1971–1975</td>
<td>3.8 ± 1.5</td>
<td>3.6 ± 0.4</td>
</tr>
<tr>
<td>1981–1986</td>
<td>4.9 ± 0.8</td>
<td>3.2 ± 0.3</td>
</tr>
<tr>
<td>1987–1992</td>
<td>4.8 ± 0.9</td>
<td>1.9 ± 0.2</td>
</tr>
<tr>
<td>1993–1998</td>
<td>4.6 ± 0.9</td>
<td>3.7 ± 0.3</td>
</tr>
<tr>
<td>1999–2004</td>
<td>5.3 ± 0.9</td>
<td>3.3 ± 0.9</td>
</tr>
<tr>
<td>2005–2011</td>
<td>5.2 ± 0.5</td>
<td>3.6 ± 0.31</td>
</tr>
</tbody>
</table>
Figure 3.20 Annual summer (June – September) Secchi transparency at 0612, 1964–2011, compared with the lake management goal of 4.0 meters.

Figure 3.21 Annual summer epilimnetic (June – September) chlorophyll a at 0612, 1964–2011, compared with the lake management goal of 2.8 meters.

Transparency was 1.1 m less than expected with the observed chl a and TP over the past 40 years based on relationships from Carlson (1977) (Figure 3.22). Since the late 1990s, transparency has exceeded what would be expected from the Carlson relationship based on chl a by up to 2 meters (Figure 3.22). In contrast, chl a and transparency in Lake Washington more dramatically followed the Carlson relationship during pre- and post-diversion (Figure 3.22).
The changes in plankton populations in Lake Washington were complicated. The most pronounced shift in transparency occurred as a result of a restructuring of the plankton community as a result of the decrease in P; with the decrease in available P, conditions were no longer ideal for Oscillatoria, which diminished entirely in 1976 (Edmondson 1991). This decline in Oscillatoria coincided with an increase in the filter feeding planktonic crustacean Daphnia, which can quickly reduce algae abundance, thus increasing water transparency. In addition, there was a reduction in the population of Neomysis mercedis, a planktonic crustacean that was shown to have a strong feeding preference for Daphnia. Nevertheless, the net result was close agreement with the Carlson relationship both before and after diversion.

![Figure 3.22 Chlorophyll and transparency summer means in compared to a typical Carlson relationship.](image)

Why transparency was greater than would be expected in Lake Sammamish given the summer epilimnetic TP and chl \( a \), may be due to a decrease in non-algal turbidity. An increase in water residence time, which has occurred, would tend to increase transparency. A change in the plankton community structure can also affect transparency. A shift from unicellular algae to colonial algae such as cyanobacteria can change the scattering of light. Colonial cyanobacteria tend to fuse together and thus yield a relatively higher transparency per unit chl \( a \). Although chl \( a \) changed little in the early years following diversion, the cyanobacteria fraction had decreased by 46 percent during 1970–1975 after diversion (Welch, 1977). Chrysophytes (mostly diatoms) have always made up most of the biovolume with peaks in early spring or late fall–sometimes both (Figure 3.23). Cyanobacteria (blue-green algae) usually occur in largest concentration during mid to late
summer, which was the case during all years monitored, except for 2010 when the maximum occurred in the February followed by another bloom in June. The highest biovolumes of cyanobacteria occurred in September 2012. Nevertheless, there is no evidence that colonial cyanobacteria have increased in recent years to explain increased transparencies and such an increase would be unexpected given the consistency in TP.

![Figure 3.23 Biovolume of phytoplankton from 2003 to 2013. Cyanophyta referes blue-green algae. Chrysophyta are mostly diatoms. Other includes chlorophyta and Euglena. Note: data not recorded for 2008.](image)

Maximum spring-bloom chl $a$, highly variable year-to-year due largely to local climate effects on water column mixing and stability, also has shown no trend since the 1960s (Figure 3.24). What may have changed is the timing of the spring-bloom maximum, which has shifted earlier ($\rho=0.07$), especially since 2000 when all maximums except one occurred before 100 days (Figure 3.24). Before and including 2000, maximums occurred on or later than 100 days. That tendency may be related to an earlier onset of stratification as observed in Lake Washington (Winder and Schindler, 2004), though at this point there is not a temperature dataset long enough to verify this in Lake Sammamish (Section 3.2).
Figure 3.24 Maximum spring bloom chlorophyll $a$ and timing of spring bloom maximum chlorophyll $a$ (March – May) since the 1960s.

3.4.4 Areal Hypolimnentic Oxygen Demand

The hypolimnion in Lake Sammamish routinely goes anoxic during summer stratification. That is, respiration exceeds the initial supply of DO that exists when stratification seals off the hypolimnion from contact with the atmosphere. The areal hypolimnetic DO deficit, or AHOD, is used as an index of trophic state, like TP, chl $a$, and transparency. The oligotrophic-mesotrophic boundary has been defined at 0.4 g/m$^2$ per day (Nürnberg, 1996). AHOD is the earliest established index of eutrophication (Mortimer, 1941; Hutchinson, 1957). The areal rate (normalized for depth) allows comparison of lakes with different hypolimnetic depths. For example, AHOD in Lake Washington is similar in magnitude ($\sim 0.5$ g/m$^2$ per day) to that in Lake Sammamish despite hypolimnetic DO
concentrations remaining above 2 mg/L in Lake Washington, while in Lake Sammamish DO is completely depleted in the lower hypolimnion during stratification (King County DNRP, 2003).

3.4.4.1 AHOD Response to Diversion

With less algal organic matter enriching sediments, due to less external P loading, AHODs should decrease following wastewater diversion. That has been documented in other lakes; e.g., Lake Washington (Lehman, 1988; King County DNRP 2003), Lake Onondaga, New York (Matthews and Effler, 2006), and Long Lake Spokane (Patmont, 1987, and recent data) (Table 3.7). Also, AHOD has been correlated with TP loading and retention (Welch and Perkins, 1979; Cornett and Rigler, 1979), and chlorophyll (Walker, 1986). Also, AHOD increased with non-point TP loading to Tenkiller Reservoir, OK. Nevertheless, AHODs in Lake Sammamish apparently did not decrease following wastewater diversion, as had been suggested in the early 1980s (Welch et al., 1986). If anything, reanalysis shows that the rates were slightly higher in the 1970s, just after diversion, but declined to about 0.5 g/m² per day during the 1980s and early 1990s (Table 3.7). There is some indication that AHOD was probably much lower in 1913 (0.13-0.26 g/m² per day), based on a comparison of DO remaining below 15 m on August 13, 1913, versus on August 12, 1964 and 1965, and AHODs in 1965 (Isaac et al., 1966).

Table 3.6 Response of areal hypolimnetic deficit rate (AHOD) to changes in TP input during pre- and post-diversion of wastewater, except the change in TP input to Lake Tenkiller was an increase from agricultural nonpoint runoff.

<table>
<thead>
<tr>
<th>Lake/Reservoir</th>
<th>Pre</th>
<th>Post-1</th>
<th>Post-2</th>
<th>%Δ</th>
<th>%Δ TPᵢ</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lake Spokane</td>
<td>2.2-6.3¹ (1971-1977, n=7)</td>
<td>1.8-2.6¹ (1978-1985, n=8)</td>
<td>0.73 ± 0.11 (2010-2013, n=4)</td>
<td>66-88</td>
<td>-83¹</td>
</tr>
<tr>
<td>(WA, reservoir)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lake Tenkiller</td>
<td>0.60² (1960)</td>
<td>1.5 ± 0.4² (1986-2006, n=13)</td>
<td></td>
<td>+250</td>
<td>+200²</td>
</tr>
<tr>
<td>(OK, reservoir)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lake Onondaga</td>
<td>2.12 ± 0.2² (1978-1986, n=6)</td>
<td>1.08 ± 0.1² (1997-2002, n=6)</td>
<td></td>
<td>-49²</td>
<td>-44²</td>
</tr>
<tr>
<td>(NY)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lake Washington</td>
<td>0.71 ± 0.1⁶ (1957-1969, n=8)</td>
<td>0.58 ± 0.05⁶ (1970-1983, n=14)</td>
<td>0.47 ± 0.09⁶ (1993-2001, n=9)</td>
<td>-34</td>
<td>-75⁶</td>
</tr>
<tr>
<td>(WA)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lake Sammamish</td>
<td>0.50 ± 0.2⁶ (1957, 1964-1965, n=3)</td>
<td>0.60 ± 0.15⁶ (1971-1975, n=4)</td>
<td>0.52 ± 0.10⁶ (1981-1994, n=13)</td>
<td>NS</td>
<td>-33⁷</td>
</tr>
<tr>
<td>(WA)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

¹Patmont (1987) – only ranges were given;
²data from Summers (1961);
³Cooke et al. (2011);
⁴Matthews and Effler (2006);
⁵in-lake concentration, inflow unavailable, Matthews and Effler (2006);
⁶Lehman (1988), rate in 1933 was 0.42 g/m² per day, similar to recent recovered rate;
⁷King County (2003);
⁸Issac et al. (1966), UW (1971-1975) and King County (1981-1994) data;
⁹Welch et al. (1980).
AHOD determination is sensitive to several variables. Sparseness of data before diversion and in the 1970s and the time interval selected for calculating AHOD may be part of the reason for not detecting a pre- and post-diversion difference. Rates using two separate time intervals differed by 0.1 g/m² per day or more. Also, year-to-year variation, even using a constant time interval, was often on the order of 0.3 g/m² per day. The two pre-diversion values reported by METRO differ by a factor of two. The range in AHOD in Lake Washington during the 1990s was also of the same order (King County DNRP, 2003). Some of the year-to-year variation probably resulted from differences in climatic conditions and how they affect the time and intensity of stratification. AHOD determination is also sensitive to temperature and influx of DO from the epilimnion due to mixing, which can vary greatly from year-to-year. The range in AHODs in Lake Onondaga over a six-year period was 42 percent of the mean (Effler, 1996). This observed variation is greater than the magnitude of change that could be expected in Lake Sammamish from only a 35 percent reduction in TP loading. Thus, the failure to detect a reduction in AHOD following wastewater diversion, in contrast to the other indicators (TP, chl a, transparency and sediment P release rate), is not surprising.

### 3.4.4.2 AHOD Response to Development

From 1981 to 2011, AHODs ranged from 0.41 to 0.86, with a mean of 0.56 ± 0.11 g/m² per day (Table 3.8). This is only slightly higher than the 1981–1994 mean of 0.52 ± 0.10 g/m² per day. That difference since the 1980s–early-1990s is not significant, given the large year-to-year variability. Thus, AHOD in Lake Sammamish has not changed appreciably over the past four decades, despite the diversion of wastewater, in contrast to other lakes. The reason may be that algal biomass (and hence productivity), which is the source of organic matter and DO demand, has also changed little since diversion. The decreases in AHODs following wastewater diversion (or P removal) in other cases were associated with large decreases in TP algal biomass, such as Long Lake Spokane and Lake Washington (Table 3.7).
Table 3.7  Summer (from May through when DO reaches ≤1 mg/L) mean volume-weighted DO concentration, (± 1 SD), in the hypolimnion below 15 m, and AHOD, 1981-2011. A hypolimnetic mean depth of 8.17 m and volume of 98 x 10^6 m^3 was used.

<table>
<thead>
<tr>
<th>Year</th>
<th>End date</th>
<th># days</th>
<th>Hypolimnetic Mean Volume-weighted DO (May –DO ≥ 1mg/L)</th>
<th>SD</th>
<th>n=</th>
<th>AHOD g/m² per day</th>
</tr>
</thead>
<tbody>
<tr>
<td>1981</td>
<td>13-Aug</td>
<td>89</td>
<td>6.1</td>
<td>2.3</td>
<td>7</td>
<td>0.581</td>
</tr>
<tr>
<td>1982</td>
<td>10-Aug</td>
<td>87</td>
<td>6.5</td>
<td>2.0</td>
<td>7</td>
<td>0.480</td>
</tr>
<tr>
<td>1983</td>
<td>9-Aug</td>
<td>90</td>
<td>6.7</td>
<td>1.8</td>
<td>7</td>
<td>0.428</td>
</tr>
<tr>
<td>1984</td>
<td>No May or June data</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>1985</td>
<td>24-Jul</td>
<td>75</td>
<td>8.5</td>
<td>2.5</td>
<td>6</td>
<td>0.722</td>
</tr>
<tr>
<td>1986</td>
<td>6-Aug</td>
<td>89</td>
<td>7.2</td>
<td>1.8</td>
<td>7</td>
<td>0.412</td>
</tr>
<tr>
<td>1987</td>
<td>8-Jul</td>
<td>62</td>
<td>7.6</td>
<td>1.4</td>
<td>5</td>
<td>0.462</td>
</tr>
<tr>
<td>1988</td>
<td>17-Aug</td>
<td>104</td>
<td>7.0</td>
<td>2.6</td>
<td>8</td>
<td>0.544</td>
</tr>
<tr>
<td>1989</td>
<td>6-Sep</td>
<td>116</td>
<td>7.7</td>
<td>3.3</td>
<td>6</td>
<td>0.589</td>
</tr>
<tr>
<td>1990</td>
<td>11-Jul</td>
<td>63</td>
<td>7.7</td>
<td>1.7</td>
<td>4</td>
<td>0.518</td>
</tr>
<tr>
<td>1991</td>
<td>7-Aug</td>
<td>82</td>
<td>7.5</td>
<td>1.8</td>
<td>5</td>
<td>0.467</td>
</tr>
<tr>
<td>1992</td>
<td>7-Jul</td>
<td>62</td>
<td>5.5</td>
<td>1.4</td>
<td>4</td>
<td>0.442</td>
</tr>
<tr>
<td>1993</td>
<td>3-Aug</td>
<td>67</td>
<td>6.3</td>
<td>1.7</td>
<td>4</td>
<td>0.451</td>
</tr>
<tr>
<td>1994</td>
<td>6-Jul</td>
<td>63</td>
<td>8.4</td>
<td>2.2</td>
<td>4</td>
<td>0.692</td>
</tr>
<tr>
<td>1995</td>
<td>10-Jul</td>
<td>68</td>
<td>7.5</td>
<td>1.9</td>
<td>4</td>
<td>0.476</td>
</tr>
<tr>
<td>1996</td>
<td>24-Jul</td>
<td>76</td>
<td>7.5</td>
<td>2.5</td>
<td>6</td>
<td>0.728</td>
</tr>
<tr>
<td>1997</td>
<td>6-Aug</td>
<td>89</td>
<td>7.0</td>
<td>2.4</td>
<td>4</td>
<td>0.504</td>
</tr>
<tr>
<td>1998</td>
<td>6-Jul</td>
<td>61</td>
<td>6.1</td>
<td>1.9</td>
<td>3</td>
<td>0.510</td>
</tr>
<tr>
<td>1999</td>
<td>3-Aug</td>
<td>88</td>
<td>7.4</td>
<td>2.7</td>
<td>6</td>
<td>0.632</td>
</tr>
<tr>
<td>2000</td>
<td>8-Aug</td>
<td>96</td>
<td>6.4</td>
<td>2.5</td>
<td>6</td>
<td>0.548</td>
</tr>
<tr>
<td>2001</td>
<td>2-Jul</td>
<td>60</td>
<td>7.2</td>
<td>2.2</td>
<td>5</td>
<td>0.747</td>
</tr>
<tr>
<td>2002</td>
<td>1-Jul</td>
<td>54</td>
<td>7.6</td>
<td>2.3</td>
<td>5</td>
<td>0.862</td>
</tr>
<tr>
<td>2003</td>
<td>22-Jul</td>
<td>76</td>
<td>6.7</td>
<td>2.2</td>
<td>6</td>
<td>0.626</td>
</tr>
<tr>
<td>2004</td>
<td>20-Jul</td>
<td>77</td>
<td>6.5</td>
<td>2.0</td>
<td>6</td>
<td>0.538</td>
</tr>
<tr>
<td>2005</td>
<td>18-Jul</td>
<td>75</td>
<td>6.1</td>
<td>1.6</td>
<td>6</td>
<td>0.459</td>
</tr>
<tr>
<td>2006</td>
<td>24-Aug</td>
<td>111</td>
<td>7.1</td>
<td>2.4</td>
<td>8</td>
<td>0.482</td>
</tr>
<tr>
<td>2007</td>
<td>24-Jul</td>
<td>69</td>
<td>6.1</td>
<td>1.5</td>
<td>6</td>
<td>0.486</td>
</tr>
<tr>
<td>2008</td>
<td>12-Aug</td>
<td>89</td>
<td>6.8</td>
<td>2.2</td>
<td>7</td>
<td>0.548</td>
</tr>
<tr>
<td>2009</td>
<td>13-Aug</td>
<td>91</td>
<td>7.3</td>
<td>2.5</td>
<td>7</td>
<td>0.618</td>
</tr>
<tr>
<td>2010</td>
<td>14-Jul</td>
<td>63</td>
<td>6.7</td>
<td>1.9</td>
<td>5</td>
<td>0.599</td>
</tr>
<tr>
<td>2011</td>
<td>10-Aug</td>
<td>90</td>
<td>7.5</td>
<td>2.2</td>
<td>6</td>
<td>0.538</td>
</tr>
</tbody>
</table>
4.0 DISCUSSION

The important question posed by this assessment of long-term data is why mean annual whole-lake TP concentrations remained stable despite increased development? Based on predictions in 1994, TP loading was expected to increase to a slightly higher level (by 8 percent) than before wastewater diversion with full build-out and no controls in TP. Even with TP controls, the annual whole-lake and summer epilimnetic TP was expected to be around 24 and 17 μg/L, respectively.

The reason for no increase may be multifaceted: (1) decreasing hypolimnetic TP, due to reduced sediment P recycling as a result of continued recovery from wastewater, (2) watershed protections recommended in basin planning efforts and the lake management plan in the mid-1990s to reduce runoff P from land-use change, (3) long-term reduction in Issaquah Creek flow and increased water retention time, allowing greater settling loss of TP, and (4) particulate P increases in stormwater runoff were offset by decreases in SRP inputs in baseflow. These factors may have combined to produce no or little change in annual lake TP, despite an increase in development. These points will be discussed further.

4.1 Ongoing Recovery from Wastewater Diversion

Research in the 1970s–1980s recorded the lake’s slow recovery from wastewater diversion in 1968 (Welch et al., 1980). Hypolimnetic TP has continued to decrease significantly since the 1980s, with a weakly significant decline since 1995, reaching the lowest level in the past seven years (Figure 3.10). Overall the levels are now two thirds what they were pre-diversion. Hypolimnetic TP markedly affects annual whole-lake TP (Figure 3.14). The reduction in hypolimnetic TP mass since the 1970s has represented 64 percent of the decrease in annual, whole-lake TP mass. Without the decrease in hypolimnetic TP, annual TP would be 23 μg/L instead of 17.9 μg/L. Hypolimnetic TP has decreased because sediment-P release rate (SRR) has greatly declined since wastewater diversion (61 percent). Also, the decline in hypolimnetic SRP is consistent with reduced SRR as an explanation for most of the hypolimnetic TP decrease and annual whole-lake TP stability, since released sediment P is in soluble form.

Declining sediment-P release rate over several decades following wastewater diversion is not uncommon (Cooke et al., 2005). The P release rate decline may have been even slower in Lake Sammamish due to the observed efficient settling of TP following fall mixing resulting from high concentrations of iron, which coagulates and settles out P (Birch, 1976; King County, 1995).

4.2 Phosphorus and Development

Research in the 1970s–1980s also documented the threat of increasing P in runoff as the watershed became more developed (Welch et al., 1980). The future effects of increasing watershed development were assessed in the early 1990s as annual whole-lake total TP concentration had increased to around 20 μg/L and was predicted to increase further to
28 µg/L at build-out if no control measures were taken – nearly as high as before wastewater diversion (32 µg/L; Figure 3.9) (King County, 1995). Fortunately, annual mean TP has remained stable over the past two decades between 17–19 µg/L, well below the goal maximum of 22 µg/L set by King County in the mid-1990s, despite the increase in development and impervious area in the watershed.

Winter TP concentrations have also shown no trend over the past three decades. Winter is the high rainfall period, so lake TP during that time should have increased due to the increased runoff expected from more developed area in the watershed (Brett et al., 2005a). The rather stable TP during winter since the 1970s suggests that (1) the post-fall turnover, iron-P settling process has mitigated any increase in runoff TP that occurred, which may represent a resilience of this lake to development, and/or (2) there was no increase in inflow TP concentration during winter runoff despite a rapid increase of developed land as a result of stormwater management controls. Either or both of these effects may have been sufficient to offset increased inputs from external sources of TP.

Watershed protections instituted in the mid-1990s likely have contributed to the stable annual lake TP concentrations. Those protections involved retaining forest and requiring P retention from developed land as well as multiple specific short-term actions (Section 1.3.4). Runoff TP would be expected to be even greater now because impervious land cover has increased dramatically, unless runoff controls were effective. Yet the TP concentrations in Issaquah Creek, which is 70 percent of the inflows to the lake, have not changed significantly since 1972 (Table 3.1). Nor has the TP concentration increased significantly in any of the other tributaries monitored, except Tibbetts Creek. In fact there was a significant decrease in SRP concentrations in Issaquah, Idylwood, and Pine Lake creeks. Nevertheless, TP inputs from stormwater pipes, like those monitored in 1976–1977, may be greater than in natural streams. Runoff TP was much higher from the west-side stormwater conduits in 1976–1977 than in Issaquah Creek, based on intensive monitoring of three relatively large stormwater inputs, (Welch et al., 1980). Annual average flow-weighted west-side inflow TP was 109 µg/L compared to 74 µg/L in Issaquah Creek during 1973–1975 (Rock, 1974; Birch, 1976). There are many storm runoff conduits entering the lake besides natural streams. Moon (1973) identified 45 inputs and there have been others since then. For example, stormwater from the Timberline Ridge neighborhood was piped directly to the lake in the 1990s, after severe stream erosion (Booth and Henshaw, 2000).

Brett et al. (2005a) found that urban streams in the region had on average 95 percent higher TP and 122 percent higher SRP than most forested streams but that nutrient concentrations in Seattle regional urban streams were significantly less than previously reported for agricultural streams. Therefore one would expect an increase in P when catchments change from forested to urban, but not necessarily when changing from agricultural to urban. Agricultural land use was 171 hectares in 1994 on the east side and 872 hectares in the Issaquah basin, based on tax records (King County, 1995). By 2003 the reported animal units (AU) averaged over the entire Issaquah Creek basin were zero.
(Ellison and Brett, 2006)² Reduced agricultural land use in these drainages over the long-term may have offset the expected TP load from development. Stream TP has decreased elsewhere (Ontario) as agricultural land has been reduced by urbanization (Raney and Eimers, 2014).

Also, instituted watershed controls may have reduced TP contributions from developed land beyond that predicted, which assumed that the ratio of biological available phosphorus (similar to SRP) to TP would remain constant as reported for Issaquah Creek (Horner et al., 1987; King County, 1995). The SRP/TP declined significantly in Pine Lake, Issaquah, and Idylwood creeks while TP had not changed in any tributary except Tibbetts Creek (Table 3.1). The SRP/TP ratio in Issaquah Creek has significantly decreased since 1980, although the mean of 0.34 is within the range 0.3 to 0.4 observed in 1987 (Horner et al., 1987).

Not so surprising is that annual, whole-lake, alkalinity and conductivity substantially increased by 15 percent and 23 percent, respectively, over the past three decades, in response to increased watershed development. Unlike TP these constituents are conservative, meaning they tend to stay in solution and, unlike TP, a portion is not settled out and entrained in stormwater ponds or in lake bottom sediment. In-lake conductivity, which rose by 20 µmhos/cm (23 percent) between 1980 and 2011 (Figure 3.8), and it rose significantly in the lake while the increase in Issaquah Creek was only weakly significant (Table 3.1). That indicates the important effect of non-conservative constituents from the essentially built-out east and west sides, given that fraction of the watershed contributes only 30 percent of the lake’s inflow.

Conductivity had increased in east and west side creeks during 1995–2008 (Table 3.1, Figure 4.1). Ebright and Eden creeks show increases greater than 30 percent, and Lewis and Pine Lake creeks show weak increases of around 10 percent. Idylwood Creek conductivity increased significantly, but only 8 percent, during this 13-year period, which is not surprising since most of the development in that basin occurred before 1995 (Figure 1.2). Annual mean conductivity in Issaquah Creek did not increase significantly during this same time period (Figure 4.2). Much of that watershed has remained in forest (73 percent; Ellison and Brett, 2006). However, the smaller tributary creeks, except Tibbetts, do not show a development-related increase in TP inputs to the lake, despite the marked increase in conductivity.

There are several possible explanations for TP remaining stable in the face of increasing development, while conservative substances have increased. The increase in conservative alkalinity and conductivity is probably due to the dissolution of soil minerals through disturbance and corrosion of calcium–based structures in urbanized watershed (EPA, 2012). Although TP concentration also increases with development, it is non-conservative and subject to removal processes in stormwater treatment facilities and in-lake processes, while alkalinity and conductivity stay in solution as inflow water moves through the lake.

² Animal Units per km² were determined from a dairy farm dataset specifying location and number of animal units per farm compiled by the Washington State Department of Ecology (WA DOE, 2003).
Figure 4.1  Conductivity in Lake Sammamish tributaries; Lewis, Tibbetts, Pine Lake, Eden, Idylwood and Ebright creeks.
**4.3 Trophic State**

Phosphorus (P) concentration in the surface water stratum—the epilimnion—during spring and summer determines the lake’s production and biomass of plankton algae, and ultimately fish production. The resulting concentration of algae (and other dissolved and particulate substances) determines the amount of light attenuated and, thus, the depth of transparency. If the algae concentration is low, transparency will be high—greater transparency means fewer smaller particles allowing more blue color to penetrate and be reflected back to the observer. So P not only determines the lake’s productivity but also how it looks to lake recreational users. And, as mentioned earlier, the areal hypolimnetic DO deficit, or AHOD, is used as an index of trophic state, like TP, chl a and transparency.

**4.3.1 TP and Chl a**

The conventionally calculated indicators of trophic state are summer (June-September) mean concentrations of TP and chl a in the epilimnion, and transparency. Summer mean TP has been consistently around 12 µg/L and chl a around 3 µg/L since the 1970s. Based on these levels, Lake Sammamish is borderline mesotrophic–oligotrophic.
Chl $a$ has varied more than TP, with lower levels in the 1980s, and closer to 3.5 µg/L over the past 20 years. The long-term ratio of chl $a$:TP of 0.26 is slightly less than expected; the average for world lakes at those TP concentrations which is 0.35 (Welch and Jacoby, 2004). Chl $a$ concentrations are naturally low during summer, because the lake is thermally stratified, rainfall/runoff is usually low, and the epilimnion continually loses its TP, the determinant of chl $a$, through settling. Also, there is little resupply of TP from inflows during summer and the strong thermocline largely blocks transport of P from the hypolimnion until fall turnover. However, SRP concentrations tend to be higher in the dry periods when ground water dominates stream flows (Brett et al., 2005a). SRP concentrations in the whole lake annually and the hypolimnion during stratification, have declined but not in the epilimnion (Figure 3.15) and the SRP/TP ratio in Pine Lake, Issaquah, and Idylwood creeks have declined (Table 3.1), without a corresponding reduction in TP and chl $a$ in the lake.

The highest chl $a$ concentrations have occurred during the spring diatom bloom, usually in March/April, when chl $a$ has reached 30 µg/L in response to high TP residuals from fall turnover and winter rain/runoff. Average winter TP concentrations were usually double the summer epilimnetic levels. Subsidence of the spring bloom effectively strips much of the TP from the epilimnion, which is not replenished until late summer-early fall with entrainment from the hypolimnion as the thermocline sinks and/or fall rain/runoff occurs. A rainfall event and subsequent runoff may have helped cause the toxic cyanobacteria bloom that occurred in September 1997 (Johnston and Jacoby, 2003). Stormwater would tend to enter the lake’s epilimnion during May-October due to similar temperatures and densities. However, such events tend to be unusual during the relatively dry summers in the Pacific Northwest. That climate situation accounts for much of the observed long-term stability in summer TP and chl $a$. Changing climate (i.e., warming) may be contributing to the stability of summer TP by lengthening the stratified period, which increased by three weeks in Lake Washington between 1964 and 2000 (Winder and Schindler, 2004), and is predicted to increase 12–17 days in Lake Sammamish by 2040 (King County, 2013).

Transparency, the other indicator of trophic state, has also remained at a summer mean of about 5 m over the past three decades, with means slightly deeper than 5 m since 2000, and many values greater than 6 m. That depth of visibility rates Lake Sammamish as oligotrophic and of exceptional quality.

The relation between chl $a$ and transparency is nonlinear; very small changes in chl $a$ can produce large changes in transparency, over a range of relatively low chl $a$ (Figure 3.21). The values from Lake Sammamish generally adhere to the relationship of Carlson (1977), although transparency since 2000 was greater than expected from the chl $a$ levels. The points from Lake Washington before and after wastewater diversion more closely follow the Carlson curve. The greater than expected transparency in Lake Sammamish since 2000 may have been caused by longer water residence time and a longer stratified period that would allow greater settling of non-algal particle matter, and may be an effect of climate change. Although not statistically significant, the spring bloom maximum does appear to have been occurring earlier over the past 40 years. That suggests a climate change effect of earlier onset of stratification allowing enough available light for phytoplankton to increase.
Nevertheless, the nature of the chl \( \alpha \)-transparency relationship during summer indicates that the lake’s current high transparency is dependent on summer TP remaining at a low level. The vulnerability of the lake’s transparency to only small increases in TP, and hence chl \( \alpha \), should be carefully weighed when any plans are considered that loosen protections against runoff from development.

### 4.3.2 DO Demand

Oxygen is another indicator of trophic state, either as an areal rate of DO depletion in the hypolimnion (AHOD) or as a minimum concentration. The calculated AHODs averaged 0.56 ± 0.11 g/m\(^2\) per day over the past three decades and showed no trend. As noted in Section 4.2.2, that rate characterizes Lake Sammamish as eutrophic based on its DO depletion rate.

The year-to-year variability in AHOD averaged ± 20 percent, probably due to differences in productivity, temperature and the possible effect of climatic conditions on length and intensity of stratification (discussed in Section 3.5). Recognizing any trend toward lower rates of depletion as the lake recovered following wastewater diversion was difficult given the high year-to-year variability. Lake Washington did show significant recovery over three decades following wastewater diversion (Table 3.6), because wastewater loading was much higher than to Lake Sammamish, resulting in a reduction in TP loading that was double that for Lake Sammamish. Also, Lake Washington AHOD has recovered to near its pre-wastewater rate, while there is indication that Lake Sammamish has not (Table 3.6). That is probably due to the much larger decrease in algal biomass in Lake Washington following diversion (Figure 3.22) and also the ratio of epilimnion:hypolimnion. Lake Washington has a larger hypolimnion and therefore a much larger reserve of DO (Section 3.5.4). Also, the efficient settling and anoxic condition in Lake Sammamish may have resulted in a slower rate of organic matter loss than in Lake Washington. Given the slow decline in sediment-P release rate, AHOD may yet show a significant decrease, since its productivity (chl \( \alpha \)) is now about the same as that in Lake Washington.

An AHOD of 0.5-0.6 g/m\(^2\) per day and the relatively shallow hypolimnetic depth (8.17 m) results in most of that layer going anoxic (0 to -1 mg/L) by mid-summer. Further, the lake volume below about 13 meters is uninhabitable by salmonids from late July to October, because there is less than 4 mg/L DO (Berge 2009). Also, summer warming of the surface layer results in the water with temperature less than 17°C, necessary for growth, being confined to depths greater than 9–10 m. These conditions squeeze salmonid habitat to a layer only 3 to 4 m in thickness during a substantial part of the summer growing season (Figure 4.3). Such conditions are expected to reduce growth and possibly survival. Berge (2009) found that this DO-temperature squeeze reduced favorable habitat for salmonids by up to 90 percent during that critical period. Tracking tagged fish showed that both cutthroat and kokanee moved to this metalimnetic depth interval causing a predation risk to juvenile kokanee as well as a slower growth rate. So, while stability in summer epilimnetic waters results in less available summer TP for algal blooms, it has an adverse effect to fisheries by squeezing the summer habitat for salmonids and it is expected to worsen as the stratified period lengthens with climate change and lake warming (King County, 2013).
Annual average lake temperature has increased at about one fourth a degree C per decade similarly in lakes Sammamish and Washington. Modeling of the lake’s temperature to predict climate change effects indicates that stratification should begin 6 to 7 days sooner by 2040 and 16 to 23 days sooner by 2080 (King County, 2013). A longer period of stratification could negatively affect the DO/temperature habitat of cold water fish species. The earlier spring algal (chl a) bloom maximums, which depend on stratification as well as seasonally increasing solar radiation supports this.

![Figure 4.3](Image)

**Figure 4.3** Temperature and DO isopleths in Lake Sammamish during 2002 showing the depth restriction ("squeeze") for salmonids (from Berge, 2009).
5.0 CONCLUSIONS AND RECOMMENDATIONS

The nearly 50 years of observation, from before wastewater diversion and through increasing development in the watershed, suggests that Lake Sammamish water quality management strategies have been effective. First, the lake recovered (though delayed) from diversion of wastewater, which effectively eliminated one third of the external TP load in the late 1960s. Second, management actions instituted in the 1990s have contributed to maintaining the TP level in the lake despite a substantial increase in watershed development, particularly on the east side and the lower reach of the Issaquah Creek basin. Summer chl a and TP have actually changed little over the 50-year period, nor has the hypolimnetic oxygen demand (AHOD). The lake’s hypolimnion has regularly gone anoxic by mid-summer, but the rate of sediment-released P has declined—indicating less P-rich sediment, in contact with overlying water.

The Pacific Northwest climate generally has wet fall-winters and dry summers, so that most particulate P enters the lake via stormwater during the time period when algae are not growing as rapidly. By summer, stratification most of the P entering the lake with stormwater has settled from the water column. During baseflow conditions, TP inputs are primarily in soluble form as SRP when groundwater dominates the system. Tributaries to the lake have not shown a change in TP, but SRP in several—Issaquah, Pine Lake, and Idylwood creeks—has significantly decreased. Although not certain, it is possible that the decline in SRP may be due to an overall reduction in the total number of onsite septic systems and the connection of new homes and businesses to the regional wastewater system—transporting a substantial portion of P out of the watershed. The decline may also be from the conversion of agricultural (dairy farms) to single and multi-family residences over the last several decades.

Development in the area was projected in 1994 to substantially increase TP contribution to the lake (King County, 1995). Management actions and stormwater controls placed on new development have prevented some of the TP increase in the lake and/or its tributaries. On the other hand, while non-conservative TP has not increased, conductivity, which is conservative, has increased in the lake and inflow streams over the past 30 years (Figure 4.2 and 4.3), providing supporting evidence that stormwater treatment facilities may have retained much of the particulate P in runoff from new development. Such treatment would not have retained conservative substances that account for conductivity.

Two of the important reasons annual, whole-lake TP has remained stable, and well below the goal maximum, over the past three decades—reductions of hypolimnetic TP and SRP in inflow streams—may not exist in the future. The sediment P release rate has reached a very low rate (~ 2 mg/m$^2$ per day), about one-fifth of pre-diversion, so hypolimnetic TP will probably not decrease further given that late summer anoxia is likely to remain constant as shown by the AHOD. Stream SRP concentrations are not expected to continue to decrease, because most agricultural land has been urbanized, and sewering has eliminated most on-
site treatment systems in the urban areas. Moreover, forest cover within the urban growth boundary is likely to decline from current levels, especially in the Issaquah Creek basin. Tree cover is an important impediment to storm runoff and institutional rules for their protection on private property are weak at best. These prospects suggest heightened importance of the ongoing or enhanced TP removal from stormwater, erosion control best management practices during site development, forest retention requirements, and institutional protections for shoreline integrity along tributary streams and the lake itself. In addition, salmon recovery efforts in the watershed to improve habitat may also reduce P loads to surface waters. All of the provisions above should remain strongly in force if the lake’s water quality is to remain high for this and future generations to enjoy.

Based on this investigation, the following are recommended to ensure that Lake Sammamish water quality is maintained:

**Enforce Management Strategies**

- Continue to enforce strategies to control phosphorus input and instituted land-use zoning, which were put into place by multi-jurisdictional efforts that resulted in the Lake Sammamish Initiative in 1995.

**Monitoring**

- Continue to collect and evaluate lake water quality data. As this report demonstrates, a consistent long-term data set to evaluate trends over time is essential to evaluate the effectiveness of management strategies in response to the lake’s condition, and make adjustments as needed.

- Continue to monitor tributary water quality. A five-year hiatus (2009-2013) in water quality data in some of the Lake Sammamish tributaries occurred due to budgetary constraints. A consistent long-term data set to evaluate trends over time in loading of pollutants to the lake is essential to evaluate the effectiveness of management strategies.

- Continue to collect routine lake phytoplankton (algae) and zooplankton data. King County began collecting phytoplankton data in 2003 and has intermittently collected zooplankton data. Having a consistent long-term data set on the plankton community structure will enable future analyses of food web (includes impacts to kokanee and other species) interactions and their response to regional climate changes, which may have already affected the lake.

- Maintain and expand the flow-monitoring gaging network for tributaries to Lake Sammamish. Flow is currently measured in Issaquah Creek, Lewis Creek, and Laughing Jacobs Creek. Installing gages at other tributary mouths where water quality data are collected (e.g., Idylwood Creek, Eden Creek, Ebright Creek, Pine Lake Creek, and Tibbetts Creek) would allow comparison of flow and phosphorus (and other constituent) loading in the future.
Outreach

- Conduct outreach to jurisdictions and stakeholder groups within the Lake Sammamish watershed to maintain efforts to protect the lake and ensure that water quality improvements from diversion and other planning efforts are not overshadowed by future development.
6.0 REFERENCES


Berge, H.B. and B.V. Mavros. 2001. King County bull trout program, 2000 bull trout surveys. King County Department of Natural Resources, Seattle.


Buehler, Joanna. Founder and former president of Save Lake Sammamish. Personal communication email via Frank Lill. April 3, 2014.


Davis, J. I., and R.G. Swartz. 1981. An evaluation of fifteen lakes in King County. METRO, Water Quality Division, Seattle, WA


Effler, S.W. 1996. Limnological and Engineering analysis of a polluted urban lake. Springer-
Verlag, N.Y.


Isaac, G.W., R.I. Matsuda, and J.R. Walker. 1966. A limnological investigation of water quality conditions in Lake Sammamish. METRO, Seattle, WA.


King County Surface Water Management Division. September 1994. Issaquah Creek Basin and Nonpoint Action Plan. Report prepared by King County (lead agency) with Issaquah/East Lake Sammamish Watershed Committee participants; City of Issaquah, King Conservation District, King County, Muckleshoot Indian Tribe, Washington Department of Natural Resources, as well as the Basin Advisory Team. Funding provided in part by the Washington Department of Ecology.
Lake Sammamish Water Quality Response to Land Use Change

King County Surface Water Management Division. December 1994. East Lake Sammamish Basin and Nonpoint Action Plan. Report prepared by King County (lead agency) with Issaquah/East Lake Sammamish Watershed Management Committee participants; City of Issaquah, Washington Department of Natural Resources, King Conservation District, King County, Muckleshoot Indian Tribe, as well as the Basin Advisory Team. Funding provided in part by the Washington Department of Ecology.

King County. 1995. Lake Sammamish total phosphorus model. Prepared by W.W. Perkins for METRO-King County Surface Water Management.


King County Department of Natural Resources and Parks (DNRP). 2003. Lake Washington Existing Conditions Report. Prepared by Tetra Tech ISG, Inc and Parametrix, Inc. Seattle for King County Department of Natural Resources and Parks as part of the Sammamish/Washington Analysis and Modeling Program (SWAMP).


Larson, T.V. Professor, Department of Civil and Environmental Engineering. University of Washington. Seattle, WA. Personal communication with Gene Welch. April 7, 2014


Partners for a Clean Lake Sammamish. July 10, 1996. Lake Sammamish Initiative Report and Recommendations. Report produced by the Partners for a Clean Lake Sammamish with the support of the cities of Bellevue, Issaquah, and Redmond; and the King County Surface Water Management and Water Pollution control Divisions.


income. Ph.D. Dissertation, University of Washington, Seattle, WA.


