

Stave Falls Project Water Use Plan

Pelagic Monitor (Nutrient Load/Total Carbon Levels)

Implementation Year 10 (Meta-Analysis)

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Pelagic Productivity of Stave Lake Reservoir since Water Use Plan Implementation (1999-2014)

Study Period: 1999-2014

Final Report

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A Meta-Analysis of Monitoring Data and Assessment of WUP Environmental Management Questions (SFLMON #1)

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Executive Summary

During the Stave Lake Water Use Planning (WUP) process, several difficulties were encountered when trying to assess the impacts of facility operations on the pelagic productivity of Stave and Hayward reservoirs, including linkages to littoral productivity. As a result, “pelagic immunity” was assumed and the potential for impacts of alternative operating strategies on pelagic productivity was not considered in the WUP. The WUP Consultative Committee acknowledged there was uncertainty with this assessment. To address this uncertainty, the WUP CC developed management questions with corresponding impact hypotheses. Since a direct measurement of total annual production (i.e., flora and fauna growth as indicated by carbon assimilation rates) was beyond the budgetary scope of this monitoring program, the CC accepted the use of primary production (i.e., phytoplankton) and zooplankton growth as alternative index measures. This document reports on a series of meta-analyses of data collected during the 2000-2014 monitoring program. Issues with data quality and experimental design were identified and overcome at the cost of statistical power in the analyses. Nevertheless, the impact hypotheses were successfully tested, if not statistically, then by inference, reasoning or weight of evidence (see Table A).

Results determined that pelagic primary productivity varied both seasonally and across all years of the monitoring program, regardless of the metric used (i.e., Chl_a, ¹⁴C radio assay, pico-cyanobacteria bio-volume, or phytoplankton bio-volume). Seasonal variance in primary production followed a predictable cycle with peak values occurring in September. The high inter-annual variance in primary production did not appear to follow a particular trend over time. The primary drivers for this seasonal cycling pattern appeared to be the availability of light and to a lesser extent water temperature. Reservoir hydrology did not appear to be a factor in primary production, though the variables used to describe it (i.e., inflow discharge, water retention time, and reservoir elevation) were correlated with light and water temperature. It would appear that primary production for the most part is independent of Combo 6 reservoir operations.

Average annual zooplankton biomass was relatively stable over time, though there was considerable intra-annual variance. Like with primary production, this intra-annual variance was not correlated with reservoir operations. The data do suggest however, that the September drawdown operation may flush individuals out of Stave Lake reservoir. The impact of this on overall reservoir productivity remains uncertain and will require a separate study to resolve. However, the September drawdown operation is required to minimise downstream flooding.

Both the Stave Lake and Hayward reservoirs are severely nutrient poor and are considered ultra-oligotrophic systems. Given that pelagic productivity is for the most part independent of reservoir operations, it is considered unlikely that any kind of change to the Combo 6 operating alternative would lead to measurable changes in trophic status.

Table A. Summary of impact hypothesis outcomes arising from analysis of the 2000-2014 pelagic productivity monitoring study.

Impact Hypothesis	Description	Status	Rationale
H ₀₁	Average reservoir concentration of Total Phosphorus (TP), an indicator of general phosphorus availability, does not limit pelagic primary productivity.	Rejected	TP < 3 µg/L; Ultra-oligotrophic
H ₀₂	Relative to the availability of phosphorus as measured by the level of total dissolved phosphorus (TDP), the average reservoir concentration of nitrate (NO ₃) does not limit pelagic primary productivity.	Not Rejected	NO ₃ < 200 µg/L; Ultra-oligotrophic, but not as limiting as TP
H ₀₃	Water retention time (τ _w) is not altered by reservoir operations such that it significantly affects the level of TP as described by Vollenweider's (1975) phosphorus loading equations (referred to here as TP(τ _w)).	Not Rejected	TP independent of reservoir operations in Stave lake Reservoir, inconclusive in Hayward Reservoir
H ₀₄	Water temperature, and hence the thermal profile of the reservoir, is not significantly altered by reservoir operations.	Rejected	Thermal profile breakdown from September drawdown
H ₀₅	Changes in TP as a result of inter annual differences in reservoir hydrology (i.e., TP(τ _w)) are not sufficient to create a detectable change in pelagic algae biomass as measured by levels of chlorophyll a (Chl _a).	Not Rejected	No statistically significant relationships were detected in the data
H ₀₆	Independent estimates of algae biomass based on TP(τ _w) and Secchi disk transparency (SD) prediction equations are statistically similar, suggesting that neither non-algal turbidity, nor intensive zooplankton grazing, are significant factors that influence standing crop of pelagic phytoplankton (Carlson 1980, cited in Wetzel 2001)	Not Rejected	Lack of available nutrients is a much stronger limitation to growth than either non-algae turbidity or zooplankton grazing
H ₀₇	The effect of non-algal turbidity on pelagic algae biomass, as indicated by the difference in independent predictions of TSI(Chl _a), TSI(TP) and TSI(SD) (Carlson 1980, cited in Wetzel 2001), does not change as a function of reservoir operation (TSI = Trophic Status Index).	Not Rejected	Non-algal turbidity was for the most part unrelated to reservoir operations
H ₀₈	The ratio of ultra-phytoplankton (< 20 µm in size) to micro-phytoplankton (20-200 µm in size) abundance is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.	Rejected	Change in average phytoplankton size detected; but uncertain that the change is linked to reservoir operations
H ₀₉	The size distribution of the pelagic zooplankton population (an indicator of fish food bioavailability as larger organisms tend to be preferred over small ones) is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.	Not Rejected	No temporal change in zooplankton biomass detected, but time series too short to be definitive.
H ₀₁₀	Primary production, as measured through ¹⁴ C inoculation, is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.	Not Rejected	Hypothesis could not be directly tested, but weight of evidence indicate not rejected

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1 Introduction

During the Stave Lake Water Use Planning (WUP) process, several difficulties were encountered when trying to assess the impacts of facility operations on the pelagic productivity of Stave and Hayward reservoirs, including linkages to littoral productivity. These difficulties stemmed from the lack of productivity data specific to these reservoirs and resources needed to fill these data gaps. A simple assessment using general models of lake-ecosystem function, general knowledge of ecosystem impacts arising from impoundment practices, published data from other reservoirs throughout North America, and limited available reservoir-specific data indicated that pelagic productivity would remain largely unaffected by changes in the reservoir operations being investigated. As a result, “pelagic immunity” was assumed and the potential for impacts of alternative operating strategies on pelagic productivity was not included in the WUP, however, the WUP Consultative Committee acknowledged there was uncertainty within their assessment.

The first of these uncertainties was implication that if pelagic immunity assumption was indeed valid, any changes in overall reservoir productivity would solely be the result of changes in littoral productivity. Another uncertainty was the method used to estimate carbon assimilation (^{14}C ; an indicator of reservoir production) and the underlying assumption that it would serve as a reasonable indicator of fish production potential. Annual ^{14}C assimilation rate was estimated from a linear regression equation developed from lake data collected throughout BC (J. Stockner Pers. Comm.). The data set did not include storage reservoirs and its application to a reservoir setting was considered suspect, including its use as an overall indicator of reservoir productivity. Also contributing to the uncertainty was the large error associated with the predictions made with this equation (Failing 1999). Finally, it was generally acknowledged that the assumed link between carbon production and fish production was a rather tenuous one and fraught with uncertainty. Its use in the WUP decision making process was driven primarily by the absence of any other kind of production-based information.

In recognition of these uncertainties, the WUP CC recommended that they be addressed in a comprehensive monitoring program designed to improve the decision-making process in future WUPs. WUP CC acceptance of the “Combo 6” alternative (Table 1) as the preferred operating strategy for the Stave Lake generation facility was conditional on the design and implementation of such a monitoring program (Failing, 1999). Associated with the monitoring program were a series of management questions and impact hypothesis that were to be addressed through the monitoring program (BC Hydro, 2004).

This document reports on a series of meta-analyses of data collected during the 2000-2014 monitoring program and attempts to resolve at least some of the uncertainties described above. This includes addressing the management questions posed by the WUP CC, and testing the corresponding impact hypotheses. Because a direct measurement of total annual production (i.e., direct measure of flora and fauna growth as indicated by ^{14}C assimilation rates) was beyond the budgetary scope of this monitoring program, the CC accepted the use of primary production (the annual production of phytoplankton) as an alternative index measure. As a result, the monitoring results presented here are focused primarily on this trophic level of production, with some analysis on zooplankton growth.

Table 1. Summary of operational constraints that form the Combo 6 operating alternative recommended by the WUP CC (BC Hydro 2003).

Constraint	Implementation Dates	Description
1	Oct 15 - Nov 30	Minimum tailwater elevation 1.8m. Initially 1.7m minimum tailwater elevation rest of year, but later changed to 1.8m year round.
2	Oct 15 - Nov 30	Limited Fall Block Loading. Ruskin output is subject to weekly block loading where generation has a set output for a 7-day period. When the discharge from Ruskin is < 100 m ³ /s, the discharge must remain constant for a minimum of 7 days after each change in discharge and each new block load on the plant must be greater than the previous block load. When the weekly block load flow > 100 m ³ /s, peaking operations at Ruskin above 100 m ³ /s is permitted.
3	Feb 15 - May 15	Limited Spring Block Loading. Ruskin is subject to daily block loading, defined as a maximum of one plant load change each day. When the daily block load flow > 100 m ³ /s, peaking at Ruskin is permitted provided a minimum flow of 100 m ³ /s is maintained for the day.
4	Oct 15 - Nov 30, Feb 15 - May 15	Flow Ramping the Spring and Fall Block Loading. The rate of flow reduction from Ruskin, when discharge is < 100 m ³ /s, will be less than 35 m ³ /s at intervals > 10 minutes. When discharge is greater than 100 m ³ /s, rate of flow change will be < 113 m ³ /s at intervals > 30 minutes.
5	Jan 1 - Mar 31	Archaeological Drawdown. Stave Lake Reservoir drawdown below the licensed minimum reservoir level of 73.0 m require for a minimum of 6 week, one out of 3 years on average. The lowest elevation above which access will be provided is 72 m.
6	May 15 - Sep 7	Recreation Season Targets on Stave Lake Reservoir. Preferred elevation of Stave Lake Reservoir for recreational activities is between 80.0 and 81.5 m. During this period, the level of Stave Lake Reservoir will be targeted at 76 m or higher, and will be targeted between 80.0 and 81.5 m for a minimum of 53 days.
7	Oct 15 - Nov 30, Feb 15 - May 15	Hayward Reservoir Operations. During the spring and the fall block load periods, the normal minimum operating level at Hayward Reservoir will be 39.5 m. At other times, the normal minimum operating level at Hayward Lake Reservoir will be 41.08 m.

1.1 Management Questions

The consultative committee identified four key management questions pertaining to the pelagic productivity of Stave and Hayward reservoirs (BC Hydro, 2004):

- a) What is the current level of pelagic productivity in each reservoir, and how does it vary seasonally and annually as a result of climatic, physical and biological processes, including the effect of reservoir fluctuation?
- b) If changes in pelagic productivity are detected through time, can they be attributed to changes in reservoir operations as stipulated in the WUP, or are they the result of change to some other environmental factor?
- c) To what extent would reservoir operations have to change to 1) illicit a pelagic productivity response; and 2) improve or worsen the current pelagic state of productivity?
- d) Given the answers to the management questions above, to what extent does the Combo 6 operating alternative improve reservoir productivity in pelagic waters, and what can be done to make improvements, whether they be operations based or not.

1.2 Impact Hypotheses

A total of 10 hypotheses were identified for the present monitor. Collectively, they form an impact hypothesis model that explores the interrelationship of various environmental factors on productivity, as well as inter-trophic interactions. The impact hypotheses, expressed here as null hypotheses, were tested separately for each reservoir and related primarily to changes in primary productivity (BC Hydro, 2004).

- H₀1: Average reservoir concentration of Total Phosphorus (TP), an indicator of general phosphorus availability, does not limit pelagic primary productivity.
- H₀2: Relative to the availability of phosphorus as measured by the level of total dissolved phosphorus (TDP), the average reservoir concentration of nitrate (NO₃) does not limit pelagic primary productivity. Nitrate is the dominant form of nitrogen that is directly bio-available to algae and is indicative of the general availability of nitrogen to pelagic organisms.
- H₀3: Water retention time (τ_w) is not altered by reservoir operations such that it significantly affects the level of TP as described by Vollenweider's (1975) phosphorus loading equations (referred to here as TP(τ_w)).
- H₀4: Water temperature, and hence the thermal profile of the reservoir, is not significantly altered by reservoir operations.
- H₀5: Changes in TP as a result of inter annual differences in reservoir hydrology (i.e., TP(τ_w)) are not sufficient to create a detectable change in pelagic algae biomass as measured by levels of chlorophyll *a* (Chl_{*a*}). [This hypothesis can only be tested if H₀3 is rejected]
- H₀6: Independent estimates of algae biomass based on TP(τ_w) and Sechi disk transparency (SD) prediction equations are statistically similar, suggesting that neither non-algal turbidity, nor intensive zooplankton grazing, are significant factors that influence standing crop of pelagic phytoplankton (Carlson 1980, cited in Wetzel 2001).
- H₀7: The effect of non-algal turbidity on pelagic algae biomass, as indicated by the difference in independent predictions of Chl_{*a*} by TP(τ_w) and SD (Carlson 1980, cited in Wetzel 2001), does not change as a function of reservoir operation.
- H₀8: The ratio of ultra-phytoplankton (< 20 μ m in size) to micro-phytoplankton (20-200 μ m in size) abundance is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.
- H₀9: The size distribution of the pelagic zooplankton population (an indicator of fish food bioavailability as larger organisms tend to be preferred over small ones) is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.
- H₀10: Primary production, as measured through ¹⁴C inoculation, is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.

1.3 Objective and Scope

In the absence of reliable information on the effect of facility operations on Stave and Hayward reservoir productivity, it was assumed that pelagic productivity would remain unchanged over the

spectrum of feasible reservoir operating strategies. However, the WUP CC recognised that there was considerable uncertainty in this assumption, but had no information with which to form other, more probable outcomes. The overarching objective of this monitoring program was to (BC Hydro, 2004):

- a) Test the validity of this assumption of no operational impact, and confirm that pelagic conditions have not worsened with the new Combo 6 operating strategy.
- b) Provide the information necessary to promote a better understanding of the pathways by which operational changes can affect primary productivity, and in turn provide better predictions of operational impacts for future WUP reviews.
- c) Develop a better linkage between the effect of reservoir operations on primary productivity and the potential for fish production.

More specifically, the objective of this monitor was to collect the data necessary to test the impact hypotheses outlined above and hence, address the management questions. The following aspects defined the scope of the study:

- a) The study area consisted of Stave Lake and Hayward Lake Reservoirs.
- b) Data was collected at two sites; one on Stave Lake reservoir, and one on Hayward Lake reservoir.
- c) The program was carried out in two phases; an initial 3-year high intensity sampling program (Beer 2004), and a subsequent longer term, base level sampling program.
- d) The monitoring program was to last for at least 10 years, or until the next WUP review period.
- e) The monitoring program was focused primarily on variables associated with measures of pelagic primary productivity, a component of reservoir productivity that was assumed to be a reasonable indicator of overall productivity.

2 Methods

2.1 Study Area

The Stave Falls power project consists of three reservoirs, four dams and three powerhouses (Figure 1) and is located roughly 15 km west of Maple Ridge BC. The uppermost reservoir, Alouette Lake Reservoir, feeds into Stave Lake Reservoir via a diversion tunnel located at its northern end. Outflows from Alouette Lake Reservoir are directed either through a powerhouse or a bypass tunnel that can be used in conjunction with the powerhouse to help regulate Alouette Lake Reservoir elevations. Use of the powerhouse however, has been infrequent in recent years due to reliability issues, thus the majority of outflows have been through the bypass tunnel. Stave Lake Reservoir can be viewed as being comprised of two basins; an upper basin that includes a deep-water lake (Stave Lake) fed by the outflows of Alouette Reservoir and the majority of the large tributaries in the watershed (including upper Stave River); and a lower basin that consists mainly of flooded river channel and is therefore much shallower than the upper basin. Both Alouette Lake and Stave Lake reservoirs have the capability to store and release water for power generation and downstream flood control. Hence, both have highly variable water levels that can change seasonally in response to local inflows, power generation demands, and recreational requirements. The lower basin leads to Stave Falls Dam, which releases

water to Hayward reservoir. Like the lower basin of Stave Lake Reservoir, Hayward Reservoir is largely a flooded river channel with little capacity for storage. However, because of its length, it can be fairly deep at its downstream end and for the purposes of this study, was considered to be 'lake-like'.

This monitoring program was focussed primarily on the hydrology and productivity of Stave Lake and Hayward reservoirs. The hydrology of Alouette Lake Reservoir was taken as another source of inflow to the Stave Lake Reservoir, though its water quality may be unique due to its reservoir ecology and long running fertilization program (Harris et al. 2010). Stave Lake Reservoir was considered the treatment system with its broad range of water level fluctuations - sometimes exceeding 8 m. Hayward Reservoir was deemed the control system with relatively stable (variance < 1.5m) with periodic but short-duration drawdowns to carry out maintenance activity (generally every 2 years). In 2006, maximum Hayward Reservoir elevation was dropped from roughly 42.7 m to 41.3 m to mitigate seismic concerns related to Ruskin Dam's aging infrastructure resulting in a more stable reservoir elevation (variance < 0.5m) outside the typical summer time drawdown periods. In 2012, there were increases in Hayward Reservoir draw down depth and duration as a result of work related to Ruskin Dam and Powerhouse upgrades.

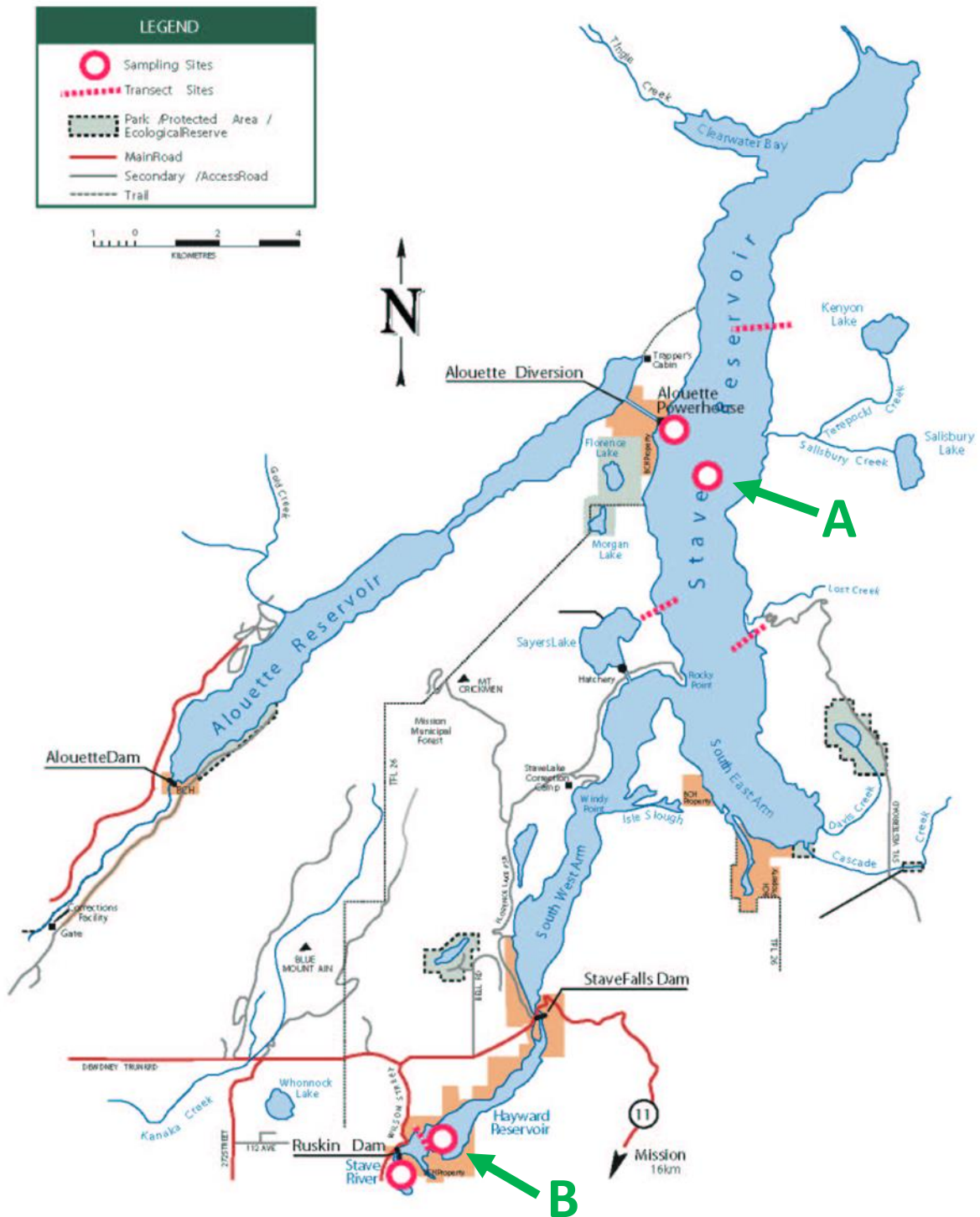


Figure 1. Map of Stave Lake and Hayward Lake reservoirs showing monitor sampling locations. Red circles identify the location of water quality sampling sites while the dashed red line denotes the location of periphyton sampling transects. Only the open water sites A and B were used in the present study.

2.2 General Approach

A single site was established in the open waters of each reservoir where multiple water quality and biological parameters were sampled every 4 to 8 weeks depending on prevailing sampling conditions and logistical constraints. Sampling was started in July 2000 and was most intensive in the first 3 years of the monitoring program (Stockner and Beer 2004; Beer 2004). Preliminary results lead to refinement in sampling methodology and changes to the way biological parameters were characterised for longer term monitoring. As a result, not all parameters were sampled in all years. A list of parameters sampled is provided in Table 2. The monitoring program ended November 2014.

Table 2. List of water quality and biological parameters sampled during the 2000 to 2014 monitoring period.

Water Quality Sampling	Biological Sampling
Reservoir Hydrology	Primary Production
Inflow Discharge (Q , m^3/s)	General
Water surface elevation (WSE, m)	Chlorophyll a (Chl_a , $\mu g/L$)
Water Retention Time (τ_w , days)	^{14}C Assimilation Rate ($mg/m^3/h$)
Light	Bacteria
Light extinction coefficient (k , m^{-1})	Pico Cyano-Bacteria (mm^3/L)
Secchi Depth (SD, m)	Heterotrophic Bacteria (mm^3/L)
Water Temperature	Phytoplankton
Average Epilimnion Temperature (T_{Epi} , $^{\circ}C$)	Edible Nano Phytoplankton (mm^3/L)
Epilimnion Depth where $dT/dz > 1^{\circ}C$ (D_{Epi} , m)	Edible Pico Phytoplankton (mm^3/L)
Metalimnion Depth where $dT/dz = Max$ (D_{Meta} , m)	Edible Macro Phytoplankton (mm^3/L)
Nutrients	Inedible Macro Phytoplankton (mm^3/L)
Total Dissolved Phosphorus (TDP, $\mu g/L$)	In-ed./edible Macro Phytoplankton (mm^3/L)
Total Phosphorus (TP, $\mu g/L$)	Zooplankton
Nitrate (NO_3 , $\mu g/L$)	Zooplankton Abundance (count/L)
	Zooplankton Biomass (mg/L)

All data were arranged in a time series and assigned an ordinal date value, year and sampling location (Appendix 3). The ordinal date values were used to explore seasonal trends in the data, while the year and sampling location attributes were used to explore annual and spatial trends through post-hoc, two-way analysis of variance (ANOVA). Relationships between variables were explored using multiple correlation and regression techniques, including forward stepwise regression techniques where required. Details on how the data were collected and later processed for analysis are provided in the sections that follow. It should be noted that all *in situ* sampling in both reservoirs was carried out on the same day.

2.3 Reservoir Hydrology

All inflow discharge and water surface elevation (WSE) data for each reservoir were provided by BC Hydro. Daily average inflow to Stave Lake reservoir included the outflow discharge from the Alouette Powerhouse facility (combined powerhouse and spillway discharges). Reservoir storage curves were also provided by BC Hydro to determine the proportion of total inflow stored versus released and to calculate daily water retention times (τ_w) to capture the joint effects of daily varying inflows and WSEs. It should be noted that the storage curves only provide information on active or live storage (i.e., the proportion of total reservoir volume that lies within the limits of maximum and minimum reservoir operating limits) and not total storage. As a result, all τ_w calculations should be treated as indicator values rather than absolute measurements.

2.4 Water Quality Sampling

2.4.1 Light

Photosynthetically active radiation (PAR; $\mu\text{mol}/\text{m}^2/\text{s}$) was measured at 1-metre intervals to a depth where it is diminished to less than 1% of surface values (the compensation depth). A LiCor Li-250A light meter equipped with a Li-192SA submersible quantum sensor was used to collect the PAR data at both the water surface and underwater. A small weight was attached to the sensor to keep it vertical while taking deep water measurements, and care was taken to ensure that the boat did not cast a shadow over the sensor. Each measurement was taken as a 15 second average, with a typical accuracy of $\pm 0.6\%$ (LiCor 2006). A single light profile was collected at each sampling site in the Stave Lake and Hayward reservoirs (Figure 1). At the same time, Secchi depth (SD) readings were also taken. On the shaded side of the boat, a 20-cm dia. Secchi disk was lowered until it disappeared from view and the depth was noted. The Secchi disk was then raised until it came into view again, and this depth also noted. The two depth measurements were averaged to obtain the SD reading.

Vertical profiles of PAR were log transformed and then regressed against water depth to estimate the light extinction coefficient 'k' (slope of the regression) and PAR intensity at the water surface (intercept of the regression). Similarly, SD data were used to confirm accuracy of the light extinction coefficients. All analyses, which included use of regression and ANOVA techniques to assess temporal and spatial trends, were focused primarily on the light extinction coefficient data. In cases where normality or homoscedasticity assumptions could not be met, the coefficients were transformed to 1% light compensation depths (m) by solving each regression equation for water depth assuming a 99% loss in intensity from measured surface PAR readings.

2.4.2 Water Temperature

Water temperature was measured either by an integrated temperature sensor on the Li-250A light meter or an Oxyguard Handy Beta to the maximum depth of the probe; approximately 25 m. In both cases, the temperature sensor was kept vertical using a small weight and maintaining constant boat position under windy conditions. The Li-250A light meter was able to measure temperature to the nearest $\pm 0.1^\circ\text{C}$ while accuracy of the Oxyguard Handy Beta unit was closer to $\pm 0.2^\circ\text{C}$. The data were collected at 1 m intervals and the probe was typically allowed to equilibrate to ambient conditions for a minimum 5 minutes before taking a reading. The temperature depth profiles were characterised by three statistical attributes: 1) average epilimnion temperature (T_{Epi}) measured at depths of 1, 3 and 5 m; 2) if present, the depth limit of the epilimnion defined as the first 1 m depth interval encountered with a

water temperature difference > 1 °C change; and 3) if present, the depth of the metalimnion defined as the depth interval with the maximum change in water temperature of at least 1°C.

2.4.3 Nutrients

Water quality samples (~ 500 ml) at each of the sample sites on Stave Lake and Hayward reservoirs were collected by a vertical, non-metallic Van Dorn sampler at 1, 3 and 5 m depths below the water surface. Samples were combined into a large (2 L) dark bottle to serve as a single representative sample for the site. All sub-samples used to test for water quality and biological parameters (Table 2) were drawn from this mixed epilimnetic sample.

For TP sample collection, test tubes and caps (one per site) were first rinsed with the sampled water before being filled, capped and labelled. Neither the mouth of the bottle or the inside of the cap was touched to avoid contamination. The samples were placed in a cooler on ice and then refrigerated until analysed. Two sample bottles of double distilled water (DDW) were prepared as blanks for comparison purposes.

For TDP and NO₃ sample collection, sub-samples were filtered in the field using a 47-mm filtering manifold equipped with an ashed GF/F filter. Prior to sampling, the filter was rinsed with 180 ml of DDW followed by 180 ml of the sampled epilimnetic water. Plastic 120 ml sample bottles were rinsed by filtering 60 ml of the sampled epilimnetic water into each bottle. All filtrate to this point was discarded. The rinsed sample bottles were then filled with filtered epilimnetic sample water, capped and immediately placed in a cooler for transport. TDP samples were refrigerated until processing while the NO₃ samples were frozen. Two sample bottles of DDW were prepared as blanks for comparison purposes.

All samples were immediately sent to SPAChemtest (DFO Laboratory at Cultus Lake, BC) for chemical analysis. Due to the closure of SPAChemtest in 2012, the 2013 and 2014 samples were analysed by the ALS Laboratory in Burnaby, BC. Details on the laboratory procedures used are provided in Appendix 2.

2.4.4 Non-Algal Turbidity Index

A comparison of trophic status index (TSI) values calculated separately from SD and Chl_a values is used to derive a non-algal turbidity index value that relates to the amount of dissolved organic matter and/or fine sediment turbidity in a lake system (Carson 1980). The TSI were calculated as follows (Carson 1980):

$$\begin{aligned} \text{TSI}(\text{SD}) &= 60 - 14.41 \cdot \ln(\text{SD}) \\ \text{TSI}(\text{Chl}_a) &= 9.81 \cdot \ln(\text{Chl}_a) + 30.6 \end{aligned}$$

The non-algal turbidity index was calculated as the difference between TSI(SD) and TSI(Chl_a). This made for a more intuitive interpretation of the index where a larger difference in TSI indicated a greater magnitude of non-algal turbidity. Negative differences (i.e., TSI(Chl_a) > TSI(SD)) were interpreted as being an indicator of zooplankton grazing.

TSI values using TP were also calculated, but was only used to assess TP limitations to algal growth than assess the effects of non-algal turbidity (Carson 1992). TSI(TP) was calculated as follows:

$$\text{TSI}(\text{TP}) = 14.42 \cdot \ln(\text{TP}) + 4.15$$

2.5 Primary Production

2.5.1 General

Primary production was measured in terms of Chl_a concentration and ^{14}C assimilation rate. To determine Chl_a concentration, dark 120 ml sample jars were filled with mixed epilimnetic water, capped, labelled and placed in a cooler on ice for transport. All samples were immediately sent to SPAChemtest or ALS for pigment extraction and quantification. Details on the laboratory procedures used are provided in Appendix 2.

^{14}C assimilation rate was determined by ^{14}C radio assay of water samples collected by Van Dorn sampler at 1, 3 and 5 m depths. Water samples at each depth were decanted into two separate 200 ml graduated cylinders and then poured into separate 250 ml Biological Oxygen Demand (BOD) bottles; one of clear glass, the other dark. The bottles were then inoculated with 1 ml of $\text{NaH}^{14}\text{CO}_3$, the equivalent of $5 \mu\text{Ci } ^{14}\text{C}$, and allowed to incubate *in situ* at the depth of water collection for a 2 to 4-hour period. All incubations were carried out in the hours between 11 am and 4 pm. At the end of the incubation period, the samples were removed from the incubation site and placed in a cooler on ice to slow the incubation process.

In the laboratory, samples were filtered in succession into three size classes (20, 2 and $0.2 \mu\text{m}$) using a 47mm polycarbonate filter (pore size corresponding to size class) with a 10 cm·Hg vacuum differential to avoid cell damage in the filtrate (Joint and Pomroy 1983). This was done in a semi dark environment to avoid further photosynthesis. Each filter was placed into a 7-ml scintillation vial with the additions of 200 μL of 0.5N HCl to eliminate unincorporated inorganic $\text{NaH}^{14}\text{CO}_3$ and 5 ml of Ecolite scintillation cocktail. The vial was capped and stored in the dark for another 24 hours before being radio-assayed at UBC Risk Management. All radio assays were completed using a Beckman LS1801 scintillation counter; operated in an external standard mode to correct for quenching (Pieters et al. 2000).

Primary production was estimated by converting the scintillation counts (disintegrations per minute or DPM) in each of the light and dark BOD bottles into a measure of ^{14}C assimilation rate ($\text{mg}/\text{m}^3/\text{hr}$) using the following formula:

$$C = B * A_t * (V_i / V_a) * 1.064 * 1000 / (S * T) \quad (\text{Eq. 1})$$

where;

- B = Average DPM for ^{14}C incubated sample (in either clear or dark bottles)
- A_t = Total inorganic ^{12}C ($\text{mg}\cdot\text{L}^{-1}$) in inoculant
- V_i = Volume of incubation bottle (100 ml)
- V_a = Volume of acidified aliquot (30 - 100 ml)
- 1.064 = Isotropic preference factor
- 1000 = Factor to convert $\text{mg}\cdot\text{L}^{-1}$ to $\text{mg}\cdot\text{m}^{-3}$
- S = Average DPM of reference vials
- T = Incubation time (hr)

^{14}C assimilation rate from the dark bottle (a measure of the sample's non-photosynthetic ^{14}C assimilation rate) was subtracted from that of the light bottle (a measure of the sample's total ^{14}C assimilation rate) to obtain the sample's photosynthetic ^{14}C assimilation rate (Neilson 1960).

When using Eq.1, total inorganic ^{12}C in the inoculant (A_t) was assumed to be $2.54 \text{ mg}\cdot\text{L}^{-1}$ for the entire experimental period based on the work of Bruce and Beer (2014). To account for the specific activity of the ^{14}C stock used for inoculation, a standard radio-assay was performed on a sample of the $\text{NaH}^{14}\text{CO}_3$ used in the study to determine the total radioactivity (S) that was added to each of the BOD bottles. $100 \mu\text{L}$ of the $\text{NaH}^{14}\text{CO}_3$ solution was added to a scintillation vial containing 5 ml Ecolite scintillation cocktail and radio-assayed using the same scintillation counter used to measure radioactivity in the pelagic samples.

All samples collected at the 1, 3 and 5 m depths were averaged to obtain a depth-integrated estimate of ^{14}C assimilation for each sampling date. Samples of multiple size classes were summed to obtain estimates of ^{14}C assimilation that were comparable to those that were not size fractionated. Size fractionated samples were also analysed separately to assess relative contributions to total reservoir productivity.

2.5.2 Bacteria

To estimate the bio-volume of pico-cyano bacteria and heterotrophic bacteria, a 50 to 100 ml subsample of the mixed epilimnetic water was poured into a 200-ml sample jar rinsed once with ethanol and three times with DDW. Four to six drops of dilute glutaraldehyde (25%) was then added as a preservative. Samples were generally processed within a few days to prevent degradation of the chlorophyll pigments that allow for autofluorescence during the counting procedure (Maclsaac and Stockner 1993). Samples were frozen if the time to processing was longer.

In the laboratory, the bio-volume of pico-cyano bacteria was obtained by filtering 15 ml of the water subsample through a $0.2\mu\text{m}$ black polycarbonate filter with a 10 cm-Hg vacuum differential. The filtrate was placed in a covered counting chamber on a microscope slide and mounted on an epifluorescent microscope at 1000x magnification. Bacterial counts were completed in 8 to 32 fields (depending on bacteria density) to determine a number of cells per liter and then multiplied by an average cell volume of $5 \times 10^{-6} \text{ mm}^3$ to obtain a bio-volume estimate in units of mm^3/L .

The same procedure was used to estimate the bio-volume of heterotrophic bacteria. A 5 ml water subsample was filtered through a $0.2\mu\text{m}$ black polycarbonate filter and the filtrate treated with DAPI dye to target heterotrophic bacteria for counting (Maclsaac and Stockner 1993).

2.5.3 Phytoplankton

To measure phytoplankton abundance and bio-volume, a 50 to 100 ml subsample of the mixed epilimnetic water was poured into a 200-ml sample jar rinsed once with ethanol and three times with DDW. Once filled with the subsample of water, each phytoplankton sample was preserved by adding 10 ml Lugol's iodine preservative (iodine + 10% acetic acid) and refrigerated until analysis. All samples were enumerated using the Utermohl (1958) method for micro-phytoplankton to the nearest species taxon level. Prior to quantitative enumeration, samples were gently shaken for 60 seconds, poured into 25 mL settling chambers and allowed to settle for a minimum of 24 hours. Counts were done using a Carl Zeiss inverted phase-contrast plankton microscope. Counting followed a 2-step process:

- Random fields (5 -10) were examined at 250X magnification (16X objective) and large micro-phytoplankton ($20\text{-}200\mu\text{m}$; e.g., diatoms, dinoflagellates, filamentous blue-greens) were enumerated, and

- All cells within a random transect (ranging from 10 to 15mm) were counted at 1560X magnification (100X objective). This high magnification permitted quantitative enumeration of most, autotrophic picoplankton cells (0.2-2.0µm) [Class Cyanophyceae], and small auto-, mixo- and heterotrophic nano-flagellates (2.0-20.0µm) [Classes Chrysophyceae and Cryptophyceae].

In total, random transects are repeated until 250 to 300 cells were enumerated in each sample to assure statistical accuracy (Lund et al. 1958). The compendium of Canter-Lund & Lund (1995) was used as the taxonomic reference. Counts are reported as abundance (cell/ml) and converted to estimates of bio-volume (mm³/L) using species specific conversion factors (provided by J. Stockner, EcoLogic, Pers. Comm).

2.6 Zooplankton

Zooplankton were sampled as a vertical tow (approximately 0.5 m/s) starting at 20 m depth in Stave and 15 m in Hayward with a 30-cm diameter, 90 cm long, 80 µm mesh plankton net. Samples were preserved at the lab using a small aliquot of sugar prior to the addition of formalin (37% formaldehyde solution) for a final concentration of approximately 10% formalin. Techniques used to subsample, count, and measure zooplankton were adopted from Wetzel and Likens (2000), which used length–weight relationships developed by McCauley (1984) and Koenings et al. (1987).

Preserved samples were transported to AMA Group for counting. Samples were filtered through a 0.45 µm mesh net and rinsed with settled tap water (approximately 12 hours). The filtered sample was transferred into a beaker and re-suspended using 60 to 100 ml of settled tap water. After agitating the re-suspended sample, a 2 ml sub-sample was collected with a Hensen-Stempel pipette and placed into a circular counting disk. The entire sub-sample was counted under a Meiji dissecting microscope at 30X magnification. The macro zooplankton were identified to genus or species (Thorpe and Covich 2001). Sub-sampling was continued (minimum of 2) until a minimum of 200 individuals per sample was counted. Total count was used to calculate density per unit volume as described in McCauley (1984).

The Phase 2 monitoring program Terms of Reference (BC Hydro 2004) outlined collection of zooplankton once per season on each reservoir, with a focus on late summer when reservoir levels are relatively constant. In 2006, zooplankton sampling was increased to every 4 -8 weeks to provide enumeration on an annual basis. In 2009, all collected samples were enumerated but lengths of species were not measured. As a result, average lengths of species collected in 2010 data were used to estimate 2009 biomass. In March 2010, the number of samples on each reservoir was increased to 5 per sampling trip to provide replication, which was continued to the end of the monitoring period.

2.7 Statistical Treatment

A statistical summary (sample size minimum, maximum, median, mean, and skewness) was calculated for each variable. If the distribution of data was skewed, a log transformation was used prior to analyses to establish normality. Effectiveness of the transformation procedure was informally evaluated based on a comparison between the site median, mean and geometric mean statistics. A between-site comparison of means over the entire time series was done using a t-test. Statistical time series analysis could not be done to identify the likelihood of a potential seasonal trend (e.g., periodicity, amplitude) since sampling frequency was not consistent across and within (i.e., concentrated on growing season) monitoring years. As a result, the presence of a seasonal cycle was subjectively

evaluated by plotting the data as a function of ordinal date and reporting peak values and general timing patterns.

Comparisons between variables within and across years were completed using correlation analyses. Outlier values that had undue influence on correlation coefficient determinations were identified so that inferences could be tempered accordingly. Within year comparison of variables treated all samples as independent observations in order to assess effects on variable interrelationships. All correlations between independent and dependent variables were calculated using data collected on the same date of sampling. For environmental variables potentially affecting planktonic growth, a second correlation included a time lag on the independent variables equivalent to the time interval from the previous sampling date to investigate conditions experienced during growth prior to time of sampling. Because growth is exponential, growing conditions prior to planktonic sampling tend to have greater influence on the standing stock of organisms than conditions at the time of sampling. Since time intervals between sampling were highly variable (2 – 142 days) no predictive equations were possible. Instead, variables that appeared to lead to higher growth prior to sampling were identified. Across year comparisons used annual mean values to assess variable interrelationships on a between-year basis to identify trends overtime (i.e., ascending, descending, neutral). Regression analysis was used to determine whether slope values significantly deviated from unity (i.e., the line of equality), a specific causal relationship needed to be described, or multiple regression techniques were needed to evaluate the joint effects of two or more independent variables on a single dependent variable. In the latter case, forward stepwise regression was used unless a specific hypothesis concerning a set of independent variables was being tested.

All correlations were compared between the Stave Lake and Hayward reservoirs using analysis of covariance (ANCOVA). Because only two sites were being compared, ANCOVA results were reported in terms of the T distribution. To compare sample means between sites, simple t-tests were used. In all instances, distributions were checked for normality, homoscedasticity and linearity. Where necessary, the data were log transformed to meet these assumptions.

All meta-analyses were carried out using the dataset compiled in Appendix 3. Statistical analyses were carried out using both Microsoft Excel (version 1701) and UniStat (version 6.5).

3 Results

3.1 Hydrology

3.1.1 General Trends

The total volume of discharge into Stave Lake Reservoir (i.e., the sum of local inflows and total Alouette Powerhouse releases, including the adit gate) varied from year to year with the highest value (4.95 Gm³) occurring in 2007, the lowest (3.71 Gm³) in 2010, and a mean of 4.13 Gm³ (Table 1). Total volume of discharge released from the Stave Falls facility into Hayward Reservoir was similar to the Stave Lake Reservoir total inflows ($R^2 = 0.940$) with a shared mean of 4.13 Gm³. Although unexpected, the shared mean could be the result of variable storage use in Stave Lake Reservoir between years resulting in variable outflow volumes. This appears to have occurred as the coefficient of variation (CV) in total reservoir outflow was greater than the CV of total inflow (Table 3).

Table 3. Yearly total discharge volume (Gm³) and geometric mean water retention times (days) for the Stave Lake and Hayward reservoirs over the course of the pelagic productivity monitoring program (2000 to 2014).

Year	Stave Lake Reservoir			Hayward Reservoir	
	Inflow (Gm ³)	Outflow (Gm ³)	Retention (days)	Outflow (Gm ³)	Retention (days)
2000	3.77	3.96	37.27	3.96	1.82
2001	3.80	3.59	48.29	3.58	2.06
2002	4.08	4.25	36.94	4.24	1.80
2003	4.06	4.00	32.99	3.97	1.99
2004	3.91	3.78	40.00	3.67	2.04
2005	3.89	3.81	41.02	3.70	1.85
2006	4.13	4.23	33.02	4.19	1.55
2007	4.95	5.06	28.80	5.01	1.13
2008	3.74	3.68	32.50	3.62	1.61
2009	3.71	3.59	43.06	3.59	1.69
2010	4.20	4.27	34.64	4.21	1.37
2011	4.54	4.53	31.30	4.44	1.27
2012	4.69	4.80	32.30	4.82	0.91
2013	3.87	3.85	33.44	3.95	0.92
2014	4.65	4.57	37.71	4.57	1.33
Mean	4.13	4.13	36.22	4.10	1.56
SD	0.39	0.45	5.16	0.46	0.39
CV	0.10	0.11	0.14	0.11	0.25

A comparison between total outflow volumes of Stave Falls and Ruskin facilities showed a high degree of correspondence ($R^2 = 0.986$). The slope of the relationship approached unity (1.002), while the intercept was -0.039. The negative value of the intercept suggested that there was less outflow from Ruskin Dam than was entering Hayward Reservoir from Stave falls Dam. This is reflected in the yearly mean total outflow volumes at each facility (Table 3). Because there are local inflows to Hayward reservoir, the opposite was expected. The difference between facilities however, was not statistically significant ($t = -0.279$, $P = 0.784$), thus it would appear that the effects of local inflow to Hayward reservoir are negligible compared to outflows from Stave Falls Dam.

The hydrology of Stave Lake inflows was dominated by frequent high inflow events that occur at all times of the year, but tended to be more concentrated during the winter months (October to the end of January; Appendix 1A). Also occurring each year was a prolonged period of high inflow due to spring and summer freshet (April to the end of August). Inflows to Hayward reservoir were far less volatile, predominantly due to its regulated source from the Stave Falls powerhouse (Appendix 2C). Periods of greater volatility were either due to higher local inflows relative to Stave Falls powerhouse outflows, or infrequent spill events from Stave Falls Dam.

Daily water retention times in Stave Lake Reservoir averaged 36.2 days ranging between 4.3 and 11533 days overall (Table 3; Appendix 1D). The distribution tended to be logarithmic in nature with a number of outliers occurring when total daily discharges out of Stave Falls Dam approached zero. Such instances were generally rare and lasted no more than a few days. In 99% of the cases, water retention times did not exceed 152.3 days. A strong seasonal trend was observed with low retention rates during the winter and early spring months (October to May; 4.13 to 70.7 days, geometric mean = 28.7 days) and higher retention rates in late spring and summer (20.8 to 438.6 days, geometric mean = 70 days;

Daily water retention times were lower in Hayward Reservoir (geometric mean = 1.6 days) and less variable between years (CV = 25%; Table 3). Retention rates ranged from 0.02 to 5.22 days 99% of the time with no evidence of a seasonal trend (Appendix 1G). Outlier values tended to be concentrated in the summer time period (May to September) when outflows through the Ruskin facility occasionally approached near zero values. Low retention rates (<1.6 day) were observed throughout the year with the lowest retention rates in November.

Maximum annual elevations of Stave Lake reservoir varied little across all years (EI 81.06 to 82.02 m, mean = 81.63 m) while minimum yearly elevations were far more variable (EI 71.98 to 75.97 m, mean = 74.51) in response to inflow conditions, downstream outflow requirements and power demands (Table 4; Figure 3). The average range of fluctuation was 7.11 m with a CV = 21.5%, while median elevation averaged EI 78.34 m with a CV of only 1% across all years (Table 4). Stave Lake Reservoir elevation statistics (minimum, median and maximum) were generally uncorrelated with total inflow volumes ($r = 0.165, 0.088$ and -0.357 , respectively) and total outflow volumes ($r = 0.097, 0.084$ and -0.283 respectively).

Hayward Reservoir elevation reflected its status as a run-of-the-river system (Appendix 1E). Aside from the planned drawdowns (as discussed in Section 2.1; Table 4; Figure 3), fluctuations in elevation had an amplitude of no more than 1.25 m depending on time of year. The most stable periods on record were midsummer and mid-winter during the last three years of the monitoring program where fluctuations were typically < 0.25 m in amplitude.

Table 4. Yearly summary statistics of Stave Lake and Hayward reservoir elevation (m) during the pelagic productivity monitoring program (2000 to 2014)

Year	Stave Lake Reservoir Elevation (m)			Hayward Reservoir Elevation (m)		
	Minimum	Median	Maximum	Minimum	Median	Maximum
2000	75.79	78.29	82.02	41.35	42.34	42.75
2001	75.97	79.47	81.81	33.12	42.29	42.80
2002	73.15	78.64	82.00	41.41	42.42	42.74
2003	72.15	77.63	81.58	32.95	42.43	42.71
2004	75.28	78.19	81.79	41.51	42.46	42.73
2005	75.29	78.61	81.39	33.08	42.24	42.72
2006	71.94	79.04	81.38	40.09	41.19	41.35
2007	74.90	78.64	81.73	32.72	41.23	41.40
2008	71.98	77.23	81.86	40.30	41.23	41.38
2009	75.26	78.58	81.59	34.47	41.22	41.37
2010	75.47	78.29	81.06	34.47	41.21	41.34
2011	75.16	77.81	81.73	34.43	41.19	41.34
2012	75.11	78.53	81.52	33.06	40.95	41.33
2013	74.38	77.81	81.85	33.11	39.23	41.34
2014	75.86	78.40	81.10	36.15	41.19	41.32
Mean	74.51	78.34	81.63	36.15	41.52	41.91
SD	1.46	0.56	0.29	3.63	0.87	0.71
CV	0.02	0.01	0.00	0.10	0.02	0.02

3.1.2 Time of Sampling

Inflow discharge to Stave Lake Reservoir at the time of pelagic sampling ranged from 8 to 1027 m³/s (median = 82 m³/s, geometric mean = 82 m³/s) with a seasonal pattern that was consistent with the daily average inflow discharge data (Figure 2; Appendix 1A).

Inflow discharge to Hayward Reservoir at the time ranged from 0 to 451 m³/s (median = 109 m³/s, geometric mean = 109 m³/s) with no significant seasonal trend; consistent with the daily inflow discharge data (Figure 2; Appendix 1C). However, there was increased variance in inflow discharge at the time of pelagic sampling during the early winter period. Inflow discharge in Hayward Reservoir was significantly higher than in Stave Lake Reservoir ($t_{2,181} = 3.3634$, $P = 0.0009$). The reason for this difference is uncertain, but it is at least in part related to the use of reservoir storage in Stave Lake Reservoir.

Slave Lake water retention times at time of sampling as a function of ordinal date revealed a strong seasonal pattern that was consistent with the general pattern observed over the course of the full monitoring period (Figure 4; Appendix 1D). As seen in all daily water retention times for Hayward reservoir, there was little seasonality variance in the data at time of sampling (Figure 4; Appendix 1G). There was a clear difference in the water retention times of Hayward reservoir compared to the Stave Lake Reservoir (Figure 4).

In general, the magnitude and seasonality of hydraulic parameters measured at the time of pelagic sampling appeared to be representative of daily average values overall and captures the range of conditions typically observed at both reservoirs.

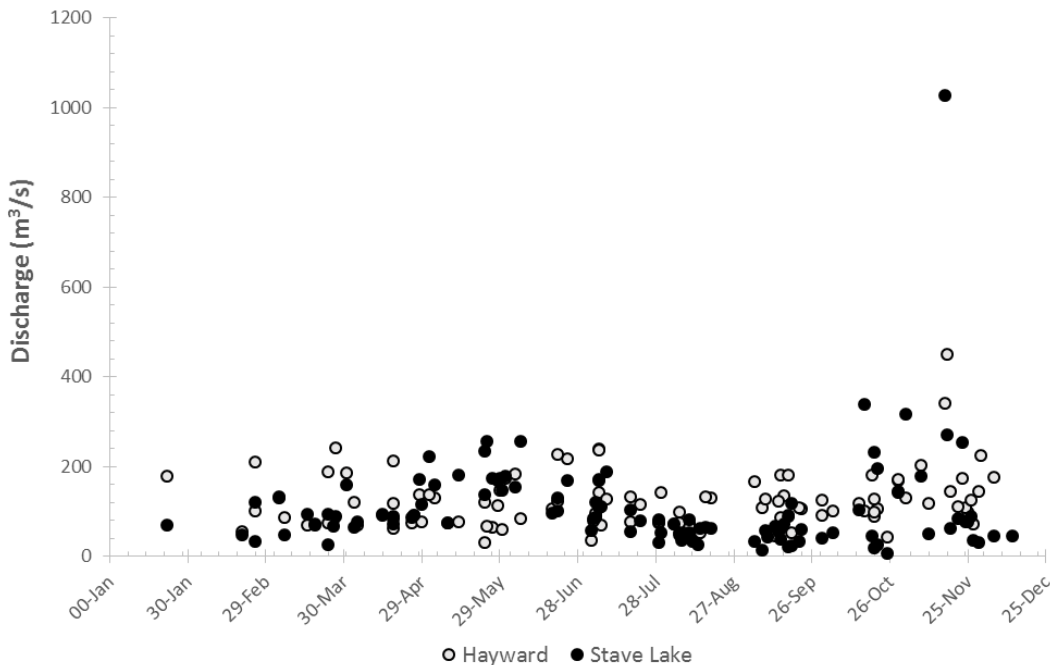


Figure 2. Ordinal date plot of daily average inflows to Stave Lake and Hayward reservoirs at the time of pelagic sampling over the course of the 2000 to 2014 monitoring period.

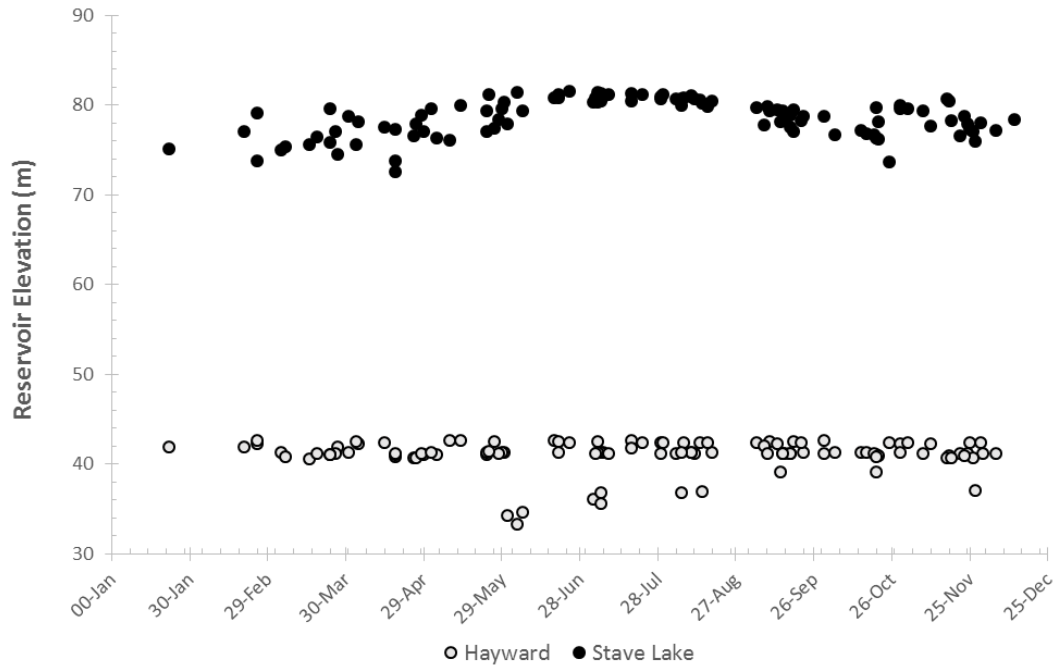


Figure 3. Ordinal date plot of water surface elevation at the time of pelagic sampling in Stave Lake and Hayward reservoirs over the course of the 2000 to 2014 monitoring period.

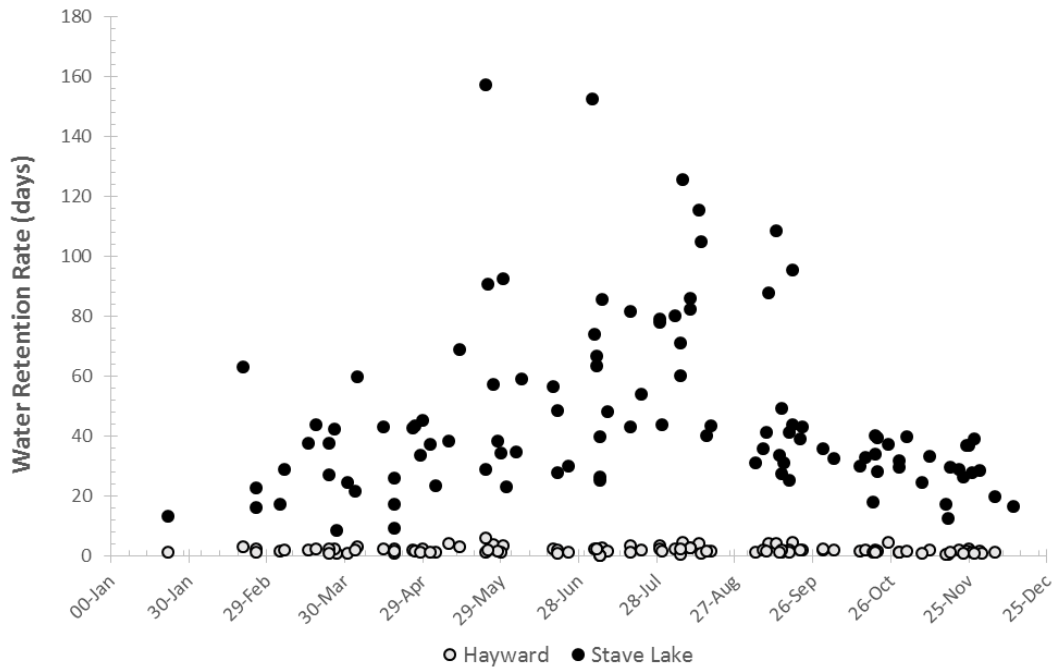


Figure 4. Ordinal date plot of instantaneous water residence times (on a daily time step) at the time of pelagic sampling in Stave Lake and Hayward reservoirs over the course of the 2000 to 2014 monitoring period.

3.2 Water Quality

3.2.1 Light

Light data were collected in 200 of the 218 sampling occasions. Estimates of surface PAR intensity at the time of pelagic sampling, obtained by regression analysis of the paired depth at PAR intensity data, ranged from 1.9 to 1876 $\mu\text{mol}/\text{m}^2/\text{s}$ in both reservoirs (median = 336 $\mu\text{mol}/\text{m}^2/\text{s}$; Figure 5). PAR values measured at the Stave Lake and Hayward sampling sites were correlated with one another ($r = 0.754$), but not as strongly as expected. The geometric mean PAR intensity at the Stave Lake site was 250 $\mu\text{mol}/\text{m}^2/\text{s}$, which was slightly higher than 184 $\mu\text{mol}/\text{m}^2/\text{s}$ found at the Hayward site ($t_{198} = 1.528$, $P = 0.128$; not significant). Plot of the surface PAR data as a function of ordinal date indicates that the distribution of PAR intensities follow a seasonal pattern, but only in terms of maximum potential light intensity. On any given day, light intensity can vary considerably below the maximum. It should be noted that the between site differences were likely due to the different times in the day that light intensity was measured (i.e., measurements were not made in tandem, but in sequence several hours apart), and likely included the effects of within-day changes in local weather.

Light extinction coefficients ranged from 0.18 m^{-1} to 0.94 m^{-1} in both reservoirs (Figure 6). The geometric mean of Hayward light extinction coefficients (0.41 m^{-1}) was significantly greater than in Stave Lake Reservoir (0.36 m^{-1} , $t_{198} = 4.474$, $P < 0.0001$). A plot of the light extinction coefficients as a function of ordinal date showed that the between-site difference appeared to be seasonal, where Hayward reservoir values were generally much higher in the spring and fall months, but very similar during the summer months (Figure 6). Stave Lake Reservoir values tended to be relatively constant over the entire year, with the exception of a few sampling periods in late fall.

The seasonal trend in light extinction coefficients was reflected in the SD data as both parameters were correlated (Figure 7). The relationship however, was not linear; requiring the SD data to be log

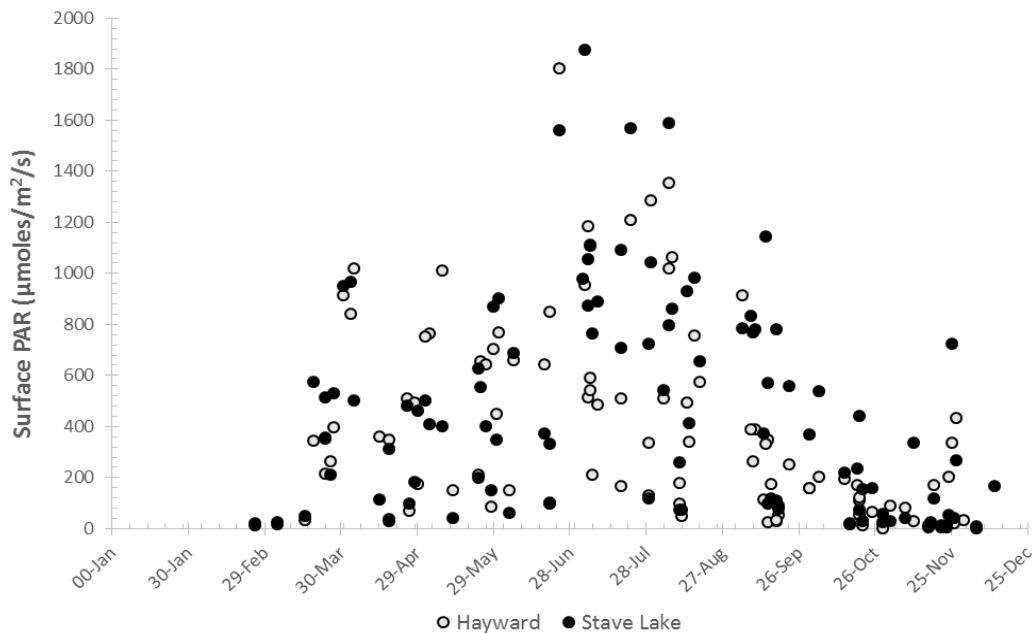


Figure 5. Surface light intensity (PAR) at the time of pelagic sampling plotted against ordinal date for the Hayward and Stave Lake reservoir sites.

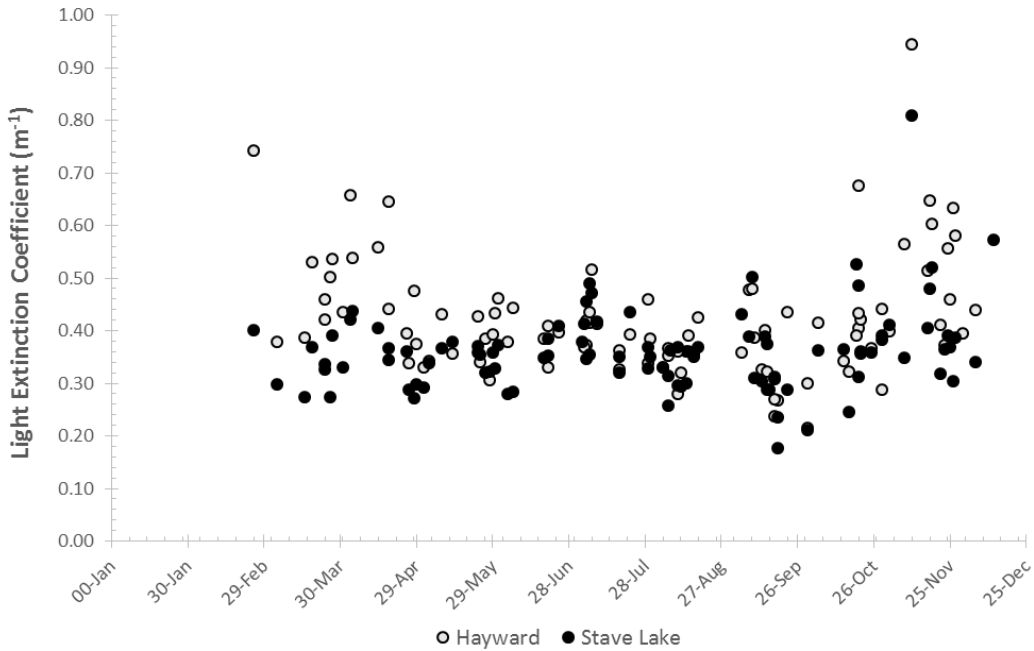


Figure 6. Light extinction coefficients (k) at the time of pelagic sampling plotted against ordinal date for the Hayward and Stave Lake reservoir sites.

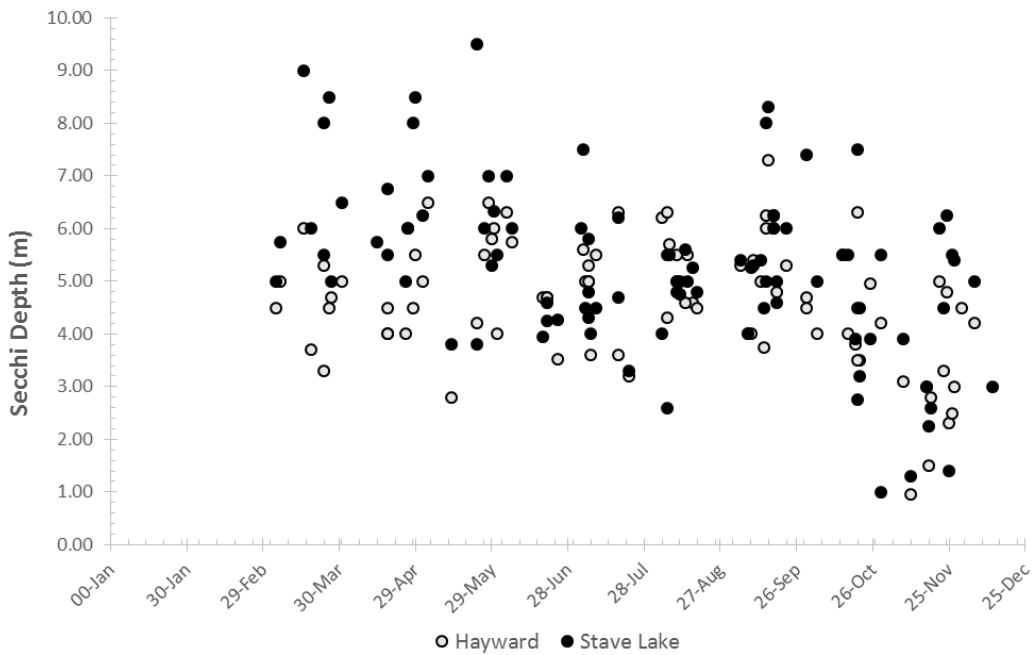


Figure 7. Secchi disk depths at the time of pelagic sampling plotted against ordinal date for the Hayward and Stave Lake reservoir sites.

transformed to achieve linearity ($r = -0.668$, $P < 0.0001$). As the negative correlation indicates, low light extinction coefficients were associated with high SD (i.e., greater light transparency). Average SD of Hayward (4.66 m) and Stave Lake (5.25 m) were significantly different ($t_{167} = -2.827$, $P = 0.0053$). The penetration of light into the upper water layers of Hayward Reservoir tended to be lower compared to that in Stave Lake Reservoir. Hayward SD were more stable through spring and summer, and dropped further in early winter. Stave Lake SD peaked in the spring and followed a gradual decline over time.

Annual geometric mean of light extinction coefficients varied from year to year but correlation analysis indicated no temporal trend at either site (Stave: $r = -0.404$, $P = 0.152$; Hayward: $r = -0.043$, $P = 0.885$; Figure 8a). Annual light extinction coefficients were consistently higher in Hayward Reservoir than Stave Lake Reservoir in all years. Conversely, annual geometric mean of SD was found to increase significantly over time at both sites (Stave: $r = 0.638$, $P = 0.019$; Hayward: $r = 0.652$, $P = 0.016$; Figure 8b). Annual SD were consistently higher in Stave Lake Reservoir compared to Hayward Reservoir. This is consistent with the negative correlation between SD and light extinction coefficients noted earlier.

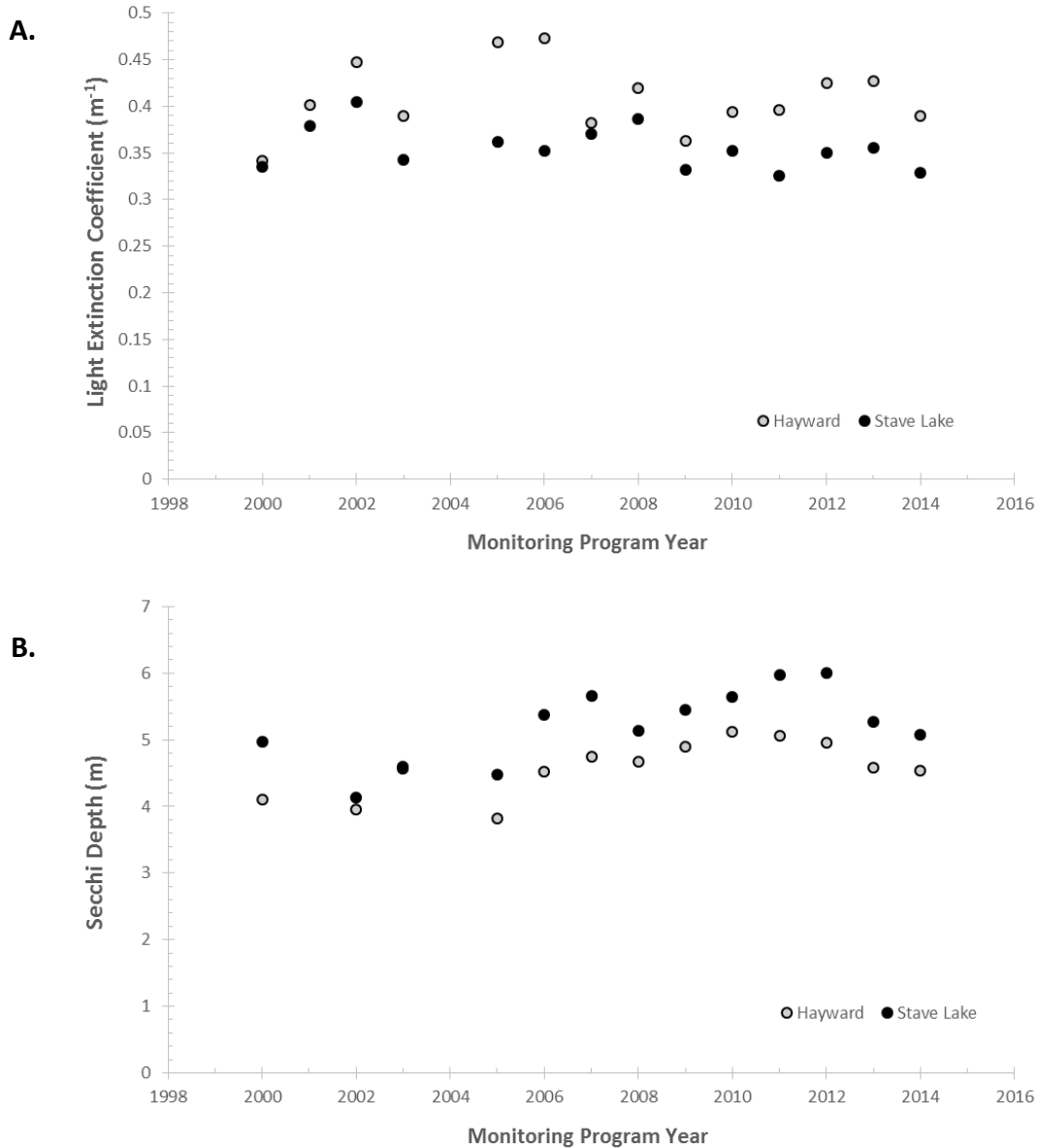


Figure 8. Plot of yearly geometric mean light extinction coefficients (A) and mean Secchi disk depths (B) over the course of the 2000 to 2014 monitoring period for each of the Hayward and Stave Lake reservoirs.

3.2.2 Water Temperature

No water temperature data were collected in years 2000 and 2001, nor were there consistent observations in years 2002 and 2003. Thus, water temperature data were only collected on 178 occasions for each reservoir (Figure 9). A comparison of epilimnion water temperatures (T_{Epi}) between sampling sites found no difference in average values, which were 12.9 and 12.8°C for the Stave Lake and Hayward reservoirs, respectively. Regression analysis of T_{Epi} between reservoir sites indicated that they were highly correlated with one another ($r = 0.964$, $P < 0.0001$), and that temperatures in Hayward Reservoir were within $\pm 2.6^\circ\text{C}$ of Stave Lake reservoir 95% of the time. It would appear that Hayward

reservoir temperatures were generally driven by conditions in Stave Lake reservoir. This is consistent with the very short water retention times in Hayward reservoir, which would limit the effects of such factors as solar heating or convective cooling.

A plot of T_{Epi} as a function of ordinal date revealed a strong seasonal pattern (Figure 9). Temperatures tended to rise steadily through the spring and summer months, reach a peak in mid-July (approximately 20°C), and then gradually fell through the late summer and fall months to winter lows (approximately 6°C). Notable in the ordinal plot is sudden drop in epilimnion temperature occurring in mid-September. This coincides with the period of rapid drawdown of Stave Lake Reservoir in preparation for winter operations (Appendix 1B).

A comparison of epilimnion water depths and the depth of the metalimnion showed little difference in the two metrics. Thus, all further analysis was focused solely on the epilimnion depth observations. A plot of these data as a function of ordinal date showed a stark contrast between sites (Figure 10). In Hayward Reservoir, there were very few occasions (8 in total) where water temperatures changed more than 1°C per meter interval of depth. In 5 of the 8 observations, the change occurred within the top 1 m of water. In the other three occasions, the changes occurred at depths ranging from 17 to 26 m, and were within 1 to 2 m of the reservoir bottom. In none of these cases was the change in water temperature indicative of metalimnion formation. Rather they appeared to be more in response to the short-term effects of solar heating or rapid flushing where pockets of cooler water close to the reservoir bottom were left undisturbed.

Epilimnion depths in the Stave Lake Reservoir showed a very clear increasing trend that typically started at the beginning of June. With the exception of a few outlier values, the increasing trend was

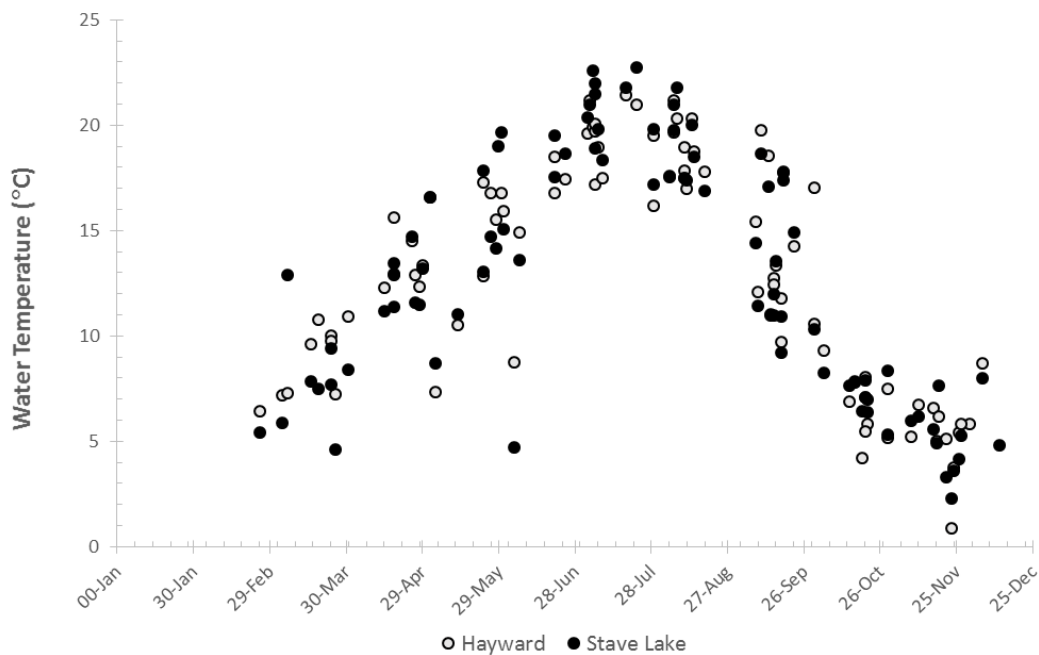


Figure 9. Epilimnion water temperature (average of temperature measurements taken at 1, 3 and 5 m depths) plotted as a function of ordinal date of sampling for the Hayward and Stave Lake reservoir sites.

linear, reaching an average maximum depth of approximately 12 m by the middle of September. After this period, there appeared to be an immediate and complete breakdown in strong thermal gradients, indicating that turnover events had been initiated where epilimnetic and hypolimnetic waters mixed and water temperatures became more uniform top to bottom. It is interesting to note that the breakdown of thermal gradient coincides with the rapid draw down of the reservoir that typically starts in the first week of September, and in turn explains the rapid drop in epilimnion temperatures observed at that time in Figure 9.

Annual mean T_{Epi} varied considerably from year to year with a high degree of correspondence between sites (Figure 11). There appeared to be no consistent downward or upward trend in the data over the course of the monitoring period (Stave: $r = -0.430$, $P = 0.163$; Hayward: $r = -0.367$, $P = 0.240$). It should be noted however, that sampling periods were not consistent between years, and given the strong seasonal pattern in the data, created a high degree of sampling error that could have masked a persistent trend in the data.

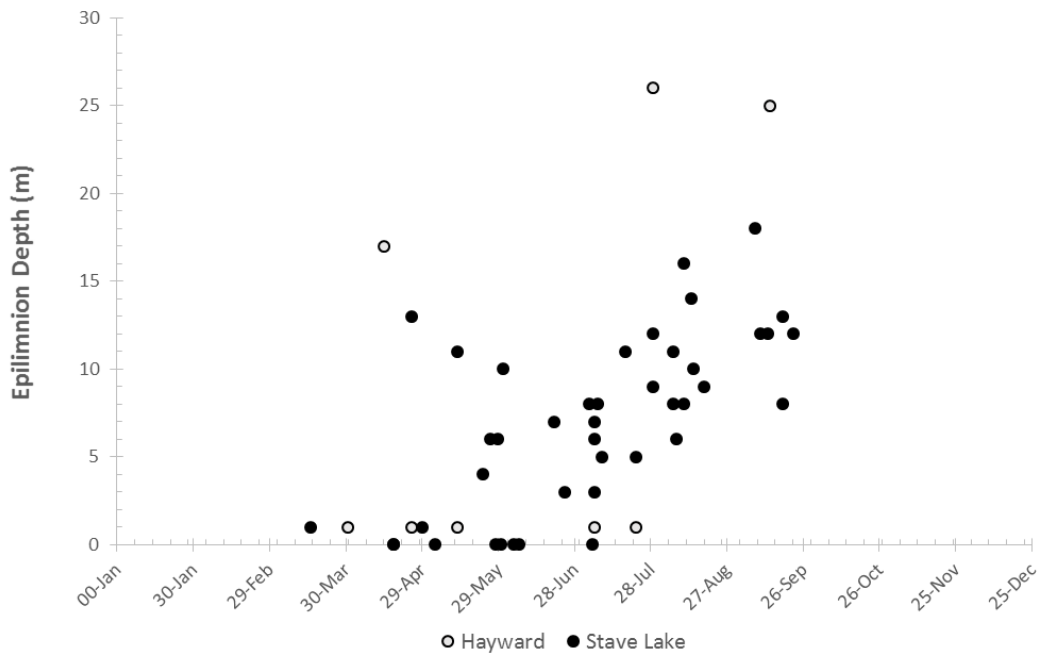


Figure 10. Depth of the epilimnion, defined here as the depth where the change in water temperature for a given 1 m depth interval is $> 1^{\circ}\text{C}$, plotted as a function of ordinal date of sampling for the Hayward and Stave Lake reservoir sites.

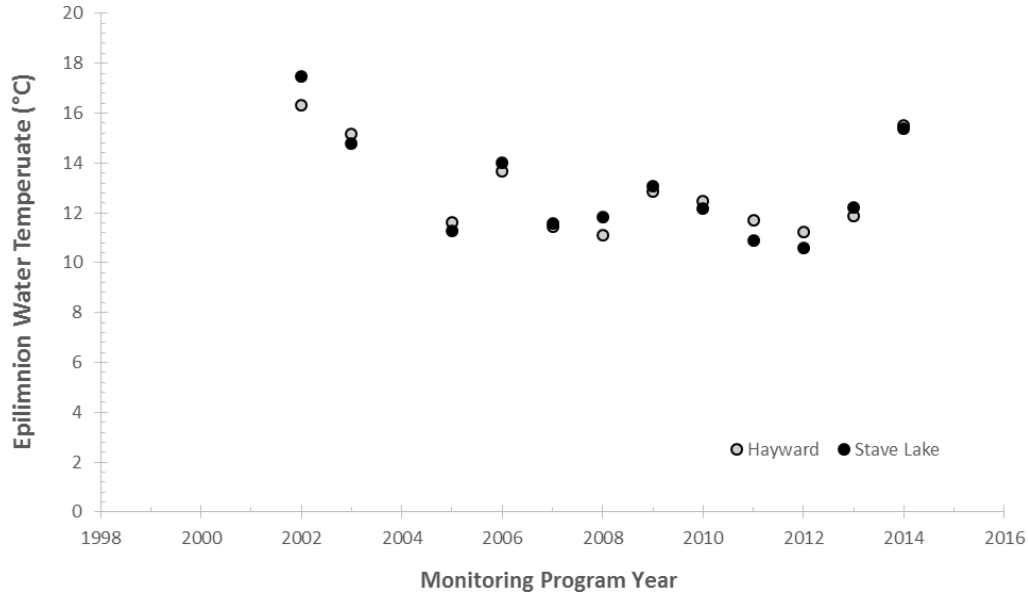


Figure 11. Plot of yearly average epilimnion water temperatures (average of temperature measurements taken at 1, 3 and 5 m depths) over the course of the 2000 - 2014 monitoring period.

3.2.3 Nutrients

Total Phosphorous

Of the total phosphorus (TP) measurements collected (n=180), seven yielded results that were below detectable limits (< 1 µg/L in those instances) and were simply excluded from the dataset. TP values ranged from 0.5 to 19.4 µg/L at the Stave Lake site and had a geometric mean of 2.0 µg/L. At the Hayward site, TP ranged from 0.8 to 23.8 µg/L and had a geometric mean of 2.5 µg/L. TP at both sites were correlated with one another ($r = 0.661$, $P < 0.0001$), though as the site geometric means indicate, Hayward reservoir values tended to be higher by an average 0.5 µg/L across the range of values ($t_{2,86} = 2.552$, $P = 0.012$).

No seasonal pattern was observed for TP concentrations of either reservoir (Figure 12). There were however, two anomalous observations that occurred late in the fall season. The first of these was on October 29, 2003 (Hayward: 23.8 µg/L; Stave: 19.4 µg/L) and the other on November 25, 2003 (Hayward 11.5 µg/L; Stave: 7.3 µg/L). The fact that these anomalous concentrations were observed at both sites and on two separate occasions suggested this was unlikely due to sampling error and was more likely the result of some localised event. The cause for these anomalous observations is uncertain, though it may be related to a very large inflow event that had occurred several weeks earlier in Stave Lake Reservoir (Appendix 1A). This inflow event caused a rapid rise in reservoir elevation to near maximum values (Appendix 1B), as well as a significant spill event out of Stave Falls Dam and into Hayward Reservoir (Appendix 1C).

A strong declining trend in TP concentrations occurred over the course of the Monitoring Program (Stave: $r = -0.680$, $P = 0.010$; Hayward: $r = -0.812$, $P = 0.0007$; Figure 13). Generally, yearly mean TP concentrations were higher at the Hayward site compared to the Stave Lake site.

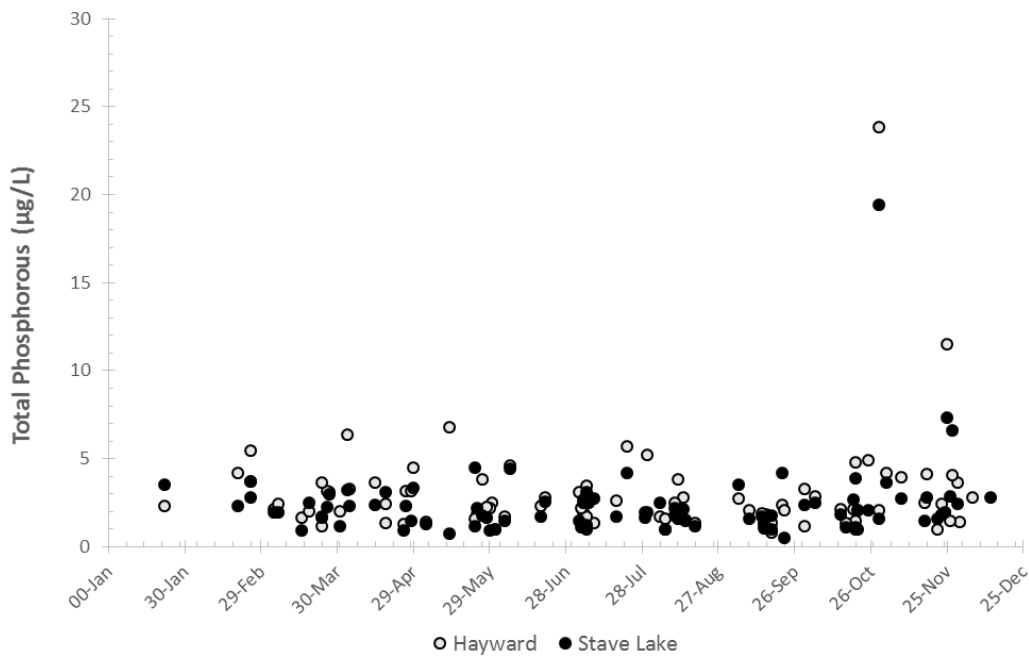


Figure 12. Plot of Total Phosphorous concentration as a function of ordinal date for the Hayward and Stave Lake reservoir sampling sites.

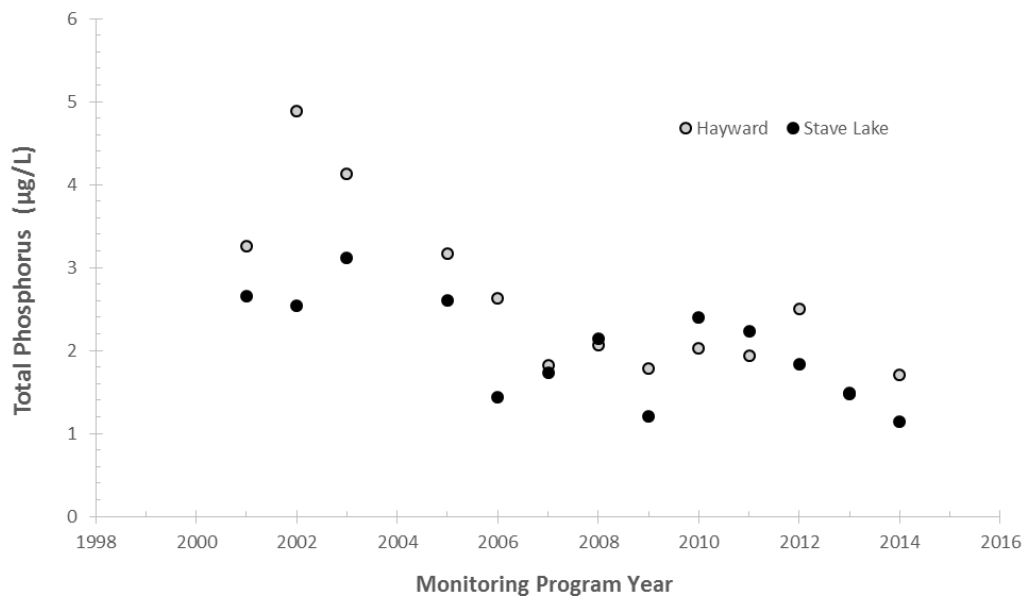


Figure 13. Plot of yearly geometric mean total phosphorous concentrations over the course of the 2000 - 2014 monitoring period for the Stave Lake and Hayward reservoir sites.

Total Dissolved Phosphorus

Of the total dissolved phosphorous (TDP) measurements collected (n=175), 16 observations had TDP concentrations below detectable limits (< 1 µg/L in those instances) and were excluded from analysis. TDP concentrations ranged from 0.1 to 6.1 µg/L overall and had median values of 1.2 and 1.3 µg/L at the Hayward and Stave Lake sites, respectively. Site means were not significantly different from one another ($t_{86} = -0.363, P = 0.717$).

TDP concentrations between sites were highly correlated ($r = 0.728, P < 0.0001$), more so than observed with the TP data. The TP and TDP data were significantly correlated ($r = 0.489, P < 0.0001$) with no strong seasonal trend observed with either variable. However, there was a slight tendency for TDP concentrations to be higher in April and May and lower in August and September (Figure 14). There was no evidence of higher TDP concentrations during the two anomalous TP sampling periods (October 29 and November 25, 2003; Figure 12), indicating that the high TP values were likely the result of increased suspended organic particulate matter, likely brought on by the rise and increased flushing of Stave Lake Reservoir just prior to sampling.

A downward temporal trend was apparent in the yearly geometric means of TDP concentrations (Figure 15). The trend was highly significant at the Hayward site ($r = -0.754, P = 0.0029$), but not so at the Stave Lake site ($r = -0.533, P = 0.0745$).

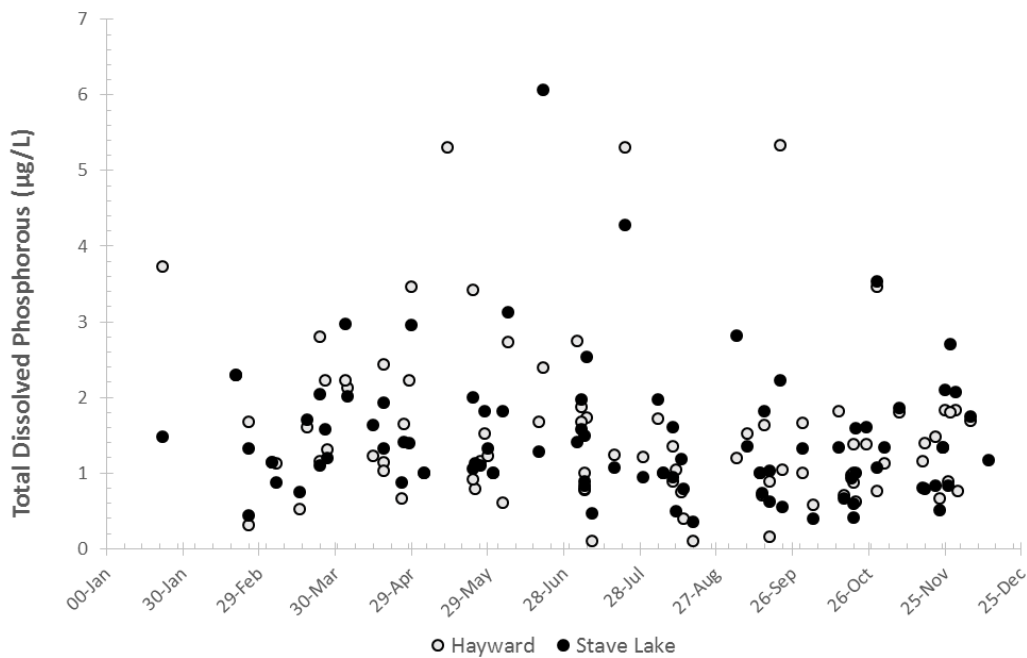


Figure 14. Plot of total dissolved phosphorous concentration as a function of ordinal sampling date for the Hayward and Stave Lake reservoir sites.

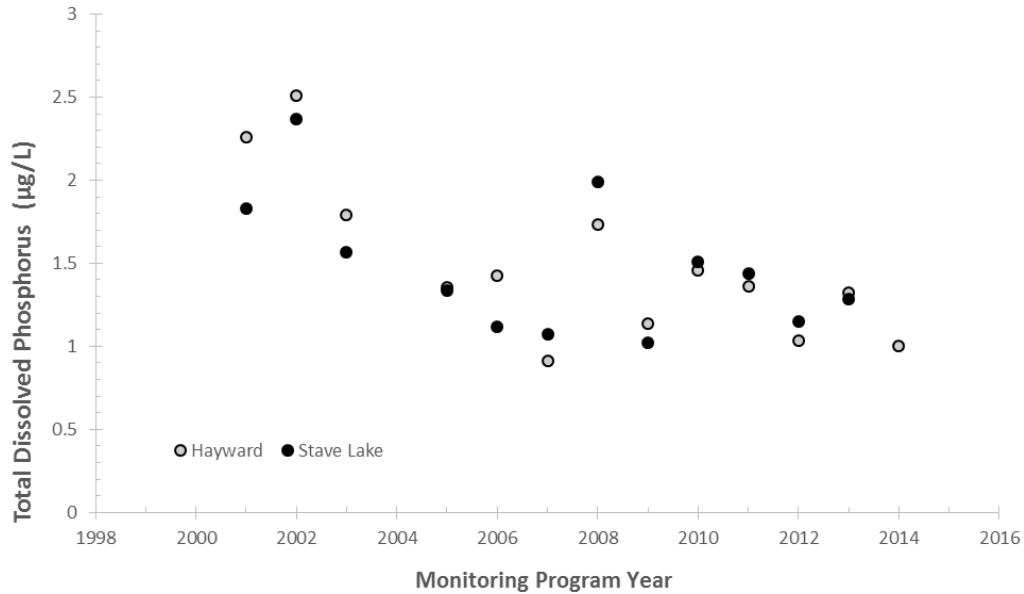


Figure 15. Plot of yearly average total dissolved phosphorous concentration over the course of the 2000 – 2014 monitoring period for the Hayward and Stave Lake reservoir sites.

Nitrate

Nitrate (NO_3) concentrations of the two reservoirs ranged from 10.4 to 162.6 $\mu\text{g/L}$ with no significant difference between means (Stave: 92.5 $\mu\text{g/L}$; Hayward: 95.7 $\mu\text{g/L}$; $t_{96} = 0.405$, $P = 0.686$). NO_3 levels were highly correlated between sites ($r = 0.976$, $P < 0.0001$) with the same strong seasonal pattern observed at both sites (Figure 16). NO_3 values were high throughout the winter season (roughly centered 140 $\mu\text{g/L}$), declining steadily after the first week of April through to the end of summer (roughly centered about 45 $\mu\text{g/L}$). This observed seasonal pattern was similar to the weak trend of the TDP data (Figure 14). As seen in the TP data, two outlier values in NO_3 concentrations were observed on October 29 and November 25, 2003, and therefore also appeared to have been impacted by the 2003 high inflow events. Drops in NO_3 concentrations relative to winter highs in lakes systems are commonly associated with increases in planktonic productivity (Wetzel 2001), suggesting that the rise in TP may have been the result of a planktonic bloom very late in the growing season. Yearly mean NO_3 concentrations showed a very strong decline over the course of the monitoring period at both sites (Stave: $r = -0.869$, $P < 0.0001$; Hayward: $r = -0.877$, $P < 0.0001$; Figure 17).

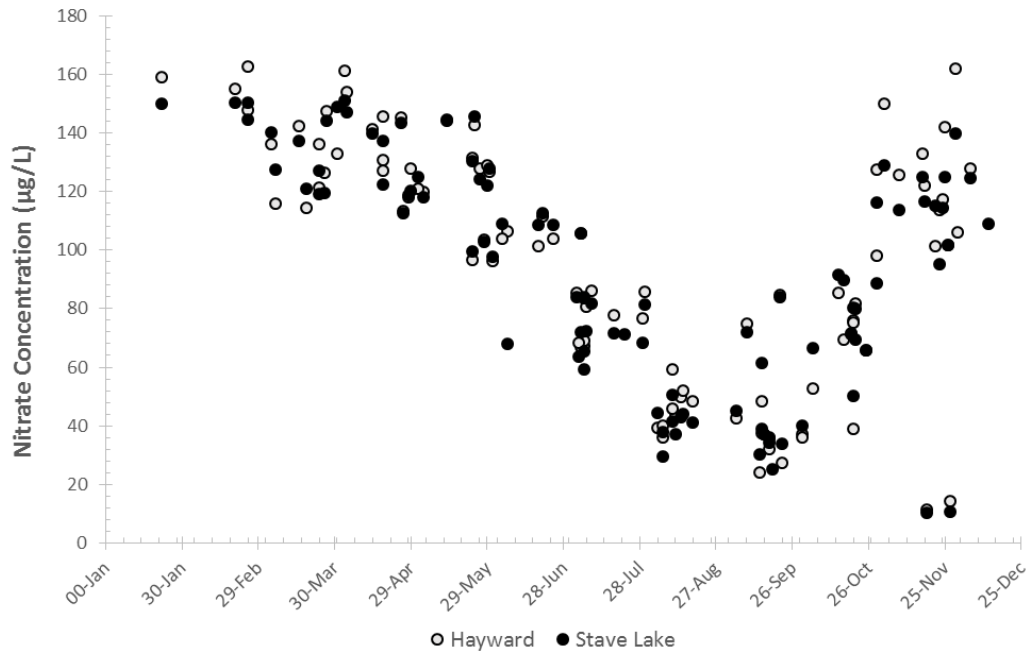


Figure 16. Nitrate concentrations plotted as a function of ordinal sampling date illustrating seasonal trends at the Hayward and Stave Lake reservoir sites.

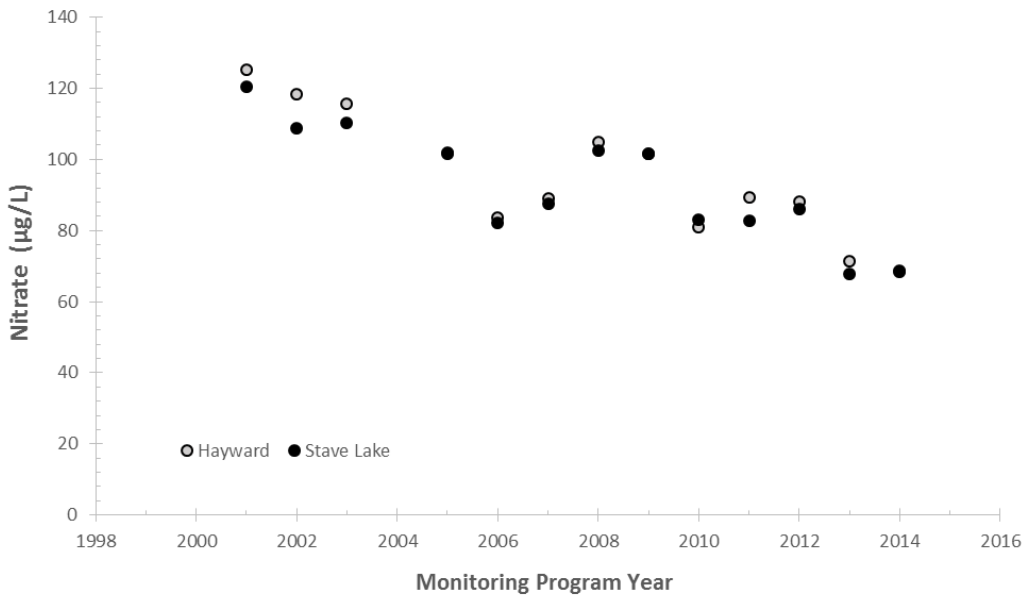


Figure 17. Plot of yearly average Nitrate concentrations over the course of the 2000 – 2014 monitoring period for the Hayward and Stave Lake reservoir sites.

Table 5. Correlation matrix of physical variables used to describe prevailing environmental conditions at the time of biological sampling over the course of the 2000 – 2014 monitoring period. Correlation coefficients in the upper diagonal area correspond to the paired variable comparisons at the Hayward site, the bottom diagonal area for the Stave lake site. Shaded values indicate correlation coefficients that are statistically significant (given the number of paired values in the comparison). Correlation coefficients highlighted in red correspond to those that are greater than 0.5.

		Hayward Reservoir										
		Inflow	Elevation	Retention	Temp	Epilim	PAR	k	Secchi	TP	TDP	NO ₃
Stave Lake Reservoir	Inflow		-0.116	-0.704	-0.353	-0.346	-0.078	0.243	-0.304	0.032	-0.124	0.174
	Elevation	0.216		0.471	-0.030	0.132	0.025	-0.041	-0.133	0.196	0.171	0.214
	Retention	-0.103	0.464		0.396	0.270	0.097	-0.178	0.168	0.071	0.255	0.067
	Temp	-0.175	0.560	0.564		-0.062	0.553	-0.340	0.381	-0.097	0.075	-0.266
	Epilim	-0.522	0.232	0.436	0.262		-0.469	0.437	-0.112	-0.303	-0.428	-0.632
	PAR	-0.194	0.332	0.239	0.619	0.074		-0.062	0.133	-0.038	0.224	0.027
	k	0.034	0.063	-0.168	-0.112	-0.150	0.120		-0.655	0.068	0.052	0.368
	Secchi	-0.163	-0.258	0.001	0.017	-0.422	0.063	-0.523		-0.398	-0.260	-0.231
	TP	0.006	0.007	-0.040	-0.112	-0.418	-0.100	0.182	-0.456		0.413	0.275
	TDP	-0.085	0.081	0.073	0.159	-0.339	0.167	0.083	-0.133	0.444		0.335
	NO ₃	0.272	-0.298	-0.197	-0.307	-0.493	-0.203	0.083	0.070	0.112	0.141	

3.2.4 Water Quality and Hydrology Correlations

Time of Sampling

All of the physical variables measured at the time of sampling were compared to one another to determine the extent with which they were correlated (Table 5).

Many of the variables were indeed correlated with one another, but in most instances the strength of the correlation was generally low to moderate. Few variable pairs had correlation coefficients greater than 0.5 in Hayward (n=3) and Stave Lake (n=5) reservoirs. Both datasets had a common high negative correlation between SD and light extinction coefficients, highlighting the fact that both variables describe the penetration of light into the epilimnion. There was also a common high correlation between PAR intensity and water temperature at the two sites. This was not unexpected as both variables are highly seasonal and interrelated. Unique to Hayward Reservoir was a very strong correlation between inflow discharge and water retention times, easily explained by the reservoir’s small size and relatively stable reservoir elevation compared to that of the Stave Lake Reservoir. In Stave Lake Reservoir, water retention time was highly correlated to water temperature, as was reservoir elevation. It would appear that the rise and fall in reservoir elevation to meet seasonal recreational targets (Failing 1999), along with seasonal fluctuations in reservoir inflow, collectively impact water retention times. This in turn varies in concert with water temperature. Whether changes in retention times have a causal impact on water temperature is uncertain, and cannot be determined from the present data. The last correlation of interest is between inflow discharge at Stave Lake Reservoir and the depth of epilimnion. The negative correlation suggested that high inflows tended to either prevent the formation or cause the breakdown of the thermal gradients that help define epilimnion depth.

Overall, results of the correlation analysis were consistent with the seasonal patterns and variable inter-relationships described in previous sections. In fact, many of the variables were significantly correlated simply because they shared a common underlying seasonal pattern and were not necessarily causally related. When attempting to establish relationships, or draw inferences with biological metrics later in the analysis, these variable inter-correlations must be taken into account.

Yearly Values

All of the nutrient variables appeared to have declined significantly over the course of the monitoring period. To determine whether these declines could be directly related to inter-year differences in reservoir hydrology, these yearly average data were compared to corresponding hydrological data in a series of correlation analyses (Table 6). Immediately apparent was the lack of significant correlations with any of the Stave Lake Reservoir variables, suggesting that yearly average nutrient concentrations and SD were largely independent of yearly reservoir conditions. The correlation between NO₃ concentration and each of the three yearly summary hydrology statistics did however approach significance and was much stronger than that of the phosphorus metrics. This pattern was consistent with the time-of-sampling data (Table 5), and may partially reflect the overlap in seasonal cycles among these variables. Overall however, there did not appear to be a strong causal link between

Table 6. Correlations between yearly average nutrient and light related variables and reservoir hydrology. A distinction is made between the average of all mean daily hydrological variables and the average of sampled values (i.e., Sampled Q, EI and τ_w)

Parameter	k	Secchi	T	TP	TDP	NO ₃
Stave Lake Reservoir						
Inflow (Gm ³)	-0.093	0.490	-0.156	-0.220	-0.346	-0.499
Outflow (Gm ³)	0.000	0.460	-0.108	-0.204	-0.294	-0.506
Retention (days)	0.002	-0.454	0.290	-0.034	0.195	0.519
Min EI (m)	-0.456	0.335	-0.360	-0.379	-0.308	-0.198
Median EI (m)	0.127	0.001	0.194	-0.249	-0.151	0.211
Max EI (m)	0.304	-0.234	0.055	0.255	0.490	0.435
Sample Q	-0.340	0.463	-0.150	-0.469	-0.361	0.108
Sample EI	-0.324	0.040	0.175	-0.191	-0.025	0.153
Sample τ_w	-0.060	-0.018	0.192	0.245	0.103	0.522
Hayward Reservoir						
Inflow (Gm ³)	0.000	0.314	-0.012	-0.154	-0.474	-0.472
Outflow (Gm ³)	-0.017	0.314	0.010	-0.166	-0.475	-0.497
Retention (days)	-0.015	-0.573	0.488	0.645	0.682	0.811
Min EI (m)	0.105	-0.424	0.406	0.146	0.437	0.115
Median EI (m)	-0.054	-0.522	0.465	0.735	0.560	0.779
Max EI (m)	0.032	-0.799	0.471	0.826	0.724	0.799
Sample Q	0.062	-0.352	-0.212	-0.271	-0.410	-0.359
Sample EI	-0.054	-0.513	0.569	0.618	0.594	0.660
Sample τ_w	0.222	-0.171	0.504	0.703	0.610	0.676
Critical r, $\alpha = 0.05$	0.532	0.553	0.576	0.553	0.553	0.532

reservoir hydrology and water quality within the Stave Lake Reservoir that could explain the downward trends in yearly values over time.

This did not appear to be the case for in Hayward Reservoir. All three of the nutrient variables, as well as SD, were correlated with reservoir hydrology and in particular water retention times (Table 6). Strong correlations were apparent in both yearly summary and average values at the time of sampling, indicating that both longer term and short term (i.e., seasonal) effects were involved. Perhaps most telling was the very strong relationship with maximum reservoir elevation, which was lowered by 1.4 m in year 2006 because of dam safety concerns (Table 4). This drop in maximum elevation had the effect of lowering the median reservoir elevation and in turn decreasing water retention times in the reservoir (by reducing the storage volume of the reservoir). At first glance, the data would suggest that this change in Hayward operation may have had an impact yearly average nutrient concentration and SD. However, the fact that similar declining trends in yearly nutrient values were also observed in Stave Lake Reservoir (Figures 13, 14, 15) suggests that these relationships may have been more coincidental than causal in nature.

3.2.5 Non-Algal Turbidity Index

Non-Algal Turbidity Index values in the two reservoirs ranged from 3.97 to 62.98 with no significant difference between geometric means (Stave: 18.79; Hayward: 18.45 $\mu\text{g/L}$; $t_{73} = 0.168$, $P = 0.867$). No negative values were encountered to indicate the potential for zooplankton over grazing.

A seasonal pattern to non-algal turbidity was observed in both reservoirs with low variability in summer and high variability in spring and fall (Figure 18). In Stave Lake reservoir, non-algal turbidity was uncorrelated with all three reservoir hydrology metrics (Table 7), indicating that it fluctuated

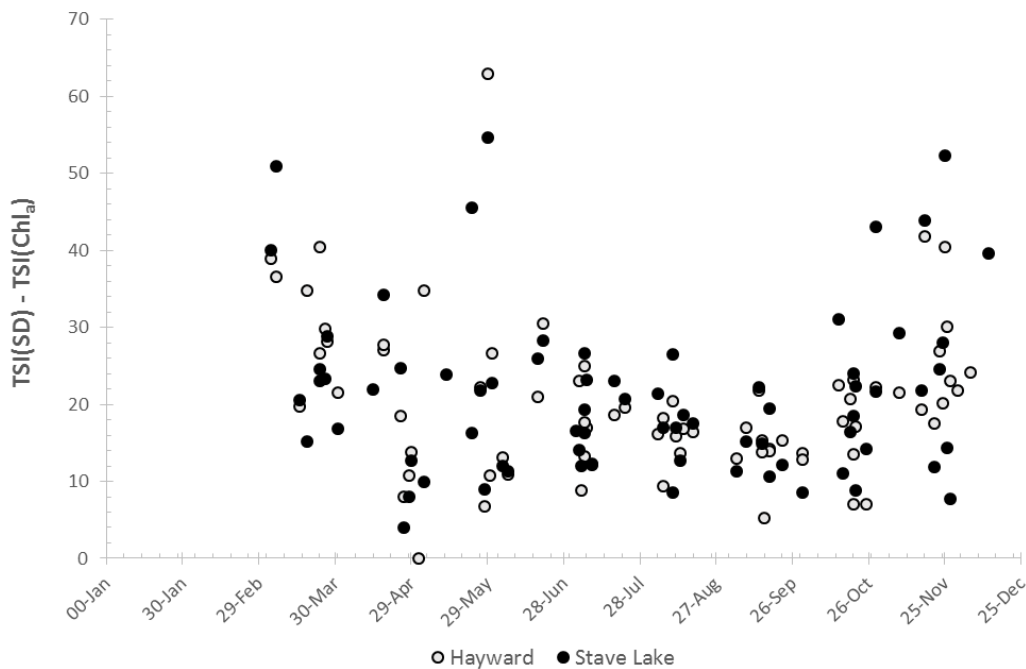


Figure 18. Index of non-algal turbidity (derived as per Carlson 1980) in Stave Lake and Hayward reservoirs plotted a s function of ordinal date of sampling over the course of the 2000 to 2014 monitoring period.

Table 7. Correlation coefficients between reservoir hydrology metrics and the index of non-algal turbidity as derived from Carlson (1980). The coefficients were calculated separately for each reservoir and timeframe. For the Hayward Reservoir, the correlation coefficients were calculated using the residuals from a regression between Stave Lake and Hayward reservoir index values, thus taking into account that Stave Lake non-algal turbidity that would likely influence values in Hayward Reservoir.

Parameter	Stave Lake Reservoir		Hayward Reservoir*	
	r	Prob.	r	Prob.
Time of Sampling				
Inflow (m ³ /s)	0.093	0.370	0.103	0.413
Reservoir Elevation (m)	-0.050	0.632	0.026	0.833
Water Retention Time (days)	0.062	0.552	-0.144	0.265
Sampling Interval Time Lag				
Inflow (m ³ /s)	-0.056	0.637	0.181	0.146
Reservoir Elevation (m)	-0.262	0.025	0.047	0.710
Water Retention Time (days)	-0.231	0.050	-0.110	0.393
Yearly Means				
Inflow (m ³ /s)	-0.270	0.395	-0.409	0.187
Reservoir Elevation (m)	-0.058	0.859	-0.165	0.608
Water Retention Time (days)	-0.184	0.567	0.104	0.748

independently of reservoir operations at the time of sampling. Significant negative correlations did occur with reservoir elevation and water retention time variables when re-examined with an interval lag. The latter suggests that non-algal turbidity was lower during summer when reservoir elevations were at their targeted high values and water retention times were at their longest. Whether this indicates causality is uncertain. This could be a coincidence because rainfall events also tend to be less frequent at this time. Though not directly implicated for the Stave Lake Reservoir, McNair (2008) did note significant increases in turbidity following rain storm events in Hayward Reservoir. This likely occurred in the Stave Lake system as well, and may have been exacerbated by rain on snow events during the spring runoff period. Unfortunately, direct measurements of turbidity were not collected in the Stave Lake Reservoir to test this hypothesis, though such increases in turbidity have been noted anecdotally by the monitoring field crew (J. Beer, Ness Environmental, Pers. Observation). Finally, correlations between year mean values over the course of the monitoring period showed that the between-year differences in in reservoir hydrology were uncorrelated with the annual mean non-algal turbidity index values. This further validates the notion that non-algal turbidity is unlikely to be related to reservoir operations in Stave Lake Reservoir.

With Hayward Reservoir being downstream of Stave Lake Reservoir, it's not surprising to find non-algal turbidity highly correlated between the two reservoir systems ($r = 0.749$, $P < 0.0001$). To account for this strong relationship, regression analysis was used to derive predicted Hayward Reservoir index values, which were then used to derive a set of residual values (similar in technique to a forward stepwise regression analysis). These residuals were considered to reflect the variance in Hayward reservoir non-algal turbidity values unexplained by influences from Stave Lake Reservoir. Correlation analyses to examine operational influences on non-algal turbidity in Hayward reservoir were done using this residual dataset. This analysis revealed no significant correlation between reservoir hydrology

metrics and non-algal turbidity, regardless of the time frame used in the analysis (Table 7). This outcome would suggest that Hayward reservoir operations had little effect on non-algal turbidity. However, as noted by McNair (2008), storm activity can increase reservoir turbidity for short periods of time, particularly in the fall and winter months. Given the short water retention times of the reservoir, such events would be short lived, only to be flushed out by Stave Lake Reservoir waters and again influenced by its turbidity profile.

3.3 Primary Production

3.3.1 General

Chlorophyll a

Chlorophyll *a* (Chl_a) concentrations of the two reservoirs ranged from 0.002 to 1.354 µg/L with no significant difference between means (Stave, 0.311 µg/L; Hayward, 0.352 µg/L; $t_{95} = 1.297$, $P = 0.196$). Chl_a concentrations between the Hayward and Stave lake sites were highly correlated ($r = 0.802$, $P < 0.0001$). A weak seasonal trend was observed in Chl_a concentrations (Figure 19). For both sites, values were high in August through October, peaking at the end of September.

Correlation analyses between Chl_a concentrations and variables that describe environmental conditions at the time of sampling revealed few statistically significant correlations (Table 8). Likely due to the mirrored seasonal patterns, significant correlations between Chl_a and NO₃ concentrations was observed for both reservoirs. The negative correlation is consistent with the general notion that NO₃ concentrations decline in summer as a result of a growing population of primary producers increasing consumption (Wetzel 2001). A significant negative correlation was between Chl_a concentration and light extinction coefficient was also observed in Hayward reservoir. This is consistent with the general notion

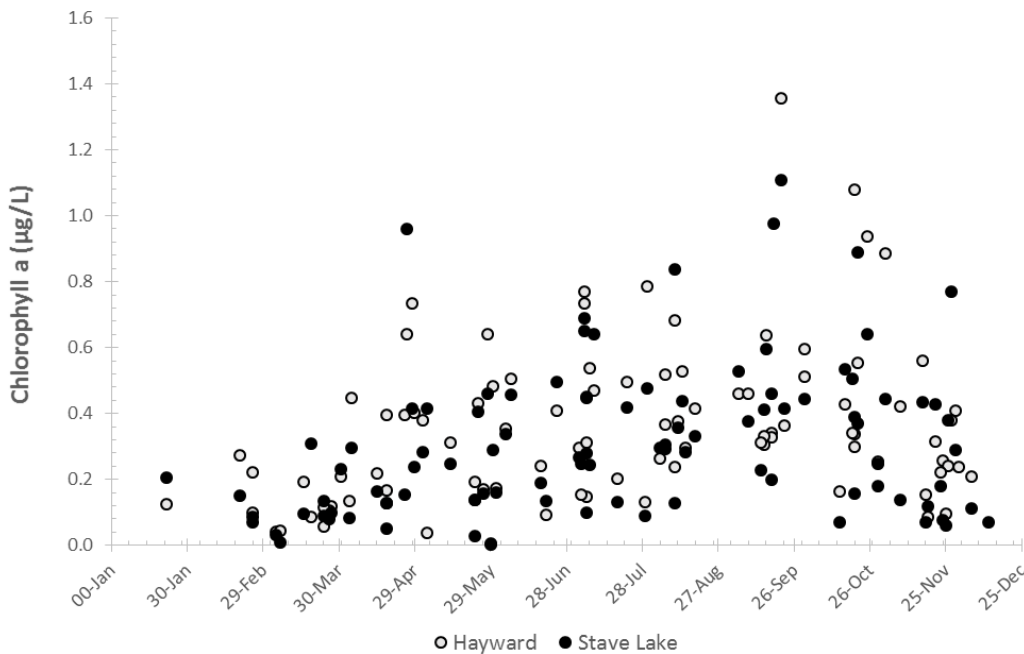


Figure 19. Plot of Chlorophyll a concentration as a function of ordinal sampling date illustrating the potential for seasonal trends at the Hayward and Stave Lake reservoir sites.

that higher concentrations of primary producers reduce the transmission of light to deeper depths (Wetzel 2001). This was not a factor in Stave Lake reservoir.

When the correlations were re-examined with a time lag equal to the last sampling period (hereinafter an interval lag), a weaker correlation between NO₃ and Chl_a concentrations was observed in both reservoirs, however, still significant (Table 8). The negative correlation between Chl_a concentration and light transmission in Hayward reservoir was also observed with the interval lag. In addition, Chl_a concentration was positively correlated with SD (Table 8), suggesting that shading also had an impact on past growing conditions. This effect was not observed in Stave Lake site. For both reservoirs, an interval lag on water temperature and light availability had a significant influence on Chl_a concentration. This was expected given the delayed response of Chl_a (peak during August to October) to peak water temperatures (July to August) and peak light intensity (June to July). This relationship of light and temperature on primary production has been well documented (Wetzel 2001). A significant correlation was also observed between Chl_a concentration and the depth of the epilimnion in Hayward Reservoir. However, this correlation was due to the presence of a single outlier value and was lost with the removal of this outlier value. Therefore, the relationship was considered spurious and not included in further inference analyses. Water retention times and change in reservoir elevations were significantly correlated with Chl_a concentration only at the Stave Lake site. This may be due to the reservoir operations creating a seasonal cycle (June to August) that, like water temperature and light intensity, precedes peak primary production. This cycle is not present in the hydrology of the Hayward Reservoir. This observed increase in significant correlations with an introduction of an interval lag suggests Chl_a concentration was influenced more by past growing conditions than at the time of sampling (Table 8).

Table 8. Correlation coefficients between Chlorophyll a concentration and environmental descriptor variables at the time of sampling and at the previous sampling interval (interval lag) for the Hayward and Stave Lake reservoir sampling sites. Statistical significance (P > 0.05) is indicated by shading.

Parameter	Stave Falls Reservoir				Hayward Reservoir			
	Sampling Date		Interval Lag		Sampling Date		Interval Lag	
	r	Prob	r	Prob	r	Prob	r	Prob
Q	-0.009	0.9309	-0.050	0.6333	-0.156	0.1287	-0.255	0.0121
El	0.131	0.2061	0.274	0.0073	0.017	0.8670	-0.039	0.7042
R	0.062	0.5520	0.333	0.0011	0.099	0.3500	0.128	0.2252
T	0.125	0.2706	0.284	0.0111	0.206	0.0683	0.360	0.0011
Z_epilim	0.068	0.6964	0.005	0.9769	-0.320	0.4841	0.888	0.0180
PAR	-0.010	0.9277	0.304	0.0040	0.050	0.6479	0.358	0.0007
k	-0.173	0.1073	-0.173	0.1064	-0.238	0.0263	-0.261	0.0146
Secchi	-0.027	0.8098	-0.014	0.9004	0.128	0.2583	0.289	0.0089
TP	0.034	0.7579	-0.045	0.6934	-0.039	0.7165	0.121	0.2671
TDP	0.051	0.6402	0.170	0.1366	0.162	0.1320	0.179	0.1046
NO ₃	-0.404	0.0000	-0.259	0.0147	-0.247	0.0151	-0.228	0.0306

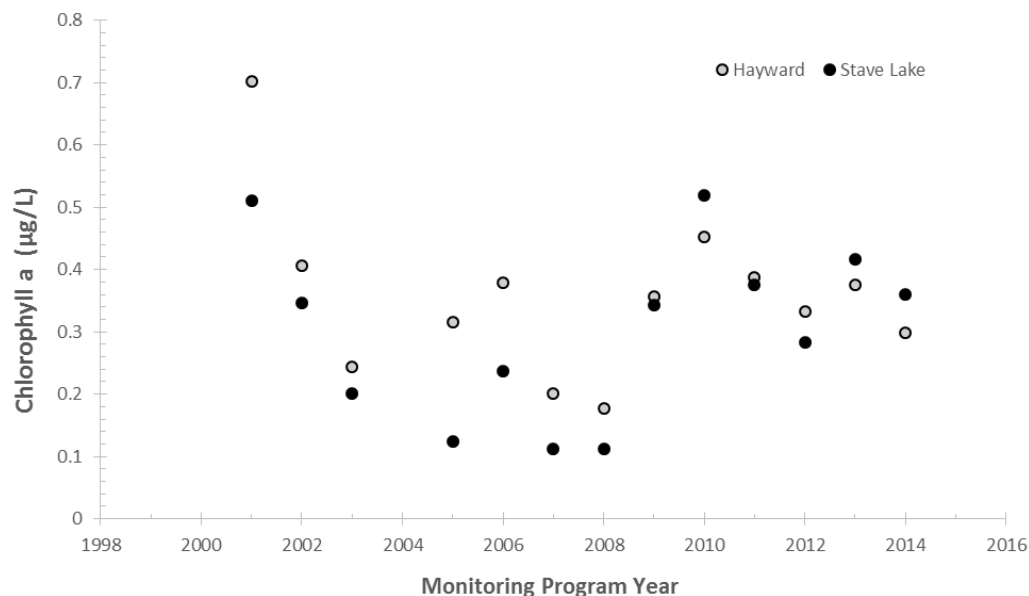


Figure 20. Yearly average Chlorophyll a concentration plotted as a function of time illustrating the lack of a temporal trend over the course of the 2000 to 2014 monitoring period for both the Hayward and Stave Lake reservoir sampling sites.

No significant temporal trend was observed in mean annual Chl_a concentrations (Stave: $r = 0.164$, $P = 0.591$; Hayward: $r = -0.297$, $P = 0.323$; Figure 20). Correlation analysis revealed no significant relationships with any of the yearly average environmental descriptor variables.

¹⁴C Radio Assays

There were 24 successful ¹⁴C radio assay surveys at each site. One outlier value was deleted from each site. It would appear that the ¹⁴C inoculant used for this particular radio assay was contaminated, yielding unreliable results (J. Beer, Ness Environmental, Pers. Comm.). Primary production measured as the rate of ¹⁴C assimilation in the light incubated assay ranged from 0.072 to 2.794 ¹⁴C mg/m³/hr at both sites with no significant difference between geometric means (Stave, 0.537 ¹⁴C mg/m³/hr; Hayward, 0.582 ¹⁴C mg/m³/hr; $t_{2,46} = 0.320$, $P = 0.750$; Figure 21). The light incubated production estimates were highly correlated between the two sites ($r = 0.962$, $P < 0.0001$). Moreover, the slope of the relationship was not significantly different from unity (95% confidence interval (CI) = 0.857 to 1.101) and the intercept values were not significantly different from zero (95% CI = -0.079 to 0.154 ¹⁴C mg/m³/hr) indicating that there were no systematic differences in production rates between sites.

Geometric mean of dark ¹⁴C assimilation rates was not significantly different between sites (Stave, 0.113; Hayward, 0.123 ¹⁴C mg/m³/hr; $t_{2,46} = 0.292$, $P = 0.794$). Dark incubated production was found to be highly correlated with light incubated production (Stave: $r = 0.823$, $P < 0.0001$; Hayward: $r = 0.796$, $P < 0.0001$). Regression coefficients for each site were very similar to one another (-0.084 vs -0.079 for intercept and 0.382 vs 0.380 for slope at the Stave Lake and Hayward sites, respectively). When pooled, the regression data indicated that dark incubation production generally accounted for 38% of the total production measured when samples are incubated with light. Intercept of the pooled regression was not significantly different from zero, indicating that the difference in production between incubation conditions was entirely proportional.

The difference between dark from light incubated production estimates is generally considered to be an estimate of photosynthetic production. In the present study, estimates ranged from – 0.062 to 1.441 ¹⁴C mg/m³/hr with geometric means of 0.424 and 0.459 mg C/m³/hr for Stave Lake and Hayward reservoirs, respectively. Values at the Stave Lake site were highly correlated with those at the Hayward site ($r = 0.912$, $P < 0.0001$). The slope was not significantly different from unity (95% CI = 0.757 to 1.133), and the intercept was not significantly different from zero (95% CI = -0.085 to 0.168) indicating no significant differences in photosynthetic production between sites.

Primary production estimates were highly variable and appeared to have a seasonal cycle in the range of possible values (Figure 21). There was a general tendency for production to reach a maximum in August to October, though low values were recorded during this time as well. Averaging the three highest values recorded in August through October, peak primary production was estimated to be 1.26 and 1.43 mg C/m³/hr in Stave Lake and Hayward Reservoirs, respectively.

Collectively, primary production at the two sites was only weakly correlated with Chl_a concentration ($r = 0.518$, $P = 0.0005$, Figure 22). The power function describing the regression was determined to be:

$$1.06 \cdot Chl_a^{0.80} \quad (R^2_{Adj} = 0.268)$$

The regression intercept was not significantly different from unity ($t_{2,35} = 0.232$, $P = 0.817$), indicating the relationship was entirely exponential in nature. There was a high degree of overlap in the site-specific plots of primary production and Chl_a (Figure 22), indicating no between-site differences in their relationship. This was also captured in the lack of a significant difference in site-specific exponential coefficients ($t_{2,35} = 0.589$, $P = 0.561$).

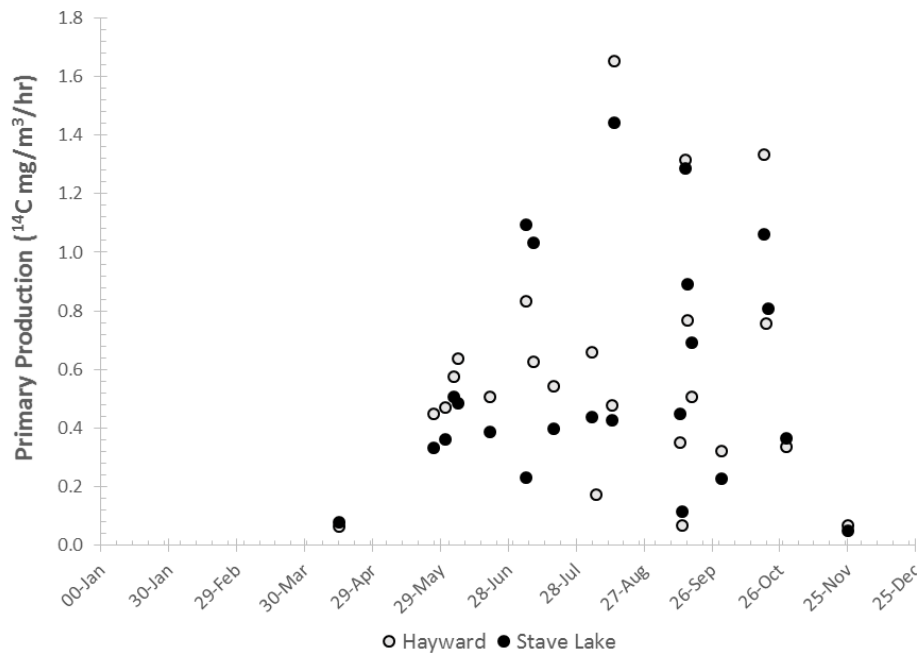


Figure 21. Plot of primary production estimates as a function of ordinal date showing a seasonal pattern in production values in both Stave Lake and Hayward reservoirs. Data are from samples collected in 2003 and 2010 to 2013.

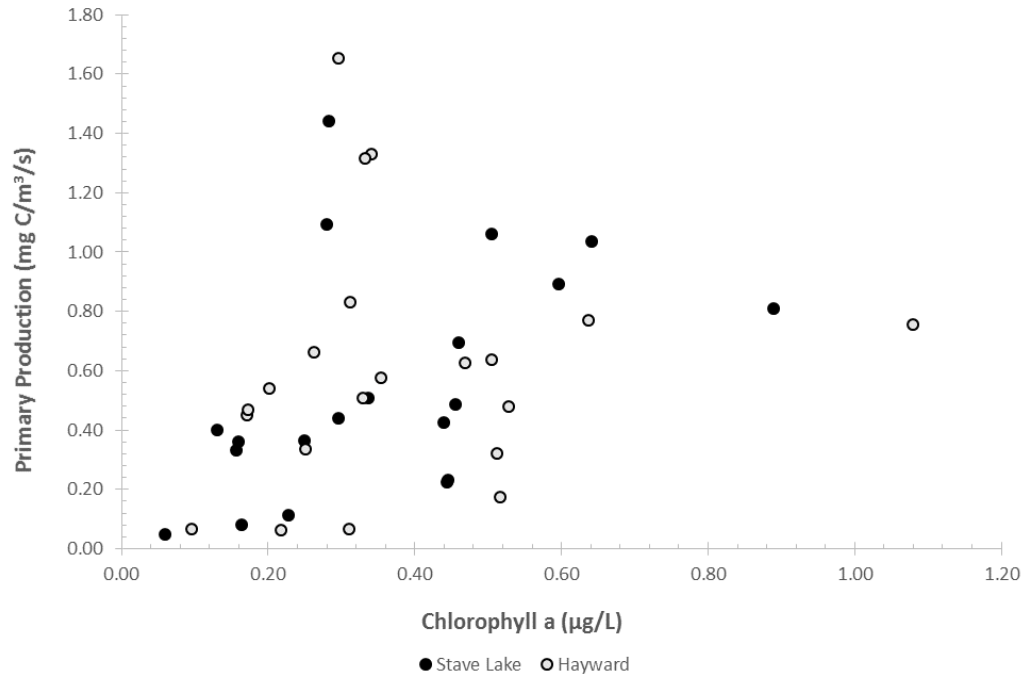


Figure 22. Primary production as estimated through ^{14}C radio assay plotted as a function of Chlorophyll a concentration measured on the same day of sampling for both Stave Lake and Hayward reservoirs.

Generally, primary production was uncorrelated with environmental variables at both sites, regardless of size fraction of organisms (Table 9). A weak positive correlation between organisms $> 20 \mu\text{m}$ and light extinction coefficients was observed in Stave Lake. This is believed to be spurious in nature since similar trends were not observed for other size fractions, in Hayward reservoir and conflicts with documented trends of higher production associated with lower light penetration due to shading (Wetzel 2001). Introducing a sampling interval lag did not significantly improve any of the correlation statistics, and rendered a non-significant correlation between light extinction coefficient and primary production of all sized organisms. The ^{14}C radio assays only reflect ^{14}C assimilation rates measured at the time of sampling.

The ratios of primary productivity among the three size fractions did not differ significantly between Stave Lake and Hayward reservoirs ($\chi^2 = 0.003$, $P = 0.998$). Primary production was dominated by $< 0.2 \mu\text{m}$ organisms (i.e., picoplankton; 57%) and less so by organisms of size 0.2 to $2 \mu\text{m}$ (33%) and 2 to $20 \mu\text{m}$ (10%). There were insufficient data to assess temporal trends over the course of the monitoring period as ^{14}C radio assay data were only collected in years 2003 and 2010 through 2013.

Table 9. Correlation between variables describing environmental conditions at the time of sampling and ¹⁴C primary production estimates. The correlations are broken down by size fraction, as well as the sum of all size fractions. Samples sizes refer to size fractionated samples.

Parameter	n	Correlation by size fraction			
		20 μm	2 μm	0.2 μm	Sum
Stave Lake Reservoir					
Avg. Discharge (m ³ /s)	19	-0.078	-0.070	-0.147	-0.015
Avg. Elevation (m)	19	-0.239	-0.098	-0.114	-0.005
Avg. Retention Rate	19	-0.343	0.198	0.052	0.039
Epilimnion Temp (°C)	19	-0.130	0.020	-0.124	-0.053
Epilimnion Depth (m)	19	-0.192	-0.164	-0.148	-0.200
PAR (μmoles/m ² /s)	11	-0.281	-0.210	-0.333	-0.192
Light Extinction Coeff. (m ⁻¹)	18	0.471	0.166	0.253	0.266
Secchi Depth (m)	18	0.017	0.303	0.157	0.219
TP (μg/L)	19	-0.127	-0.278	0.003	-0.151
TDP (μg/L)	17	-0.250	-0.366	-0.210	-0.329
NO ₃ (μg/L)	17	-0.085	-0.132	-0.096	-0.254
Hayward Reservoir					
Avg. Discharge (m ³ /s)	19	0.065	-0.289	-0.225	-0.116
Avg. Elevation (m)	19	0.096	0.051	0.008	-0.122
Avg. Retention Rate (days)	16	-0.108	0.040	-0.078	-0.121
Epilimnion Temp (°C)	19	-0.263	-0.157	-0.201	-0.156
Epilimnion Depth (m)	-	-	-	-	-
PAR (μmoles/m ² /s)	18	-0.205	-0.106	-0.230	-0.121
Light Extinction Coeff. (m ⁻¹)	18	0.080	0.128	0.074	-0.088
Secchi Depth (m)	18	-0.078	0.076	0.050	0.235
TP (μg/L)	18	-0.155	-0.313	-0.077	-0.245
TDP (μg/L)	17	-0.266	-0.277	-0.114	-0.237
NO ₃ (μg/L)	18	-0.053	-0.178	-0.098	-0.266

3.3.2 Bacteria

Picoplankton data were only collected in years 2010 to 2013. Bio-volumes of pico-cyanobacteria (B_{Cyano}) in both reservoirs ranged from 0.026 mm³/L to 0.984 mm³/L with no significant difference between site geometric means (Stave, 0.135 mm³/L; Hayward, 0.136 mm³/L; $t_{2,56} = 0.108$, $P = 0.914$; Figure 23). A significant annual cycle was observed peaking near the end of September and through October (Figure 23). Given the high degree of correlation between Stave Lake and Hayward reservoir bio-volumes ($r = 0.925$, $P < 0.0001$), there appeared to be no difference in cycle timing or amplitude between the two reservoir sampling sites. Heterotrophic bacteria (B_{Hetero}) had bio-volumes ranging from 0.040 mm³/L to 0.378 mm³/L in both reservoirs with no significant difference in site geometric means (Stave, 0.109 mm³/L; Hayward, 0.114 mm³/L; $t_{2,56} = -0.336$, $P = 0.738$; Figure 24). A more subdued cycling pattern was observed for heterotrophic bacteria where the range of possible values tended to increase rather than a shift in central tendency alone (Figure 24). Site specific values were highly correlated ($r = 0.939$, $P < 0.0001$) with little difference in cycle timing or amplitude.

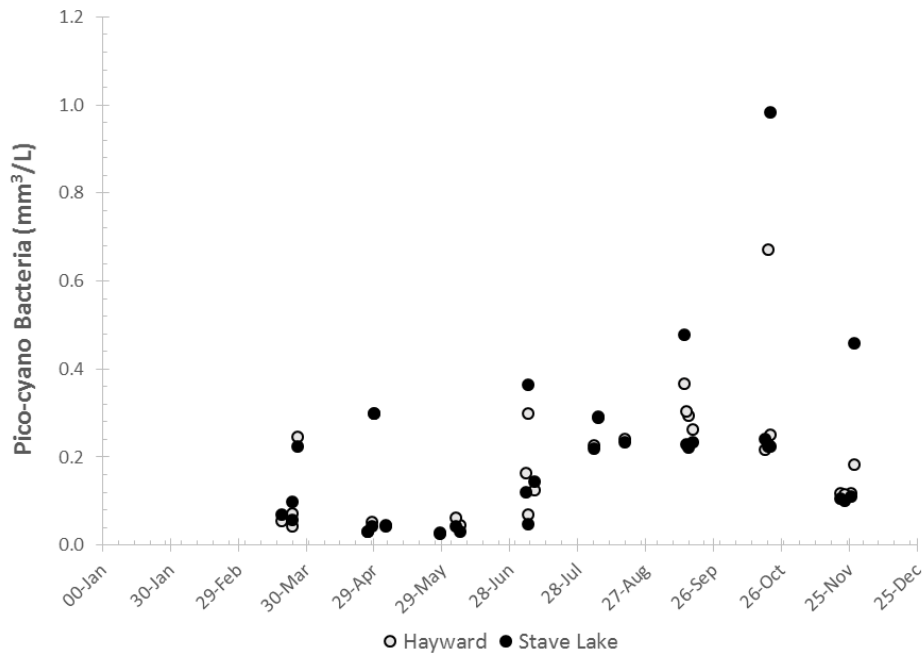


Figure 23. Bio-volume of pico-cyanobacteria plotted as a function of ordinal date showing the seasonal cycle in data collected at the Stave Lake and Hayward reservoir sites in years 2010 to 2013.

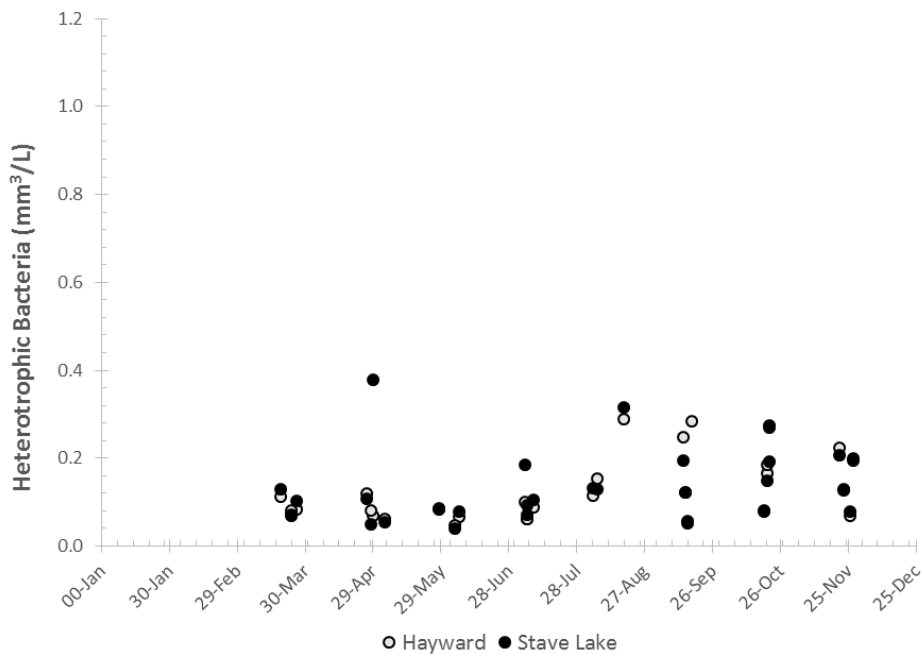


Figure 24. Bio-volume of heterotrophic bacteria plotted as a function of ordinal date showing the seasonal cycle in data collected at the Stave Lake and Hayward reservoir sites in years 2010 to 2013.

Similar periodicity in cycle timing between bacterial forms was confirmed by regression analysis where a significant correlation was observed between the log transformed bio-volumes of the two bacteria types ($r = 0.499$, $P < 0.0001$). The relationship was best described by the following power function:

$$B_{\text{Hetero}} = 0.202 \cdot B_{\text{Cyano}}^{0.306} \quad (R^2_{\text{Adj}} = 0.236)$$

The regression exponential coefficients were found to be similar between the Stave Falls and Hayward sites (0.277 and 0.332, respectively; $t_{2,53} = 0.996$, $P = 0.339$), suggesting that the ratio between bacterial types was consistent between sites and tended to vary in tandem over time. The ratio of bacterial forms itself however, was not constant over time; varying in a predictable fashion as a function of pico-cyanobacteria bio-volume over time. The $B_{\text{Cyano}}:B_{\text{Hetero}}$ ratio tended to be largest during the late summer/fall growing season, while it was much narrower in spring and early summer.

A strong correlation between pico-cyanobacteria bio-volume and the primary production rates of 0.2 μm organisms measured by ^{14}C radio assay was expected, but none was found (Stave: $r = 0.092$, $P = 0.775$; Hayward: $r = 0.191$, $P = 0.532$). One possibility is that the species composition of each sample was highly variable (See section 3.3.3 below). With each species potentially having a different ^{14}C assimilation rate, this could have introduced considerable variability in the production estimates of each sample. Combined with measurement error, this variance could have masked the potential for detecting a significant correlation.

A significant correlation was found between Chl_a concentration and pico-cyanobacteria bio-volume at the Hayward site ($r = 0.389$, $P = 0.045$). This correlation however, appeared to be driven by a single outlier value. With the removal of this outlier, the correlation was no longer significant ($r = 0.035$, $P = 0.860$). Chl_a concentration and pico-cyanobacteria bio-volume were not correlated at the Stave Lake site ($r = 0.286$, $P = 0.126$). The lack of a significant correlation was likely due to the different size fractions that were accounted for in each metric. The Chl_a metric was not size fractionated, while the bio-volume data only considered organisms $< 0.2 \mu\text{m}$ in size.

The bio-volume of pico-cyanobacteria in Stave Lake reservoir was negatively correlated with outflow discharge from Stave Lake reservoir (Table 10); however, this correlation appeared to be driven by a single outlier value. Without outlier, the correlation was no longer significant ($r = -0.340$, $P = 0.071$.)

Significant negative correlations were also observed between NO_3 concentration and pico-cyanobacteria bio-volumes in both reservoir systems (Figure 25). The strength of these correlations increased when the data were re-analysed with an interval lag (Table 10). A between-site comparison of regression slope coefficients and intercepts showed that they were not significantly different from one another (slope: $t_{2,55} = 0.523$, $P = 0.602$; intercept: $t_{2,55} = 0.371$, $P = 0.712$) and could be pooled to form the following exponential regression function:

$$B_{\text{Cyano}} = 0.689 \cdot e^{-0.019 \cdot \text{NO}_3} \quad (R^2_{\text{Adj}} = 0.524, P < 0.0001)$$

The pooled regression without the time lag is as follows:

$$B_{\text{Cyano}} = 0.454 \cdot e^{-0.015 \cdot \text{NO}_3} \quad (R^2_{\text{Adj}} = 0.318, P < 0.0001)$$

Table 10. Correlation coefficients between variables describing environmental conditions at the time of sampling and the bio-volume of pico-cyanobacteria and heterotrophic bacteria. Also included are the correlation coefficients when a time lag equivalent to the duration of the previous sampling interval was introduced to the analysis. The later highlights the relative importance of growing conditions leading to the measured bio-volumes rather than at the time of sampling. Shaded values highlight those correlations deemed statistically significant (P < 0.05)

Parameter	n	Time of Sampling		Sampling Interval Time Lag	
		Pico-cyano	Heterotrophic	Pico-cyano	Heterotrophic
Stave Lake Reservoir					
Discharge (m ³ /s)	30	-0.395	-0.146	-0.236	-0.028
Elevation (m)	30	-0.224	-0.142	0.166	0.063
Retention Rate	30	-0.083	0.216	0.092	0.040
Epilimnion Temp (°C)	30	-0.043	0.028	0.169	0.080
Epilimnion Depth (m)	11	0.458	0.221	0.294	0.178
PAR (μmoles/m ² /s)	29	0.146	0.069	0.454	0.172
Light Extinction Coeff. (m ⁻¹)	29	0.246	0.104	0.011	0.024
Secchi Depth (m)	30	-0.250	-0.203	-0.124	-0.037
TP (μg/L)	29	0.077	0.131	-0.244	-0.040
TDP (μg/L)	29	-0.097	0.087	-0.161	-0.249
NO ₃ (μg/L)	30	-0.453	-0.210	-0.592	-0.239
Hayward Reservoir					
Discharge (m ³ /s)	30	-0.082	-0.143	-0.105	-0.002
Elevation (m)	30	0.006	0.189	-0.233	0.072
Retention Rate (days)	28	-0.139	-0.075	-0.213	-0.286
Epilimnion Temp (°C)	30	-0.051	-0.166	0.323	0.263
Epilimnion Depth (m)	-				
PAR (μmoles/m ² /s)	29	-0.046	-0.159	0.268	0.186
Light Extinction Coeff. (m ⁻¹)	29	-0.073	-0.035	-0.202	-0.188
Secchi Depth (m)	30	-0.048	-0.260	0.200	0.087
TP (μg/L)	30	-0.172	-0.251	-0.336	-0.387
TDP (μg/L)	29	-0.145	-0.402	-0.356	-0.616
NO ₃ (μg/L)	30	-0.507	-0.502	-0.699	-0.541

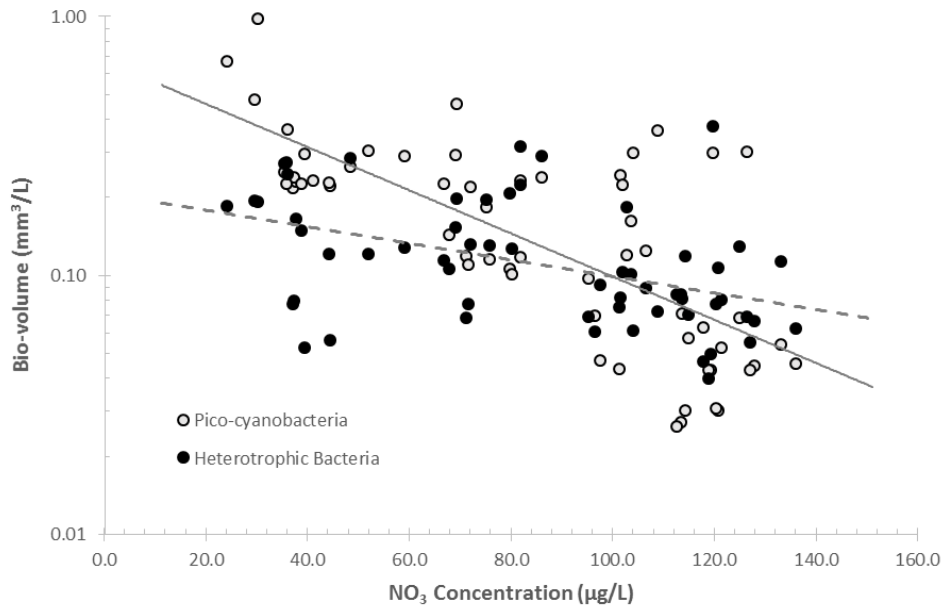
NO₃ concentration was also correlated with heterotrophic bacteria bio-volume in Hayward Reservoir. The relationship with NO₃ at the Stave Lake site only approached statistical significance ($R^2_{Adj} = 0.092$, $P = 0.061$). Regression slope coefficients were similar to one another between sites ($t_{2,54} = -0.618$, $P = 0.539$), as were the intercepts ($t_{2,54} = 0.472$, $P = 0.639$). Pooling the data from both sites resulted in the following exponential regression function relating NO₃ to B_{Hetero} applicable to both sites:

$$B_{Hetero} = 0.207 \cdot e^{-0.007 \cdot NO_3} \quad (R^2_{Adj} = 0.186, P = 0.0003)$$

The pooled regression without the time lag is as follows:

$$B_{Hetero} = 0.186 \cdot e^{-0.006 \cdot NO_3} \quad (R^2_{Adj} = 0.142, P = 0.0019)$$

Figure 25. Pico-bacteria bio-volumes plotted as a function of NO₃ concentration at the time of the previous sampling interval at both the Stave Lake and Hayward reservoir sites. Solid line represents best fit exponential regression for pico-cyanobacteria, while the dashed line is that of the heterotrophic bacteria.



It is clear from the regression coefficients that the variance in NO₃ was most strongly associated with pico-cyanobacterial bio-volume. This was confirmed by multiple regression analysis where the bio-volumes of both bacterial forms were used as predictor variables for NO₃ concentration, which revealed pico-cyanobacteria bio-volume as the dominant consumer of NO₃ ($P < 0.0001$). The role of heterotrophic bacteria was not statistically significant ($P = 0.278$) compared to that of the pico-cyanobacteria. This was consistent with the general notion that in oligotrophic lakes, NO₃ uptake occurs primarily in photosynthetic organisms (second to ammonia), while in heterotrophic organisms, NO₃ uptake is primarily through assimilation of more complex organic compounds often released by cyanobacteria (Wetzel 2001). This inter relationship may explain the observed correlation in bio-volume between the two bacteria types, and hence the weak correlation with NO₃ and heterotrophic bacteria.

In Stave Lake Reservoir, a significant positive correlation was determined between the bio-volume of pico-cyanobacteria and PAR intensity measured with an interval lag (Table 10). No such correlation was found with the heterotrophic bacteria. Given that the cyanobacteria are photosynthetic and heterotrophic bacteria are not, such an outcome was expected. This relationship to PAR intensity however, was absent in the Hayward data set. It is possible that retention time in Hayward reservoir was too short for measurable growth to occur in the reservoir itself and that the population in Hayward lake was governed more by the recruitment of organisms from Stave Lake Reservoir. The fact that the bio-volumes of either pico-cyanobacteria or heterotrophic bacteria did not differ significantly between sites suggests that this may indeed be the case.

A negative correlation between the bio-volume of heterotrophic bacteria and the two time metrics of TDP concentration was found in Hayward Reservoir (Table 10). As with the NO₃ data, it would

appear that growing conditions prior to sampling had a greater role in predicting bacteria bio-volumes (or vice versa) than at the time of sampling. The fact that these significant correlations were only observed in the heterotrophic bacteria is consistent with the general notion that these organisms are typically responsible for the majority share of inorganic phosphorus uptake in freshwater lakes (Kirchman 1994, Wetzel 2001). Regression analysis found that the relationship of time lagged TDP concentration to heterotrophic bacteria bio-volume was best described by the following power function (Figure 26):

$$B_{\text{Hetero}} = 0.107 \cdot \text{TDP}^{-0.422} \quad (R^2_{\text{Adj}} = 0.482, P < 0.0001)$$

A similar relationship was found for Stave Lake however a statistically significant trend was not detected. A significant correlation between heterotrophic bacteria bio-volume and interval lagged TP concentration in Hayward reservoir was likely a function of the high correlation between TP and TDP.

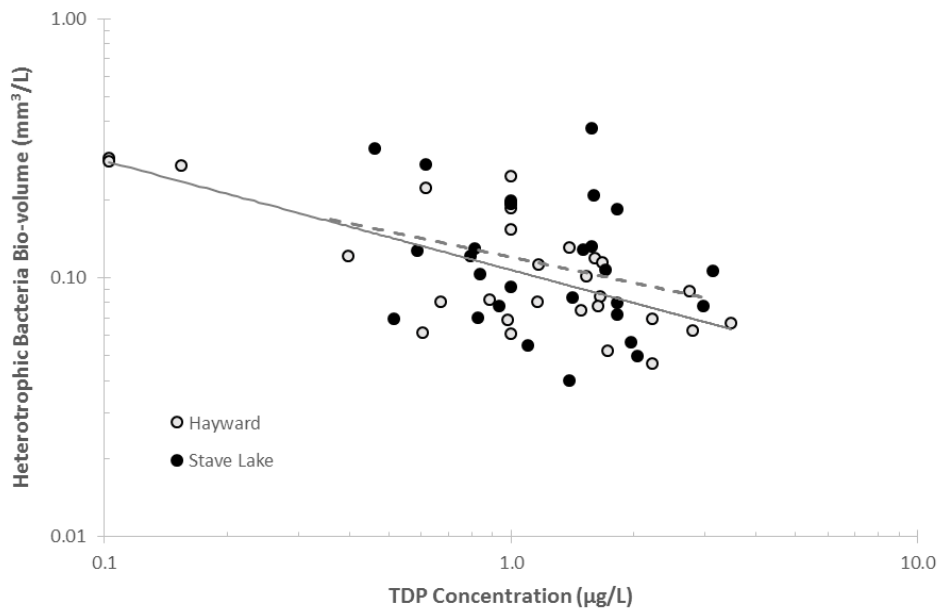


Figure 26. Heterotrophic bacteria bio-volume plotted as a function of total dissolved phosphorous concentration for the Hayward and Stave Lake reservoir sampling sites. Solid line represents the best fit power regression at the Hayward site, while the dashed line represents that of the Stave Lake site.

3.3.3 Phytoplankton

Bio-Volume

The measurements of phytoplankton bio-volume began in earnest in 2003 (n=167) including two measurements taken in Hayward reservoir during fall, 2002. Values from both reservoirs ranged from 0.066 to 1.580 mm³/L with no significant difference between geometric means between sites (Stave, 0.291 mm³/L; Hayward, 0.296 mm³/L; $t_{2,162} = 0.322$, $P = 0.747$; Figure 27). A strong seasonal pattern peaking in early September was observed in both reservoirs (Figure 27). The two datasets were highly correlated ($r = 0.818$, $P < 0.0001$) with a regression intercept of 0.038 mm³/L that was not significantly different from zero ($t_{2,81} = 1.303$, $P = 0.196$) and a slope approaching unity ($b = 0.929$, $t_{2,81} = 12.721$, $P <$

0.0001) suggesting that phytoplankton bio-volumes were very similar between sites, varying in tandem over time.

Correlation analysis between phytoplankton bio-volume and the suite of environmental variables measured at the time of sampling revealed a number of interesting trends (Table 11). In Stave Lake Reservoir, a strong negative correlation with NO_3 and a positive correlation with PAR intensity was observed. Weaker positive correlations were found with Chl_a and reservoir elevation. The correlation with reservoir elevation may have been driven by the strong seasonal trend and the strong correlation with NO_3 concentration. In Hayward Reservoir, associations between phytoplankton bio-volume and reservoir elevation and PAR intensity were absent. The seasonal cycle in Hayward Reservoir was largely absent, while water retention time may have been too short for appreciable growth to occur in response to changing PAR levels. Correlations between phytoplankton bio-volume and NO_3 concentration and Chl_a were of similar magnitude to that in Stave Lake reservoir. This latter observation is consistent with the results showing that phytoplankton bio-volumes are very similar between the Stave Lake and Hayward reservoir sampling sites.

An interval lag revealed additional associations of interest (Table 11). In Stave Lake Reservoir, PAR became the dominant correlated variable, followed by epilimnion water temperature. In both cases, the correlation was positive. Nitrate depletion as a result of this growth was still evident in the data, though the strength of correlation was lower. The correlation with Chl_a concentration was lost, though this was not unexpected as this variable is an indirect measure of present phytoplankton abundance (Wetzel and Likens, 2000). Correlations were also observed with reservoir elevation and water retention times. However, both variables are highly correlated with epilimnion water temperature (Table 5), therefore, may be a reflection of the interrelationship between those variables.

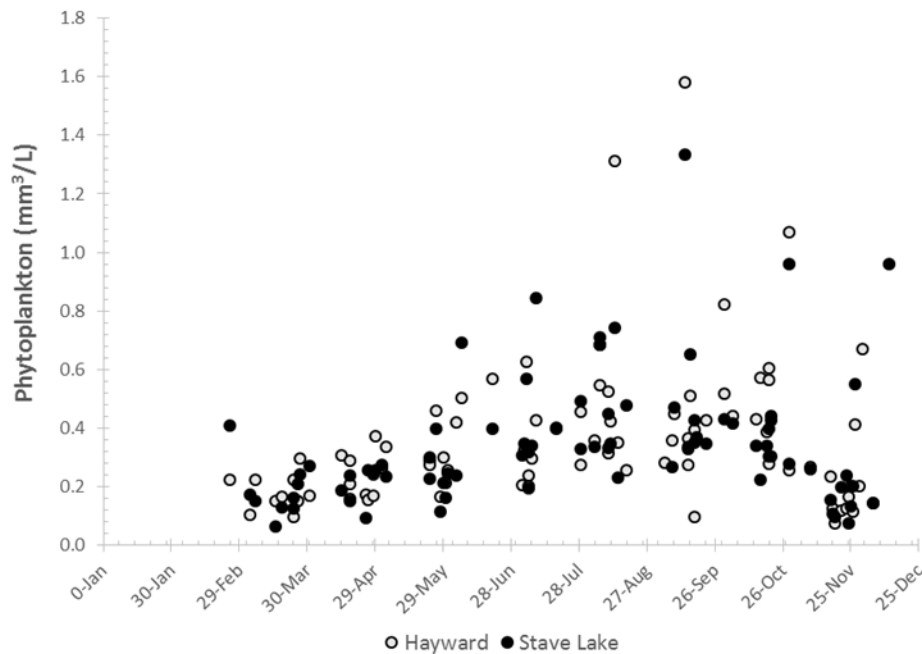


Figure 27. Plot of Phytoplankton bio-volume as a function of ordinal date showing a seasonal cycle in values at both the Stave Lake and Hayward reservoir sites.

Table 11. Correlation coefficients between phytoplankton bio-volume and environmental descriptor variables at the time of sampling and at the previous sampling interval (interval lag) for the Hayward and Stave Falls reservoir sampling sites. Statistical significance ($P > 0.05$) is indicated by shading.

Parameter	n	Stave Lake		Hayward	
		Sampling Date	Interval Lag	Sampling Date	Interval Lag
Discharge (m ³ /s)	84	-0.161	-0.133	-0.126	-0.243
Elevation (m)	85	0.241	0.332	-0.025	-0.091
Retention Rate	80	0.139	0.322	0.109	0.240
Epilimnion Temp (°C)	79	0.167	0.397	0.162	0.419
Epilimnion Depth (m)	6	0.225	0.087	0.655	0.732
PAR (μmoles/m ² /s)	78	0.323	0.479	0.123	0.316
Light Extinction Coeff. (m ⁻¹)	78	0.108	-0.130	-0.204	-0.317
Secchi Depth (m)	77	-0.129	0.052	0.031	0.329
TP (μg/L)	78	0.004	-0.020	-0.068	-0.074
TDP (μg/L)	76	-0.009	0.146	-0.114	0.062
NO ₃ (μg/L)	83	-0.479	-0.359	-0.418	-0.297
Chlorophyll a (μg/L)	82	0.246	0.045	0.278	0.120

In Hayward reservoir, phytoplankton bio-volume was positively correlated with discharge and water retention rate (Table 11), however, these variables are also highly correlated with water temperature. Water temperature was in turn highly correlated with PAR, though both factors have the potential to directly impact phytoplankton growth (Wetzel 2001). The significant negative correlation with light extinction coefficient and the positive correlation with SD are opposite of what would normally be expected. Similar to Stave Lake, phytoplankton bio-volume relationship with NO₃ concentration and Chl_a increased with an interval lag, further strengthening the notion that the phytoplankton bio-volumes at both sites are very similar to one another and appear to vary in tandem.

Typically, increased plankton density is associated with greater shading of incoming light, leading to increased light extinction rates and hence shorter SD. In Hayward Reservoir, the data suggested otherwise and that outside factors may be playing a role in water transparency and that this in turn could have impacted phytoplankton growth in Hayward reservoir. Given the short water retention times, and hence short retention times for the phytoplankton, it seems unlikely that such changes in lake turbidity would have much of an effect on growth, as there would be too little time in the reservoir for an effect to be realized. Thus, the correlation was likely coincidental, and that high turbidity events were occurring at times of lower phytoplankton output from Stave Lake Reservoir. It should be noted that both light extinction and SD were significantly correlated with water discharge out of Hayward Reservoir. It is possible that increased discharges from Stave Falls Dam or higher local inflows during the winter rainy season may be associated with increased within-reservoir turbidity. In fact, McNair (2008) was able to show significant changes in reservoir turbidity in response to local, precipitation driven, high flow events, with turbidity values often exceeding 1 NTU and at times rising above 5 NTU.

Annual geometric mean phytoplankton bio-volumes were found to vary considerably over the course of the monitoring period (Figure 28). Values ranged from 0.195 to 0.391 mm³/L (median = 0.295 mm³/L) and 0.240 to 0.413 mm³/L (median = 0.0278 mm³/L) in Stave Lake and Hayward reservoirs, respectively. Though highly variable, there was no consistent trend in bio-volume over time at either

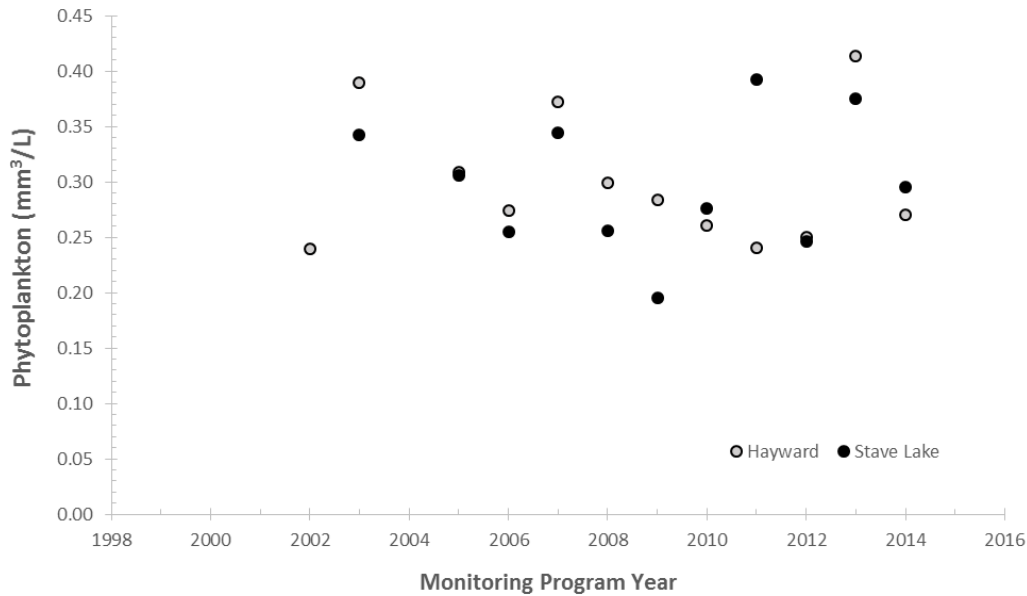


Figure 28. Plot of annual geometric mean phytoplankton bio-volume over the course of the 2000 to 2014 monitoring period for each of the Stave Lake and Hayward reservoir sites.

site (Stave: $r = 0.042$, $P = 0.903$; Hayward: $r = -0.087$, $P = 0.788$). No significant difference between-site geometric means was found (Stave, $0.293 \text{ mm}^3/\text{L}$; Hayward, $0.295 \text{ mm}^3/\text{L}$; $t_{2,19} = 0.200$, $P = 0.843$).

Correlation analysis between annual phytoplankton bio-volume geometric mean and the environmental variable's annual averages revealed no statistically significant associations at both sites (Table 12). Though a number environmental factors appeared to play a role in phytoplankton growth (or vice versa) within years, it would seem these effects were related more to the seasonal cycling of phytoplankton abundance rather than explaining across year variance.

Species Diversity

There were 101 species of phytoplankton identified in both Stave Lake and Hayward Reservoirs over the course of the 2000 – 2014 monitoring period. Eighty-five of these were common to both reservoir sites. Stave Lake Reservoir had 5 unique species, while Hayward Reservoir had 11 unique species. Species unique to either site were generally rarer with 1 to 2 cells/sample (or 10 to 20 cells/L) versus common species of both sites measured at 2280 cells/sample (or 23110 cells/L). A strong seasonal pattern was observed in the number of species encountered peaking in mid-August at a median of 25 species (maximum = 37) and decreasing to a median of 15 species in the winter for both sites (Figure 29).

Table 12. Correlation matrix of annual geometric mean phytoplankton bio-volumes over the course of the 2000 to 2014 monitoring period in Stave Lake a Hayward reservoir and annual summary metrics of the environmental conditions. Sample size for the Stave Lake reservoir correlations was 11 years and that for the Hayward reservoir was 12 years.

Parameter (Annual Average)	Stave Lake		Hayward	
	r	Prob.	r	Prob.
Outflow Discharge (m ³ /s)	0.280	0.404	-0.144	0.655
Elevation (m)	-0.378	0.252	-0.292	0.356
Retention Rate	-0.503	0.115	-0.063	0.846
Epilimnion Temp (°C)	-0.129	0.704	-0.130	0.688
PAR (µmoles/m ² /s)	0.455	0.160	-0.064	0.843
Light Extinction Coeff. (m ⁻¹)	-0.085	0.803	-0.138	0.669
Secchi Depth (m)	-0.048	0.889	-0.115	0.722
TP (µg/L)	0.340	0.306	-0.105	0.745
TDP (µg/L)	0.142	0.697	-0.146	0.651
NO ₃ (µg/L)	-0.295	0.378	-0.035	0.913
Chlorophyll a (µg/L)	0.048	0.889	-0.464	0.129

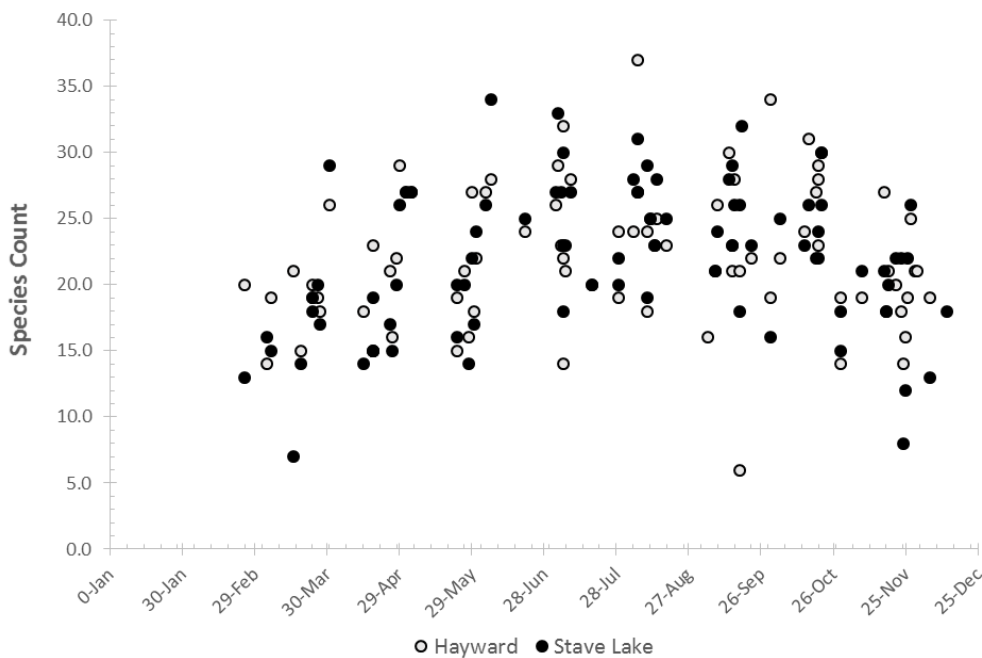


Figure 29. Median species count for a given sample plotted as a function of ordinal sampling date illustrating a seasonal cycle in species diversity in both the Stave Lake and Hayward reservoirs.

For each site, only a small fraction of the total species identified ($n=101$) was collected per sample (6 to 37 species; median of 22) with considerable differences in species composition and relative abundance. 76% of the total cell count was comprised of 7 species (dominated by the blue-green algae *Cyanophyceae* and *Cryptophyceae* flagellate bacteria). Three quarters of the phytoplankton bio-volume was comprised of 15 species, including the seven species of highest cell count. Species diversity index (H; Zar 1974) was not significantly different between sites when measured by species abundance (Stave, 1.123; Hayward, 1.153; $t_{2,200} = 0.294$, $P = 0.769$) or species bio-volume (Stave, 1.483, Hayward, 1.491; $t_{2,200} = 0.106$, $P = 0.915$) despite the fact that some species appeared to be unique to each site.

Species abundance between sites was highly correlated ($r = 0.991$, $P < 0.0001$; Figure 29). Regression analysis showed that cell abundances in Hayward reservoir were generally 92% of that observed in the Stave Lake Reservoir (slope $b = 0.916$, intercept = 74.10). Species bio-volume between sites was also correlated, though the strength of the correlation was not as strong ($r = 0.962$, $P < 0.0001$). Slope of the bio-volume regression ($b = 1.03$) did not differ significantly from unity ($t_{2,102} = 1.089$, $P = 0.279$) and the intercept ($a = 4.66$) did not differ significantly from zero ($t_{2,102} = 0.529$, $P = 0.598$).

A seasonal cycle was observed in both species diversity (Figure 29) and total phytoplankton bio-volume (Figure 27). The periodicity of each cycle differed with peak species diversity occurring 2 to 4 weeks earlier than peak phytoplankton bio-volume. This was observed in both Stave Lake and Hayward Reservoir sites, suggesting that these cycles occurred in tandem. Furthermore, the strong correlation in species specific abundance between sites suggested that the mix of species present in both reservoirs at any given time tended to be similar as well. This was further confirmed by the strong correlation between species count and bio-volume (Stave: $r = 0.585$, $P < 0.0001$; Hayward: $r = 0.549$, $P < 0.0001$). Species count tended to increase logarithmically as sample bio-volume increased, with slopes and intercepts that were not significantly different at both sites. Slope of the logarithmic regressions were 5.59 and 4.91 ($t_{2,163} = 0.572$, $P = 0.568$) and intercepts were 28.75 and 28.05 ($t_{2,163} = 0.434$, $P = 0.665$) at the Stave Lake and Hayward sites, respectively.

Correlation analysis between species count and the set of environmental metric measured at the time of sampling (Table 13) revealed a similar pattern found with the phytoplankton bio-volume data when analysed with an interval lag (Table 11). This similarity would account for the 2 to 4 week lag in peak species count compared to the time of peak bio-volume analyzed with an interval lag. This similarity was especially pronounced with the Stave Lake dataset. The exception was the strong correlation between species count and Chl_a concentration, which was absent in the bio-volume analysis. This association between species count and Chl_a concentration was likely driven by the strong correlation of each variable with phytoplankton bio-volume.

Stave Lake and Hayward reservoirs found species count to have significant positive correlations with epilimnion temperature, PAR and Chl_a concentration and a negative correlation with NO_3 concentrations (Table 13). Stave Lake species count was also positively correlated with water retention

Table 13. Correlation coefficients between phytoplankton species count and environmental descriptor variables at the time of sampling and at the previous sampling interval (interval lag) for the Hayward and Stave Falls reservoir sampling sites. Statistical significance ($P > 0.05$) is indicated by shading.

Parameter	n	Stave Lake		Hayward	
		Sampling Date	Interval Lag	Sampling Date	Interval Lag
Discharge (m ³ /s)	84	0.045	-0.086	-0.097	-0.186
Elevation (m)	85	0.380	0.379	-0.323	-0.318
Retention Rate	80	0.237	0.325	-0.054	0.030
Epilimnion Temp (°C)	79	0.342	0.456	0.212	0.268
Epilimnion Depth (m)	6	0.133	-0.065	-0.085	0.120
PAR (μmoles/m ² /s)	78	0.360	0.427	0.337	0.313
Light Extinction Coeff. (m ⁻¹)	78	-0.063	-0.163	-0.061	-0.258
Secchi Depth (m)	77	-0.057	0.131	0.061	0.335
TP (μg/L)	78	-0.160	-0.021	-0.191	-0.093
TDP (μg/L)	76	0.045	0.112	-0.029	0.037
NO ₃ (μg/L)	83	-0.530	-0.319	-0.293	-0.210
Chlorophyll a (μg/L)	82	0.460	0.245	0.333	0.105

time. Significant correlation was found with water surface elevation in Stave Lake (positive) and Hayward (negative) reservoirs, however in opposite directions. This could be explained by the seasonal variation of elevation in Stave Lake Reservoir coinciding with the seasonal changes in species count while Hayward Reservoir elevation was relatively constant. Introducing an interval lag to the analysis did not alter the above associations for Stave Lake. An interval lag resulted in non-significant correlations with NO₃ and Chl_a concentrations and significant correlations with light extinction coefficients (negative) and SD (positive) for Hayward reservoir, as seen with bio-volume (Table 13). The direction of correlations indicated that both phytoplankton bio-volume and species counts were generally higher at times of higher light transmissivity. This is opposite of what is generally thought to occur in lake systems, where high phytoplankton densities are thought to hinder light transmission (Wetzel 2001). One possible reason for this correlation in Hayward reservoir may be the strong seasonal cycle in light transmissivity (see Figure 7) that doesn't occur in Stave Lake Reservoir. The cause of this cycling is uncertain, but may be associated with rising water turbidity during the winter season due to storm activity (McNair 2008). This correlation is likely to be more coincidental than causal in nature given the low growth of phytoplankton in general during the winter season.

A plot of annual median species count over the course of the monitoring period revealed a strong rising trend over time (Figure 30). This trend was highly significant (Stave: $r = 0.890$, $P = 0.0002$; Hayward: $r = 0.820$, $P = 0.0011$) and did not appear to differ significantly between sites as both slope ($t_{2,19} = 1.223$, $P = 0.236$) and intercept coefficients ($t_{2,19} = 1.223$, $P = 0.236$) were similar to one another.

Correlation analysis between annual phytoplankton species count and the suite of yearly average environmental metrics revealed strong negative correlations with NO₃ concentration at both sites (Table 14). Whether the relationship is casual or coincidental is unclear, though with greater species diversity, there is the possibility that some of the 'newer' species are more efficient consumers of NO₃. This would explain the decreasing trend in NO₃ concentration over time, as well as the fact that annual mean bio-volume did not change significantly over time (Figure 28).

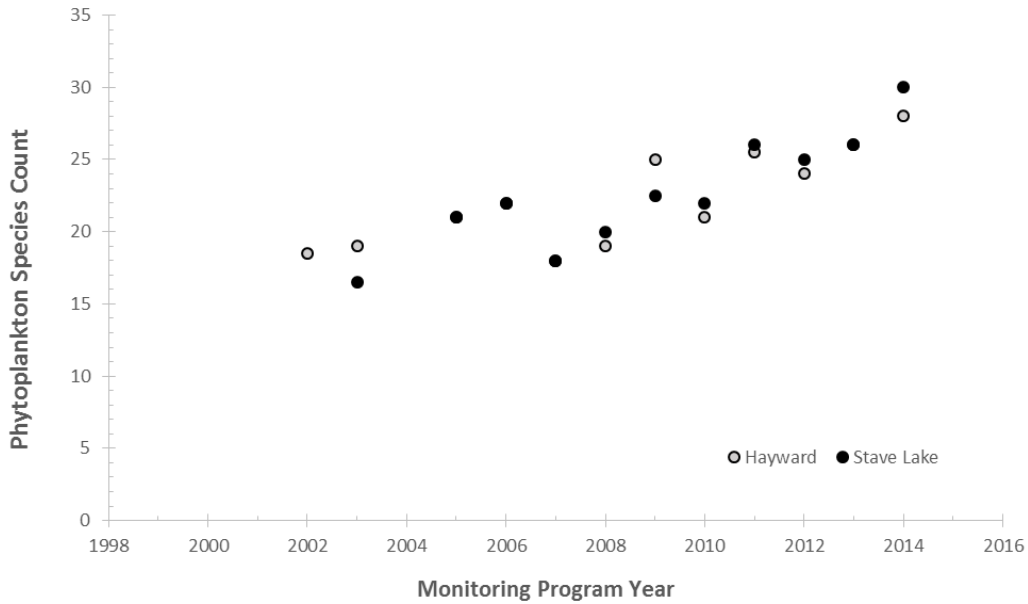


Figure 30. Plot of annual median species count over the course of the 2000 to 2014 monitoring period at the Stave Lake and Hayward reservoir sites.

Table 14. Correlation matrix of annual median phytoplankton species count over the course of the 2000 to 2014 monitoring period in Stave Lake and Hayward reservoirs and annual summary metrics of the environmental conditions. Sample size was 11 years for the Stave Lake Reservoir and 12 years Hayward Reservoir. Shading identifies those correlations that are statistically significant ($P < 0.05$).

Parameter (Annual Average)	Stave Lake		Hayward	
	r	Prob.	r	Prob.
Outflow Discharge (m^3/s)	0.180	0.596	0.059	0.855
Elevation (m)	0.109	0.749	-0.572	0.052
Retention Rate	0.204	0.547	-0.469	0.124
Epilimnion Temp ($^{\circ}C$)	0.058	0.865	-0.027	0.934
PAR ($\mu moles/m^2/s$)	0.566	0.070	0.261	0.413
Light Extinction Coeff. (m^{-1})	-0.538	0.088	-0.195	0.544
Secchi Depth (m)	0.325	0.329	0.306	0.334
TP ($\mu g/L$)	-0.594	0.054	-0.574	0.051
TDP ($\mu g/L$)	-0.239	0.507	-0.528	0.077
NO ₃ ($\mu g/L$)	-0.778	0.005	-0.683	0.014
Chlorophyll a ($\mu g/L$)	0.605	0.049	0.368	0.240

The strong correlation between annual Chl_a concentrations and species count in Stave Lake reservoir suggested the increase in species diversity across years may have been more efficient at photosynthesis than the lesser diverse phytoplankton community early in the monitoring period. This would be consistent with the NO_3 correlation noted above. Annual species count in Stave Lake was negatively correlated with mean annual light extinction coefficients and positively correlated with PAR intensity; however, neither of these were significant. Collectively, the data suggest that species count may be related to changes in the depth and brightness of the pelagic photic zone. Lack of correlation in Hayward reservoir may be explained by the short water retention times limiting the effect of external factors on phytoplankton growth.

Annual average size of organisms (i.e., annual average of the bio-volume conversion factors) in the phytoplankton community decreased over the course of the monitoring period for both sites; however, the decline was not significant (Stave: $r = -0.504$, $P = 0.113$; Hayward: $r = -0.379$, $P = 0.249$; Figure 31). Regression slopes at the two sites were not significantly different ($t_{2,18} = 0.461$, $P = 0.650$), nor were the intercepts ($t_{2,18} = 0.460$, $P = 0.651$), allowing the two data sets to be pooled. The resulting correlation between pooled annual mean phytoplankton size and year of the monitoring program was statistically significant ($r = -0.442$, $P = 0.039$). Thus it would appear that the increase in species count (and hence diversity) involved the introduction of smaller organisms and/or the loss of larger individuals.

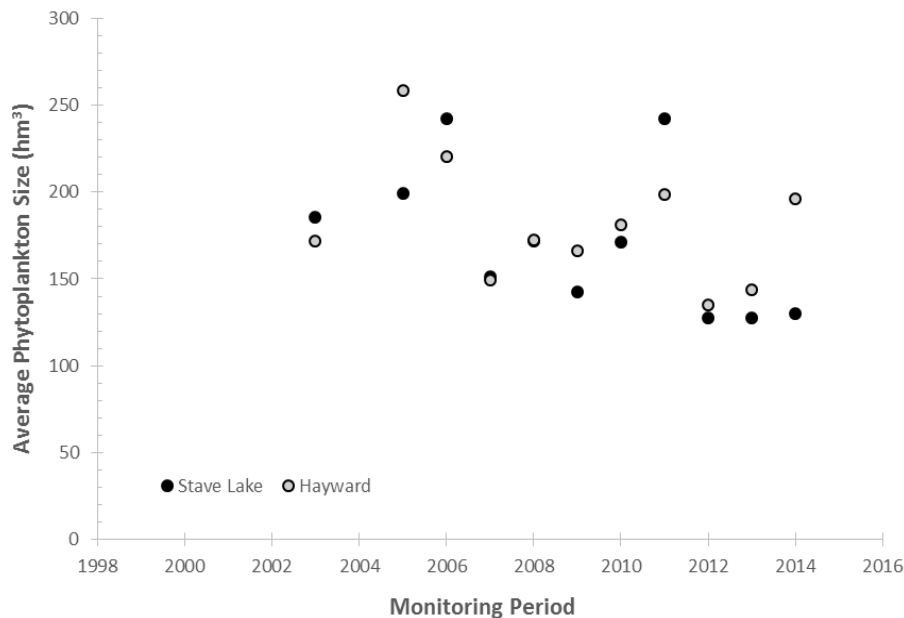


Figure 31. Plot of annual mean phytoplankton size over the course of the 2000 to 2014 monitoring period at the Stave Lake and Hayward reservoir sites.

3.4 Zooplankton

Zooplankton biomass at both sites ranged from 13.9 to 66.5 $\mu\text{g/L}$ with no significant difference in geometric mean biomass between reservoirs (Stave, 29.08 $\mu\text{g/L}$; Hayward, 27.27 $\mu\text{g/L}$; $t_{2,106} = 1.120$, $P = 0.233$; Figure 32). A strong seasonal pattern of zooplankton biomass peaking in late September was observed at both reservoir sites (Figure 32). Cycle periodicity was similar between sites with slightly higher values occurring in Stave lake reservoir. Moreover, regression analysis of the non-transformed data was able to show that the two datasets were highly correlated with one another ($r = 0.660$, $P < 0.0001$), with a slope that was not significantly different from unity ($t_{2,52} = 0.586$, $P = 0.560$) and an intercept that was not significantly different from zero ($t_{2,52} = 0.021$, $P = 0.983$). Thus, values between sites tended to vary in tandem, but often expressed a high degree of between-site residual error.

Correlation analysis of zooplankton biomass against the suite of environmental variables measured at the time of sampling, including metrics related to phytoplankton production, revealed a strong negative correlation with NO_3 concentration at both the Stave Lake and Hayward reservoir sites (Table 15). Since zooplankton are not direct consumers of NO_3 , this was likely an indirect relationship through the consumption of phytoplankton (Table 11). This is supported by the significant positive correlations with phytoplankton bio-volume at both sites and with phytoplankton species count in Stave Lake. A strong positive correlation between zooplankton biomass and depth of the epilimnion was also present in Stave Lake reservoir at time of sampling and with an interval lag; however, it may be coincidental due to the strong season cycles seen in both metrics. Alternatively, it is possible that the increase in epilimnion depth provides an increase in usable habitat with the ability to support a larger population of zooplankton. Such effects of deepening epilimnion levels have been reported in a number

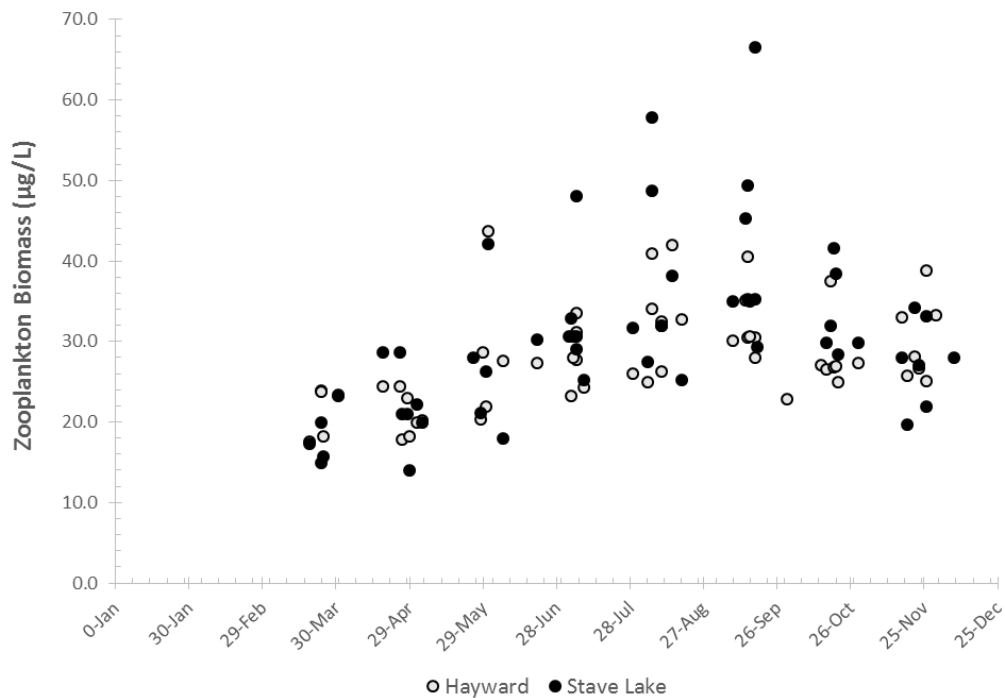


Figure 32. Zooplankton biomass plotted as a function of ordinal sampling date illustrating seasonal patterns over the course of the 2000-2014 monitoring period at the Stave Lake and Hayward reservoir sites.

Table 15. Correlation coefficients between zooplankton biomass and environmental descriptor variables at the time of sampling and at the previous sampling interval (interval lag) for the Hayward and Stave Falls reservoir sampling sites. Shaded areas indicate statistical significance ($P > 0.05$).

Parameter	n	Stave Lake		Hayward	
		Sampling Date	Interval Lag	Sampling Date	Interval Lag
Discharge (m ³ /s)	54	-0.144	-0.187	0.087	0.012
Elevation (m)	54	0.229	0.402	-0.427	-0.372
Retention Rate	51	0.075	0.193	-0.369	-0.180
Epilimnion Temp (°C)	52	0.245	0.550	0.141	0.310
Epilimnion Depth (m)	23/4	0.542	0.447	0.785	0.374
PAR (μmoles/m ² /s)	53	0.221	0.413	0.161	0.175
Light Extinction Coeff. (m ⁻¹)	53	0.053	0.133	0.017	-0.280
Secchi Depth (m)	53	-0.238	-0.076	-0.041	0.310
TP (μg/L)	51	-0.218	-0.341	-0.277	-0.238
TDP (μg/L)	48	-0.251	-0.121	-0.336	-0.352
NO ₃ (μg/L)	54	-0.560	-0.416	-0.521	-0.334
Chlorophyll a (μg/L)	54	0.091	-0.075	0.141	0.077
Phytoplankton Bio-volume (mm ³ /L)	54	0.331	0.141	0.303	0.376
Phytoplankton Species Count	54	0.281	0.264	0.182	0.284

of lake systems, though none have specifically examined this relationship in reservoirs (Berger et al. 2006, Cantin et al. 2011, Kuns and Sprules 2000, Longhi and Beisner 2009).

A similarly strong positive correlation between zooplankton biomass and epilimnion depth was found in Hayward Reservoir, but because of the low sample size ($n = 4$), it was not statistically significant. Thermoclines rarely developed in Hayward reservoir during the monitoring period, causing the low sample size. Given the short retention time of water in the system, a causal relationship was considered unlikely. Rather, the development of a thermocline in Hayward reservoir was often linked with strong thermocline development in Stave Lake Reservoir. It would seem more probable that the high biomass in Hayward Reservoir was due to a concurrent rise in biomass in Stave Lake Reservoir at those times, and subsequent immigration to the reservoir.

Also significant was a negative correlation between zooplankton biomass and reservoir elevation in Hayward Reservoir. Scatterplot analysis of the data showed that the correlation was driven mostly by samples taken when the reservoir was below 40m for maintenance purposes (Figure 33A). One possible explanation is that the drop in water surface elevation reduced reservoir volume, causing a higher concentration of zooplankton. For this to occur however, the zooplankton must be sufficiently mobile to swim against the prevailing current and not get flushed out of the reservoir over time. Studies carried out by Wicklum (1999) showed that this was likely the case, where in small shallow lakes systems similar in size to Hayward Reservoir, zooplankton actively avoided near shore areas (active avoidance hypothesis; Siebeck 1969, 1980). Moreover, some species were found to show active avoidance of outlet areas that was stronger in response than simple shoreline avoidance. Both shoreline avoidance, and outlet avoidance behaviours in zooplankton could help explain the concentrating effects of reservoir drawdown, and hence the observed increase in biomass. It should be noted however that the mobility of zooplankton is limited and that it only can slow the rate of emigration from a lake system (Wicklum

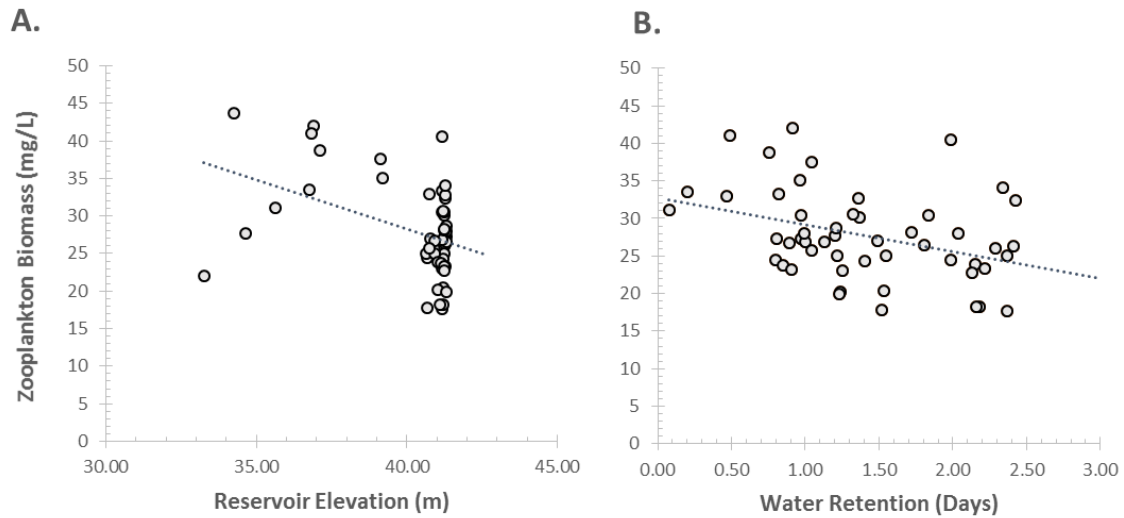


Figure 33. Plot of zooplankton biomass as a function of reservoir elevation (A) and water retention time (B) at the time of sampling for Hayward Reservoir, 2007 to 2014.

1999), even though swimming behaviours can respond to increased water velocity and turbulence (François-Gaël et al. 2015).

As expected, a significant negative correlation between water retention time and zooplankton biomass was also observed (Figure 33B). Unlike the reservoir elevation data, the bi-variate distribution of water retention time data was more uniform throughout the range of water retention values, suggesting that other factors may be involved. In early September when zooplankton biomass was still high in both reservoirs (Figure 32), water retention times in Hayward reservoir were particularly low due to high Stave Falls Dam outflows for operational purposes (Figure 4). These high discharges appeared to be flushing Stave Lake's high summer biomass of zooplankton into Hayward Reservoir. When forward stepwise regression was used to account for the linkage between Stave Lake and Hayward Reservoir zooplankton biomasses ($R^2_{Adj} = 0.430$, $P < 0.0001$), the only other variable found to make a significant contribution was reservoir elevation ($R^2_{Adj} = 0.159$, $P = 0.0020$). Water retention no longer was a significant explanatory factor. The two variables together were able to explain 59% of the variance in Hayward Reservoir zooplankton biomass. This outcome supports the hypotheses that water retention was a coincidental variable linked by seasonal timing, while reservoir elevation appeared to be more of a causal factor.

If large scale emigration of zooplankton occurs from Stave Lake Reservoir, particularly during the late summer when outflow discharges are typically high, a negative correlation between biomass and outflow discharge would be expected. This however, was not apparent in the data. Active avoidance behaviours likely play a role, but only to a limited extent. Alternatively, the longer water retention time in Stave Lake Reservoir (36 days) provides enough time for sufficient zooplankton growth to replenish the lost biomass, and during the summer growing period, increase overall biomass. Thus, the longer water retention time in Stave Lake Reservoir, and hence the potential for population growth, likely masks the effect of varying outflow discharge.

An interval lag reduced the strength of significant correlations with zooplankton biomass at time of sampling and in the case of phytoplankton bio-volume in Stave Lake Reservoir, changed the

significance of the relationship (Table 15). Interval lag also introduced positive correlations with water temperature and PAR intensity in Stave Lake, suggesting that these variables were important factors influencing zooplankton growth leading up to the time of sampling. The effect of PAR intensity was likely through its role in governing phytoplankton growth, which is consistent with the positive correlation observed between zooplankton biomass and phytoplankton bio-volume at the time of sampling. Water temperature was likely both a direct and indirect factor influencing zooplankton growth (Wetzel 2001). Reservoir water surface elevation was positively correlated in Stave Lake and negatively correlated in Hayward Reservoir. This difference in correlation direction is likely a cause of the targeted rise in reservoir elevation each year (defined in the WUP) timed with the seasonal growth in zooplankton biomass (Table 15). The significant negative correlation with TP concentration in Stave Lake was likely indirectly linked to zooplankton biomass through the effects of heterotrophic bacteria growth (Section 3.3.2). The heterotrophic bacteria, like the phytoplankton, would be vulnerable to zooplankton feeding, such that increases in their number would lead to greater foraging opportunity. The presence of greater heterotrophic bacteria numbers may also have facilitated the growth phytoplankton (Wetzel 2001).

Similar patterns of change were observed in Hayward Reservoir when an interval lag was added to the correlation analyses (Table 15). In general, correlations that were significant at the time of sampling tended to have lower correlation coefficients while factors associated with phytoplankton growth and TDP gained prominence. As was seen in Stave Lake Reservoir, epilimnion temperature became significantly correlated with zooplankton biomass in Hayward reservoir. However, because of the short water retention time, this and correlations with phytoplankton growth were likely coincidental in nature. Significant correlations with light extinction coefficients (negative) and SD (positive) was determined. As was seen in section 3.3.3, the direction of correlation with each variable was opposite than expected if increased phytoplankton growth was the main causal factor. Rather phytoplankton growth, and hence zooplankton growth, seemed to do more poorly with the loss of light transmissivity. This was likely due to precipitation, local turbidity events, and short retention times for both phytoplankton, and to a lesser extent, zooplankton.

Over the 8 years of zooplankton data collection, there was no annual trend in the highly variable data (Figure 34). Regression analysis determined the slope of zooplankton biomass was not significantly different from zero at both sites (Stave: $t_{2,6} = 0.440$, $P = 0.675$; Hayward: $t_{2,6} = 0.158$, $P = 0.879$). Annual mean biomass values at the two sites appeared to vary in concert, and were therefore highly correlated ($r = 0.913$, $P = 0.0015$). Slope of the regression did not differ significantly from unity ($t_{2,6} = 0.076$, $P = 0.941$), but a paired comparison of values showed that there was a persistent difference across all years ($t_{2,6} = 6.054$, $P = 0.0005$). On average, annual mean zooplankton biomass in Hayward Reservoir was 9% less than Stave Lake Reservoir.

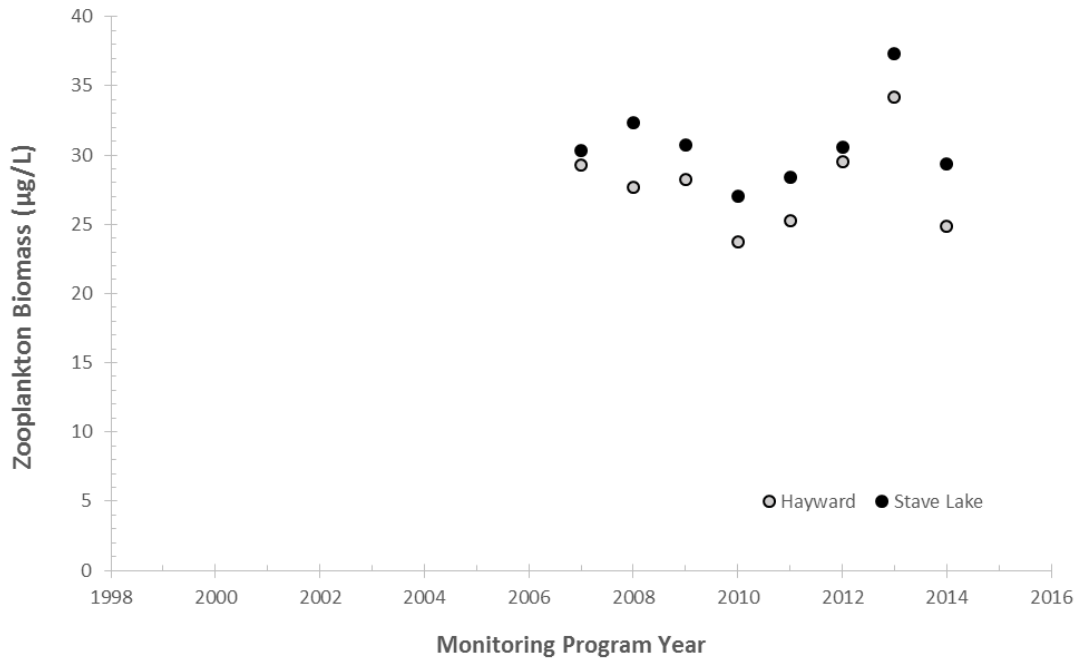


Figure 34. Annual mean zooplankton biomass over the course of the 2000 to 2014 monitoring period for the Stave Lake and Hayward reservoir sites.

A significant negative correlation between average zooplankton biomass and yearly average Hayward Reservoir elevation was found, suggesting that this could be a factor explaining the between-site difference (Table 16). This correlation however appeared to be driven by a single outlier reservoir elevation value, and was therefore potentially spurious in nature. This was confirmed by forward stepwise regression where the residual values of a between-site regression analysis were used as a dependent variable against reservoir elevation. The analysis found no relationship between the two variables ($R^2_{Adj} = -0.002$, $P = 0.917$), clearly showing that reservoir elevation did not modulate the differences in zooplankton biomass between sites on an inter-annual scale. Exploration with other environmental variables found the annual mean TDP concentration in Stave Lake Reservoir was the best predictor for between-site differences ($R^2_{Adj} = -0.738$, $P = 0.0063$). Together, Stave Lake Reservoir zooplankton concentration and annual mean TDP concentration were able to explain 96% of the variance in Hayward Reservoir annual mean zooplankton biomass.

Also in Hayward Reservoir, a positive correlation was found between annual mean zooplankton biomass and annual mean phytoplankton bio-volume (Table 16). The time that these organisms spend in Hayward Reservoir is likely too short for this to be a causal relationship within the reservoir itself, and is more suggestive of a predator-prey relationship in Stave Lake Reservoir. Within Stave Lake Reservoir however, there was no indication of such a relationship as annual mean zooplankton biomass was not correlated with any of the environmental or biological variables (Table 16). It is possible that the observed correlation in Hayward Reservoir was simply spurious in nature.

Table 16. Correlation matrix of annual mean zooplankton biomass over the course of the 2000 to 2014 monitoring period in Stave Lake and Hayward reservoirs and annual summary metrics of the environmental conditions. Sample size for the Stave Lake reservoir correlations was 8 years and for both sites. Shading identifies those correlations deemed statistically significant ($P < 0.05$).

Parameter (Annual Average)	Stave Lake		Hayward	
	r	Prob.	r	Prob.
Outflow Discharge (m ³ /s)	-0.438	0.278	-0.102	0.811
Elevation (m)	-0.373	0.363	-0.797	0.018
Retention Rate	-0.052	0.902	-0.532	0.175
Epilimnion Temp (°C)	-0.039	0.927	-0.414	0.308
PAR (μmoles/m ² /s)	0.429	0.289	0.038	0.928
Light Extinction Coeff. (m ⁻¹)	0.356	0.387	0.447	0.267
Secchi Depth (m)	-0.425	0.293	-0.454	0.259
TP (μg/L)	-0.373	0.362	-0.271	0.517
TDP (μg/L)	-0.010	0.982	-0.140	0.742
NO ₃ (μg/L)	-0.184	0.663	-0.057	0.892
Chlorophyll a (μg/L)	-0.174	0.681	-0.183	0.665
Phytoplankton Bio-volume (mm ³ /L)	0.200	0.634	0.786	0.021
Phytoplankton Species Count	0.042	0.922	0.024	0.954

Dividing the zooplankton biomass data by corresponding estimates of density yielded some insight on the average size of the zooplankton over the course of the monitoring period (Figure 35). The geometric mean size of zooplankton across all sampling periods was 3.85 μg and 3.86 μg for Stave Lake and Hayward reservoirs, respectively. In Stave Lake Reservoir, there was a general tendency for average size to decline over the course of the 7-year period ($r = -0.681$, $P = 0.043$) where no change was detected in Hayward reservoir ($r = -0.041$, $P = 0.923$). Generally, Hayward followed a similar trend to Stave Lake except for year 2013 when zooplankton size had increased substantially compared to the sizes seen in Stave Lake reservoir. A review of the species-specific biomass and density data revealed there was a greater frequency of larger individuals caught in the Hayward samples that year. Given the small number of samples collected each year and the heavily skewed distribution of size data, it's possible that this may have simply been a sampling anomaly. The pooled correlation coefficient was $r = -0.368$, which was no longer statistically significant ($P = 0.239$).

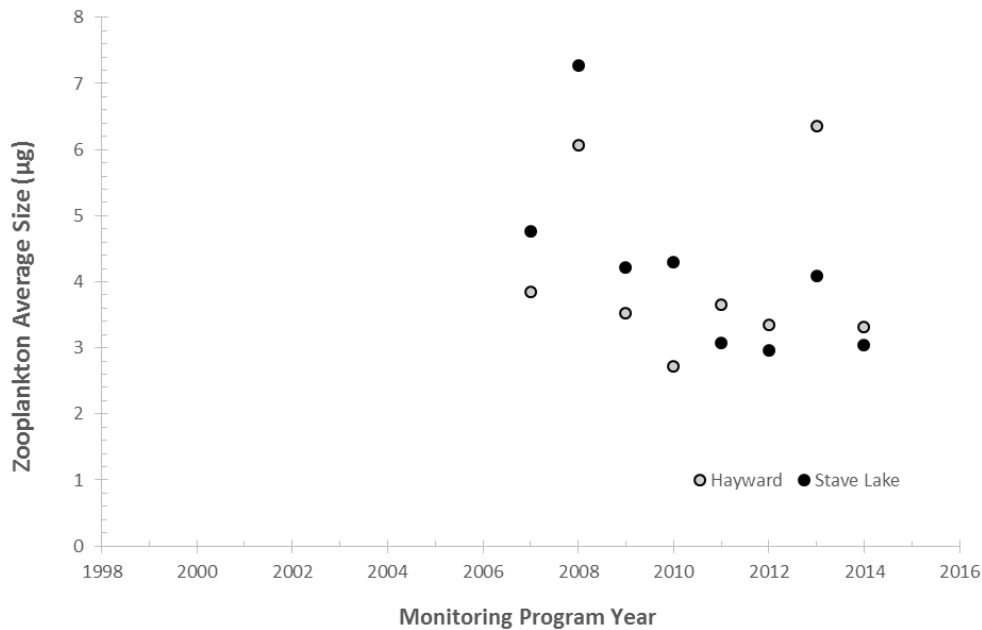


Figure 35. Plot of annual geometric mean zooplankton size, expressed in terms of mass (μg) for the Stave Lake and Hayward reservoir over the course of the 2000 to 2014 monitoring period.

4 General Discussion

4.1 General

Analysis of the data revealed a number of interesting trends, both within years and between years, which allowed one to address the impact hypotheses and management questions identified by the WUP CC. However, because of low statistical power and study design issues, there is significant uncertainty associated with all of the study outcomes discussed in Section 4.2.

Results determined that environmental variables measured at the time of biological sampling were not necessarily indicative of past growing condition. Because growth is exponential, environmental conditions early in a growth period are likely to have a greater influence on a population metric than conditions closer to the time of biological sampling. This effect was crudely accounted for by testing for significant correlations with environmental data collected during the previous sampling period. However, time intervals between samples were highly variable, ranging from 2 to 178 days. Thus, interpretation of these statistical tests was limited to a binary outcome, i.e., whether past environmental conditions were more closely associated with changes to bio-volume or biomass metrics than those measured at the time of sampling.

Over the study period, reservoir hydrology did not vary appreciably from year to year during the summer growing season. As a result, there was insufficient variance in reservoir hydrology to illicit a measurable environmental or biological response, particularly if the growth or productivity changes were subtle. This lack of between-year variance severely reduced statistical power of the monitoring study.

4.2 Impact Hypotheses

A total of 10 hypotheses were identified for testing in the present monitoring program. Collectively, they formed a general impact hypothesis model that attempted to describe the interrelationship of various environmental factors on primary and secondary production, as well as inter-trophic interactions (BC Hydro 2004). This model served as the backdrop for decision making during the WUP process (Failing 1999). One of the goals in the present monitoring study was to validate this model, and hence provide some empirical support to the expected neutral or positive benefits of WUP implementation. The impact null hypotheses are individually discussed below in light of the monitoring study results.

H₀1: Average reservoir concentration of Total Phosphorus (TP), an indicator of general phosphorus availability, does not limit pelagic primary productivity.

Over the course of the 14-year monitoring study, TP concentrations (geometric mean 2.2 µg/L) were among the lowest observed in BC lakes (Stockner and Beer 2004), limiting pelagic primary production and indicating an ultra-oligotrophic system (TP values < 3 µg/L; Wetzel 2001). Further, the ratio of nitrogen to phosphorus (N:P) was nearly 47:1 (NO₃ to TP); much greater than the threshold ratio of 23:1 that is indicative of a severe phosphorus limitation (Wetzel 2001). A downward trend in annual TP concentration suggested that this limitation had worsened over time. The cause for this declining trend is unclear as it was not correlated with any of the measured hydrological or environmental variables. This was particularly the case for the Stave Lake Reservoir. Hayward TP levels did vary slightly from Stave Lake levels in a predictable way, generally increasing as maximum reservoir elevations increased. This was particularly evident prior to 2006 when the maximum water surface elevations in Hayward Reservoir were roughly 1.4 m higher. The reason for the association could not be determined with the data at hand, though its effect on water retention times may be a possible factor. The only other metric found to be uniquely correlated to TP in Hayward reservoir was the bio-volume of heterotrophic bacteria, which are effective consumers of inorganic phosphorus. However, the bio-volume of heterotrophic bacteria was not correlated with water retention time, contradicting the hypothesis that these organisms may have played a role. Unfortunately, TP data was only collected at a single location mid reservoir. As a result, possible impacts associated with phosphorus loading in the reservoir could not be assessed.

There were a few occasions when TP concentrations were an order of magnitude higher than average levels. These tended to occur following large storm related flow events and appeared to be of very short duration. There was no evidence in the data showing that these TP pulses had a measurable effect on the productivity of either reservoir.

Despite the differences between reservoir sampling sites and occasional TP pulses, TP concentrations overall were at levels that could severely limit primary production in both Stave Lake and Hayward reservoir. H₀1 can be rejected.

H₀2: Relative to the availability of phosphorus as measured by the level of total dissolved phosphorus (TDP), the average reservoir concentration of nitrate (NO₃) does not limit pelagic primary productivity. Nitrate is the dominant form of nitrogen that is directly bio-available to algae and is indicative of the general availability of nitrogen to pelagic organisms.

For both reservoirs, TDP concentrations (1.3 µg/L; geometric mean) were indicative of an ultra-oligotrophic system with severe nutrient limitation (Wetzel 2001). TP and TDP values were significantly correlated with one another, thus shared many of the same trends with other environmental metrics, including reservoir hydrology, and also declined over the course of the monitoring period. With the TDP data, between reservoir differences were no longer apparent, and the relationship to heterotrophic bacterial bio-volume was much stronger. This in turn led to a much stronger seasonal cycle in TDP values than was observed with the TP metric.

In both reservoirs, variance in NO₃ concentrations was the result of a strong seasonal cycle that was highly correlated with phytoplankton bio-volume at time of sampling. NO₃ concentrations were highly correlated between reservoirs with no differences in mean concentrations. Nitrate concentrations (94.1 µg/L; mean) were consistently below the 200 µg/L threshold commonly used to define an ultra-oligotrophic system (Wetzel 2001). Thus, the availability of nitrogen in both Stave Lake and Hayward reservoirs was a strong impediment to pelagic primary production. Furthermore, NO₃ steadily declined over the course of the monitoring period, indicating that the severity of this nitrogen limitation increased over time.

As noted in H₀1, the N:P ratio was determined to be 47:1 when calculated with TP. This ratio increased to 72: 1 when TDP was substituted for TP. In both cases, the ratio was much greater than the 23:1 threshold indicative of a severe phosphorus limitation when compared to the availability of nitrogen. Thus, the availability of phosphorus is much more limiting to primary production in the two reservoirs than the availability of nitrogen, even though both nutrients are in very short supply. H₀2 cannot be rejected.

H₀3: Water retention time (τ_w) is not altered by reservoir operations such that it significantly affects the level of TP as described by Vollenweider's (1975) phosphorus loading equations (referred to here as TP(τ_w)).

Instantaneous (daily) water retention times in the Stave Lake Reservoir (39.2 days; geometric mean) was almost 25 times longer than in Hayward Reservoir (1.6 days). In Stave Lake Reservoir, both TP and TDP concentrations were uncorrelated with water retention time regardless of how the metrics were summarised over time (i.e., time of sampling, or yearly average of sampling dates). Thus, for the Stave Lake Reservoir, H₀3 cannot be rejected (this applies to NO₃ concentrations as well).

For Hayward Reservoir, the outcome was not as clear. On a mean annual basis both TP and TDP were highly correlated with both median and maximum reservoir elevation and water retention times. The strongest correlation was with maximum reservoir elevation; however, all three metrics were inter-correlated with one another making it difficult to distinguish the principle causal factor. When the data were examined across all sampling dates, none of these metrics were correlated with TP concentration, and only water retention time was correlated to TDP concentration. These contradictory outcomes make it difficult to draw a definitive conclusion. It is clear that the decision to reduce the maximum reservoir elevation in Hayward reservoir for dam safety reasons had an impact on the phosphorus cycle.

The causative agent or pathway for this impact however, remains undetermined. For Hayward Reservoir, H₀₃ can neither be rejected nor accepted without further study.

H₀₄: Water temperature, and hence the thermal profile of the reservoir, is not significantly altered by reservoir operations.

Epilimnion temperatures in Stave Lake Reservoir followed a strong seasonal pattern that was highly correlated with water surface elevation and water retention time. These correlations were likely coincidental, as the seasonal cycle for each parameter overlapped considerably. Water temperature was more strongly correlated with PAR intensity than either of the reservoir hydrology variables. Forward stepwise regression determined that with PAR intensity, reservoir elevation was no longer a significant correlate and that the positive correlation with water retention time increased.

Stave Lake Reservoir data suggested that from early spring to late summer, epilimnion temperatures increased as a result of solar heating. The rise in temperature occurred faster and reached higher peak values when water retention times were longer. Concurrently, a linear increase of epilimnion depth occurred throughout the summer. During this time, inflow discharge tended to modulate the rate at which epilimnion depth increased; the higher the inflow, the shallower the epilimnion depth. In mid-September when the reservoir was drawn down to manage the fall high inflows, water retention times also dropped, both as a function of decreased water elevation and increased Alouette powerhouse outflows. Additionally, epilimnion development appeared to suddenly stop in mid-September, triggered by a turnover event that mixed the various thermal layers in the pelagic zone. As a result of these conditions, average epilimnion water temperature also declined (Figure 9). Thus it would seem that reservoir draw down and potentially Alouette River inflows indeed have the capability of altering the thermal profile of Stave Lake Reservoir. For Stave Lake Reservoir, H₀₄ can be rejected.

Epilimnion temperature in Hayward Reservoir generally reflected that of Stave Lake Reservoir. In fact, 94% of the variance in Hayward water temperatures can be accounted for by corresponding measurements in Stave Lake Reservoir. No other reservoir metric was correlated with water temperature, suggesting that water retention time was too short for significant temperature changes to occur by conduction, convection or solar heating. As a result, reservoir operations likely do not impact water temperature in Hayward Reservoir. Thus, H₀₄ cannot be rejected for Hayward reservoir.

H₀₅: Changes in TP as a result of inter annual differences in reservoir hydrology (i.e., TP(τ_w)) are not sufficient to create a detectable change in pelagic algae biomass as measured by levels of chlorophyll a (Chl_a). [This hypothesis can only be tested if H₀₃ is rejected]

Both annual geometric mean TP and TDP concentrations were found to decline steadily over the course of the 14-year monitoring period. This decline did not appear to be related to reservoir operations, especially in Stave Lake Reservoir, failing to reject H₀₃. Nevertheless, a decline did occur and a test of H₀₅ can proceed.

Pelagic algae biomass was measured using two key metrics; Chl_a concentration and phytoplankton bio-volume. For both metrics, no significant trend over time was observed and no significant correlation with TP or TDP was determined at time of sampling or with an interval lag. Since the differences in TP and TDP were insufficient to create a detectable change in algae biomass, H₀₅ cannot be rejected.

It should be noted however, that the differences in TP and TDP were correlated with heterotrophic bacteria bio-volumes. Being more efficient consumers of inorganic phosphorus, the seasonal rise in heterotrophic bacteria bio-volumes in summer was associated with a corresponding drop in TDP concentration. This in turn drove a weaker correlation with the TP variable. Unfortunately, only 4 years of heterotrophic bacteria data were collected, which was too small of a sample size to determine if annual TP or TDP concentration had an impact on the annual mean bio-volumes of these organisms.

H₀6: Independent estimates of algae biomass based on TP and Secchi disk transparency (SD) prediction equations are statistically similar, suggesting that neither non-algal turbidity, nor intensive zooplankton grazing, are significant factors that influence standing crop of pelagic phytoplankton (Carlson 1980, cited in Wetzel 2001)

Carlson (1980, 1992) had devised a trophic assessment system devised to identify non-nutrient related limitations to lake and reservoir primary production. At the core of the assessment system are comparisons of predicted Trophic State Indices (TSI) that are calculated separately using TP, Chl_a and SD data. The TSI based on the grand mean TP was determined to be 15.1 and 17.4 for the Stave Lake and Hayward Reservoirs, respectively, which was well below the TSI threshold of 30 indicating an ultra-oligotrophic condition (Wetzel 2001). TSI(Chl_a) was 13.6 and 15.3 for Stave Lake and Hayward reservoir, respectively, which was also below the TSI threshold for ultra-oligotrophy. TSI(SD) however was more than doubled these values (Stave, 36.8; Hayward, 37.9) and was more indicative of a mesotrophic type ecosystem. The similarity between TSI(TP) and TSI(Chl_a) indicates that primary production in the both reservoirs was in line with expectations given the severe limitations in TP availability. The large difference between TSI(Chl_a) and TSI(SD) however, suggested that the presence of dissolved organic matter or fine silt turbidity may be reducing water transparency in addition to the presence of phytoplankton (See H₀7).

Stave Lake Reservoir is a glacially fed system with the potential for fine glacial silt entering the system. However, correlation analysis between SD and various metrics of phytoplankton biomass in Stave Lake Reservoir failed to establish a linkage between the two variables, regardless of time scale or whether an interval lag was introduced to the analysis. The same outcome was obtained when the analysis was carried out with light extinction coefficients, which tend to be highly correlated with SD (Wetzel 2001). Thus, for Stave Lake Reservoir, the presence of turbid conditions may indeed be a potential limiting factor to primary production, but the availability of nutrients may be such that primary production is not high enough to reach those limits. It is also possible that the variation in reservoir turbidity is not high enough to illicit a detectable response in the analyses done in the present monitoring study, but may be limiting production nonetheless. The available data do not allow one to distinguish between the two possible outcomes, though the likelihood of the latter was considered low given the severity of phosphorus limitation (see H₀1).

In Hayward reservoir, turbidity also tends to increase in the fall and winter months as a result of storm activity (McNair 2008). In this case, significant correlation with some of the phytoplankton biomass metrics was found. However, it is unclear whether these associations were causal or coincidental in nature. Given the short water retention times of the reservoir, it seems unlikely that phytoplankton would be in the reservoir long enough for a detectable response to develop. In this case, the correlation may be more a function of overlapping seasonal cycles, and thus coincidental, than causal in nature.

Given the available data, H₀6 cannot be rejected and that neither non-algal turbidity nor intensive zooplankton grazing appeared to be significant factors influencing the standing crop of pelagic phytoplankton. The potential for non-algal turbidity related limitations on primary production does exist, but it would seem that nutrient limitations prevent production levels from reaching those limits.

H₀7: The effect of non-algal turbidity on pelagic algae biomass, as indicated by the difference in independent predictions of TSI(Chl_a), TSI(TP) and TSI(SD) (Carlson 1980, cited in Wetzel 2001), does not change as a function of reservoir operation.

Overall, the data suggests that non-algal turbidity is largely unrelated to reservoir operations (Table 7). There is some evidence that reservoir conditions prior to sampling may have had some influence on non-algal turbidity in Stave Lake Reservoir, but this may be coincidental with other seasonal cycles in the reservoir. Work by McNair (2008) in Hayward Reservoir suggests that turbidity levels tend to increase with episodic storm events, which tend to occur less frequently in the summer than in the fall through spring seasons. Anecdotal observations suggest that this may apply to Stave Lake Reservoir as well, which may be exasperated by rain on snow events during periods of glacial snow melt. Given the available information, H₀7 cannot be rejected.

H₀8: The ratio of ultra-phytoplankton (< 20 µm in size) to micro-phytoplankton (20-200 µm in size) abundance is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.

Phytoplankton abundance, bio-volume and species data were collected in most years of the program, though samples were not filtered to establish size classes. The use of filters to separate size classes of phytoplankton was employed for the 5 years of C¹⁴ radio assay work. 57% of assimilated ¹⁴C was composed of phytoplankton < 0.2 µm in size (i.e., pico-phytoplankton), 33% by ultra-phytoplankton and 10% by micro-phytoplankton. The 5 years of data collection was insufficient to establish a temporal trend, though values were similar between years and nearly identical between reservoir sites.

The average size of phytoplankton was found to systematically decrease over time (Figure 31) while the average number of species counted increased each year (Figure 30). Thus, over the course of the monitoring period, while total bio volume tended to remain unchanged, species richness increased, total abundance increased, but the average size of organisms decreased. These patterns were observed at both reservoir sites. Given these strong, persistent changes through time, H₀8 can be rejected.

H₀9: The size distribution of the pelagic zooplankton population (an indicator of fish food bioavailability as larger organisms tend to be preferred over small ones) is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.

Zooplankton size data showed a significant downward trend over time in Stave Lake Reservoir (Figure 35). Hayward Reservoir followed a similar trend with the exception of 2013 where the average size was much larger compared to Stave Lake reservoir. Given that the Hayward zooplankton population was likely entrained from Stave Lake Reservoir, this discordance draws into question the validity for the Stave Lake trend. The data are inconclusive, and as a result, H₀9 cannot be rejected.

H₀10: Primary production, as measured through ¹⁴C inoculation, is not altered by reservoir operations and hence, does not change through time with the implementation of the WUP Combo 6 operating strategy.

^{14}C radio assays were only carried out in years 2003 and 2010 through to 2013. This time series was insufficient to test hypothesis H_010 . However, production estimates were correlated with Chl_a concentrations, which were collected in most years of the monitoring period. These latter data were highly variable from year to year, but there did not appear to be a consistent trend over time. This was consistent with the lack of correlation between ^{14}C production estimates and any of the measured operations related environmental variables. The outcome was the same regardless of the size fraction tested, or whether a sampling interval lag was considered. Though H_010 could not be tested directly, the weight of evidence suggests that primary production, though variable from year to year, did not change as a result of WUP implementation failing to reject H_010 .

4.3 Management Questions

The WUP CC identified four key management questions pertaining to the pelagic productivity of Stave Lake and Hayward reservoirs (BC Hydro, 2004) and are addressed as follows.

a) What is the current level of pelagic productivity in each reservoir, and how does it vary seasonally and annually as a result of climatic, physical and biological processes, including the effect of reservoir fluctuation?

Pelagic primary production in Stave Lake Reservoir, regardless of the metric used to measure it (Chl_a , ^{14}C radio assay, pico-cyanobacteria bio-volume, or phytoplankton bio-volume) was found to vary seasonally in a predictable fashion with peak values typically occurring in September. Correlation analysis found that this seasonality was most strongly correlated with the availability of light experienced at the previous sampling interval (largely driven by seasonal PAR cycles, modified by turbidity driven changes in light extinction coefficients). This is to be expected when growth is exponential; environmental conditions early in a growth period have a much larger effect on a final population size than conditions closer to the end of that period (unless there is some catastrophic impact). A coincidental rise in water temperature may also have played a role, increasing the rate of metabolic processes that drive the growth and development of individual organisms. A consequence of these seasonal peaks in primary production was an accompanying drop in NO_3 , largely due to the consumption of this nutrient being much greater than the rate of replenishment. A similar but much weaker relationship was observed between the availability of phosphorus and the growth of heterotrophic bacteria.

The seasonal cycles in light availability and water temperature tended to match that of reservoir fluctuation as determined by the WUP making it difficult to untangle the effects of operationally driven changes in reservoir hydrology from natural seasonal cycles. Despite this similarity, reservoir fluctuation, water retention times or outflow discharge was never as strongly correlated with primary production as with the non-operational variables. Though one cannot rule out the possibility of an operations-related impact on within year variance in primary productions, it was considered unlikely.

Though all metrics of pelagic primary production were highly variable from year to year, there was no significant trend over time. Nor was the inter-annual variance significantly correlated with the year to year differences in reservoir hydrology. Observed changes over time included the increase of diversity of phytoplankton species and decrease of average organism size. This too was not correlated with inter-annual variance in reservoir hydrology, though one could not rule out the possibility that the change may have been due to some other operations-related impact either occurring at a longer timescale or was not measured in this monitor. The increase in diversity however, was correlated with a

gradual decline of TP and NO₃ concentrations. The casual direction of this relationship however, is uncertain.

Zooplankton biomass was also found to vary significantly over time. Zooplankton biomass cycled in a seasonal basis, reaching peak values in September followed by a rapid decline in biomass that appeared to be timed with the rapid drawdown in reservoir elevation required each year to accept the upcoming high inflows of the fall and winter seasons. This was interpreted as possible evidence that zooplankton are flushed out of the reservoir due to reservoir operations. This flushing action however, did not appear to be a significant factor contributing to between year variance in zooplankton biomass. Though variable from year to year, zooplankton biomass did not significantly change over the course of the monitoring period. This inter annual variance was not correlated with any of the yearly reservoir hydrology metrics and appeared to be independent of reservoir operations.

Overall, results of the monitoring study indicated that reservoir operations did not have a significant impact on pelagic primary production, though it could not be ruled out entirely. If reservoir operations did have an impact, it was too small relative to the effects of other variables to be detected. Zooplankton biomass did appear to be impacted by the seasonal drawdown of Stave Lake Reservoir, flushing a proportion of the population out of the system. This however, did not translate into an impact that could explain the year to year variance in zooplankton biomass.

b) If changes in pelagic productivity are detected through time, can they be attributed to changes in reservoir operations as stipulated in the WUP, or are they the result of change to some other environmental factor?

As noted above, pelagic primary production did vary from year to year, but there was neither a significant increasing nor decreasing trend over time. Furthermore, these inter-annual differences were not correlated with reservoir hydrology and were therefore likely driven by other environmental factors. The only possible operations related impact detected in this monitoring program was a flushing of zooplankton organisms during the seasonal drawdown of Stave Lake Reservoir. This however, did not appear to contribute to inter annual variability in zooplankton biomass. Like primary production, there was no correlation with the annual metrics of reservoir hydrology. It would appear that pelagic productivity on an annual basis was independent of reservoir operations and that other environmental factors were at play (e.g., nutrient loading, availability of light, and water temperatures).

c) To what extent would reservoir operations have to change to 1) illicit a pelagic productivity response; and 2) improve or worsen the current pelagic state of productivity?

Because pelagic primary production appeared to be independent of reservoir operations, there is little that can be done to improve pelagic productivity by altering reservoir hydrology. Production in both Stave Lake and Hayward reservoirs is severely nutrient limited, particularly with respect to phosphorus concentration. If there was a causal relationship between primary production and reservoir operations, the lack of nutrients would likely mask any type of response.

The only alteration that could potentially be of benefit to the reservoir productivity would be to delay the fall drawdown period, which would in turn delay the flushing of zooplankton organisms at that time of year. This could potentially extend the period of high food availability for fish later into the fall potentially increasing overwintering survival. The likelihood that such a positive benefit would occur is uncertain, and could not be tested with the available data. There was too little between year variance in the drawdown timing and magnitude to create measurable responses. It should be noted that delaying

the fall drawdown period has been shown to increase the frequency, duration and magnitude of downstream flooding during the late fall and winter seasons (Failing 1999).

- d) Given the answers to the management questions above, to what extent does the Combo 6 operating alternative improve reservoir productivity in pelagic waters, and what can be done to make improvements, whether they be operations based or not.**

The available empirical evidence suggests that the Combo 6 operating alternative had little to no impact on the pelagic productivity of Stave Lake Reservoir. Though one could argue that the September drawdown had a potential impact on the zooplankton community, this operational requirement existed prior to the WUP and is not unique to the Combo 6 alternative.

The Stave Lake and Hayward reservoirs were shown to be ultra-oligotrophic with the lack of nutrient availability as the primary limitation to overall productivity. As has been shown in the Alouette Lake Reservoir (Harris et al. 2010), nutrient supplementation may be the only management action capable of creating a measurable increase to pelagic productivity.

5 Conclusion

The pelagic productivity monitoring program was reasonably successful in collecting data suitable for addressing the management questions and associated impact hypotheses identified by the WUP CC (Failing 1999). Issues with data quality and experimental design were overcome, but at the cost of statistical power in the analyses. Nevertheless, the impact hypotheses were successfully tested, if not statistically, then by inference, reasoning and weight of evidence (Table A).

The collected data and impact hypothesis outcomes collectively showed that pelagic primary productivity, regardless of the metric used in this study (Chl_a , ^{14}C radio assay, pico-cyanobacteria bio-volume, or phytoplankton bio-volume), varied both seasonally and across all years of the monitoring program. Seasonal variance in primary production followed a predictable cycle with peak values occurring in September. The primary drivers for this cycling pattern appeared to be the availability of light and to a lesser extent water temperature. Reservoir hydrology did not appear to be a factor, though the variables used to describe it (inflow discharge, water retention time, and reservoir elevation) were themselves correlated with light availability and water temperature. Inter-annual variance in primary production, though high, did not appear to follow a particular trend over time. Neither was the inter-annual variance correlated with any of the reservoir hydrology variables. It would appear that primary production for the most part is independent of reservoir operations as defined by the Combo 6 operating alternative.

For the limited number of years that data were collected, zooplankton biomass was also relatively stable over time, though there was considerable inter annual variance. Like with primary production, this inter-annual variance was not correlated with reservoir operations. The data do suggest however, that the September drawdown operation in preparation for the fall/winter rainy season may flush individuals out of Stave Lake reservoir. The impact of this on overall reservoir productivity remains uncertain and will require a separate study to resolve. It should be noted that the September drawdown operation is not a feature unique to the Comb 6 alternative as it is required to minimise downstream flooding regardless of operating strategy.

Both the Stave Lake and Hayward reservoirs are severely nutrient poor and are considered ultra-oligotrophic systems. Given that pelagic productivity is for the most part independent of reservoir operations, it is considered unlikely that any kind of change to the Combo 6 operating alternative would lead to measurable changes in trophic status.

6 References

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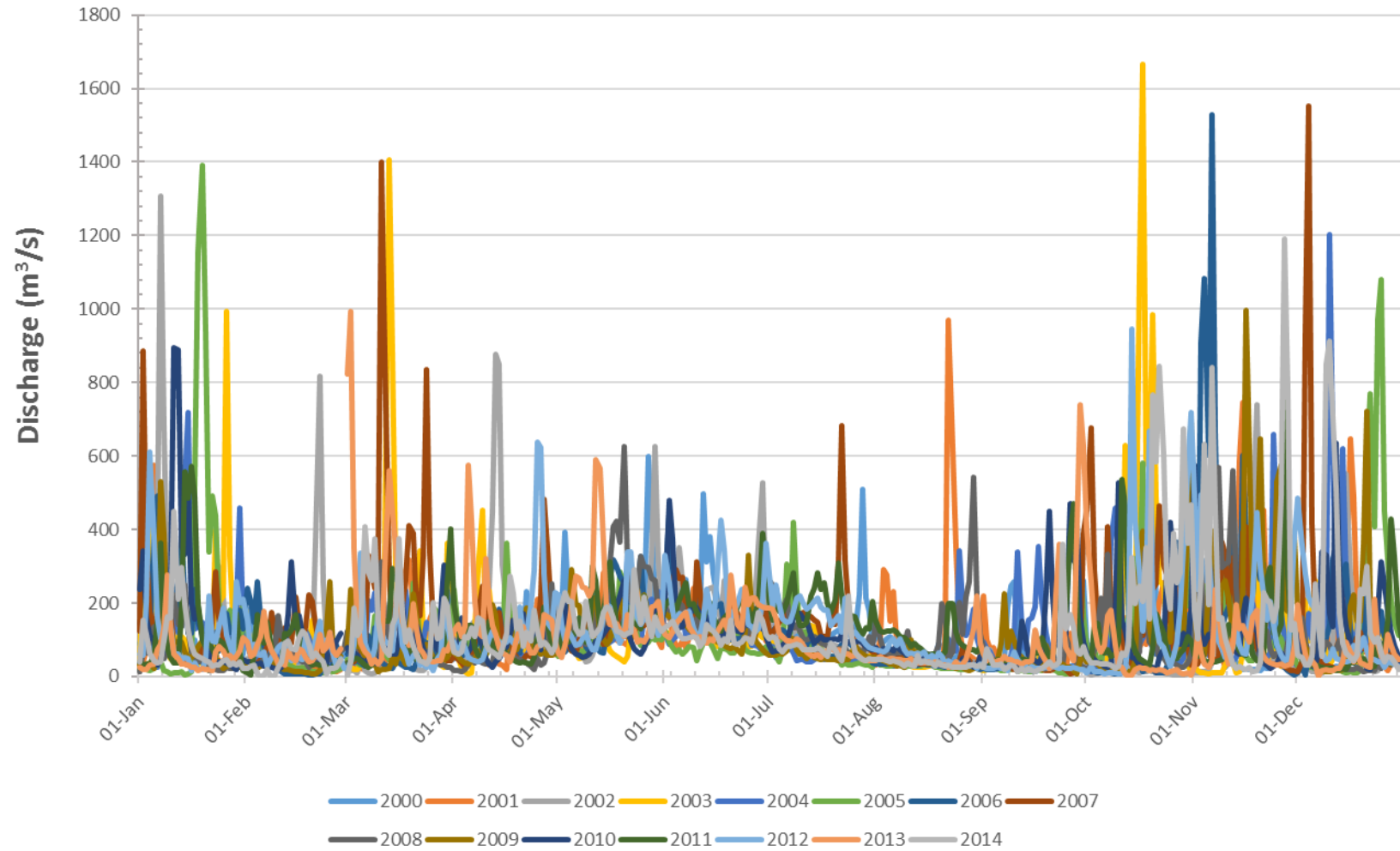
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Appendix 1: Stave Lake and Hayward Reservoir Hydrology (2000 – 2014)

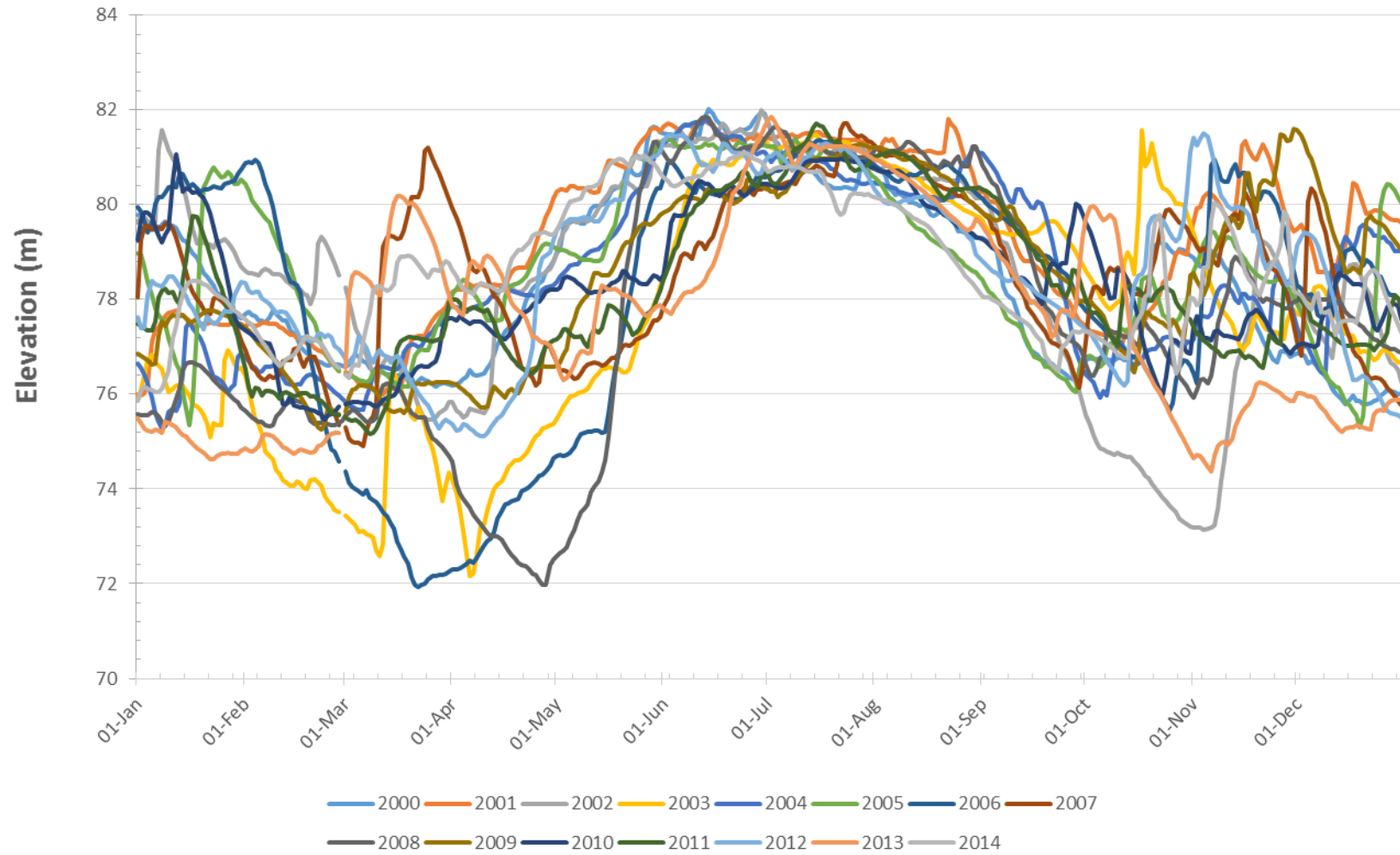
A.

Local Inflow to Stave Lake Reservoir



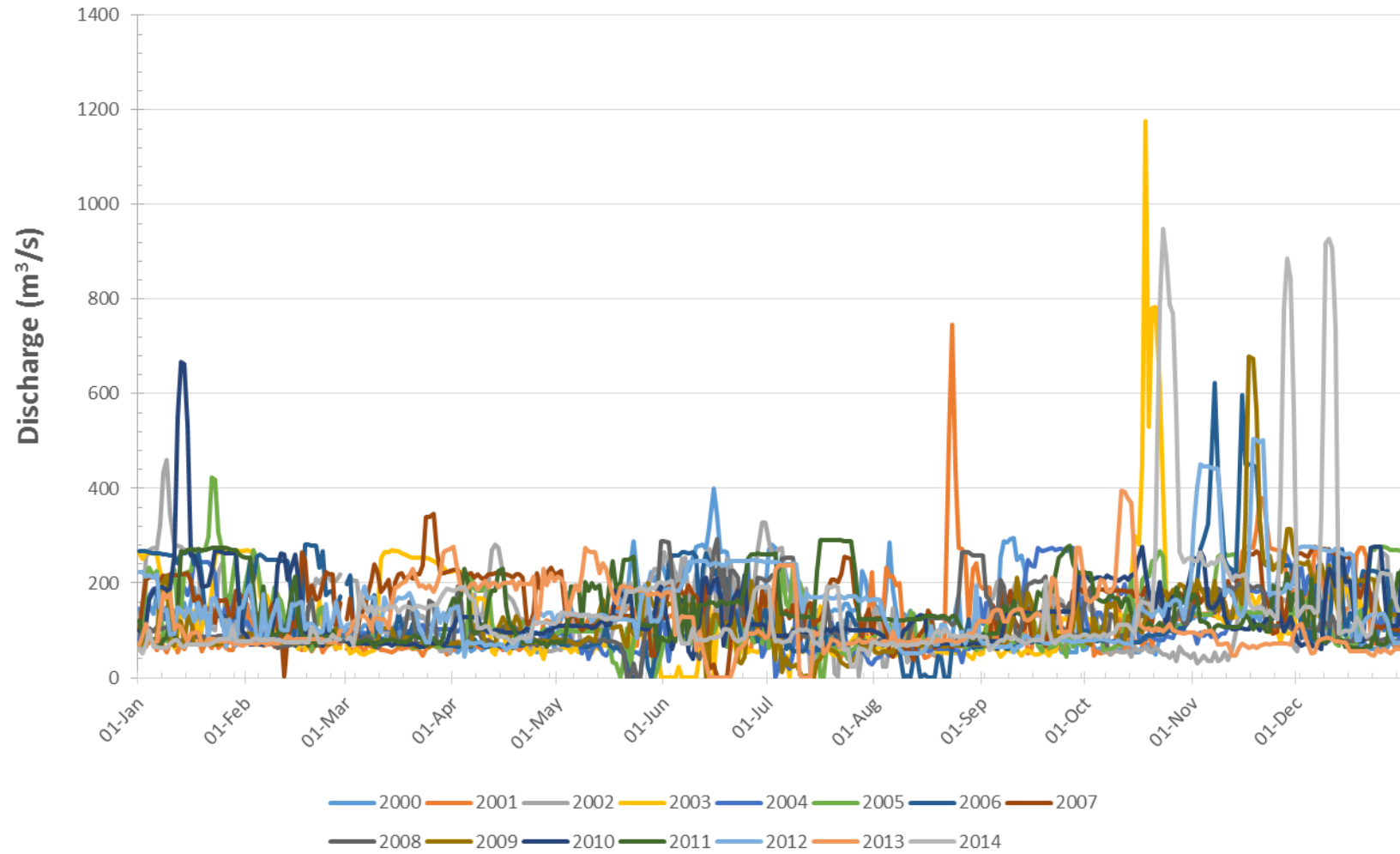
B.

Stave Lake Reservoir Water Surface Elevations



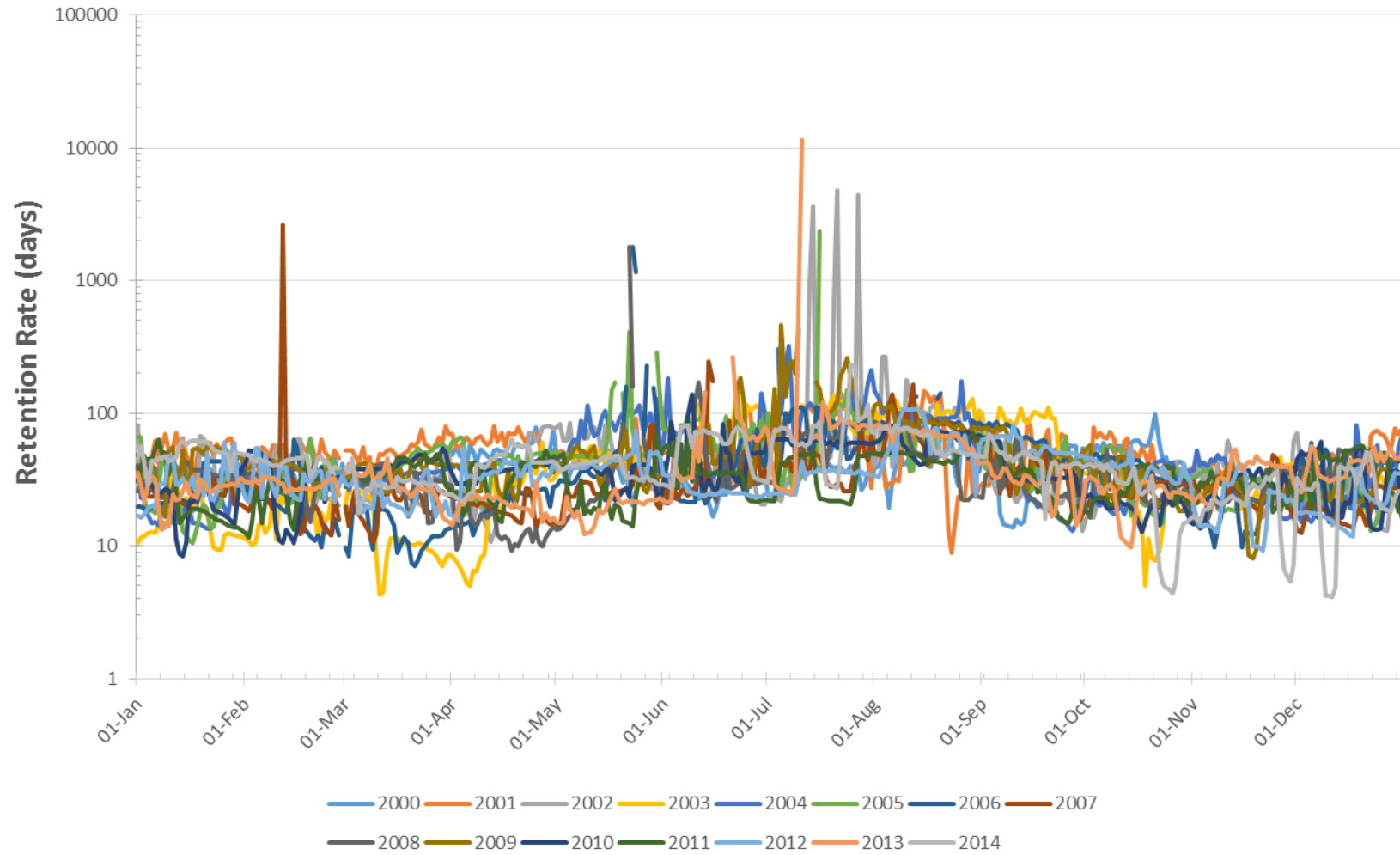
C.

Total Outflow from Stave Falls Facility to Hayward Reservoir



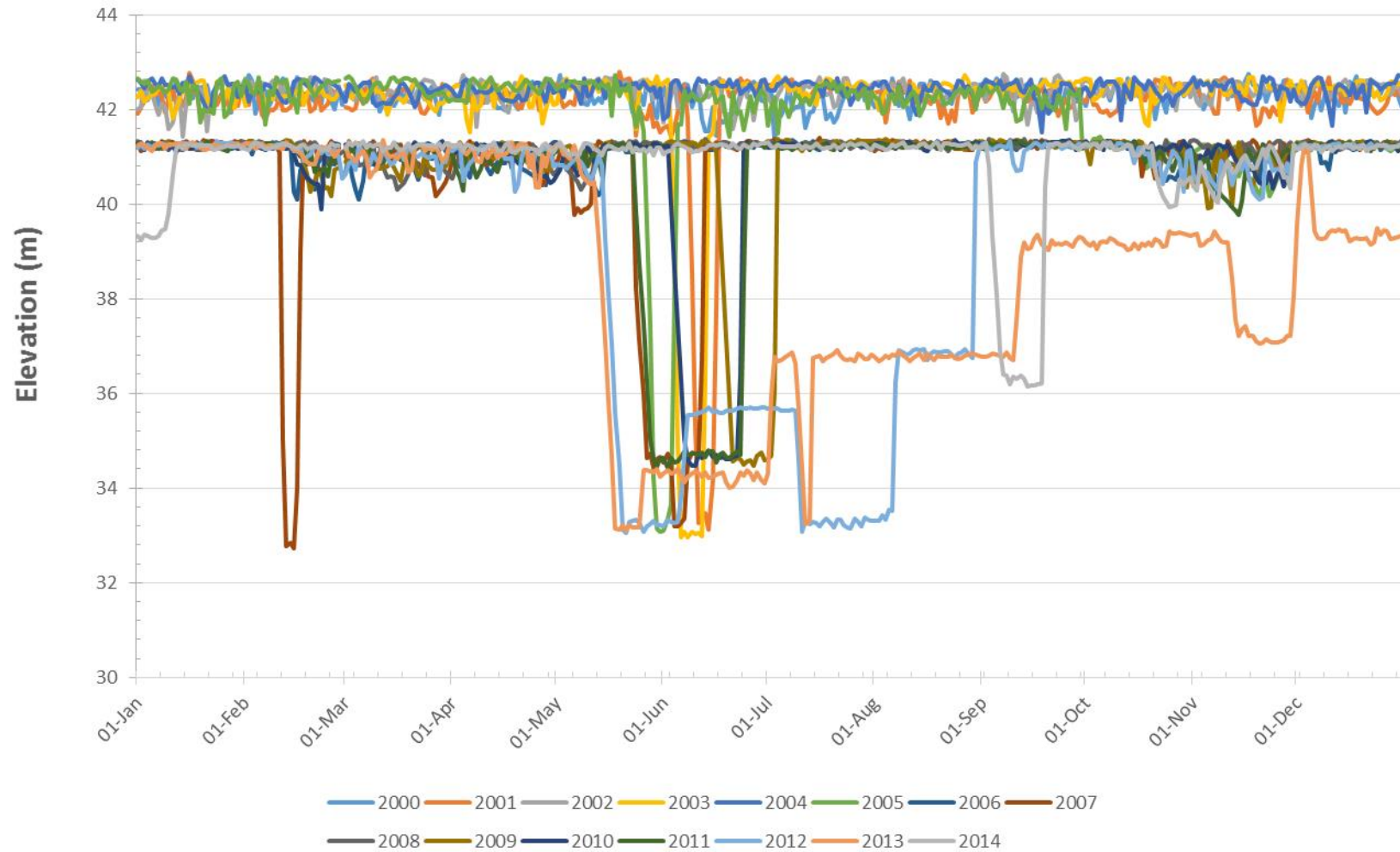
D.

Stave Lake Reservoir Water Retention Rate



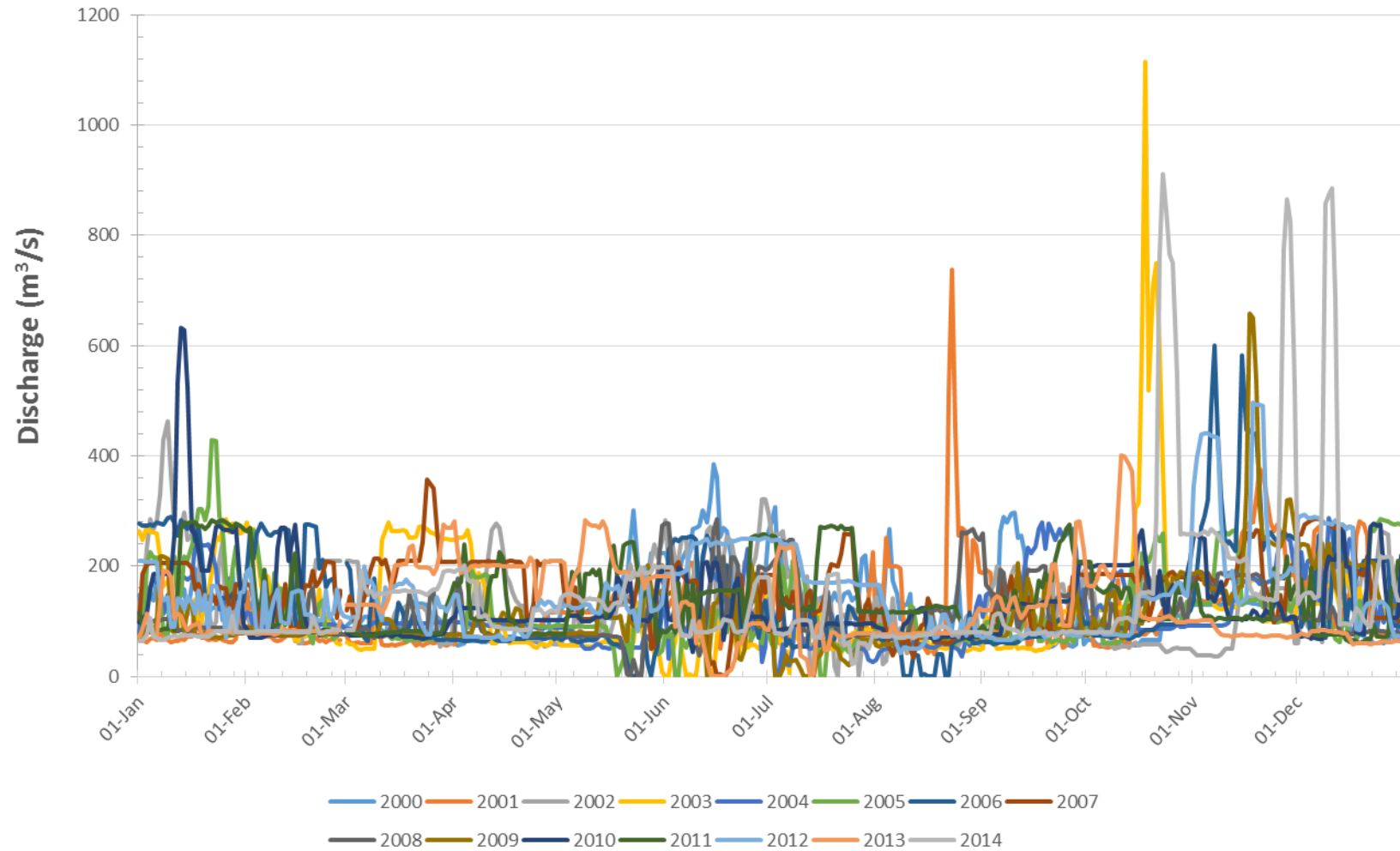
E.

Hayward Reservoir Water Surface Elevations



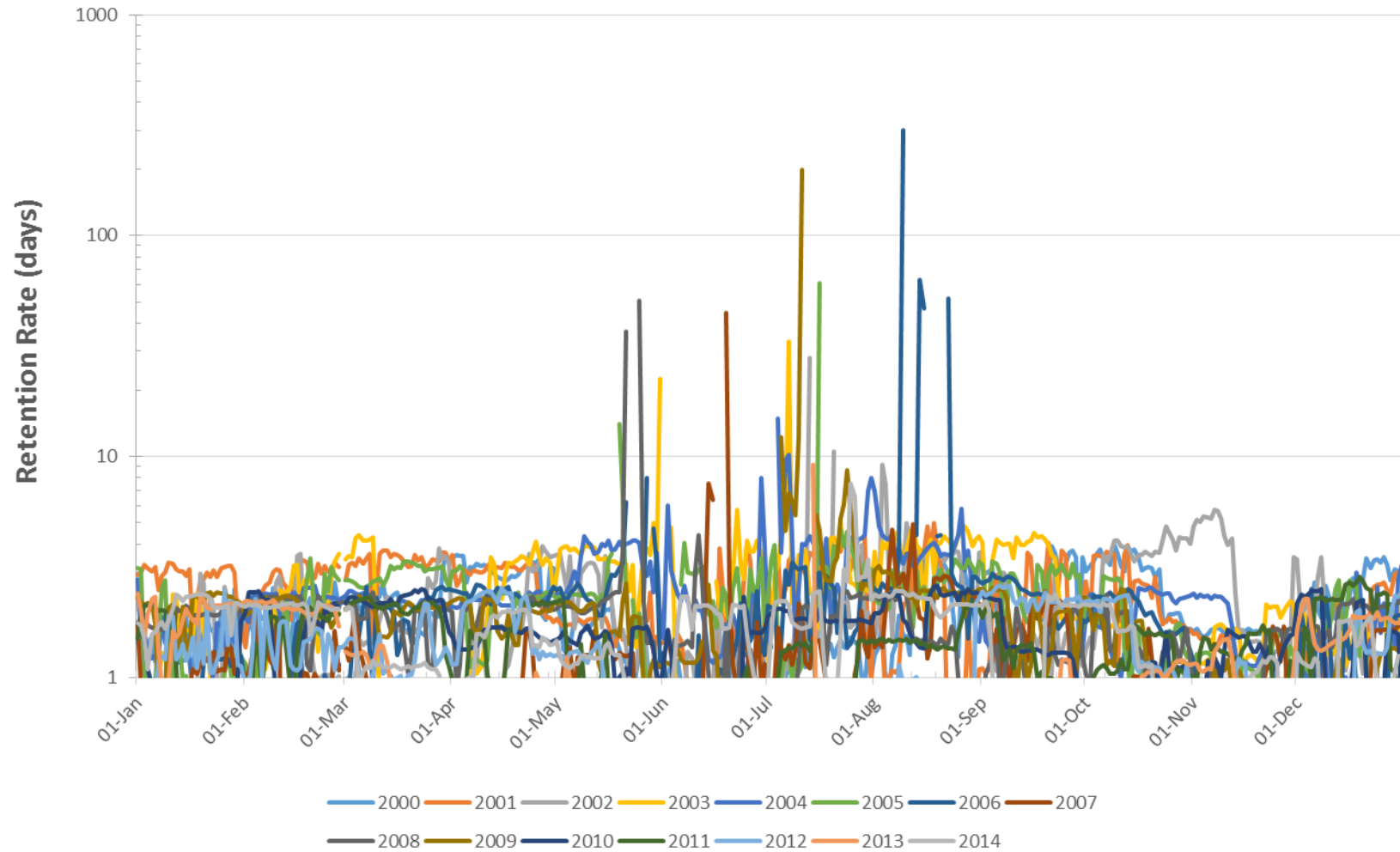
F.

Total Outflow from Ruskin Facility to Stave River



G.

Hayward Reservoir Water Retention Rate



Appendix 2: Water Chemistry Methodology

Water Chemistry Data from 2005-2012 was conducted by SpaChemtest DFO Laboratory at Cultus Lake, BC. Methods for these analyses are provided in the 2012 Pelagic Monitor and Littoral Primary Production Monitor. Water Chemistry data analyses from 2013 and 2014 were analysed by ALS Laboratory in Burnaby, BC using the methods identified below.



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Quoted Parameters with Detection Limits

Parameter	Method Reference	Report D.L.	Units
Water - Anions and Nutrients			
Alkalinity, Total (as CaCO ₃)	EPA 310.2	2.0	mg/L
Nitrate (as N)	EPA 300.0	0.0050	mg/L
Orthophosphate-Dissolved (as P)	APHA 4500-P Phosphorous	0.0010	mg/L
Phosphorus (P)-Total	APHA 4500-P Phosphorous	0.0020	mg/L
Water - Plant Pigments			
Chlorophyll a	EPA 445.0	0.010	ug

Methodology

Product	Matrix	Product Description	Analytical Method Reference
ALK-COL-VA	Water	Alkalinity by Colourimetric (Automated)	EPA 310.2
This analysis is carried out using procedures adapted from EPA Method 310.2 "Alkalinity". Total Alkalinity is determined using the methyl orange colourimetric method.			
ANIONS-NO3-IC-VA	Water	Nitrate in Water by Ion Chromatography	EPA 300.0
This analysis is carried out using procedures adapted from EPA Method 300.0 "Determination of Inorganic Anions by Ion Chromatography". Nitrate is detected by UV absorbance.			
CHLOROA-VA	Water	Chlorophyll a by Fluorometer	EPA 445.0
This analysis is done using procedures modified from EPA Method 445.0. Chlorophyll-a is determined by a routine acetone extraction followed with analysis by fluorometry using the non-acidification procedure. This method is not subject to interferences from chlorophyll b.			
P-T-COL-VA	Water	Total P in Water by Colour	APHA 4500-P Phosphorous
This analysis is carried out using procedures adapted from APHA Method 4500-P "Phosphorus". Total Phosphorous is determined colourimetrically after persulphate digestion of the sample.			
PO4-DO-COL-VA	Water	Diss. Orthophosphate in Water by Colour	APHA 4500-P Phosphorous
This analysis is carried out using procedures adapted from APHA Method 4500-P "Phosphorus". Dissolved Orthophosphate is determined colourimetrically on a sample that has been lab or field filtered through a 0.45 micron membrane filter.			
SAMPLE-DISPOSAL-VA	Misc.	Sample Handling and Disposal Fee	
VOLUME FILTERED-VA	Water	Volume Filtered (mL)	

Appendix 3: Electronic Copy of Compiled Pelagic Monitoring Data