Bridge River Water Use Plan

Seton Lake Aquatic Productivity Monitoring

Reference: BRGMON-6

Implementation Year 1

Study Period: 2014 to 2015

Authors:

Limnotek Research and Development Inc. and affiliated organizations

March 31, 2015

SETON LAKE AQUATIC PRODUCTIVITY MONITORING: PROGRESS IN 2014-15

BC Hydro project number BRGMON#6

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ST'ÁT'IMC
ECO-RESOURCES

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Cover photo: Seton Lake, April 2013: C. Perrin photo.

EXECUTIVE SUMMARY

This report provides information from the first of three years of monitoring, sample collection, laboratory work, and analysis that is required to answer four management questions addressing uncertainties about relationships between water management actions and biological production in Seton Lake.

Question 1: What is the inter-annual variation in physical conditions in the Seton Lake caused by the diversion and did the diversion change primary and secondary production in Seton Lake?

The last ~500 years of trophic changes in Seton and Anderson lakes is being reconstructed using a multi-proxy, multi-trophic paleolimnological approach using algal pigments, diatoms, cladocera zooplankton, stable isotopes, and a variety of lithological indicators. The study is intended to assess the cumulative effects of the diversion of water from Carpenter Lake to Seton Lake and climate change on primary and secondary production in Seton Lake. Anderson Lake is being used as a control lake. Cores collected in 2014 showed greater rates of sedimentation in Seton than in Anderson Lake: sediment from the year 1900 occurred at a depth of 12 cm in Anderson and 23 cm in Seton. Magnetic susceptibility was greater at sites closest to the diversion than at sites further from the diversion in Seton Lake, inferring a diversion effect on inorganic properties of the Seton sediment. Time course changes in assemblages of Cladocera and diatoms were found in Seton Lake but further analysis of the Anderson cores and application of modeling and statistics are required before causal processes explaining the temporal variation can be established.

Work in 2015 will include the following tasks:

- Completion of the core age analysis,
- Further analysis of Cladocera assemblages over time,
- Analysis of association between nutrient status and algal assemblages between blocks of years before and after the diversion of Carpenter water into Seton Lake, and
- Analysis of sediment pigment and geochemistry that is needed to define time course change in chemical attributes of habitats in each of Seton and Anderson Lakes. The pigments are a reflection of all photosynthetic organisms, but given the location of the cores in Seton and Anderson lakes, will primarily reflect changes in the pelagic environment.

Question 2: Will the selected alternative (N2-2P) increase biological production in Seton Lake?

Analysis and interpretation of hydrology data that was accessed from BC Hydro showed that N2-2P did not change available habitat and water residence time for fish and production of food for fish in Seton Lake. This finding means that measurements of primary and secondary production and attributes of the pelagic habitat will be mainly used to support interpretations of findings in other parts of the project (e.g. fish growth and migration patterns as required to answer management question 3 below).

Basic limnology findings are as follows. Stable stratification of Seton and Anderson Lakes is present in May – October with evidence of seiche activity. Turbid inflows to Seton Lake originate from the diversion of water from Carpenter Lake and from Whitecap Creek that flows into Portage Creek at the west end of Seton Lake. Inflow turbidity is greatest in the late summer and fall. That turbid flow is warm with low density and remains in the epilimnion of Seton Lake causing the lake to be more coloured in the late summer and fall than in the spring and early summer. Inflow from the diversion and from Whitecap Creek via Portage Creek has relatively low turbidity in the spring and that water flows into the hypolimnion in Seton Lake. Rates of primary production in Anderson and Seton Lakes were lower than those found in earlier measurements from 2000-2003 and they were in the middle of the range of primary production known among lakes and reservoirs of British Columbia. In contrast, the measurements from 2000-2003 were highest among lakes and reservoirs in the same comparison. Rates of zooplankton production in both lakes were within a range found in meso-oligotrophic lakes. Comparison of chemical and biological metric values with published criteria showed that both Seton and Anderson Lakes are meso-oligotrophic with respect to trophic state.

Two more years of measurements of primary and secondary production and ancillary measurements of phytoplankton and zooplankton biomass, turbidity, water chemistry, light, CTD profiles, etc. are required before analysis of the effect of N2-2P on biological production can be run. That work is scheduled for May – October of 2015 and 2016 followed by lab work, data analysis, and reporting in 2017.

Question 3: To what extent does aquatic productivity alone limit the abundance and diversity of fish populations in Seton Lake?

Hydroacoustics coupled with trawl and gill net sampling of pelagic habitat in summer and fall, 2014, followed by lab work established the fish species assemblage, age structure, stock origin of the sockeye, fry migration patterns, and fish sizes. Echograms from day time and night time surveys showed a strong diel vertical migration (DVM) pattern among the sockeye juveniles. Preliminary evidence is that during the daytime the fry are more abundant in turbid surface layers in Seton Lake than in the clearer waters of the same depth in Anderson Lake. These findings suggest that turbidity in Seton Lake may lessen the extent of vertical migration compared to that occurring in clearer Anderson Lake.

Several tasks will be completed in 2015 to continue development of answers to question 3. One is the development of a DVM model to test the hypothesis that differences in DVM between Anderson and Seton Lakes are related to differences in

water clarity. This analysis will integrate extensive physical and chemical and biological data collected for Question 2 with DVM observations. The whole-lake acoustic survey data will be used to calculate fish abundance for each seasonal survey and make comparisons to abundance data from other lakes and reservoirs. Several tasks that are outside the scope of BRGMON6 are recommended to assist with answering question 3.

Question 4: Can refinements be made to the selected alternative to improve habitat conditions or enhance fish populations in Seton Lake?

Two more years of data collection, lab work, and analysis is required before Question 4 will be answered. Multiple lines of evidence from all years of work will be used to determine if change to N2-2P will benefit fish populations.

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1 INTRODUCTION

The Bridge-Seton Water Use Plan Consultative Committee (CC) developed aquatic ecosystem objectives for Seton Lake that included efforts to maximize the abundance and diversity of fish populations while establishing flow controls for hydroelectric power generation, among other interests (Bridge River WUP CC, 2003). The Seton-Anderson watershed (Figure 1) provides habitat for a wide range of anadromous and resident fish species, which are valued from St'at'imc, commercial, recreational, and cultural perspectives. Tradeoffs occurred in the water use planning, resulting in decisions to set water elevations in reservoirs of the Bridge River watershed (Downton, Carpenter, Seton), manage spills from the reservoirs, and define flows in rivers (Middle and Lower Bridge River, Seton River). The complete package of flow controls is collectively known as N2-2P. The Bridge River WUP CC (2003) was constrained in making decisions by lack of information about the effects of change in flows on fish populations and biological production that support those populations. Despite this uncertainty, N2-2P was implemented on March 30, 2011 (Water Act Order 2011, Bridge River Power Development Water Use Plan, March 17, 2011) with a commitment to fund monitoring studies to fill data gaps and better inform people tasked with water management decisions in future years, including the St'át'imc people and St'át'imc Eco-Resources Ltd. (SER).

Much uncertainty among members of the Consultative Committee pertained to effects of the original water diversion from Carpenter Reservoir to Seton Lake on the population of sockeye salmon and Gwenis (also known as kokanee) that have provided food and shaped the cultural history of the St'át'imc Nation. A small diversion of water from the Bridge River to Seton Lake started in 1934. The diversion increased in 1954 to power four turbines at Shalalth (located on the north shore of Seton Lake, Figure 1) and it was fully developed by 1960 with the installation of four more turbines. Effects of this diversion on fish populations were first investigated by the International Pacific Salmon Fisheries Commission (Geen and Andrew 1961). Those studies suggested the diversion of cold and turbid water from the glacial Bridge River and Carpenter Lake, reduced water temperature, increased light attenuation, and decreased primary productivity in Seton Lake. While these observations imply the existence of a "footprint" impact, that impact has not been shown with a quantitative historical account. In addition, ecological links between the water diversion and biological productivity and the structure of food webs supporting anadromous and resident fish populations in the Seton-Anderson watershed are not well understood.

Several observations show this lack of understanding. It is surprising that juvenile sockeye selectively rear in Seton Lake that is affected by the diversion rather than in the upstream, hydrologically unimpacted Anderson Lake (Geen and Andrew 1961). In a comparison of limnological data between many Fraser and Skeena Basin lakes, Shortreed et al. (2001) found that photosynthetic rates in hydrologically impacted Seton Lake were similar to morphologically similar but hydrologically unimpacted lakes in the Fraser Basin. Despite this similarity, Shortreed et al. (2001) found a disproportionately low zooplankton standing crop in

Seton Lake. Another surprise is that the low zooplankton standing crop is sufficient to produce sockeye salmon smolts that are larger with expected greater overall survival rates than smolts rearing in the unimpacted Anderson Lake (Geen and Andrew 1961). This discrepancy between low availability of zooplankton and high biomass of sockeye juveniles has not been explained in data collected to date.

The CC found that these discrepancies could not be resolved with existing information and recommended studies to fill data gaps and determine what water management actions may be used to mitigate effects of the water diversion that may be found. Four management questions resulted from analysis by the CC and will be answered in this study. They are listed as follows:

1) What is the inter-annual variation in physical conditions in the Seton Lake caused by the diversion and did the diversion change primary and secondary production in Seton Lake?
2) Will the selected water management alternative (N2-2P) increase biological production in Seton Lake?
3) To what extent does aquatic productivity alone limit the abundance and diversity of fish populations in Seton Lake?
4) Can refinements be made to the selected alternative to improve habitat conditions or enhance fish populations in Seton Lake?

This report provides information from the first of three years of monitoring, sample collection, laboratory work, and analysis that is required to answer these questions.

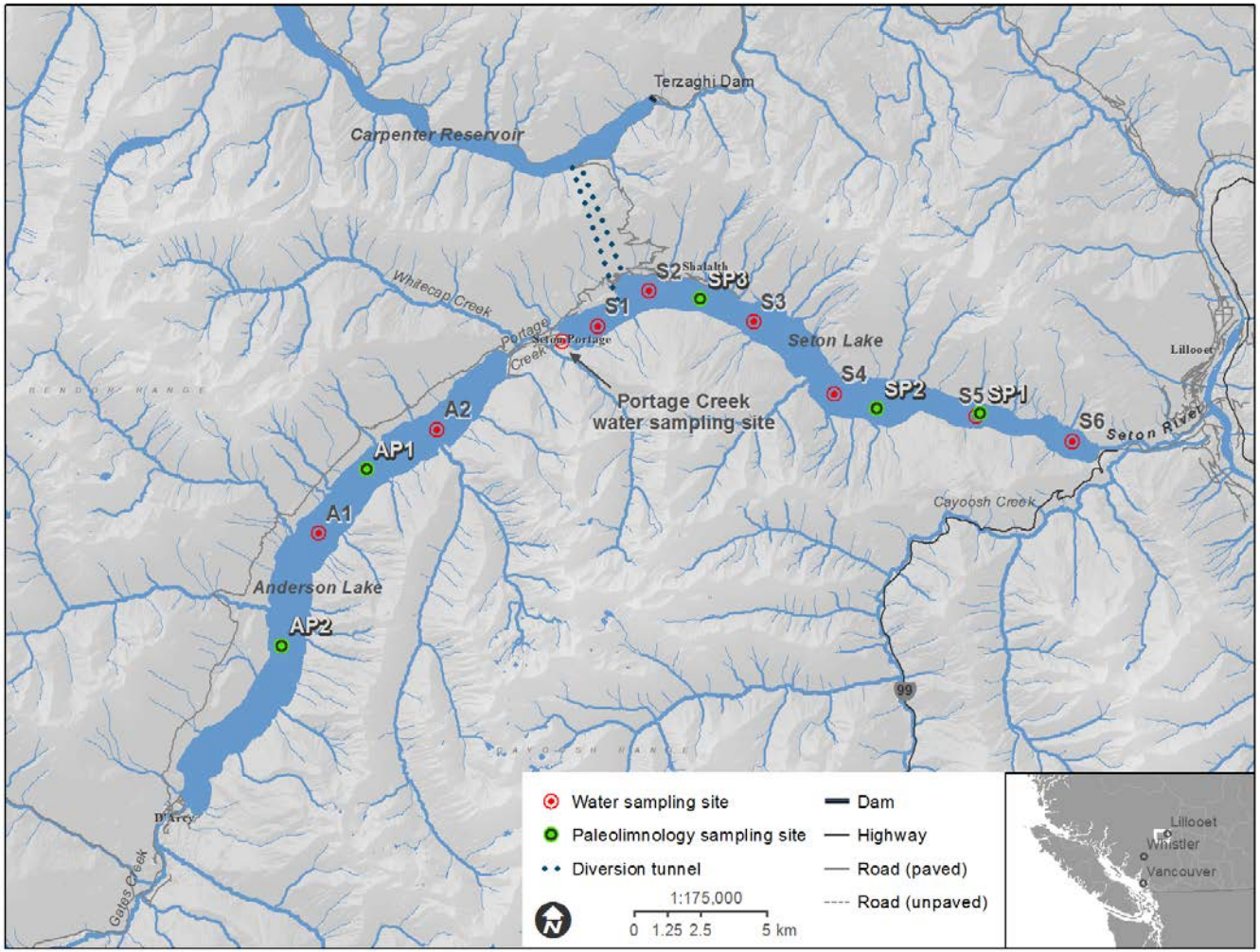


Figure 1. Study area showing Seton and Anderson Lakes, water and paleolimnology sampling sites, and the local watershed.

2 STUDY SITE DESCRIPTION

Seton Lake (N 50°41.758' W 122°08.007') is located west of Lillooet, British Columbia within the Fraser River drainage and the St'at'imc traditional territory. The geology of this region is composed of volcanic and sedimentary rock of Jurassic origin (Geen and Andrew 1961). The area is within the southern extremity of the Central Interior Ecoprovince that is characterized by a continental climate having cold winters and warm summers (Mitchell et al. 1981). The area is within the rain-shadow of the Coast Mountains but it does receive periodic moderating influences of coastal weather. It receives frequent outbreaks of artic air in winter and intense surface heating in summer. Seton Lake and Anderson Lake (N 50°38. 089' W 122°23.577'), located to the west of Seton Lake are glacially formed depressions surrounded by steep mountains reaching elevations of 2850m. Seton Lake receives flow from Anderson Lake via Portage Creek and discharge of diverted Carpenter Lake water at Shalalth. Portage Creek is a 2.9 km long stream that carries all water flowing out of Anderson Lake and discharge from Whitecap Creek that drains a valley between Carpenter Lake and Anderson Lake with alpine peaks up to an elevation of 2800 m.

Seton and Anderson Lakes are deep and long within a confined and contiguous valley (Table 1). Habitat in the lakes is mostly pelagic with steep shorelines, producing mean water depths of 85m in Seton Lake and 140m in Anderson Lake. Maximum depth is 151m in Seton Lake and 215m in Anderson Lake. Surface elevations between the two lakes differ by 21m.

Table 1. Morphometric attributes of Seton and Anderson Lakes.

Attribute*	Seton Lake	Anderson Lake
Surface area	24.6 km^2	28.6 km^2
Length	21.9 km	21.3 km
Average width	1.1 km	1.4 km
Volume	21 x 10^8 m^3	37 x 10^8 m^3
Mean depth	85 m	140 m
Maximum depth	151 m	215 m
Surface elevation	237 m	258 m
Length of shoreline	48.8 km	45.5 km

*data from Geen and Andrew (1961), which is based on survey data from International Pacific Salmon Fisheries Commission, 1953.

The source of inflow to Seton Lake changed between years before and after the onset of the water diversion from Carpenter Lake Reservoir. Before the diversion started in 1934, Anderson and Seton Lakes received most flow from Gates Creek at the south end of Anderson Lake. After 1934 and in greater amounts in 1954 and furthermore in 1960 when the diversion and power generating stations at Shalalth were fully developed, Anderson Lake continued to

receive most inflow from Gates Creek but Seton Lake received discharge from Anderson Lake and Whitecap Creek and the diverted flow from glacially turbid Carpenter Lake Reservoir via the diversion tunnel and penstocks at Shalalth (Figure 1).

Flow from Seton Lake discharges to the 5 km long Seton River which discharges to the Fraser River, 314 km upstream of the seaward edge of the Fraser River estuary. Before 1956 when flow controls were developed on the Seton River, mean annual flow from Seton Lake was estimated to be 18.7 $m^3 \cdot s^{-1}$ (Geen and Andrew 1961). After 1956, the Seton Lake outflow and lake water surface elevation has been controlled by a low head dam located 800m downstream of Seton Lake from which some water flows via a canal to a power generating station located on the banks of the Fraser River and the remaining water flows via the Seton River to the Fraser River. Cayoosh Creek that drains Duffy Lake and a valley to the south of Seton Lake flows into the Seton River 1.4 km downstream of the Seton Dam. To do so, Cayoosh Creek flows under an aqueduct of the water canal that carries water to the generating station.

The Seton-Anderson watershed is home to two sockeye salmon stocks (corresponding to DFO conservation units (CU)): Gates early summer run sockeye (Anderson-Seton-ES CU); and Portage late run sockeye (Seton-L CU). The Gates sockeye spawn in Gates Creek and since 1968 in the Gates spawning channel (Grant et al. 2011). The original summer run Portage Creek sockeye population (Seton-S CU) was extirpated in the first half of the 20th century and was replaced with transplanted sockeye from the lower Adams River (Withler et al. 2000, Grant et al. 2011), which now comprise the Portage sockeye stock. The two stocks are genetically distinct from each other (Withler et al 2000; Moreira 2014). In addition to spatial separation, the Gates and Portage stocks are separated by time of spawning. For example, in 2013 Gates sockeye spawning peaked between September 2-13 while Portage sockeye spawning peaked between October 23-29.

Sockeye spawning escapements vary considerably from year to year. Portage Creek sockeye exhibit the 4 year cycle seen in some other Fraser River stocks (Figure 2) but the dominant year changed in 1997-98. Gates sockeye were cyclic in past years but the cyclic pattern disappeared around 2000. In some years there is considerable pre-spawn mortality. To account for this mortality during fish enumerations, the number of effective female spawners (EFS) is determined through the examination of egg retention in post-spawned female carcasses. In 2013 a total of 57,209 sockeye spawned in Gates Creek and Channel with 28,948 females and 23,004 EFS. Spawning escapement to Portage Creek was much lower with only 7,509 total spawners with 4,406 females and 4,181 EFS.

Sockeye fry emerge from the spawning gravel of Gates and Portage Lakes in the spring and migrate downstream into their respective lakes. However, many fry from the Gates River and Channel migrate through Anderson Lake and down Portage Creek to rear in Seton Lake (Geen and Andrews 1961; Woodey 1975). Fry migrations through Portage Creek occur from mid-April to late June (Geen and Andrew 1961; Woodey 1975). Almost all fry rear in the lakes for one year and migrate to the ocean as age-1 smolts. On average, 99.88% of returning Gates

Creek adults and 98.80% of Portage Creek adults went to sea as age-1 smolts between 1968 and 2006 (DFO, data on file).

Other Pacific salmon and resident fish are found in the Seton-Anderson system (Geen and Andrew 1961). Pink salmon (*Oncorhynchus gorbuscha*) spawn in Seton, Cayoosh, and Portage Creeks. Coho salmon (*Oncorhynchus kisutch*) and chinook salmon (*Oncorhynchus tshawytscha*) spawn in Portage and Seton Creeks. Gwenish, or landlocked sockeye (*Oncorhynchus nerka*), are a unique variant that spawn in deep water in November-December in Seton Lake and in January in Anderson Lake (Geen and Andrews 1961; Morris et al 2003; Stables 2004). Their skin colour turns black as they mature to spawning condition (Moreira 2014). There are only moderate genetic differences between the Gwenish populations in the two lakes (Moreira 2014).Other resident fish in the two lakes include rainbow trout (*Oncorhynchus mykiss*) and bull trout (*Salvelinus confluentus*).

Seton and Anderson Lakes are considered to have underutilized rearing capacity for sockeye salmon based on a photosynthetic rate model of Shortreed et al. (2001). Shortreed et al (2001) categorized Anderson and Seton lakes as good physical environments for juvenile sockeye with relatively deep mean growing season epilimnions (18.2 m in Anderson and 22.4 m in Seton) and epilimnetic temperatures of 14ºC. Primary production in Anderson Lake with a photosynthetic rate (PR) of 276 mg $C \cdot m^{-2} \cdot d^{-1}$ is higher than in Seton Lake (219 mg $C \cdot m^{-2} \cdot d^{-1}$). Anderson Lake has an unusually high macrozooplankton biomass of 2,622 mg dry $wt \cdot m^{-2}$ (the highest of any Fraser system sockeye lake for which data are available), of which 40% is *Daphnia* (Shortreed et al 2001). In contrast, Seton Lake average zooplankton biomass of 422 mg dry $wt \cdot m^{-2}$ is lower than in most other Fraser River sockeye rearing lakes. Secondary production was not determined in the previous DFO studies.

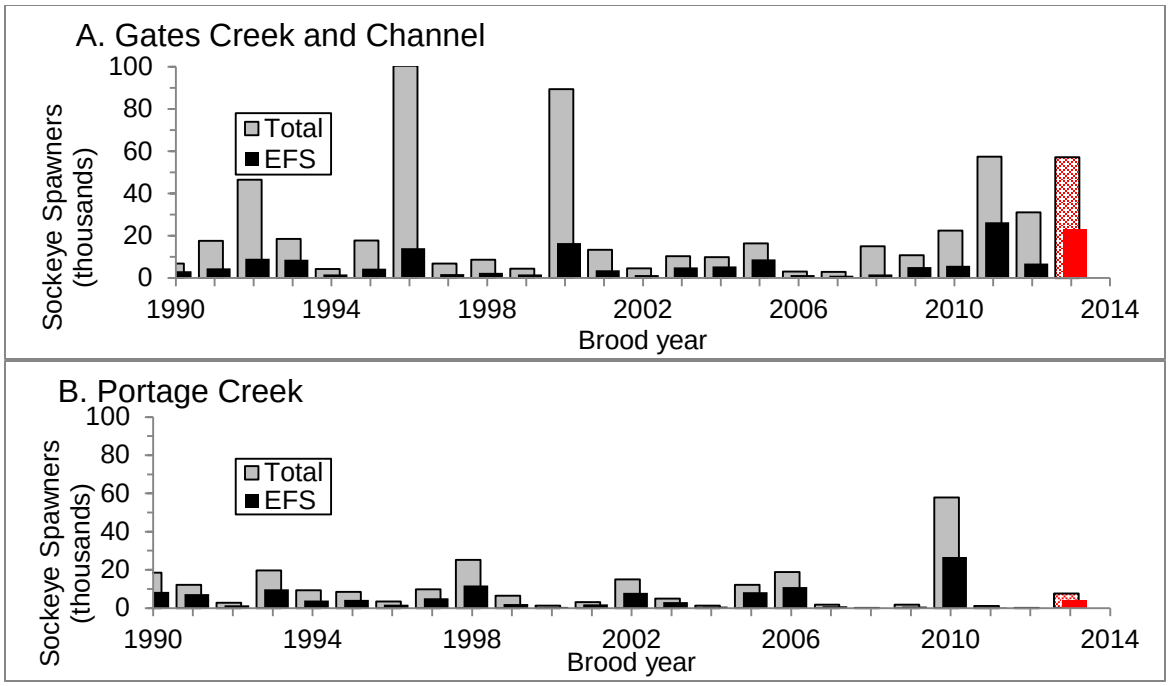


Figure 2. Number of sockeye salmon spawners and number of effective female spawners in Gates Creek and Portage Creek in brood years 1990 through 2013.

3 METHODS

3.1 Question 1: What is the inter-annual variation in physical conditions in the Seton Lake caused by the diversion and did the diversion change primary and secondary production in Seton Lake?

3.1.1 General approach

Seton Lake experienced changes related to the diversion of glacially-turbid water from Carpenter Lake Reservoir to this historically clearwater system. No hydrologic changes related to constructed works have occurred in Anderson Lake. All existing limnological information on Seton Lake represents the post-diversion period. Consequently, the extent of influence of hydropower developments on limnological conditions that could influence salmon populations is not fully understood. Similarly, how the limnological characteristics of Seton and Anderson lakes may have changed due to recent changes in climate remain largely unknown. To fill these data gaps, the last ~500 years of trophic changes in both lakes is being reconstructed using a multi-proxy, multi-trophic paleolimnological approach using algal pigments, diatoms, cladocera zooplankton, stable isotopes, and a variety of lithological indicators.

The study employs a paleolimnological Before-After-Control-Impact (BACI) design (Stewart-Oaten et al. 1986), and is intended to assess the cumulative effects of the water diversion and climate change on primary and secondary production in Seton Lake. In the BACI

design, the difference in the mean value of a given metric (physical, chemical, or biological) between cores from Seton Lake (impact lake) and Anderson Lake (control lake) before the diversion (a block of replicate years before 1934) will be tested against the difference after the diversion started (a block of replicate years after 1960). Years are replicates in this design. Average values of metrics from replicate cores in each lake can be used in a one way analysis of variance to test if the mean difference of a metric value between Anderson Lake (control) and Seton Lake (impact) in the "before" years is different from the mean difference between the two lakes in the "after" years. If the test is statistically significant, a conclusion will be that the diversion caused a change in value of the metric being tested. If the test is not significant, a conclusion will be that the diversion had no effect on the metric being tested.

Anderson Lake is being studied as well as Seton Lake to satisfy requirements of the BACI layout and understand and constrain the potential influences of climate and other factors that are unrelated to potential impacts of the diversion. In this way, we will be able to better understand potential changes to biological productivity attributable to the diversion and assess sustainability of these lakes to support populations of fish that are of particular importance to the St'at'imc people.

The science of paleolimnology uses the physical, chemical and biological information preserved in sediment cores to interpret past environmental or ecological conditions (e.g. Gross et al. 1998, Cederholm et al. 1999, Schindler et al. 2005). In this study we use a multiproxy approach based on indicators of primary (i.e. algal abundance inferred from sedimentary pigments, diatom assemblages) and secondary production (i.e Cladocera assemblages and size measurements), to better understand changes over time in Seton and Anderson lakes. These biological proxies will be used in conjunction with physical and isotopic variables to help interpret changes in nutrient source and production.

Sockeye salmon carcasses contain enriched levels of δ^{15}N (12 $^{0}/_{00}$) relative to the terrestrial and freshwater organic matter sources (~0 $^{0}/_{00;}$ Kline et al. 1993). During the spawning season, the degradation of salmon carcasses from the spawning areas enriches nursery lakes with nutrients that have an enriched isotopic ratio characteristic of the marine source from where the salmon migrated (Gross et al. 1998, Cederholm et al. 1999, Schindler et al. 2005). These marine-derived nutrients (MDN) can be a major source of nutrient subsidies to freshwater ecosystems that are linked to populations of migrating salmon (Chen et al. 2011a, 2011b). MDN can enhance the productivity of nursery lakes at each trophic level, enhancing primary and secondary production during the "fertilization period" (Selbie et al. 2009, Schindler et al. 2003, Chen et al. 2011a, 2001b). However, flushing rates of the lake can mediate overall food web attenuation of MDN (Holtham et al. 2004). The nutrient enrichment from MDN may have a positive feedback on juvenile salmon survival and growth (Naiman et al. 2002). However, salmon populations are primarily associated with the availability of spawning habitat, and this may negate the positive feedback related to MDN (Schindler et al. 2005).

Zooplankton are a major source of food for juvenile sockeye and Gwenish, and thus are important for the survival and growth of *O. nerka*. Zooplankton play a pivotal role in the food web between primary producers and planktivorous fishes including sockeye salmon and Gwenish. Zooplankton are positively influenced by nutrient inputs, but also can be negatively influenced under high predation pressures by juvenile sockeye salmon rearing in nursery lakes (Chen et al. 2011b). In addition to assemblage changes, we examined whether any change in the size of zooplankton taxa occurred over time, as zooplankton have been shown to be responsive to predation pressure through size selection due to fish predation (Sweetman and Finney 2003).

3.1.2 Core collection and processing

Sediment cores were retrieved from Seton and Anderson Lakes from August 4th to 11th, 2014. A total of 10 gravity cores were retrieved using a Glew gravity corer (Glew et al. 2001) with a 80-cm long clear core tube (internal diameter of ~7.6 cm) deployed using a winch and davit system on board a welded aluminum work boat. The winch was geared to allow fine control of the corer at great water depths close to 200m.

Five of the 10 cores were processed for analysis as listed in Table 2 and the others were stored for later use if needed. Core SP3 from Seton Lake was nearest to the discharge of water diverted from Carpenter Lake Reservoir at Shalalth while SP2 and SP1 were located at increasing distance from Shalalth. Immediately after collection, the core tubes were wrapped with aluminum foil to avoid degradation of pigments from exposure to light. The cores were taken to shore and sectioned into 0.5-cm intervals. Each section was placed into a 5x9 inch Whirlpak bag, shipped on ice to the Fisheries and Oceans laboratory at Cultus Lake or the PEARL lab at Queens University and placed in storage at 4°C. A notable observation was that Seton cores were characterized by a transition from a light grey at the top to a dark grey at the bottom but no colour transition was found in the Anderson Lake cores. The color transition is due to a change in sediment composition, which appears to be different between the two lakes. The dark grey may indicate high organic content while the light grey may show mostly inorganic content originating from the Bridge River. Grain size analysis and loss-on-ignition data to be examined in 2015 will provide more details and information about the sediment composition.

Table 2. Description of core locations in Seton and Anderson Lakes and observations upon core extrusion.

Lake Name	Core name as shown in Figure 1	Date of collection	Coordinates	Water depth at coring location (m)	Core length (cm)	Distance from top of core where transition from light grey to dark grey occurred (cm)
Seton	SP1	5/08/2014	50’40.897N	110	74.5	41.5

			122°03.081W			
Seton	SP2	7/08/2014	50°41.015N 122°06.368W	118	66.5	41.5
Seton	SP3	11/08/2014	50°43.279'N 122°11.967W	128	74	56
Anderson	AP1	09/08/2014	50°39.893N 122°22.671W	203	66.5	No colour transition
Anderson	AP2	10/08/2014	50°36.332N 122°25.241W	205	51.5	No colour transition

3.1.3 Core dating

Using the PEARL gamma counting dating facility at Queen's University, a number of isotopes were measured from selected intervals from each core to estimate the age of sediment intervals. From each core, between 14 and 30 sediment samples were analyzed for the activities of ^{210}Pb (lead), ^{137}Cs (Cesium) and ^{214}Bi (Bismuth) following the procedures outlined by Schelske et al. (1994).). ^{210}Pb is a decay of the parent radionuclide ^{226}Ra. In the sediment, the total ^{210}Pb activity has two components called unsupported ^{210}Pb and supported ^{210}Pb. The supported ^{210}Pb derives from the *in situ* decay of the ^{226}Rb in the sediment, while the unsupported ^{210}Pb derives from the atmospheric flux (^{210}Pb washed out from the atmosphere into the lake during rain fall and incorporated into the sediment). The supported ^{210}Pb is lost by radioactive decay as fast as it is renewed by the decay of ^{226}Rb. The unsupported ^{210}Pb, in the opposite, is not replaced as it decays because the radon (^{222}Rn) that produced it is in the atmosphere. Radioisotopic dating relies upon an exponential decay curve. Isotopes (for example, ^{210}Pb) laid down in sediments at a particular period then undergo predictable decay based upon known half lives. ^{210}Pb has a half life of 22.3 years. Over a number of half lives ^{210}Pb activity reaches background activity in the deepest "dateable" sediments. PEARL measured background as ^{214}Bi activity along the core section dated. So in a simplest sense, once unsupported ^{210}Pb activity reaches ^{214}Bi activity, background is reached. That time period is approximately 150 years. A rise in the concentration of ^{137}Cs activity can also be used as an independent dating marker for the ~1962-63 horizon in the sediment core because peak fallout of atmospheric nuclear testing occurred at that time. The activity of ^{214}Bi provides an estimate of supported (or background) ^{210}Pb, which is used to estimate unsupported ^{210}Pb (excess above supported). We used unsupported ^{210}Pb activities to estimate the chronology of the sediments using the Constant Rate of ^{210}Pb Supply (CRS) model. The CRS model has provided a reasonable and accurate chronology for lakes with variability in sedimentation rates (Appleby and Oldfield 1978). For Anderson Lake, every centimeter from 0 to 10cm and at 12cm, 15cm, 20cm and 25cm were subsampled and prepared for dating analysis. For Seton Lake, every two centimeters from the top to the bottom of each of the cores were subsampled and prepared. Higher sedimentation rates were suspected in Seton Lake due to settlement of glacial fines diverted from Carpenter Lake Reservoir. Hence, we expected to reach ^{210}Pb background deeper in the Seton cores than in the Anderson cores.

3.1.4 Magnetic susceptibility analysis

Changes in the magnetic susceptibility in sediment cores provided insight into the nature of inorganic sediment inputs over time (Dearing et al. 1981). Magnetic susceptibility was performed using a MS2/MS3 Magnetic Susceptibility Equipment (Bartington Instruments) and the software called Bartsoft at the Department of Fisheries and Ocean Laboratory in August, 2014. All five cores were analyzed at 0.5 cm intervals. Prior to measurement, the sediments were mixed and agglomerated in one part of each sample bag. A series of 5 measurements were taken for each interval, and the probe was calibrated after each set of samples.

3.1.5 Cladoceran analysis

Slides for the analysis of Cladocera were prepared following the standard methods outlined by Frey (1986) and Korhola and Rautio (2001). Approximately 1g of sediment was treated with 150mL of 10% KOH to deflocculate the sample. The sediment KOH mixture was then sieved through a 34 µm mesh and washed with deionized water. The material remaining on the mesh was backwashed into a 12mL glass vial and mixed with several drops of safranin glycerine solution as a dye and alcohol as a preservative. A 50µl slurry was deposited on a slide and allowed to dry. This process was repeated as necessary to concentrate the sample.

In order to calculate the concentration of Cladocera, individuals on the entire slide were counted. A minimum count of 70 individuals per sample (25 individuals for samples at very low concentration) were enumerated (Kurek et al. 2010). Standard identification keys were Szeroczyńska and Sarmaja-Korjonen (2007), Korosi and Smol (2012a), and Korosi and Smol (2012b).

The length of the mucro, antenna, and carapace of *Bosmina* spp. and the postabdominal claw of *Daphnia* spp were measured following the method outlined in Korosi et al. (2010) (Figure 3).

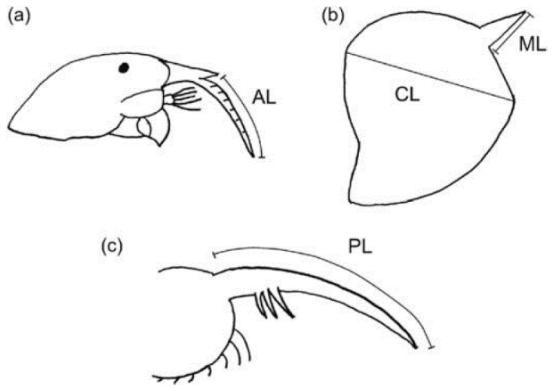


Figure 3. Schematic view of the headshield (a) and carapace (b) of *Bosmina* and the post-abdominal claw of *Daphnia* (c) from which length measurements were made. Abbreviations: AL, antennule length; CL, carapace length; ML, mucro length; PL, post abdominal claw length). The figure is reprinted from Korosi et al. 2010.

Cladocera were counted at a courser resolution (every two cm) in Seton Lake samples than in Anderson Lake samples (every 1 cm) because the sedimentation rate was expected to be greater in Seton than in Anderson.

Once section dates are known, the Cladocera enumeration and size analyses will be linked to time periods before and after the Bridge River diversion.

To date Cladocera data from Seton station SP3 are complete and are reviewed in this report.

3.1.6 Diatom analysis

For each core prepared to date, ~0.2-0.3 g of wet sediment was sub-sampled and placed in 20-ml glass vials to which a 1:1 mixture by molar weight of concentrated nitric (HNO_3) and sulphuric (H_2SO_4) acid was used to remove organic matter. The samples were allowed to settle for 24 h before the acid above the sample was removed, and the sample was rinsed with distilled water. This procedure was repeated until the sample had the same pH as distilled water (approximately eight rinses). Four successive dilutions for each sample were pipetted onto coverslips ensuring that each sample was well mixed. Samples on the coverslips were air-dried overnight, then heated on a warming plate to remove any remaining moisture, and subsequently mounted with Naphrax® onto glass microscope slides. Diatoms were identified and counted along transects on the prepared slide using a Leica (DMRB model) microscope fitted with a 100x fluotar objective (Numerical Aperture of objective = 1.3) and using differential interference contrast optics at 1000x magnification. Approximately 400 diatom valves were enumerated per slide. Diatoms were identified to the species level or lower, using the following taxonomic references: Krammer and Lange-Bertalot (1986, 1988, 1991a, b), Lange-Bertalot and Melzeltin (1996), Camburn and Charles (2000), Fallu et al. (2000) and the online database of Diatoms of the United States (westerndiatoms.colorado.edu).

Concentration of diatoms was determined using methods outlined in Battarbee and Keen (1982). An aliquot of a known concentration of microspheres was added to each of the diatom samples, prior to settling on coverslips. The microspheres were enumerated along with the diatoms and will be used to calculate estimates of number of diatoms per gram dry weight of sediment.

The diatom assemblage zones in the down-core analyses were defined by a constrained cluster analysis (Grimm, 1987), which provides a means of grouping those samples that are most similar to each other. The cluster analysis was stratigraphically constrained in order to

group the assemblages according to core depth (or core age) using a squared Euclidean similarity coefficient.

3.2 Question 2: Will the selected alternative (N2-2P) increase biological production in Seton Lake?

3.2.1 Defining the treatment to test

The "treatment" to be tested in question 2 will be change in system hydrology resulting from N2-2P that was implemented on March 30, 2011. In reality, the management of Seton Lake hydrology has been the same over the period of 2000 – 2011 so the date of implementation of N2-2P was effectively 2000. Given that lag effects of change in hydrology on biological production may occur because of an expected multiyear water residence time in Seton Lake, we assigned a period before change in hydrology to be before 2000, a transition period was 2000 – 2003 and full effect was after 2003.

Seton inflow, outflow, volume of live storage, and lake water surface elevation, in daily time steps, was accessed from BC Hydro for these three blocks of years (T. Neighbour, BC Hydro, Water License Requirements, Burnaby, B.C. Pers. Comm.) in which the before period was 1996 through 1999, the transition period was 2000 through 2003, and the after period was 2011 through 2014. Lake water residence time was calculated as total lake volume using data from Geen and Andrew (1961) (no later bathymetric data are available) divided by mean annual rate of outflow to the Seton River using mean daily flow data from BC Hydro. Comparisons of the hydrological metrics between the three blocks of years were made by one way analysis of variance (ANOVA) having three levels corresponding to the three blocks of years. The probability level was set at 0.05. The magnitude of the difference of a mean metric value between the blocks of years (where an effect is found at $p<0.05$) defined the treatment imposed by N2-2P. For example, if a significant 10% difference in water surface elevation between the block of transition years (2000 – 2003) and the "after" years (after 2003) is found, the treatment to be tested on the biological metrics will be that 10% change in water surface elevation. If no significant difference is found among any of the hydrology metrics between the "before", "transition", and "after" time periods, a conclusion will be that N2-2P did not produce a change to habitat. If this latter outcome occurred, the limnology data was used to describe biological production in Seton and Anderson Lakes. These descriptions were used to support interpretations of data from other parts of the project.

3.2.2 Monitoring layout

Regardless of whether N2-2P caused a change in the hydrology, question 2 will be answered in a before after control impact (BACI) design as described in Section 3.1.1 in which the measurements will be mean annual primary and secondary production. Data will be

available for this analysis from the "transition" period of years (2000 – 2003) and the "after" years (new data to be collected in 2014 – 2016). For this analysis the transition period will hereafter be called the "before" period. The data from 2000 – 2003 will be unpublished measurements collected by Fisheries and Oceans Canada (hereafter called DFO) from each of Anderson and Seton Lakes. Again, the "control" will be Anderson Lake and "impact" will be Seton Lake. The production data will be handled in a BACI layout as described above for Question 1 and statistical tests will be run to determine if the mean difference in primary or secondary production between Anderson Lake (control) and Seton Lake (impact) in the "before" years is different from the mean difference between the two lakes in the "after" years. If the tests show statistically significant differences, a conclusion will be that N2-2P contributed to change in primary and secondary production. If not significant, the tests will show no effect of N2-2P on primary and secondary production. If there is no hydrology "treatment" to be tested, a significant effect of statistical tests will show that something other than hydrology has influenced primary and secondary production in Seton Lake.

The layout of sampling sites in 2014 that will be repeated in 2015 and 2016 matched those used in 2000 – 2003 to facilitate a balanced design. Primary production sampling in 2014 was done at Station A1 on Anderson Lake and Station S4 on Seton Lake (Figure 1). Supplementary measurements including phytoplankton biomass measured as chlorophyll-a concentration, composition of the phytoplankton assemblage, light attenuation, turbidity, and water chemistry were also done at A1 and S4 and at A2 that was a duplicate station on Anderson Lake and at S5 that was a duplicate station on Seton Lake. Secondary production was calculated from zooplankton samples collected at those same four stations wherein A1 and A2 were considered replicate stations on Anderson Lake and S4 and S5 were considered replicate stations on Seton Lake. Sampling was conducted monthly during May through September except zooplankton that was sampled monthly during May through October.

3.2.3 Habitat attributes

Measurements were made monthly at both stations in each lake in 2014 to assist with interpretation of the biological production data and to describe the different habitats supporting phytoplankton, zooplankton, and fish in Seton and Anderson Lakes.

A key variable needed for measurement of primary production was depth of the euphotic zone, which is where photosynthetically active radiation (PAR) exceeded 1% of that at the water surface. A standard measure of water transparency was Secchi depth, determined as the mean depth of disappearance of a standard 20 cm Secchi disc when lowered through the water column and depth of reappearance of the disc when subsequently raised. These measurements were done on the shaded side of the boat. In addition, a LiCor LI250A irradiance meter equipped with a spherical quantum sensor was used to measure PAR in 1 m intervals from the surface to a depth where PAR was less than 1% of that at the surface at each of the two stations on each lake. The average depth receiving 1% of surface irradiance measured on five

dates distributed in May through October was the mean euphotic zone depth. The irradiance profiles were used to calculate the light extinction coefficient for each lake according to the following equation:

$$n = \frac{\ln(I_0) - \ln(I_2)}{z}$$ Equation 1

Where:
n is the light extinction coefficient
I_0 is irradiance at the water surface and
I_2 is irradiance at depth z

Depth of the thermocline was mean depth of the water strata where water temperature changed more rapidly with depth than it did in stable layers above (epilimnion) and below (hypolimnion). Profiles of water temperature as well as dissolved oxygen concentration, conductivity, and turbidity were measured over the water profile in May-July with a calibrated YSI Sonde model 6920. In August through October, profiles of the same parameters plus fluorescence (and indirect measure of chlorophyll concentration that is a measure of algal biomass) were completed using a Sea-Bird Electronics SBE19plusV2 CTD. CTD is a generic term given to an instrument that measures conductivity and temperature amongst other parameters over a depth profile. We changed from the YSI to Sea-Bird to facilitate profiles to the bottom of each lake (the YSI could only sample to a depth of 60m), greater sensor resolution and accuracy, and to provide more detailed data than could be achieved with the YSI. In Seton Lake during May and June the CTD casts were done at S4 and S5 but they were expanded to all six stations in July through October. In Anderson Lake the casts were always done at A1 and A2. Scripts in R were written to produce colour filled isopleths of YSI and CTD sensor data over time in both lakes and longitudinally at a given time in Seton Lake.

Total and dissolved nutrients and suspended solids concentration was measured at a depth of 1m and within the hypolimnion. The nutrient analyses included TN (total nitrogen), TP (total phosphorus), TDP (total dissolved phosphorus), nitrate-nitrogen (NO_3 -N), ammonium (NH_4-N), and SRP (soluble reactive phosphorus). Water for TDP, nitrate, ammonium, and SRP were filtered in the field at the time of collection through Waterra 0.45 µm FHT-45 polyethersulphone filters (http://www.waterra.com/pages/Product_Line/filters/filters_2011.html) using an Alexis peristaltic pump (http://pegasuspumpcompany.com/alexis-peristaltic-pumps). All samples were submitted within 24 hours to ALS labs in Burnaby for analysis using standard methods (APHA 2014).

Chemical and physical attributes of stream inflows and outflows were measured to assist with interpretations of the production and habitat data. Measurements occurred on the same monthly frequency during May – September that was applied to the lake station sampling on five dates between May and October. They included turbidity, temperature, dissolved oxygen concentration, total dissolved solids concentration, conductivity, pH, soluble reactive

phosphorus, total dissolved phosphorus, total phosphorus, nitrate, ammonium, and total suspended solids concentration. Seton sites included inflow from Carpenter reservoir at the Bridge 1 generating station at Shalalth (hereafter called BR1), the inflow to Seton Lake from Anderson Lake (Portage Creek), and the outflow in the Seton River. One Anderson site was the inflow from Gates Creek. The particle size distribution in water discharged from BR1 was determined from a water sample collected in the BR1 tailrace when all turbines were running, which occurred on the regular monthly sampling episodes in June through September. Representative aliquots were taken and diluted with background electrolyte (2% and 8% NaCl) to obtain samples for counting using a Micromeritics Elzone 280PC. Samples were tested over 2 ranges, ~1.3-25 microns and 11-200 microns. All particle size distribution analyses were run at the University of British Columbia Department of Mining Engineering.

An Onset Hobo temperature logger set to record an average measurement every two hours during May – September was installed in the tailrace at BR1 and in Portage Creek downstream of the confluence with Whitecap Creek. Temperature data for other stream sites including Gates Creek (inflow to Anderson Lake) and the Seton River (downstream of Seton Lake) were accessed from people managing other Bridge River monitoring projects.

3.2.4 Phytoplankton

Primary production was measured *in situ* as the amount of ^{14}C incorporated into particulate organic carbon, following the methods of Steemann Nielsen (1952) that were used by DFO in 2000-2003 for the "before" period. Discrete water samples collected with a Niskin water sampler from seven depths over the profile of the euphotic zone were transferred directly into two light and one dark 300 ml acid-cleaned BOD glass bottles assigned as a group of bottles to each depth, resulting in seven sets of two light and one dark bottle. Each BOD bottle was rinsed three times with sample before filling. The water samples were maintained under low light conditions during all manipulations until the start of the incubation that started within 1 h of the water collections. Water in the BOD bottles was inoculated with 0.185 MBq (5 μCi) of $NaH^{14}CO_3$ New England Nuclear (NEC-086H). The cluster of BOD bottles for each depth was attached to an acrylic plate and suspended at each of the seven depths from which the water samples were taken. The samples were incubated *in situ* for 4-5 h between the hours of 1000 and 1500 to allow the carbon uptake to proceed. Following retrieval of the incubation array, the BOD bottles were transported to a field lab on BC Hydro property at BR1 in a cool dark box. The incubations were terminated by parallel filtration of 100 ml of sample onto a 0.2 μm 47-mm diameter polycarbonate filter and a 0.75 μm 47-mm diameter glass fibre filter. The 0.75 μm pore size was required because that was the size used by DFO in 2000-2003. The 0.2 μm pore size was used to determine the amount of primary production missed when filtering through the 0.75 μm filter. Each folded wet filter and retained biomass was placed in a 7 ml scintillation vial until processing at the University of British Columbia. In the fumehood, 100 μL of 0.5 N HCl was added to each vial to eliminate the unincorporated inorganic $NaH^{14}CO_3$. The scintillation vials were left uncapped in the fumehood for approximately 48 h until dry, 5 ml of Scintisafe®

scintillation fluor was added to each vial, and they were stored in the dark for >24 hours before the samples will be counted using a Beckman® Model #LS 6500 liquid scintillation counter. Each vial was counted for 10 minutes in an external standard mode to correct for quenching. The specific activity of the stock was determined by adding 100 μL ^{14}C-bicarbonate solution to scintillation vials containing 100 μL of ethanoalamine and 5 ml Scintisafe® scintillation cocktail. Rates of carbon incorporation followed methods reported by Parsons et al. (1984) to obtain hourly primary productivity and were vertically integrated according to procedures of Ichimura et al. (1980). Daily rates of primary production were calculated by multiplying the hourly primary productivity by the incubation time and by the ratio of the solar irradiance during the incubation to the solar irradiance of the incubation day where daily solar irradiance in air was continuously measured using a Li-Cor irradiance meter and logger installed on a residence roof top at Shalalth in May through, October 2014. The difference between the ^{14}C incorporation in the light bottles (includes photosynthetic and non-photosynthetic uptake) and the ^{14}C incorporation in the dark bottle (includes only non-photosynthetic ^{14}C uptake) indicated carbon uptake by photosynthesis.

The irradiance logger at Shalalth inadvertently shut down over the period of June 18 through August 19, which required supplementary data to be used in place of the LiCor data for that period. Bench top testing of the logger suggested the shutdown was due to high air temperatures that commonly exceeded 40ºC at the Shalalth site during the shutdown period. Supplementary data was accessed from Environment Canada for the closest site where total solar radiation is measured continuously, which was Peachland. Comparison of the solar radiation data during successful logging periods at Shalalth with the Peachland data showed close association and acceptability of the Peachland data as a substitute for the lost data at Shalalth.

Chlorophyll *a* corrected for phaopigment was determined by *in vitro* fluorometry (Yentsch and Menzel, 1963) in aliquots from each of the seven discrete water samples that were used for primary production analysis at A1 and S4 and from euphotic zone depth-integrated water samples from S5 and A2. An aliquot of water from the samples was filtered at a field station using parallel filtration onto a 47-mm diameter, 0.2 μm polycarbonate Nuclepore™ filter and 0.75 μm glass fiber Advantec® filter using a vacuum pressure differential of <100 mm of Hg. Filter papers were stored in aluminum foil envelopes on dry ice during transport to the lab, and then transferred to a freezer at –20^{o}C until analysis. Chlorophyll *a* was extracted from the sample in 5 mL of 90% acetone and stored covered in the freezer for 20-24 hours. The filter was then removed and the fluorescence of the acetone extract was measured in a Turner Designs™ Trilogy fluorometer calibrated with a solution of commercially available Chl *a* before and after the addition of 100 μL of 10% HCl. Calculations for chlorophyll *a* were made using the equations of Parsons et al. (1984). The average phytoplankton biomass of the euphotic zone was determined by calculating the mean of all sampling depths.

At each of the four sampling stations (A1 and A2 on Anderson Lake and S4 and S5 on Seton Lake) a depth integrated water sample was collected with the Niskin bottle for

phytoplankton cell enumeration by species. An aliquot was dispensed to a glass amber jar, preserved with acid-Lugol's solution, and stored in a cool and dark location for later analysis in the lab. Prior to the enumeration, the samples were gently shaken for 60 seconds and allowed to settle in 25 mL chambers for a minimum of 8 hrs (Utermohl 1958). Counts of algal cells, by taxa, were done using an inverted phase-contrast plankton microscope. Cells of large micro-plankton (20-200 µm) were counted at 250X magnification. All cells within one 10-15 mm random transect were counted at 1560X magnification. This high magnification provided enumeration of small autotrophic picocyanobacteria in the size range of 0.2-2.0 µm and autotrophic and heterotrophic nano-flagellates in the size range of 2.0-20 µm. In total, 250-300 cells were counted in each sample. The biovolume of each taxa was determined as the cell count multiplied by the volume of a simple geometric shape corresponding most closely with the size and shape of the algal taxon. Taxonomic references were Canter-Lund and Lund (1995) and Prescott (1978).

3.2.5 Zooplankton

Zooplankton samples were collected in a single vertical haul of a 153 µm mesh Wisconsin net having a 30 cm intake opening at each of the two replicate stations on each lake. The depth of haul matched that used by DFO for the "before" data in 2000-2003 (typically 30m). The net was raised at a speed of approximately 0.5 $m \cdot s^{-1}$. The zooplankton was washed into the cod-end of the net and anaesthetized to prevent egg shedding in a wash of Club Soda before being added to a 10% sugared formalin solution. Each zooplankton sample was split using a Folsom plankton splitter to a subsample volume containing post-naupliar stages of >100 of the most abundant taxa of crustaceans. For each sub-sample, the species were enumerated at 5-100x magnification under a GSZ-Zeiss stereo microscope. The number of attached eggs was counted. Sub-sample counts were extrapolated to the total sample. Biomass of zooplankton was determined from length-to-weight regressions reported by McCauley (1984) using lengths measured with a digitizing system. Up to 25 random length measurements per taxon were taken per sample, and the final biomass was expressed as µg dry weight per sample. Using the known volume of water that passed through the Wisconsin net (intake opening area multiplied by depth of haul), the amount of zooplankton biomass per sample was converted to volumetric zooplankton biomass (µg dry $weight \cdot L^{-1}$). This value was corrected to the amount of biomass in a 1 m^2 column of water over the sampling depth to yield areal biomass units of mg dry $weight \cdot m^{-2}$.

Zooplankton production was measured at each of the two sampling stations on each lake. Secondary production, in this case by zooplankton (in units of $mass \cdot m^{-2} \cdot yr^{-1}$), is an indicator of food available to fish, and is the most commonly used indicator of ecological function, water quality, energy flow, disturbance, and recovery in freshwater ecosystems (Benke and Huryn 2010). Secondary production integrates several aspects of ecological performance including density, biomass, growth rate, reproduction, survivorship, and developmental time.

Zooplankton production in Seton and Anderson Lakes was determined by re-organizing the equation:

$$\frac{P}{B} = y \qquad \text{Equation 2}$$

where P is annual zooplankton production (mass·m^{-2}·yr^{-1}), B is mean annual dry weight biomass (mass·m^{-2}) of the population of interest, and y is a rate in units of yr^{-1} (Benke and Huyrn 2006). Given that biomass can be measured and y, known as a production/biomass or P/B ratio, can be found in the literature for many taxa, the product of B and y gives P.

Production of zooplankton was determined from Equation 2, but P/B was calculated from a temperature dependent model reported by Shuter and Ing (1997) and shown to work well by Clarke and Bennett (2007):

$$[P{:}\,B]_{daily} = 10^{(\alpha_{taxon} + \beta T_{daily})} \qquad \text{Equation 3}$$

where $[P{:}\,B]_{daily}$ is daily $P{:}\,B$, $\propto_{taxon}$ is -1.725 for cladocerans, -1.766 for cyclopoid copepods, and -2.458 for calanoid copepods, β is 0.044 for cladocerans, 0.040 for cyclopoid copepods, and 0.050 for calanoid copepods, and T is average water temperature (ºC) measured over the depth that zooplankton were collected on each sampling day. Zooplankton biomass and $[P{:}\,B]_{daily}$ was linearly interpolated between the six sample dates distributed between May and October, and the product of $[P{:}\,B]_{daily}$ and zooplankton biomass was summed over the sampling period May through October to estimate annual zooplankton production. In this approach, zooplankton production in the active growing season of May through October was considered to include most production for the calendar year and was called annual zooplankton production.

Measurements and calculations to determine primary production (production of phytoplankton) and secondary production (production of zooplankton) were run for data collected from Seton and Anderson Lakes in 2014 and for existing data that were collected by DFO in 2000-2003. The same will be done for zooplankton data to be collected in 2015 and 2016 to yield a complete data set with which to apply statistics to test the effect of N2-2P on zooplankton production in Seton Lake. That final analysis will be run after the 2016 data are compiled.

3.3 Question 3: To what extent does aquatic productivity alone limit the abundance and diversity of fish populations in Seton Lake?

3.3.1 Hypotheses and approach

In an uncommon early migratory behaviour, many age-0 Sockeye Salmon from Gates Creek migrate through Anderson Lake and down Portage Creek into Seton Lake during spring and early summer, where they rear until smolting the following year (Geen and Andrew 1961; Woodey 1975). This early inter-lake migration would normally be expected to have adverse growth and survival consequences because Anderson Lake is thought to be more productive and have a higher zooplankton prey density than Seton Lake (Shortreed et al 2001). In spite of the apparent growth and survival advantages of the habitat in Anderson Lake, those sockeye fry that do stay in Anderson Lake are unexpectedly smaller (mean weight 1.1g) than the sockeye fry rearing in Seton Lake (5.0 g, Shortreed et al 2001).

Since the time of the Bridge River diversion, Seton Lake has become more turbid, resulting in decreased light transmission and visibility (Geen and Andrew 1961; Woody 1975; Shortreed et al 2001). We speculate that sockeye and Gwenish grow larger in Seton Lake than in Anderson Lake because instead of avoiding predators by undergoing diel vertical migration (DVM, Clark and Levy 1988; Levy 1990; Scheuerell and Schindler 2003) the high turbidity hides the fry from predators (Gregory 1993; Gregory and Levings 1998) allowing them to graze continuously during the day in the upper water column and thereby enabling the juveniles to grow faster than in the clear waters of Anderson Lake.

Our primary hypothesis is that diel vertical migration is extensive in the relatively clear pelagic habitat of Anderson Lake but reduced or absent in the turbid water of Seton Lake. We postulate several corollaries to this hypothesis: a) age-0 *O. nerka* growth and survival is higher in Seton Lake because of the increased time spent grazing; b) juvenile sockeye are a larger component of the diet of large piscivorous predators in Anderson Lake than in Seton Lake and; c) predator density and/or condition is lower in the pelagic habitat of Seton Lake than in Anderson Lake.

We are addressing Question 3 using two sets of biological and habitat data. Firstly we are analyzing mostly unpublished limnological and pelagic fish data collected by DFO in 1975 - 1978, 1981, and 2000 - 2003, and secondly we are supplementing this analysis with new information from directed studies conducted in the summer and fall of 2014. At present, processing and analysis of historical and new 2014 data are in their early stages, so this report only describes background information, 2014 data collection, and analysis tasks completed to date. Analyses to be conducted later are described in the Remaining Analysis section of this report.

Standard pelagic survey methods were used to collect data in 2014 to estimate the growth rate, population size and density, and spatial distribution of pelagic sockeye, Gwenish, and potential piscivore predators in Seton and Anderson Lakes. Mobile acoustic surveys were performed to allow determination of the abundance, horizontal and vertical distribution, and

acoustic target size of fish in the pelagic zone (Parker-Stetter et al. 2009; MacLellan and Hume 2010). Trawling and gill netting were conducted concurrently with acoustic sampling to identify acoustic targets and provide biological information about them. Samples from trawling and gill netting were used to determine fish size, age, stock origin, and diet. Two surveys were conducted on each lake, a summer survey from July 28 to August 4, 2014 and a fall one from October 22 – 29, 2014. Each survey of each lake was composed of two parts: a whole-lake survey for developing a fish abundance estimate, and a daily vertical migration (DVM) study to quantify diel vertical migration patterns of fish.

In subsequent analysis, trawl and gill net catch data will be used to apportion the acoustic population estimates among fish species and stock origin (from DNA).

3.3.2 Whole lake acoustic surveys

Mobile acoustic sampling methods for the whole-lake surveys closely followed those of the 2000-2003 DFO surveys, and were consistent with protocols described in standard fisheries publications (Thorne 1983; Brandt 1996; Simmonds and MacLennan 2005; Parker-Stetter et al. 2009) and other sources specifically designed for surveying *O. nerka* dominated fish communities in BC Lakes (Perrin et al. 2006; MacLennan and Hume 2010). We used the stratified systematic survey design developed by DFO in 2000-2003. The lakes were divided into sections, two in Anderson and three in Seton. Within these sections three acoustic transects were established perpendicular to the long axes of the lakes for a total of 9 transects on Seton Lake and 6 transects on Anderson Lake (Figure 4). Data collection was completed during the hours of darkness (sun >18° below the horizon) in the course of one night for each survey.

The surveys were performed using an 8 m long, welded aluminum boat with a covered cabin at a transecting speed of approximately 1.5 to 2.0 $m \cdot s^{-1}$. The echo sounding system consisted of a 206 kHz Biosonics split-beam scientific echo sounder with a 6.7 degree beam transducer paired with a Garmin model 546 differential GPS. The transducer was deployed from a towfin, with the transducer face aimed vertically downwards ~ 1.0 m beneath the lake surface. The echo sounder was operated by a computer, which also served as a data logger allowing monitoring of data quality on echograms during collection. Latitude and longitude from the GPS were merged with acoustic data during logging. Data collection settings were the same as those used in the past by DFO (e.g., collection threshold of -100 dB; pulse width 0.4 ms; ping rate 5-6 pings/s). Because the night time distribution of fish is almost entirely above 80 m in most BC sockeye lakes (MacLennan and Hume 2010), data were usually collected to a depth of 80 m, with occasional sampling to greater depths to check for the presence of fish in deeper strata. Accuracy of acoustic measurements was verified by in situ TS measurements of a standard calibration sphere at least once during each survey of each lake. Passive data (with acoustic transmitter off and receiver on) were collected at least once during each survey to record background noise levels (Parker-Stetter et al. 2009).

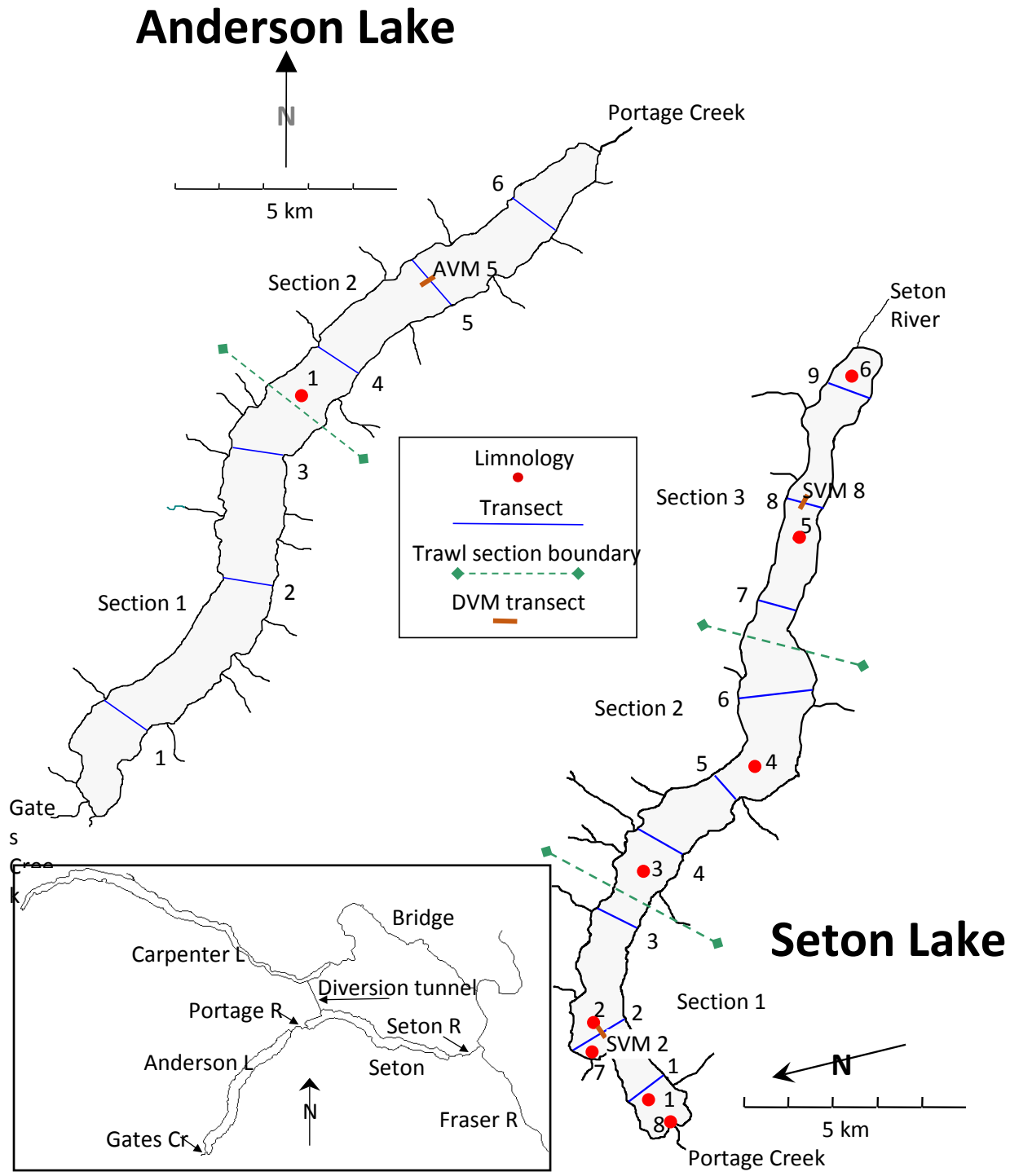


Figure 4. Maps of Anderson and Seton lakes showing hydroacoustic transects and mid water trawling sections. Limnology sampling stations from Figure 1 are shown for reference.

3.3.3 DVM acoustic sampling

The diel vertical migration (DVM) behaviour of juvenile *O. nerka* and other fish was measured by repeated mobile acoustic sampling of short transects (~ 600 m) at fixed stations where sockeye fry density was expected to be high. This sampling used the same boat and hydroacoustic system as the whole-lake acoustic surveys, and the same sampling procedures with two exceptions. For DVM sampling the maximum data collection range was 100-210 m and ping rates were 3-6 pings/s depending on fish and bottom depth at the sampling location. During each season (summer and fall), DVM sampling was performed in each lake during two daytime periods (0.5-2 hours each), two dusk periods (1-2 hours each), and two night periods (0.5 hour each), except that Anderson lake was only sampled once at night in the summer (Figures 5 and 6). DVM sampling periods were defined in relation to light conditions as: day = sunrise to sunset (we sampled with the sun above the surrounding mountains when possible), dusk = 1 hour before sunset to 0.5 hour after the end of civil twilight, and night = 0.5-1.0 hour after end of civil twilight. To enhance transducer stability and acoustic data quality under occasional rough conditions DVM transects were run parallel to the long axis of the lake, crossing the regular population estimate transects mid-lake. On Anderson Lake DVM sampling took place on transect AVM5 during both surveys (Figure 4). On Seton Lake it took place on transect SVM8 in the summer and, to better assess the effect of a longitudinal water clarity gradient in Seton Lake, on transects SVM2 and SVM8 in the fall.

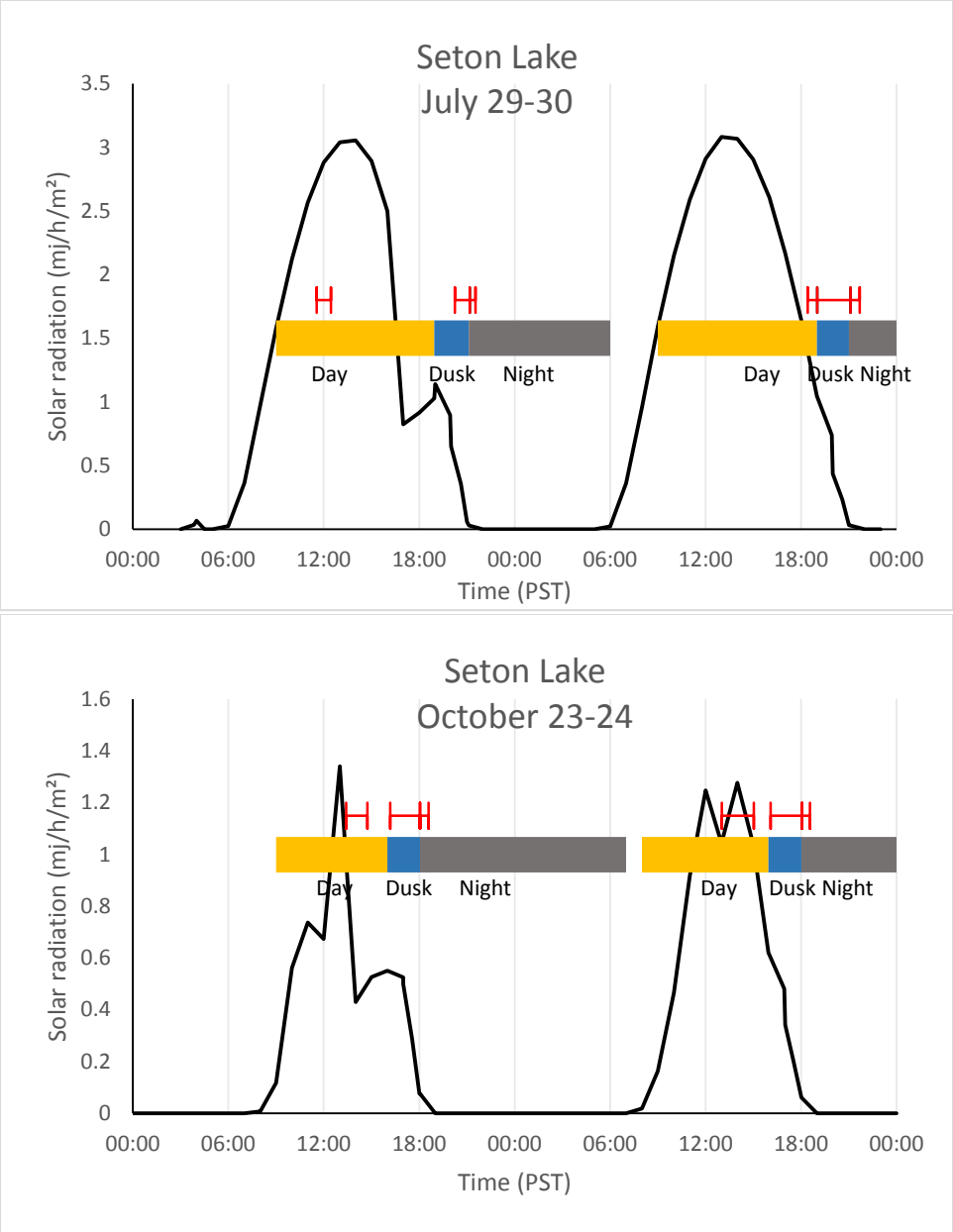


Figure 5. Diel vertical migration sampling windows (thick bars) and actual data collection periods (red lines) in relation to solar radiation (black lines) during the summer and fall surveys in Seton Lake.

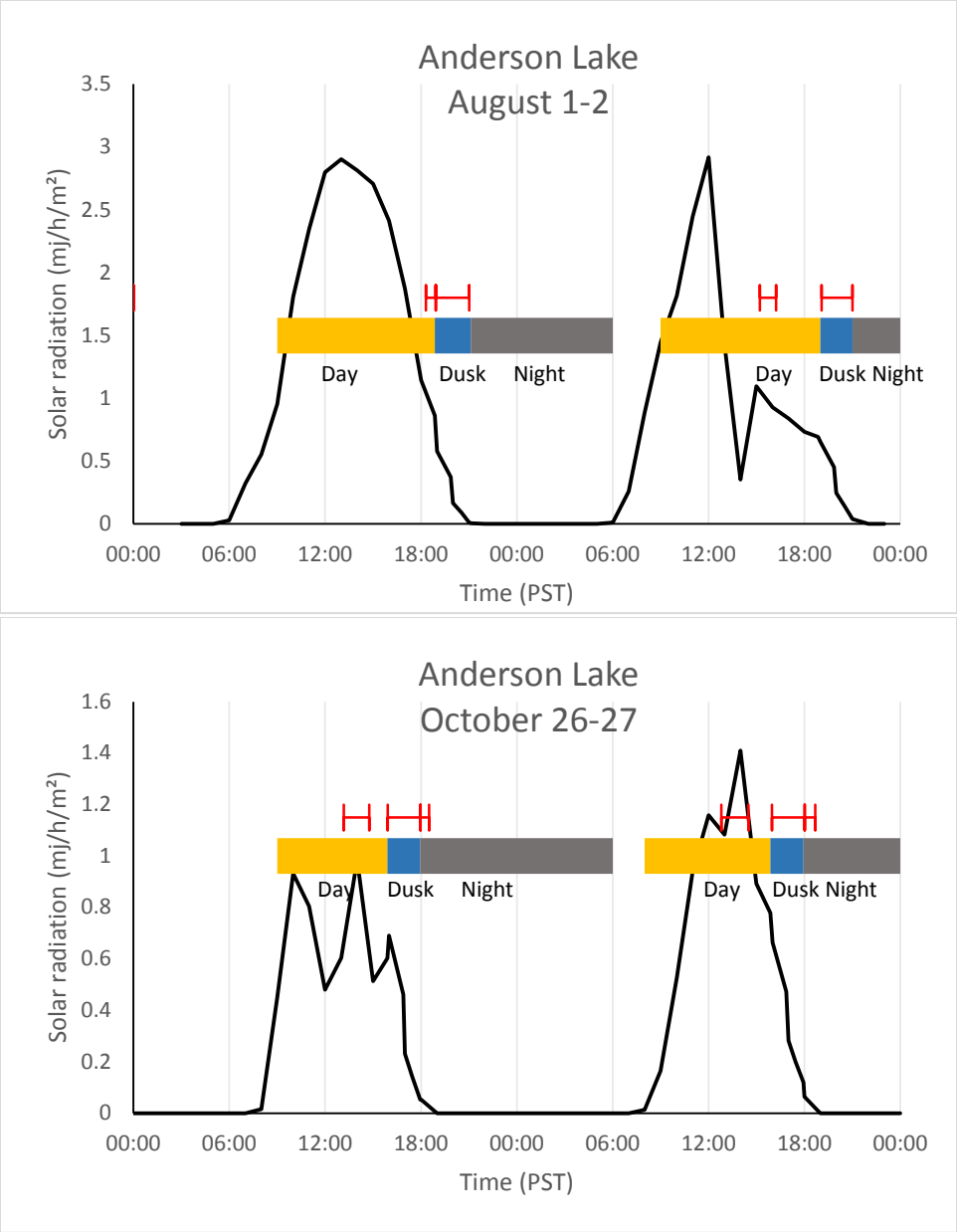


Figure 6. Diel vertical migration sampling windows (thick bars) and actual data collection periods (red lines) in relation to solar radiation (black lines) during the summer and fall surveys in Anderson Lake.

3.4 Question 4: Can refinements be made to the selected alternative to improve habitat conditions or enhance fish populations in Seton Lake?

Question 4 will be answered using multiple lines of evidence. If analyses addressing question 2 show no effect of N2-2P on primary and secondary production, changes to N2-2P will be irrelevant because it has not changed biological production in Seton Lake compared to hydrological conditions present before N2-2P was implemented. If analyses show that N2-2P has significantly altered biological production, factors potentially contributing to the change will be contrasted between the before and after periods. Those factors may be water residence time, light attenuation, turbidity, suspended sediment concentrations, water temperature, or nutrient concentrations. The influence of any temporal differences among these attributes on biological production will be explored with reference to relevant literature. Modification of N2-2P that may change attributes influencing biological production will be proposed as options for change to N2-2P to increase biological production.

All of these investigations of lines of evidence will be done once findings from all other parts of the project are complete. That will occur in the final year of work that is scheduled to be 2016-17.

4 RESULTS AND DISCUSSION

4.1 Overview

Data were collected and analysed in 2014 that will contribute to answering the management questions by the end of the three years of study. Field and laboratory work was completed in 2014 to contribute to answering the first three questions. Those results and preliminary interpretations are provided below in Sections 4.2 to 4.4. Data and interpretations to address question 4 will be done in the final year of study and thus are not discussed in this report.

4.2 Question 1: What is the inter-annual variation in physical conditions in the Seton Lake caused by the diversion and did the diversion change primary and secondary production in Seton Lake?

4.2.1 Core dating

Two cores from Anderson Lake (AP1 and AP2) and one core from Seton Lake (SP2) have been analyzed to date. Locations from where the cores were taken are shown in Figure 1. In all cores, total ^{210}Pb activity versus core depth followed exponential decay functions that were modelled by first-order polynomials (r^2= 0.87 for AP1 (Figure 7b), r^2= 0.90 for AP2 (Figure 8b), r^2=0.90 for SP2 (Figure 9b)). This response provided evidence of undisturbed sediment cores that were suitable for modeling a depth-time relationship shown in Table 3 for AP1 in Anderson

Lake and in Table 4 for SP2 in Seton Lake. Background (or supported) levels of ^{210}Pb (where total ^{210}Pb activity and total ^{214}Bi activity intersect) were reached by ~15cm in Core AP1 in Anderson Lake (Figure7a). In that same core, the cesium peak was distinguishable and reached its maximum in measured samples at 8 cm (Figure7a). Background ^{210}Pb was reached by 12.25 cm in Core AP2 from Anderson Lake (Figure 8a). The cesium activity in AP2 (Figure 8a) was not characterized by as pronounced a peak like in Core AP1 (Figure 7a) but was present and it reached a maximum activity at ~ 9 cm. In Core SP2, background ^{210}Pb was reached at ~40 cm (Figure 9a) and a distinct cesium peak was found at 24 cm (Figure 9a). Finding ^{210}Pb background deeper in the Seton core than in the Anderson cores inferred greater rates of sediment deposition in Seton Lake than in Anderson Lake. This difference is shown over time in Figure 10.

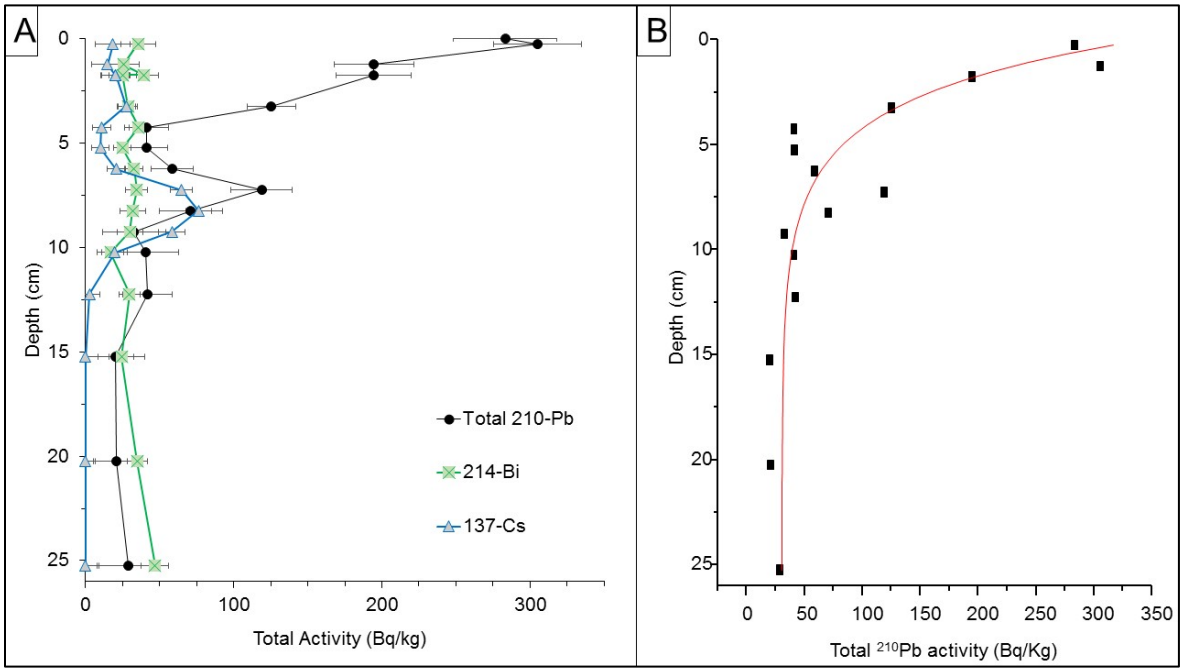


Figure 7. Measured activities of ^{210}Pb, ^{137}Cs and ^{214}Bi in Core AP1 from Anderson Lake (A) and exponential decay of ^{210}Pb activity with core depth (r^2= 0.87) (B).

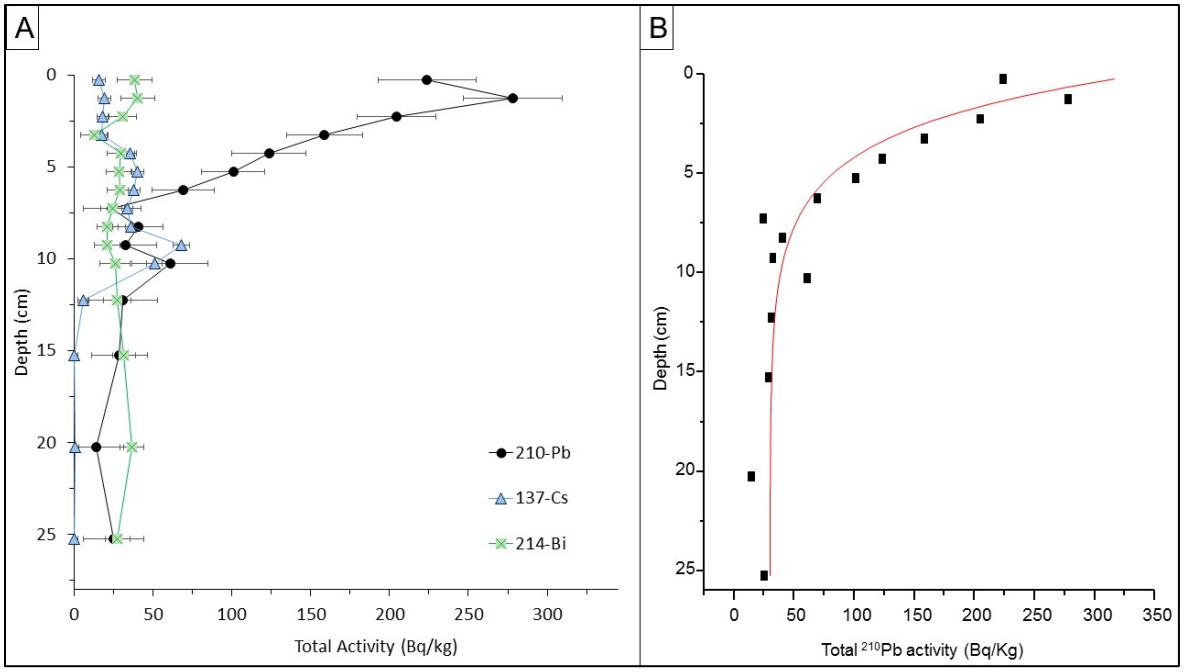


Figure 8. Measured activities of ^{210}Pb, ^{137}Cs and ^{214}Bi in Core AP2 from Anderson Lake (A) and exponential decay of ^{210}Pb activity with core depth (r^2= 0.90) (B).

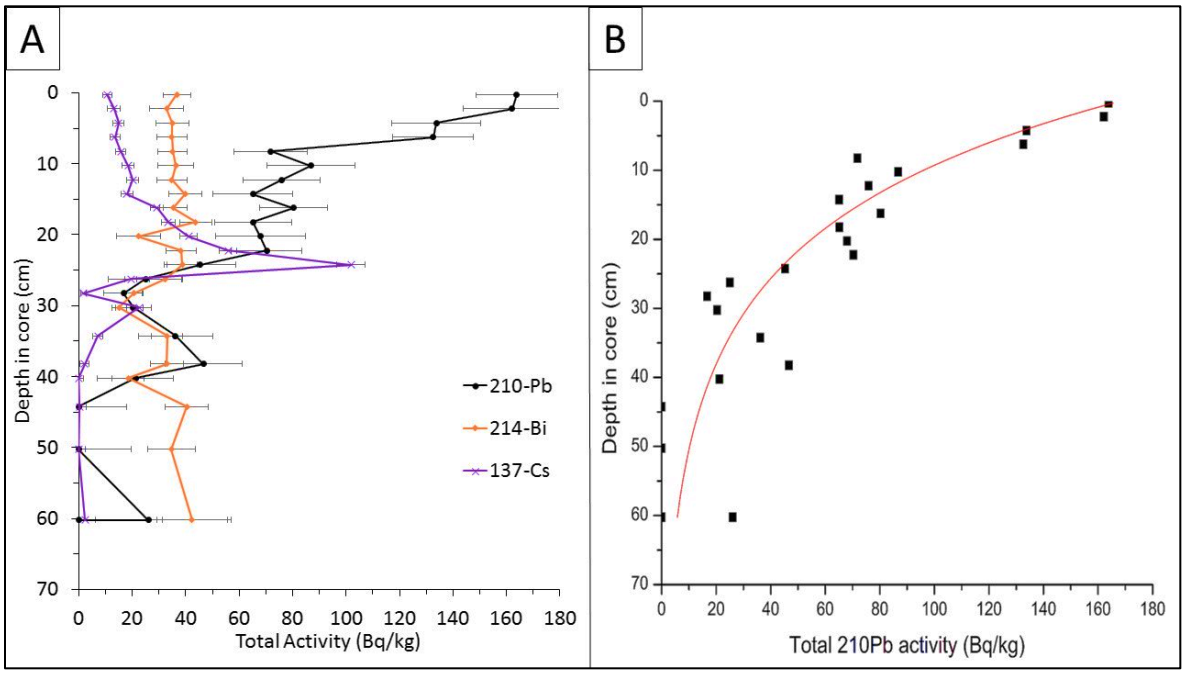


Figure 9. Measured activities of ^{210}Pb, ^{137}Cs and ^{214}Bi in Core SP2 from Seton Lake (A) and exponential decay of ^{210}Pb activity with core depth (r^2= 0.90) (B).

Table 3. Preliminary chronology for the sediment core AP1 from Anderson Lake.

Midpoint Depth (cm)	Age (CRS) (years before date of coring)*	Year C.E. (CRS)*
0	0	2014.6
0.25	2.6	2012
1.25	7.8	2006.8
1.75	12.4	2002.2
3.25	25.4	1989.2
4.25	33	1981.7
5.25	36.4	1978.2
6.25	40.6	1974
7.25	52.1	1962.5
8.25	73.6	1941
9.25	87.9	1926.7
10.25	100.3	1914.3
12.25	135.1	1879.5

*C.E refers to common era and CRS refers to an estimate of date from the constant rate of supply model

Table 4. Preliminary chronology for the sediment core SP2 from Seton Lake.

Midpoint depth (cm)	Age (CRS) (years before date of coring)*	Year C.E.(CRS)*
0	0	2014.6
0.25	1.9	2012.8
2.25	6.6	2008
4.25	12.9	2001.7
6.25	19.6	1995.1
8.25	25.8	1988.8
10.25	31.2	1983.4
12.25	37.5	1977.2
14.25	44.8	1969.8
16.25	54.5	1960.2
18.25	64.7	1950
20.25	77.8	1936.9
22.25	107.9	1906.8

*C.E refers to common era and CRS refers to an estimate of date from the constant rate of supply model

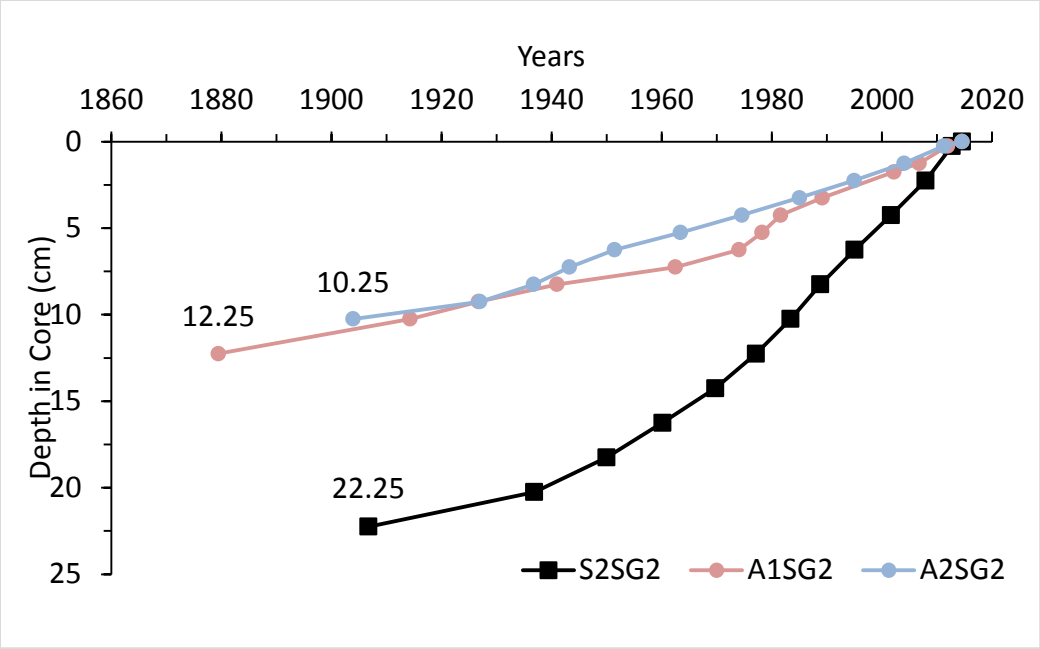


Figure 10. Variation in core age by depth between sediment cores from Anderson Lake (A1SG2 and A2SG2) and the single core from Seton Lake (S2SG2).

4.2.2 Magnetic susceptibility analysis

The magnetic susceptibility in cores from Seton Lake was episodically greater at Station SP3 (closest to Shalalth) than at stations further east (Figure 11a) within the top 20cm of core depth. Dating of cores from SP1 and SP3 is not complete but given the known dating from SP2 (Table 4), it is likely that this episodic site effect occurred after the diversion from Carpenter Reservoir started. Given that these changes occurred at SP3 and not at the more distant stations from Shalalth it is evident that change in sediment attributes was from material introduced to Seton Lake in the diverted water. It is also evident that settlement of that sediment occurred mostly at the west end of Seton Lake because coincidental changes in magnetic susceptibility did not occur at sites further east.

In the deeper sediments there were large changes in magnetic susceptibility between sites in Seton Lake. At the 28-32 cm depth interval, the magnetic susceptibility at SP2 was far greater than at SP3, which was greater than at SP1, which over the depth range of 34-55 cm the magnetic susceptibility was greater at SP3 than at the other stations. These differences were naturally occurring and unrelated to the diversion. The actual timing cannot be resolved until sediment dating is completed.

In the cores from Anderson Lake (Figure 11b), the magnetic susceptibility seems to be cyclic and generally higher than in the Seton Lake cores at SP1 and SP2 over most of the core depth. Again, dating needs to be completed before temporal and spatial comparisons can be made.

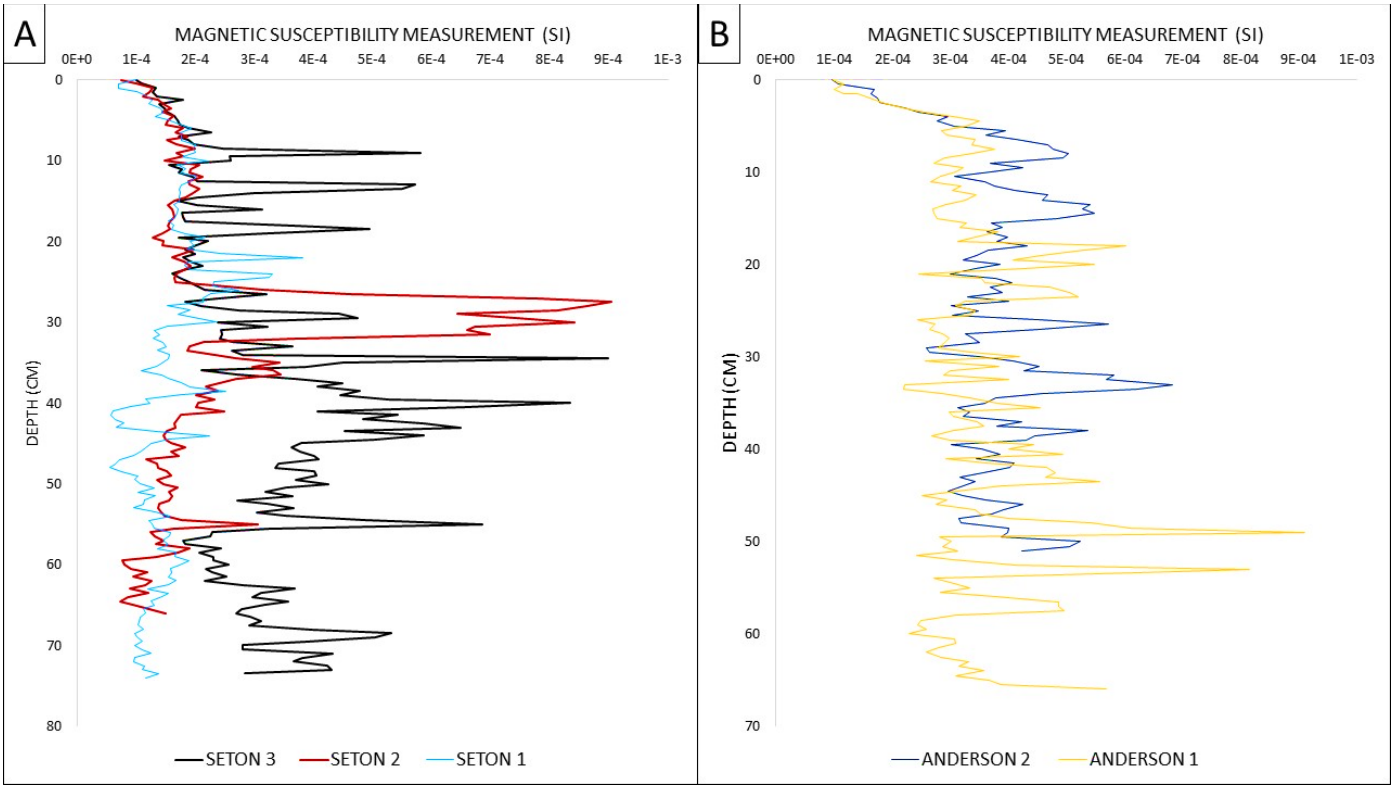


Figure 11. Comparison of the magnetic susceptibility in each core from (A) Seton Lake (cores SP1, SP2, and SP3), and (B) the cores from Anderson Lake (AP1and AP2).

4.2.3 Cladoceran analysis

Cladocera assemblages found in core SP3 from Seton Lake were dominated by *Bosmina longispina* and *Daphnia longispina* (Figure 12), with a predominance of *Bosmina spp.* (Figure 13). *Bosmina* almost represented the entire zooplankton remains from the bottom of the core to a depth of 50cm. After that point the relative abundance of *Bosmina spp.* decreased and *Daphnia longispina* increased. Both species interchangeably co-dominated at core depths of 50 cm to the surface but in the top 8 cm the relative abundance of *Bosmina spp* was greater than that of *Daphnia longispina*. There was a consistent increase in relative abundance of *Daphnia pulex* in the top 10cm of the core, which was not found in deeper sections. *Daphnia* ephippia were found in the middle of the core (between 40 and 30cm) corresponding with a progressive decline in relative abundance of *Daphnia longispina*.

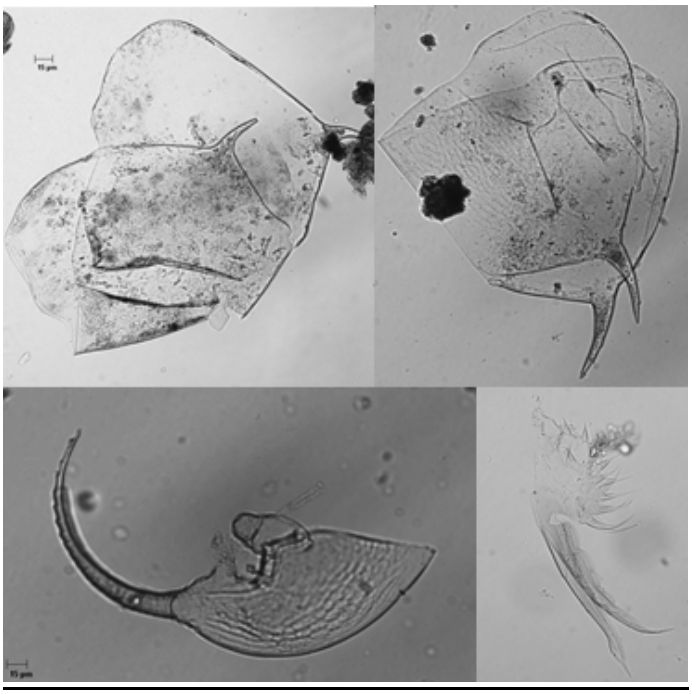


Figure 12. Samples of Cladocera remains found in core SP3 from Seton Lake (*Bosmina spp.* carapaces at the top; *Bosmina spp.* headshield bottom left; *Daphnia spp.* postabdominal claw bottom right).

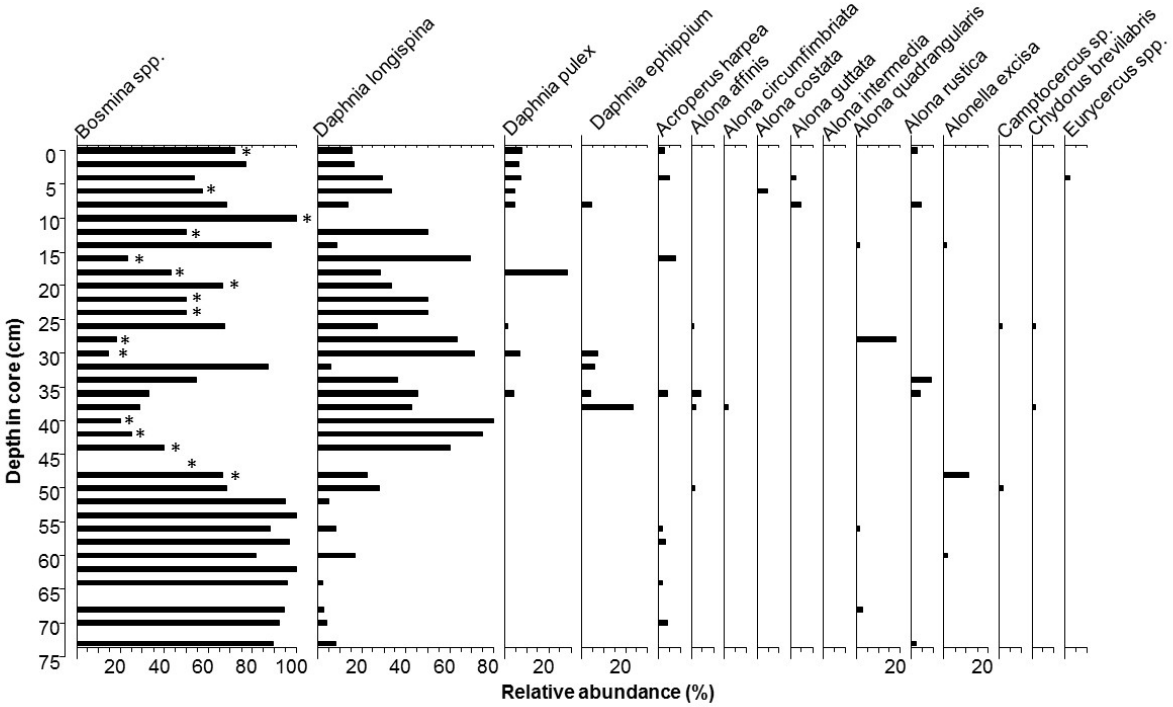


Figure 13. Relative abundance of Cladocera in core SP3 from Seton Lake. Intervals in which the minimum count has not yet been reached because of low concentrations have been marked by an asterisk. *Daphnia* ephippium is the resting stage of *Daphnia sp*.

Bosmina spp. body part lengths varied within small ranges between 15 cm to the bottom of the core but they declined in the top 15cm of the core (Figure 14). Mucro length decreased by ~26 µm in the top 3 samples from ~100 µm deeper in the core. A corresponding decline in lengths was found for antenna length and carapace length.

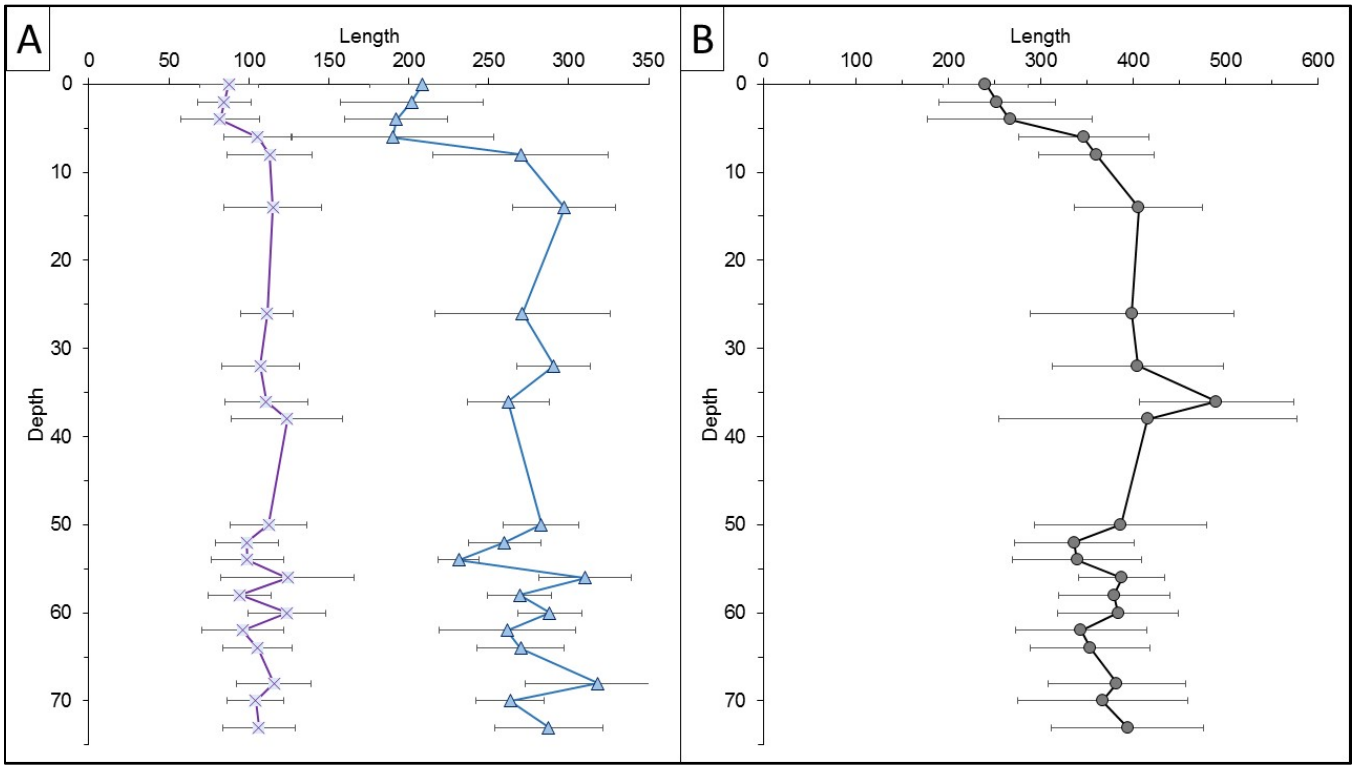


Figure 14. Changes in *Bosmina spp.* mucro length (A, purple cross) and antenna length (A, blue triangle) and carapace length (B, black circle) in core S3 from Seton Lake. *Bosmina* body length was not measured for the intervals between 16 cm and 24 cm and between 42 cm and 50 cm because of lack of measurable Cladocera remains.

4.2.4 Diatom analysis

The enumeration and identification of diatoms from core A1 from Anderson Lake has been completed on 41 samples (every 1 cm from 0-39cm (Figure 15). One-hundred and fifty-six diatom taxa were encountered. Most taxa are rare, with only 16 taxa reaching abundances >4% in at least one sample. While taxonomically-identified in the raw data, rare taxa were grouped into either genera or larger taxonomic groups (e.g. other planktonics) for presentation.

The diatom assemblages were dominated by planktonic taxa, consisting of 78-94% (mean = 88%) of the assemblage. *Discostella stelligera* was a dominant planktonic taxon throughout the core. Subdominant planktonic taxa through much of the core included *Aulacoseira subarctica*, *Cyclotella ocellata* and *Stephanodiscus minutulus*. A distinct change in

the dominant planktonic taxa occurred in the top 5 cm with the abrupt increase in *Cyclotella comensis*. A small increase in *Asterionella formosa* occurred in the top 10 cm, with small increases in *Cyclotella gordonensis* and *Fragilaria crotonensis* in the top 5 cm.

Both oligotrophic (low productivity) and eutrophic (high productivity) taxa occurred throughout the core. Fluctuations in eutrophic taxa such as *Aulacoseira subarctica* and *Stephanodiscus minutulus* with oligotrophic taxa *Discostella stelligera* and *Cyclotella ocellata* may be a reflection of changes in the abundance of salmon migration into Anderson Lake. Higher abundances of meso-eutrophic to eutrophic diatom taxa in sediment cores from salmon lakes have been shown to correspond to periods of higher salmon abundance which can lead to an enrichment of nutrients into the lake (e.g. Gregory-Eaves et al. 2003). Later isotopic analyses will be used to help determine any linkages to inferred salmon abundances.

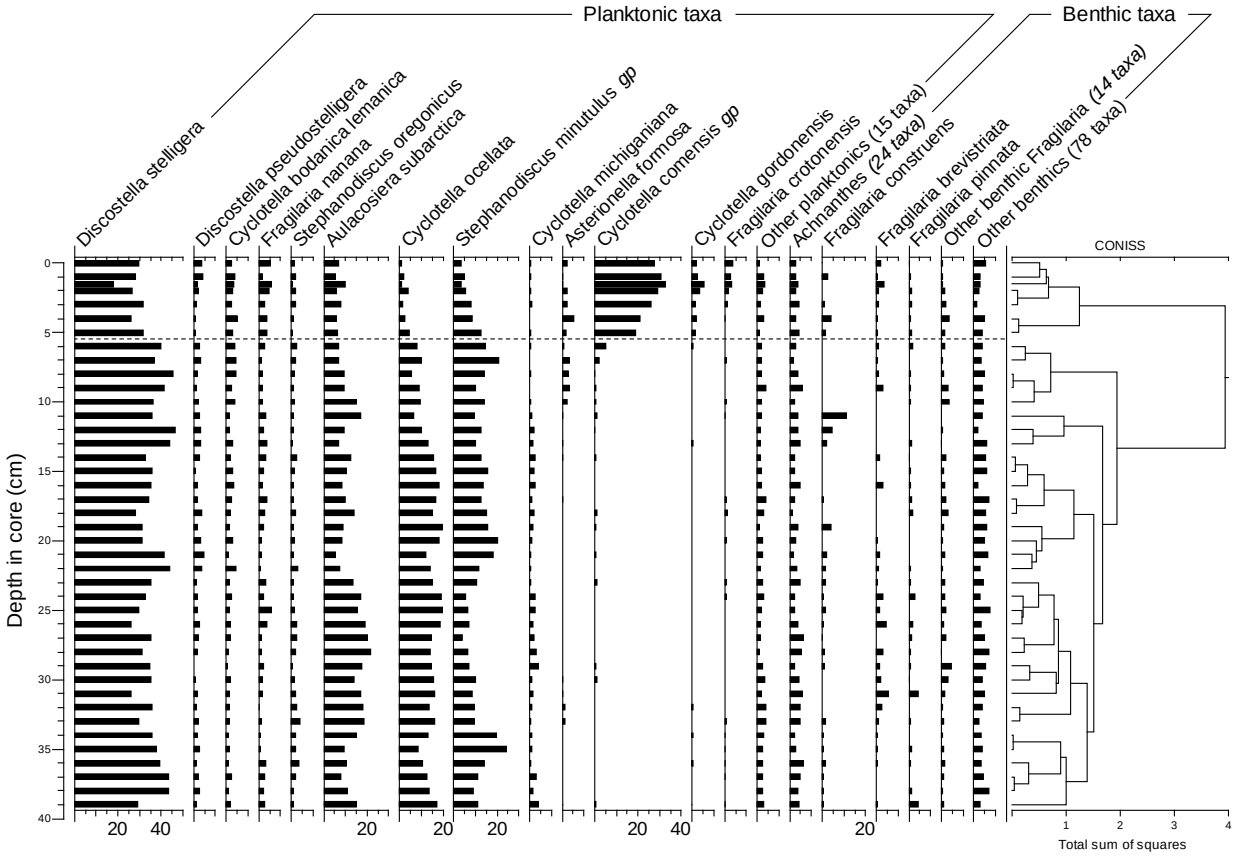


Figure 15. Diatom relative abundances of the dominant taxa in core A1 from Anderson Lake.

4.3 Question 2: Will the selected alternative (N2-2P) increase biological production in Seton Lake?

4.3.1 Defining the hydrologic treatment

A statistically significant difference (p<0.001) between the three blocks of years (before, transition, after) was found for each of the hydrologic metrics (Table 5). Water surface elevation was greatest in the after period and lowest during the transition years but the mean difference was only 5 cm. In a lake with a mean depth of 85m, this change is not biologically important. Similarly, live storage volume was greatest in the after years in association with the small rise in water surface elevation but again the difference from earlier time periods was trivial and not biologically important. These small changes occurred because of an approximate balance between a decline in diversion inflows between the before and later years and a decline in lake outflow to the Seton River mainly during the transition years. The decline in diversion inflow occurred to offset a release of water to the Lower Bridge River from Carpenter Reservoir as part of N2-2P. That release to the Lower Bridge River did not occur before N2-2P was implemented. Lake water residence time doubled from 2.4 to 4.8 years between the before and transition years due to smaller outflows to the Seton River during the transition years. Water residence time then returned to 2.5 years, which was similar to that occurring before N2-2P was implemented. The water residence times were always greater than a year, which means that the annual cycle of growth of phytoplankton and zooplankton and availability of food for fish would not have been affected by the water management.

Table 5. Mean (± SD) of hydrological metrics for Seton Lake between the "before", "transition", and "after" time periods.

Hydrological metric in Seton Lake	Mean metric value ± standard deviation			Time effect (p)
	Before (1996 - 1999)	Transition (2000 - 2003)	After (2011 - 2014)	
Water Surface Elevation (m)	236.22 ± 0.06	236.19 ± 0.12	236.26 ± 0.06	<0.001
Live storage volume (Mm^3)	22.6 ± 1.4	21.7 ± 2.8	23.5 ± 1.5	<0.001
Total lake volume (Mm^3)	2100*	2100*	2100*	Not applicable
Diversion inflow ($m^3 \cdot s^{-1}$)	102.8 ± 29.9	81.7 ± 35.1	85.6 ± 28.9	<0.001
Outflow to Seton River ($m^3 \cdot s^{-1}$)	32.2 ± 32.9	15.2 ± 9.4	26.7 ± 18.3	<0.001
Lake water residence time (years)**	2.4 ± 1.3	4.8 ± 1.8	2.5 ± 0.4	0.049

*data from Geen and Andrew (1961), which is based on survey data from International Pacific Salmon Fisheries Commission, 1953.

**calculated as total lake volume divided by mean annual rate of outflow to the Seton River

Given our interest in pelagic habitat of Seton Lake for rearing of juvenile sockeye salmon and all life stages of Gwenish, these analyses show that N2-2P did not change available habitat for these fish species and life stages in Seton Lake and residence time of water needed to support food production for fish between the three blocks of years. If a significant difference among biological production metrics is found between before and after years following all three years of data collection (2014 – 2016), it will be attributed to factors other than system hydrology.

4.3.2 Habitat attributes

4.3.2.1 Temperature

Both lakes were thermally stratified during the period of measurement in 2014 (Figure 16). In May the thermoclines were broad, showing weak resistance to mixing but with heating during spring and summer, a well-defined thermocline was established over a depth range of 20-25m in both lakes. A maximum surface temperature close to 22ºC was recorded on the August casts. Despite surface cooling in the fall, resistance to mixing remained high and a distinct epilimnion and hypolimnion remained intact in both lakes at the end of sampling in late October. At that time the surface temperatures had cooled to 12-13ºC. Hypolimnetic temperatures were 4-5ºC, the temperature at which water has highest density.

Structuring of the Seton Lake thermal data over the distance from S1 to S6 in August and October showed no disturbance of the thermal structure in Seton Lake from the inflow of Carpenter Lake water (Figure 17). No unusual pattern along boundary layers between the epilimnion and hypolimnion was found, which implies no physical disturbance that exceeded resistance to mixing was present. The August data do show an upward tilt of the thermocline west to east, which is consistent with presence of a seiche, an internal wave oscillation set up between the epilimnion and hypolimnion over the length of the lake. Seiche activity is common in long narrow lakes like Seton and Anderson Lakes. It is caused by wind that pushes water to one end of the lake. When the wind stops, the water rocks back in the opposite direction. In a temperature stratified lake, the effect can be observed as the thermocline tilting in one direction and then the other. The magnitude of oscillation appearing in Figure 17 was 5m but it could be more or less at other times in relation to the pattern of oscillation. Remnants of oscillations or inter-laminar eddies are apparent in the October isopleth by a discontinuity of the thermocline at station S5. We don't know what was the variation in amplitude or maximum amplitude of the oscillation over time but if large enough it may have contributed to intra-day 3-4ºC shifts in temperature that were observed in the Seton River (e.g. July 15, August 3, August 20, September 8 and others: Figure 18). Analysis of data from a thermistor chain that was installed near our Station 6 in Seton Lake as part of another monitoring study along with river temperature and information about flow management at the dam during the time of seiche

activity is needed to investigate links between seiche activity and river temperature in greater detail.

Water temperature in the tailrace at BR1 was 7ºC in May, it increased to 15ºC in early September and declined to 12ºC in late October (Figure 19a). These changes were interspersed with periods of higher temperature caused by periodic shutdown of the turbines, which resulted in backwatering of relatively warm lake water that produced temporary rises in temperature at the location of the temperature logger. The baseline of the curve in Figure 19a was the actual temperature of water discharged at BR1 because that baseline occurred when the turbines were operating. Water seeks similar density when it flows into a body of other water, largely defined by temperature. This basic law means that cool Carpenter water flowing into Seton Lake at BR1 and the Bridge 2 generating station called BR2 that is situated immediately west of BR1 in the spring flowed to the hypolimnion of Seton Lake where temperature was similar to that of the inflow. As temperature of Carpenter Lake water increased over the summer, it eventually reached a level that was similar to that of the epilimnion in Seton Lake and discharge from BR1 and BR2 would have flowed in the epilimnion of Seton Lake without mixing in the hypolimnion.

Portage Creek water would be expected to have flowed differently in Seton Lake. Portage temperature was 10ºC in late May, it increased to 20ºC by early August, and then declined to 13ºC by late October (Figure 19b). The peak temperature was approximately one month earlier than in diversion water from Carpenter Reservoir (Figure 19a) and it coincided with the timing of peak temperature in Anderson. At all temperatures in Portage Creek, the water would have flowed to similar temperature near the surface of Seton Lake , except possibly in the early spring when stratification was forming.

There were episodic changes in water temperature in Portage Creek (Figure 19b) that were not typical of its source in Anderson Lake where surface water temperature changed gradually over time (Figure 16b). Those changes must have been related to temporal variation in discharge from Whitecap Creek that originates in the alpine and flows into Portage Creek from the north. Flow from Whitecap Creek likely cooled Portage Creek and would have influenced its temperature and flow in association with precipitation within its drainage.

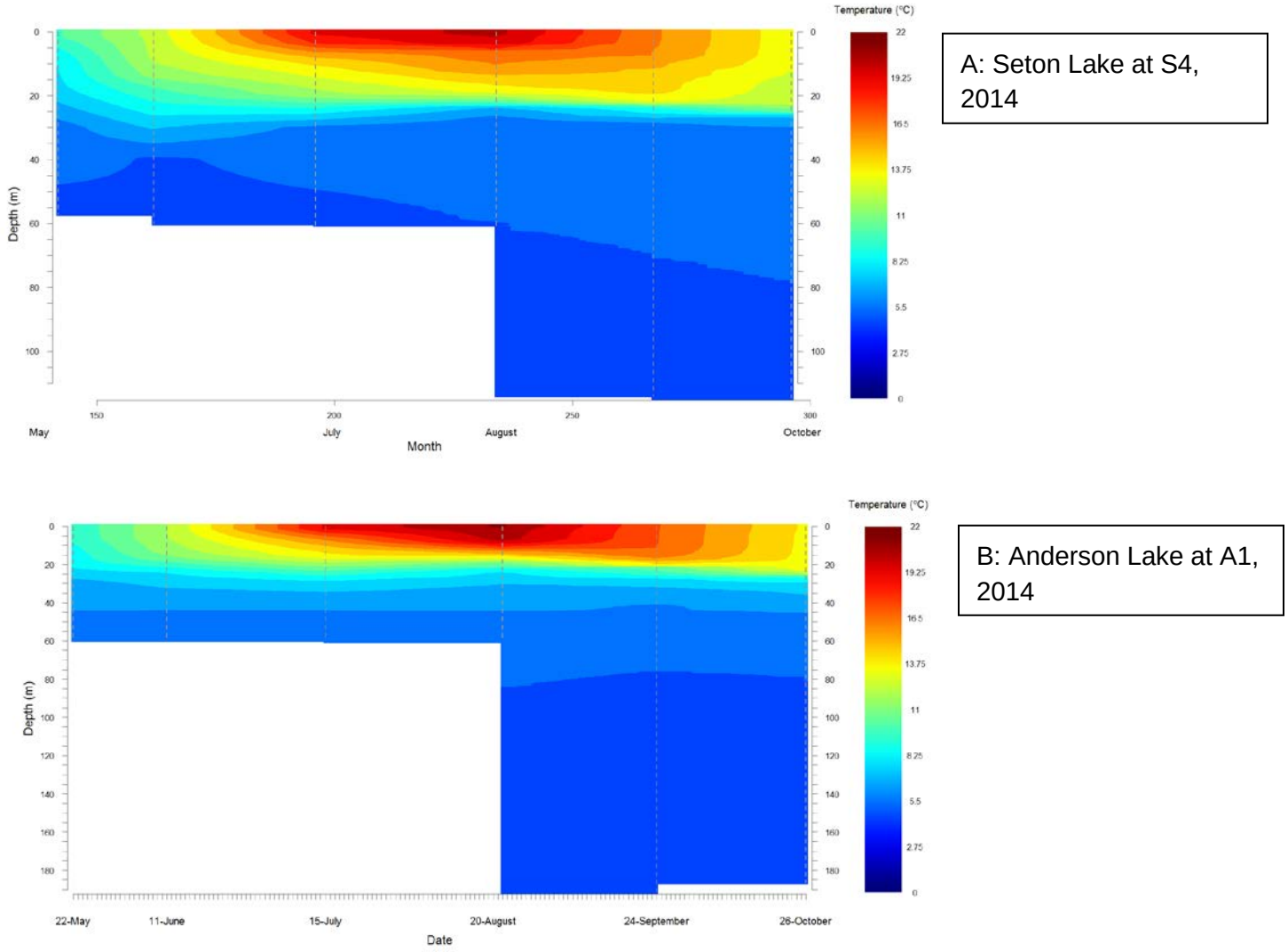
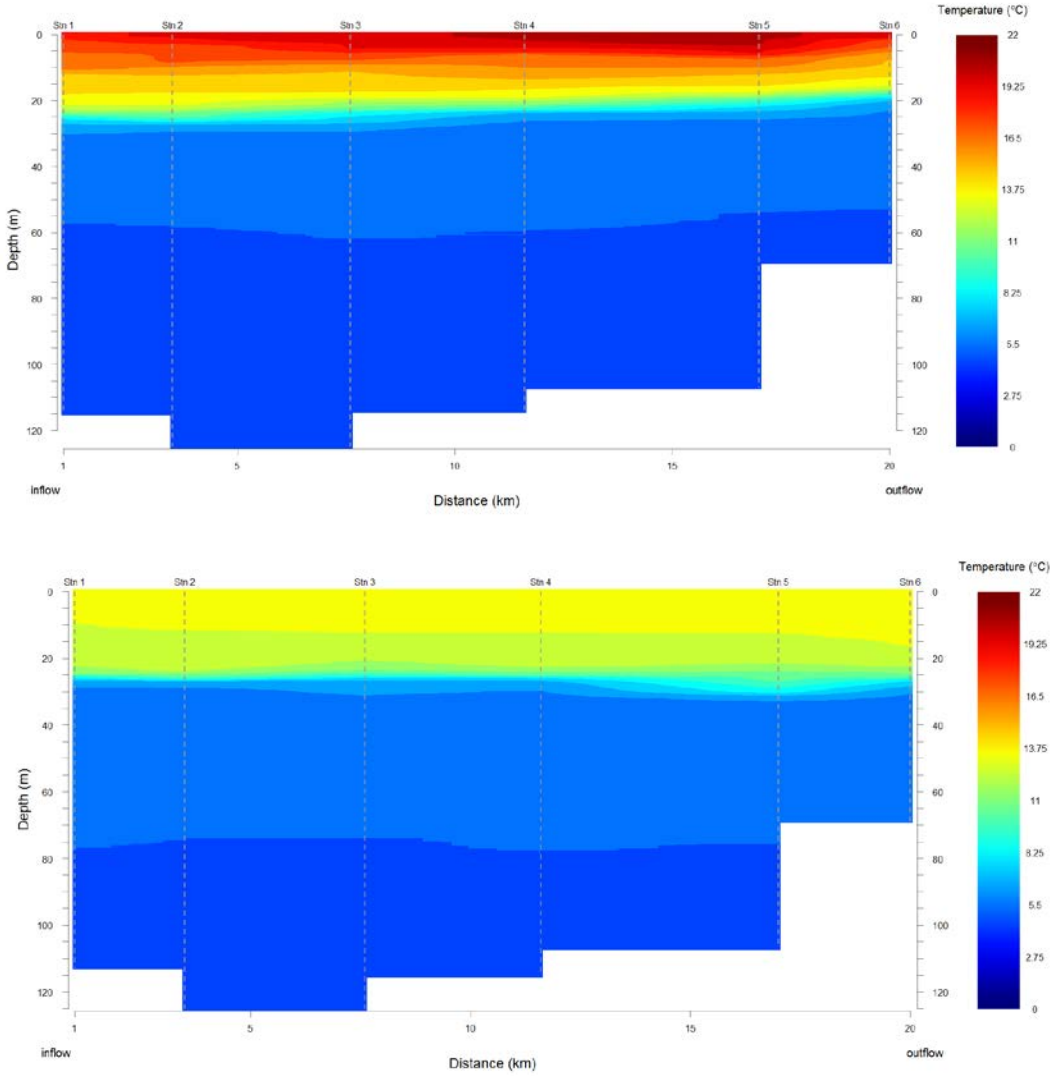


Figure 16. Water temperature during May-October, 2014 in Seton Lake at S4 (a) and Anderson Lake at A1 (b). The profiles were to 60m in May through July when the YSI Sonde was used and it extended to the lake bottom in August to October when the SeaBird CTD was used. The vertical dotted lines indicate dates of measurement. Data between those dates were linearly interpolated.



A: Seton Lake, August 20, 2014

B: Seton Lake, October 23, 2014

Figure 17. Water temperature profiles integrated between all six stations on August 20 (top) and October 23 (bottom), 2014 casts in Seton Lake. S1 is west of Shalalth and S6 is close to the outflow at the east end. Vertical dotted lines indicate stations of measurement. Data between those stations were linearly interpolated.

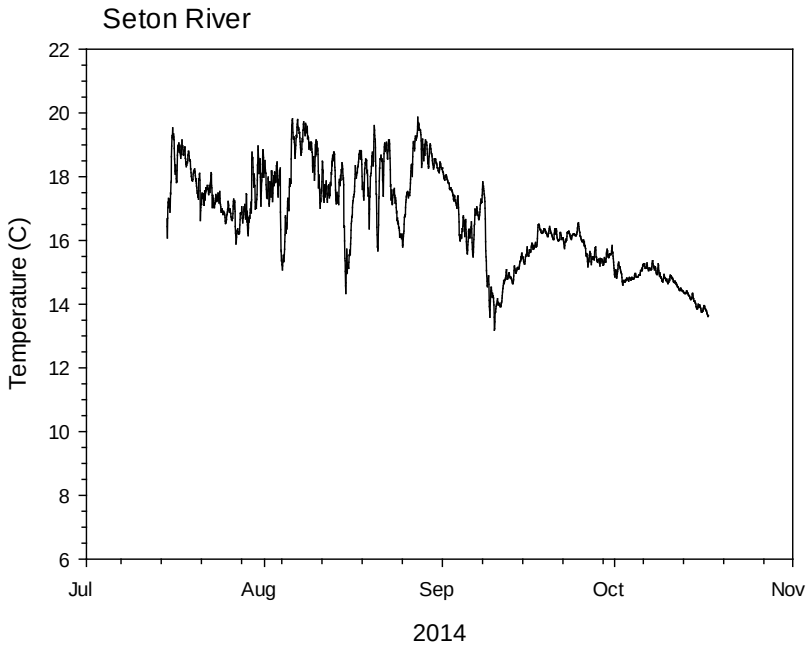


Figure 18. Hourly temperature in the Seton River 200m downstream of the Seton Dam (1 km downstream of the outflow of Seton Lake) in 2014.

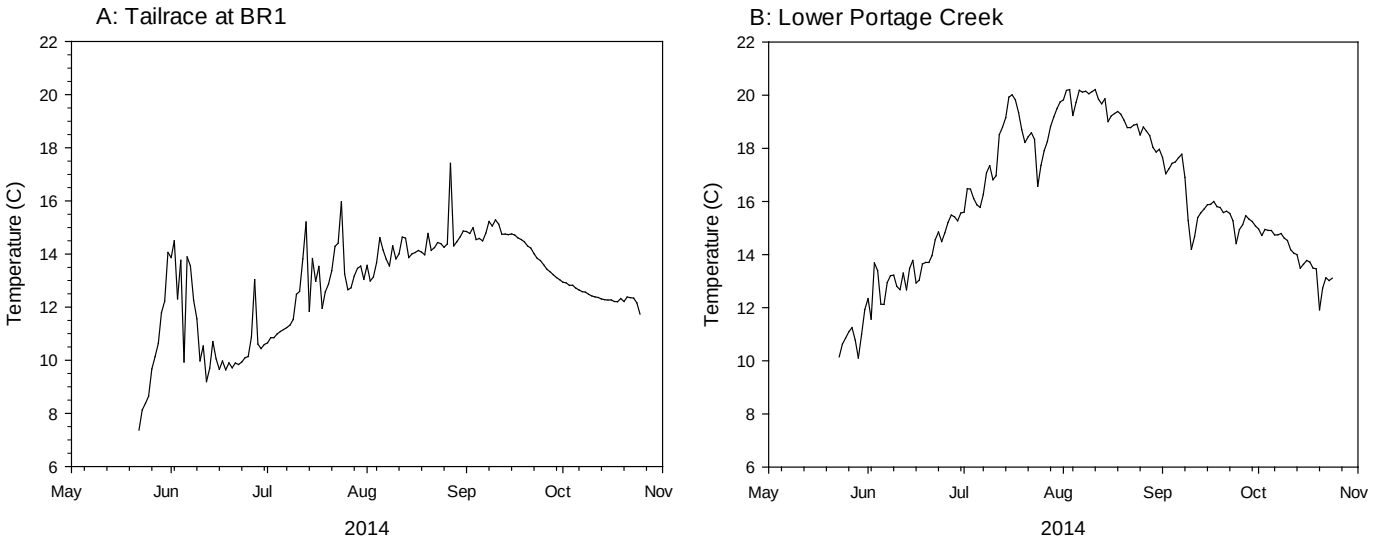


Figure 19. Mean daily temperature in the tailrace at BR1 (a) and Lower Portage Creek (b) in 2014.

4.3.2.2 Turbidity

Turbidity of the Portage and diversion inflows to Seton Lake increased over the spring through fall period while turbidity in the Seton outflow was 3.1 NTU in May, declining to ≤1.1 NTU in June through September (Figure 20). Turbidity in the BR1 tailrace was 3.5 NTU in May and it increased to 30 NTU in September. Turbidity in Portage Creek was ≤5 NTU in May through August but increased to 87 NTU in September. Given that turbidity in the upstream Anderson Lake was <1 NTU for the entire sampling period in 2014 (Figure 21), the high turbidity in Portage Creek in September must have originated as outwash of glacial fines from the alpine drained by Whitecap Creek.

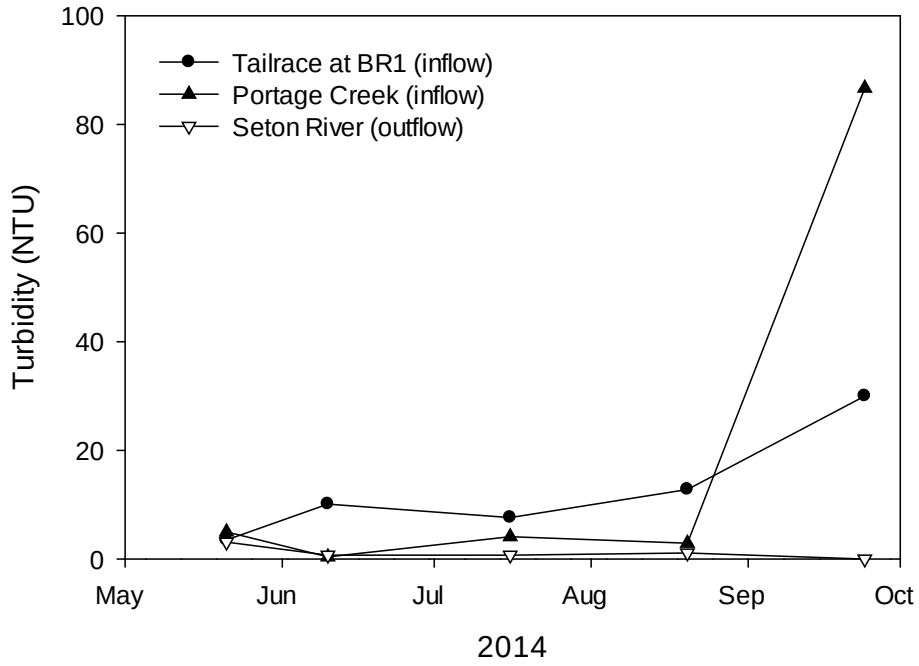


Figure 20 Monthly turbidity of Seton Lake inflow at BR1 and Portage Creek and outflow in the Seton River, 2014.

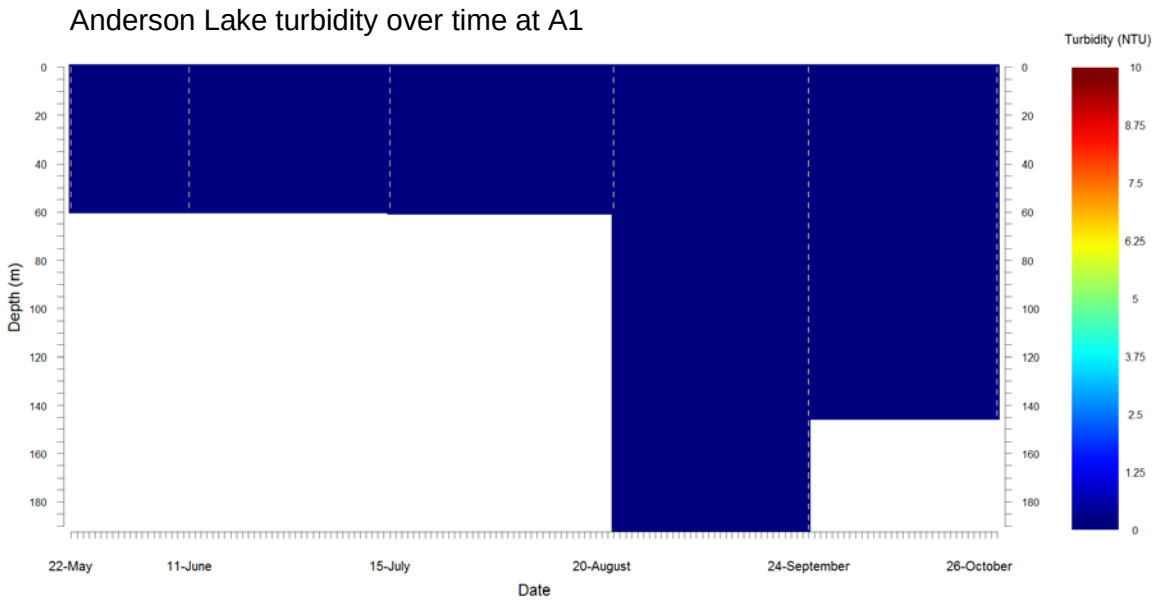


Figure 21. Turbidity in Anderson Lake at A1 during May through October, 2014.

The differences in turbidity between Seton Lake inflow and outflow (Figure 20) indicated settlement of particles in the lake. Mean particle size contributing to turbidity increased during spring through fall months but it was always <3.6 µm (Figure 22). This mean size is an overestimate because the lower half of the particle size distribution (<1.3 µm) was not detectable in all samples. Regardless, all sizes were typical of clay (Ashworth et al. 2001) that settles slower than 1 $cm \cdot hr^{-1}$ according to Stokes Law (Gee and Bauder 1986, Wetzel 2001). At that rate the particles introduced from BR1 would settle to sediment over the mean depth of Seton Lake (85 m) in not less than one year and it would take a minimum of 1.7 years for it to settle to deepest places in the lake where water depth is 151 m in the absence of short routing through density layers that may limit particle settlement. It is important to note that the differences in turbidity between the Seton Lake inflows and outflow showed that settlement of large particles did occur in the lake so not all was clay even if that material dominated the water samples. Also, the data in Figure 22 show mean particle size and did not show the full range of particle sizes, including large ones that would have settled within relatively short time periods in the lake. It is also possible that with the low frequency of sampling, particles larger than clay were present in discharge from BR1 but were missed in the monthly grab samples. A more detailed analysis of the complete size distribution of particles will be completed once more samples are collected in 2015 and 2016 to better characterize the particle size distribution.

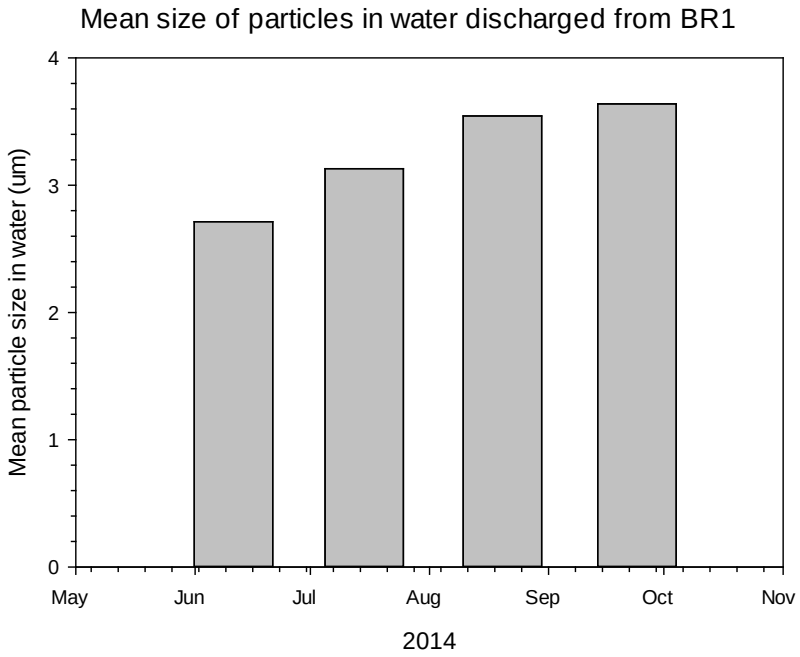
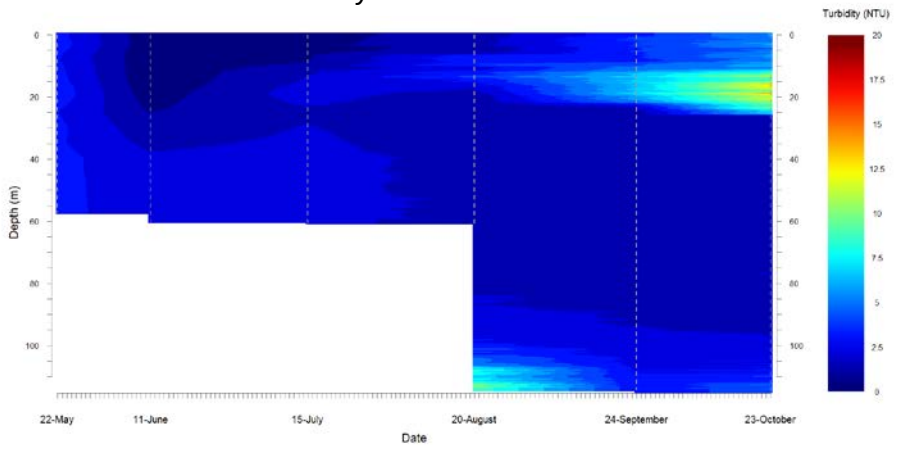


Figure 22. Arithmetic mean size of detectable particles in tailrace water at BR1 when all turbines were running and no backwatering from Seton Lake was occurring.

Changes in turbidity in Seton Lake corresponded with the phenomenon of water seeking constant density as if flows into another body of water. The increasing turbidity over time and the large increase in turbidity in the fall from BR1 and Portage Creek (Figure 20) showed in the epilimnion at S4, 9 km from BR1 and 12 km from the Portage Creek discharge (Figure 23a). Turbidity throughout the water column at S4 was <3 NTU in May. It gradually increased during the summer and fall but only in the epilimnion, reaching 14 NTU at the thermocline depth of 20m in October (Figure 23a). During that October sampling, the source of turbidity was BR1 as shown by highest turbidity among stations at S2 (18 NTU). It was also from Portage Creek because high turbidity was found at S1, which is upstream of BR1. The turbidity dissipated eastward with flow and most remained in the epilimnion (Figure 23b). At that time, the inflowing temperature was 12.3ºC in BR1 water (Figure 19a) and 13ºC in Portage Creek water (Figure 19b), both the same as lower epilimnetic temperatures in Seton Lake (Figures 16a and 17b). Flow of most turbidity along the bottom of the epilimnion shows that water from BR1 and Portage Creek flowed to similar density as defined by temperature and did not mix in the hypolimnion. Some, presumably large particles did pass to the hypolimnion based on presence of 4-5 NTU water within the hypolimnion mainly at S3. Much, however, remained in the lower epilimnion, mainly near the 20m depth. These observations show that the hypolimnion in Seton Lake has low turbidity throughout the biologically active growing season of May through October but the epilimnion became highly turbid due to density-specific flow of turbid inflows close to thermocline depth. The turbid inflow was from the generating station and from Whitecap Creek that strongly influenced the Portage Creek inflow to Seton Lake.

A: Seton Lake turbidity over time at S4



B: Seton Lake turbidity over distance in October 23

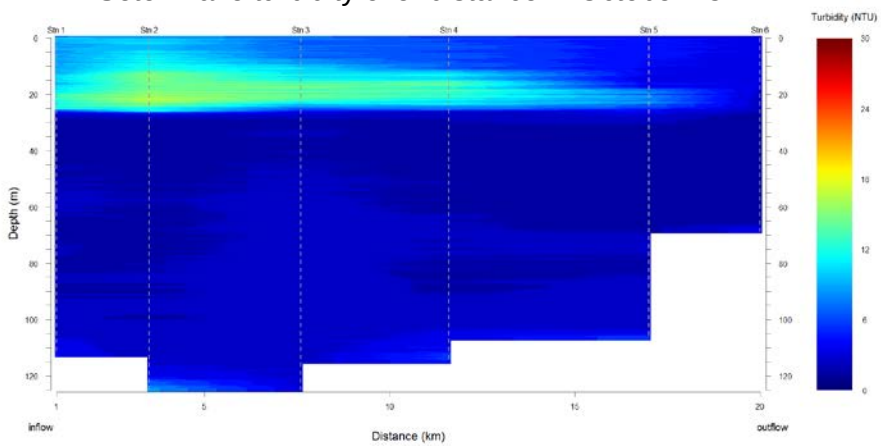


Figure 23. Turbidity in Seton Lake at S4 during May through October (A) and over the depth profile between S1 and S6 on October 23 (B). The high turbidity at the lake bottom in August (A) was due to resuspension of sediment affecting the sensor when the CTD touched bottom. The instrument did not touch bottom at other times. Vertical dotted lines indicate dates of measurement (A) or stations (B). Data between the lines were linearly interpolated.

4.3.2.3 Light

In both lakes, Secchi depth increased during May through July and declined thereafter with light attenuation being 2 to 3 times less in Anderson Lake than in Seton Lake (Figure 24). The difference between lakes can be attributed to presence of glacial turbidity in Seton but absence of glacial turbidity in Anderson. Consistency of temporal change in Secchi depth between the lakes suggests that processes contributing to that change may be different but produced the same temporal trend between the lakes. One factor may be phytoplankton biomass interacting with zooplankton grazing that may be most important in Anderson Lake. The other is glacial fines that are expected to be important in Seton. Both factors may produce the same result of increasing water clarity in spring through summer and decreasing water clarity in the fall. Over the complete growing season, Secchi depth was three times greater in Anderson Lake than in Seton Lake. That difference in light attenuation resulted in a euphotic zone depth in Anderson Lake that was twice as deep as in Seton Lake and a light extinction coefficient in Seton Lake that was double that in Anderson Lake (Table 6).

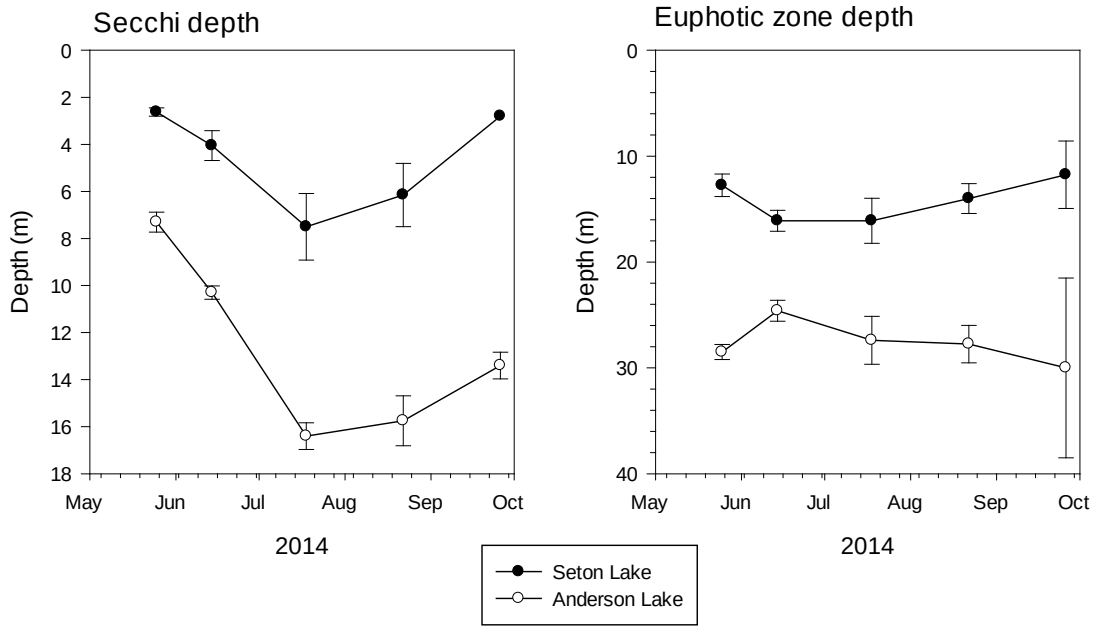


Figure 24. Mean Secchi depth (left) and euphotic zone depth (right) (±sd) in Seton and Anderson Lakes in 2014. Euphotic zone depth was the depth at which PAR was 1% of surface irradiance as measured using a LiCor LI250A irradiance meter equipped with a spherical quantum sensor. Values are from one measurement at each of 2 stations on each lake by date.

Table 6. Mean euphotic zone depth, Secchi depth and light extinction coefficient over the growing season. Values are a mean of monthly measurements at two stations on each lake over 5 months (May to September).

Metric and units	Mean light attenuation values ± sd	
	Seton Lake	Anderson Lake

Euphotic zone depth (m)	13.4 ± 3.6	28.8 ± 2.5
Secchi depth (m)	4.1 ± 1.6	12.5 ± 4.0
Light extinction coefficient	0.373 ± 0.087	0.162 ± 0.016

4.3.2.4 Chemistry

The water chemistry in Seton and Anderson Lakes was consistent with oligotrophic conditions found in earlier studies (e.g. Shortreed et al. 2001) (Table 7). The pH was slightly alkaline in both lakes. Total suspended sediment concentration was <1 $mg \cdot L^{-1}$ in both lakes at all times, indicating that the temporal and spatial variation in turbidity in Seton Lake (Figure 23) was occurring at low concentrations of suspended solids and that measurement of suspended solids at a method detection limit of 1 $mg \cdot L^{-1}$ cannot be used to assess differences in particle content between lakes. The various forms of nitrogen and phosphorus occurred at low concentrations, which is typical of nutrient deficient lakes. A difference between the two lakes is that inorganic nitrogen concentration (NO_3-N plus NH_4-N) was lower in Seton Lake than in Anderson Lake and vice versa, total phosphorus concentration was lower in Anderson Lake than in Seton Lake. These differences imply greater potential phosphorus deficiency for algal growth in Anderson Lake than in Seton Lake and greater potential nitrogen deficiency for algal growth in Seton Lake than in Anderson Lake.

The molar ratio of bioavailable N:bioavailable P in water can indicate the relative supply of N and P for phytoplankton. Bioavailable N can be approximated as the DIN concentration (NO_3-N plus NH_4-N) when detectable or TN concentration when it is not. Bioavailable P can be approximated as SRP concentration, when it is detectable or TDP if it is not or TP if nothing else works. A challenge with using molar ratios is that they often can't be compared between times or between lakes if they are calculated in different ways depending on what forms of N and P can be detected. When the bioavailable forms of N and P can be detected, Rhee (1978) showed that for a given species of algae there is a sharp transition between P-limited and N-limited growth. The particular N:P ratio at which the transition between N and P-limitation occurs is species dependent, varying from as low as 7:1 for some diatoms (Rhee and Gotham 1980) to as high as 45:1 for some blue-greens (Healey 1985). It is commonly regarded that below a molar N:P ratio of 20, growth of most algal species will be limited by N whereas P-deficient growth is prevalent at molar N:P ratios greater than 50 (Guildford and Hecky 2000). Because an optimum N:P ratio (above which P limitation occurs and below which N limitation occurs) can vary among freshwater algae, the range between 20 and 50 may be regarded as a transition range in a community where some species will be P-limited and others will be N-limited. These ratios are relevant to the epilimnion of lakes where there is photosynthetic activity.

Table 7 shows that the bioavailable forms of N (NH_4-N and NO_3-N) and P (SRP or TDP) were not sufficiently detectable in either lake for calculation of molar N:P. Even TN that contains inorganic and complex organic fractions of N that are not bioavailable

were not detectable in the epilimnion of both lakes. TP was also not detectable in the epilimnion of Anderson Lake. In this circumstance molar N:P cannot be reliably calculated. Simple review of N and P concentrations as was done above is the best option for examining potential N and P deficiency in phytoplankton. That analysis implied greater potential phosphorus deficiency for algal growth in Anderson Lake than in Seton Lake and greater potential nitrogen deficiency for algal growth in Seton Lake than in Anderson Lake.

Table 7. Mean chemical concentrations and other measures in the epilimnion and hypolimnion of Seton and Anderson Lake in 2014 (n=10).

Analyte	Seton Lake		Anderson Lake	
	Epilimnion	Hypolimnion	Epilimnion	Hypolimnion
pH	7.9 ± 0.04	7.8 ± 0.02	8.0 ± 0.13	8.0 ±0.05
TSS ($mg \cdot L^{-1}$)	<1*	<1*	<1	<1
NH_4-N ($\mu g \cdot L^{-1}$)	< 5	< 5*	< 5	< 5
NO_3-N ($\mu g \cdot L^{-1}$)	< 5**	41.4 ± 6.7	20.7 ± 18.8	69 ± 8.1
TN ($\mu g \cdot L^{-1}$)	<50*	58 ± 23	<50**	89 ±13
SRP ($\mu g \cdot L^{-1}$)	< 1	<1	<1**	<1
TDP ($\mu g \cdot L^{-1}$)	< 2**	<2**	<2**	<2**
TP ($\mu g \cdot L^{-1}$)	2.3 ± 1.2	3 ± 2.5	<2**	<2**

* 1 value greater than MDL

** at least half of values <MDL

4.3.3 Phytoplankton

Phytoplankton biomass, measured as chlorophyll-a concentration, and rates of primary production were lower in Seton Lake than in Anderson Lake and lower in 2014 in both lakes than during the transition period in 2000-2003 (Figures 25 and 26). In 2014, over the May to September growth period, the average depth-integrated phytoplankton biomass was 11.6 $mg \cdot m^{-2}$ in Seton Lake and 18.7 $mg \cdot m^{-2}$ in Anderson Lake. These values were two thirds lower than the mean depth integrated phytoplankton biomass in each lake in the transition period. In Seton Lake, the rate of primary production in 2014 (63 mg $C \cdot m^{-2} \cdot day^{-1}$) was 20% of average primary production in the transition period (318 mg $C \cdot m^{-2} \cdot day^{-1}$). In Anderson Lake, primary production was 45% lower in 2014 (144 mg $C \cdot m^{-2} \cdot day^{-1}$) compared to the earlier years (321 mg $C \cdot m^{-2} \cdot day^{-1}$).

These temporal comparisons are only descriptive at this point in the study because more samples in the after period to be collected in 2015 and 2016 are required before statistics can be run to test the before after control impact comparisons.

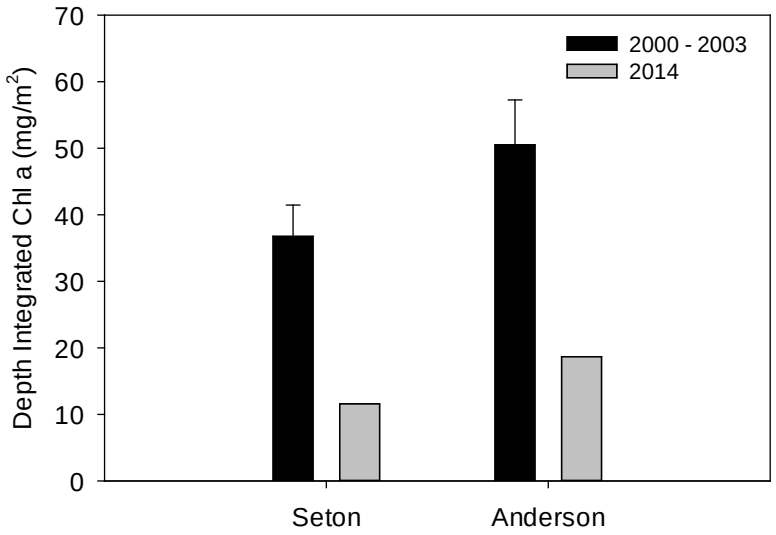


Figure 25 Mean sampling period (May to Sept) depth-integrated areal phytoplankton chlorophyll-a concentrations in Seton Lake and Anderson Lake during the transition period (2000 – 2003, n=4) and the after period (2014, n=1).

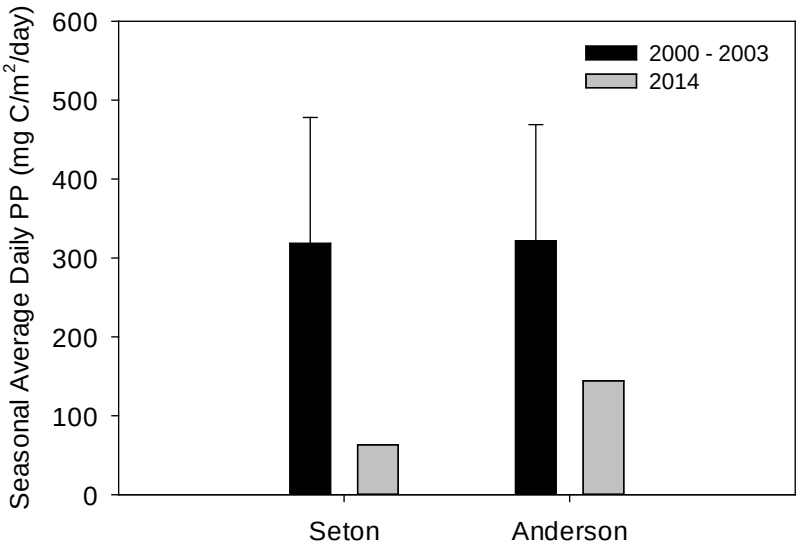


Figure 26 Seasonal average daily primary productivity (PP) in Seton Lake and Anderson Lake during the transition (2000 to 2003, n=4) and after (2014, n=1) periods.

Seasonal average daily primary production in Seton Lake and Anderson Lake in 2014 was in the middle of the range of values found among other British Columbia lakes and reservoirs (Table 8). In contrast the rates measured by DFO in 2000-2003 were the highest among the same comparisons. Those DFO values were similar to those found among fertilized lakes in the Province. Further measurements in 2015 and 2016 will be needed to determine if an actual change has occurred or whether the lower rates in 2014 were anomalous. The Seton Lake 2014 rate of primary production (63 mg C m^{-2} d^{-1}) was higher than in other reservoirs influenced by glacial turbidity including Kinbasket (55 mg C m^{-2} d^{-1}) and Revelstoke (38 mg C m^{-2} d^{-1}). If the 2000-2003 Seton data are included in this comparison, the rate is many times greater than those in other glacially influenced reservoirs. Reasons for the differences are unknown but may involve more efficient cycling of nitrogen and phosphorus in Seton Lake than in the other lakes or that nutrient loading from the glacial sources is more effectively sequestered by phytoplankton in Seton Lake than in the other lakes.

Table 8. Comparison of phytoplankton biomass and primary production among lakes and reservoirs in British Columbia.

Lake or Reservoir	Primary production (mg C m^{-2} d^{-1})	Fertilized or not	Reference
Seton Lake 2014	63	No	This report
Seton Lake mean from 2000-2003	318	No	This report
Anderson Lake 2014	144	No	This report
Anderson Lake mean from 2000-2003	322	No	This report
Kinbasket 2013	55	No	Unpublished data from MOE
Elsie Lake Reservoir	13.9	No	Perrin and Harris (2006)
Williston Reservoir	33.5	No	Harris et al. (2005)
Okanagan Lake	72.2	No	Andrusak et al. (2004)
Slocan Lake	59.3	No	Harris (2002)
Stave Reservoir	28.5	No	Stockner and Beer (2004)
Alouette Lake	140	Yes	Reddekopp et al. (2006)
Kootenay Lake 2003	303	Yes	Harris (2004)
Kootenay Lake 2013	259	Yes	Unpublished data from MOE
Kootenay Lake 2014	179	Yes	Unpublished data from MOE
Revelstoke Reservoir 2013	38	No	Unpublished data from MOE
Arrow Lake Reservoir	262	Yes	Pieters et al. (2001)

Values of primary production at a given station were compared between two procedures: one using filtration through 0.2 µm polycarbonate filters and the other following filtration through 0.8 µm glass fibre filters (Table 9). Filtration at 0.2 µm is standard among most studies but DFO used 0.8 µm filters for the 2000-2003 measurements. The same procedures were required in 2014 to support later statistical

tests in the BACI layout. Results showed that rates of primary production were higher following filtration with the 0.8 µm glass fibre filters than with the 0.2 µm polycarbonate filters. This finding is counterintuitive if the pore sizes were exact. Glass fibre filters do not have an exact pore size but rather a nominal pore size. The data show that particles smaller than the nominal pore size will be retained during filtration and potentially retain more biomass than is achieved on a filter having smaller pore size at more exact specifications. More data will be required to determine if the differences showing in Table 9 are statistically significant. Until that test is run, all filtrations in 2015 and 2016 will include both the 0.2 µm polycarbonate filters and the 0.8 µm glass fibre filters to ensure data are available for comparison of rates of primary production to other lakes and particularly to the DFO data from Seton and Anderson Lakes that was collected in 2000-2003.

Table 9. Comparison of rates of primary production in Seton and Anderson lakes between methods using filters having pore sizes of 0.2 µm and 0.8 µm.

Lake	**Seasonal mean primary production (mg C m^{-2} d^{-1})**	
	Using 0.2 µm polycarbonate filters	**Using 0.8 µm glass fibre filters**
Seton	53	63
Anderson	118	144

Phytoplankton in Seton and Anderson Lakes included diatoms (Bacillariophyceae), green algae (Chlorophyceae), flagellates (Chrysophyceae and Cryptophyceae), blue green algae (Cyanobacteria), and dinoflagellates (Dinophyceae) (Figure 27). Low biovolumes of Euglenoids (Euglenophyceae) were present in Seton Lake but not in Anderson Lake, while three other algae (Prymnesiophyceae, Eustigmatophyceae and Bicosecophyceae) were also present in both lakes. The average phytoplankton biovolume was 292 $mm^3 \cdot mL^{-1}$ in Seton Lake and 323 $mm^3 \cdot mL^{-1}$ in Anderson Lake over the growing season in 2014. The single largest phytoplankton assemblage in both lakes was the flagellated chryso-cryptophytes. Fourteen species of flagellates were present in both lakes, accounting for 53% of the average biovolume in Seton Lake and 38% of the average biovolume in Anderson Lake. Green algae (Chlorophyceae) were the second largest division in both lakes, with 13 species accounting for 32% of total biovolume in Anderson Lake, and 10 species accounting for 15% in Seton Lake. In Anderson Lake, six species of blue green algae accounted for 12% of total biovolume, while five species of diatoms accounted for 8% of total phytoplankton biovolume. In Seton Lake, four species of diatoms accounted for 10% of biovolume while three species of blue green algae accounted for 3% of phytoplankton biovolume. One species of each yellow green algae (Xanthophyta), Haptophyta and Bicosecophyceae (all shown as other algae in Figure 27) accounted for 10% and 6% of

total phytoplankton biovolume in Seton Lake and Anderson Lake respectively. There were four species of dinoflagellates in each lake accounting for 8% and 4% of phytoplankton biovolume in Seton and Anderson respectively.

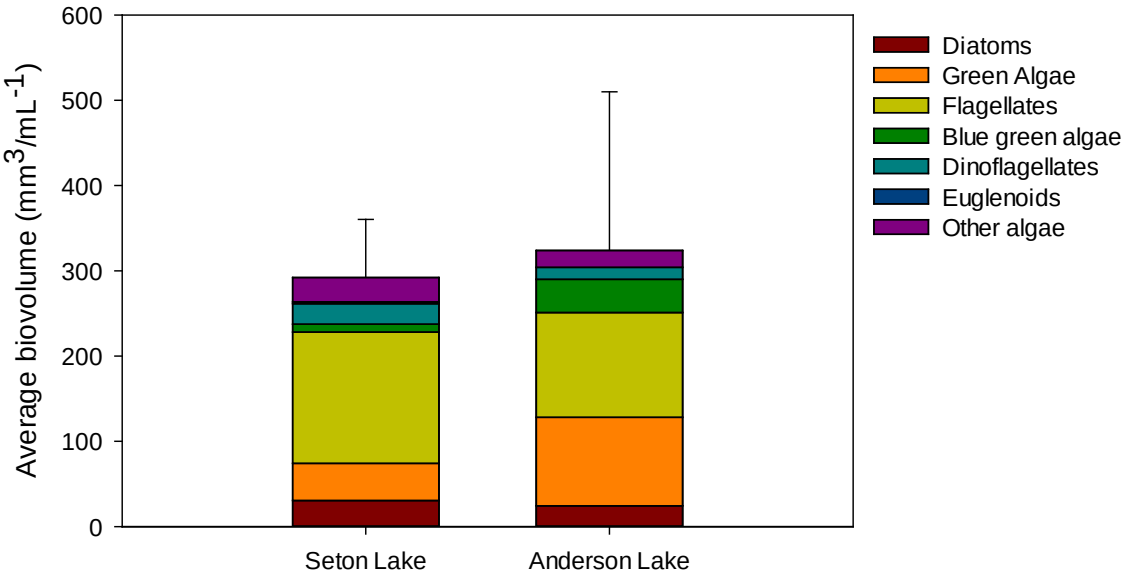


Figure 27 Mean biovolume of phytoplankton by division over the growing season in 2014 (± standard deviation) (n=10, 2 stations for each lake sampled monthly from May to September).

In lakes and reservoirs in which the supply of both N and P is low, blue green algae do not have a competitive advantage. Rather, it is the very small sized flagellates of the Chrysophytes and Cryptophytes that are favoured because they can outcompete the larger sized taxa for the available nutrients (Suttle and Harrison 1988, Suttle et al. 1991). This was the case in Seton and Anderson Lakes where the microflagellates dominated. Under these conditions, any slight addition of phosphorus can produce limitation of algal growth by nitrogen and vice versa, any slight addition of N can produce limitation by P. Under small nutrient fluxes, the phytoplankton communities would be expected to constantly respond to changing N and P deficiency, depending on processes that determine the delivery of nutrients to the euphotic zone. In Seton Lake we expect a seasonal return of nutrients to the water column during winter mixing and fluxes in availability of phosphorus in particular from the glacially turbid diversion inflows. These processes may be important in supporting phytoplankton production even if the various forms of N and P are not detectable. In systems like these two lakes, if nutrients are detectable using routine wet chemistry, it most likely means the nutrients are in excess of requirements by phytoplankton which rarely happens in lakes where there is high demand for N and P by phytoplankton. The prevalence by microflaggelates

supports the nutrient chemistry data in showing high demand for phosphorus and nitrogen in each of Seton and Anderson Lakes.

Trophic state is a sliding scale related to growth of biota or degree of carbon fixed by plant growth. In most lakes and reservoirs, including Seton and Anderson Lakes, the two critical nutrients that can limit this process are nitrogen and phosphorus. Oligotrophic and ultraoligotrophic lakes and reservoirs are those in which the supply of N and P is low enough to severely limit the growth of phytoplankton, which results in relatively low biomass measured as chlorophyll-a. At the other end of the scale, eutrophic lakes are those receiving relatively high loads of N and P that produce high biomass of algae in the ranges shown in Table 10. Mesotrophic lakes are those having a nutrient load and algal biomass intermediate between oligotrophic and eutrophic states. Of the two nutrients, phosphorus is primarily important because it can theoretically generate 500 times its own weight in algae while nitrogen can only produce 71 times its own weight in algae, meaning that algae are much more reactive to change in P supply than to change in N supply when growth is limited by either nutrient.

Wetzel (2001) produced a useful table allowing one to classify a lake or reservoir according to ranges of N and P concentrations, primary production, algal biomass, and Secchi depth. Secchi depth is less useful for trophic classification in reservoirs or lakes that receive glacial turbidity because it is influenced by non-biological particles. Wetzel surmised Secchi depth as being a useful criterion but only when it was affected by plankton, not suspended inorganic fines. Table 10 shows the Wetzel criteria for trophic state along with information for Seton and Anderson Lakes. Using these criteria, Seton Lake is classified as oligotrophic in 3 of 4 criteria (excluding Secchi depth) and potentially mesotrophic based on rate of primary production. Anderson Lake is the same. Hence, both lakes have the same trophic state that can be stated as meso-oligotrophic for purposes of comparison in other parts of this study.

Table 10. Assignment of trophic state in Seton Lake and Anderson Lake based on criteria defined by Wetzel (2001).

Parameter		**Trophic classification by Wetzel (2001)****				**Seton Lake***	**Status of Seton Lake**	**Anderson Lake***	**Status of Anderson Lake**
		ultraoligotrophic	**oligotrophic**	**mesotrophic**	**eutrophic**				
TP (μg/L)	mean		8.0	27	84	3	oligotrophic	3	oligotrophic
	range	<1 – 5	3 – 18	11 - 96	16 – 386	<2 – 9.5		<2 - 21	
TN (μg/L)	mean		661	753	1875	64	ultraoligotrophic	68	ultraoligotrophic
	range	<1 – 250	307 - 1630	361 - 1387	393 - 6100	<5 – 106		<5 - 109	
Chl-*a* (μg/L)	mean		1.7	4.7	14.3	0.8	oligotrophic	0.7	oligotrophic
	range	0.01 – 0.5	0.3 – 4.5	3 - 11	3 - 78	0.1 – 1.9		0.06 – 2.2	
Secchi depth (m)	mean		9.9	4.2	2.5	4.1	Not relevant***	12.5	Not comparable to Seton Lake***
	range		5.4 – 28.3	1.5 – 8.1	0.8 – 7.0				
Net primary production (mg C m^{-2} d^{-1})	mean					63 in 2014 318 in 2000-2003	Oligotrophic to mesotrophic	144 in 2014 322 in 2000-2003	mesotrophic
	range	<50	50 – 300	250 - 1000	>1000				

*based on sampling in May through September (nutrient concentrations are means of epilimnetic and hypolimnetic samples)

**based on annual means

***Secchi depth as a trophic indicator is not relevant in lakes like Seton Lake that are affected by glacial turbidity.

4.3.4 Zooplankton

Nine species of zooplankton were found in Seton Lake and seven species in Anderson Lake in 2014. Cladocerans common in both lakes included *Eubosmina longispina* and *Daphnia ambigua*. *Chydorous sphaericus*, *Leptodora kindtii*, and *Daphnia pulex* were also present in Seton Lake. Two Cyclopoida were present in both lakes: *Cyclops scutifer* and *Cyclops sp*. and two calanoid copepods were present including *Epischura nevadensis* and *Acanthodiaptomus denticornus*. Peak zooplankton biomass was 5,422 mg dry wt·m^{-2} on 15 July in Seton and 12,403 mg dry wt·m^{-2} on 16 July in Anderson. Cladoceran biomass accounted for 73% to 94% of total biomass in Anderson Lake (Figure 28). In Seton Lake, cladoceran biomass accounted for 49% to 83% of total biomass, with the low biomass occurring in June (58%) and September (49%) Biomass of calanoid copepods was ≤ 1% in all months, with the lowest levels in May in both lakes.

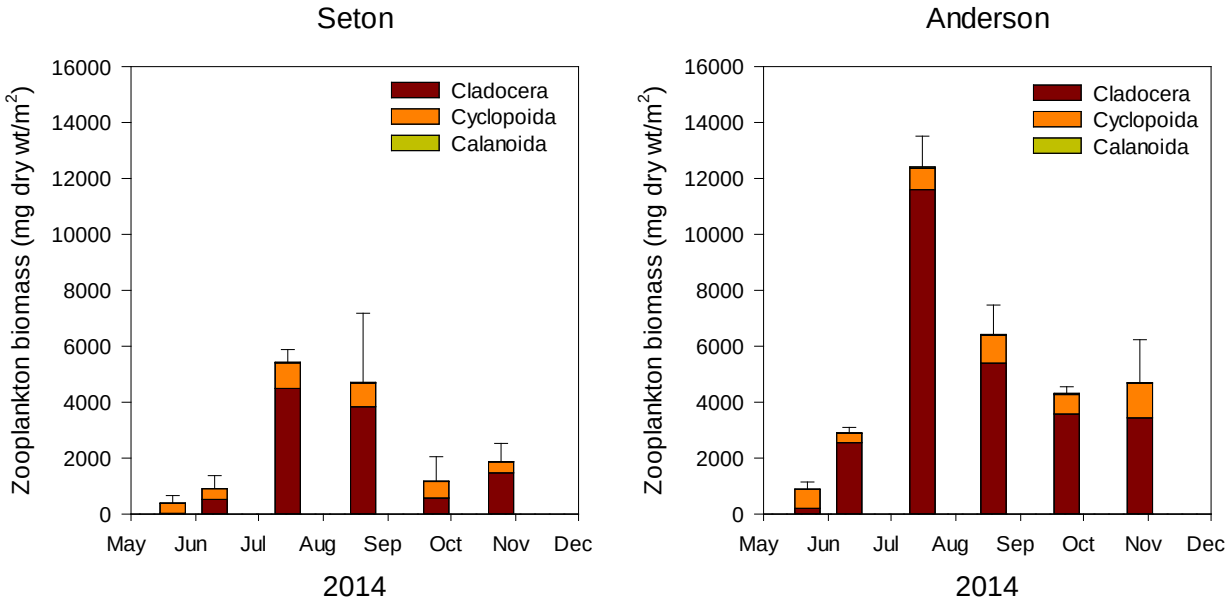


Figure 28. . Zooplankton dry weight biomass in Seton Lake (left) and Anderson Lake (right) in 2014. Data are shown for all three orders of zooplankton (Cladocera (suborder of Diplostraca), Cyclopoida and Calanoida). Data are shown as a mean and standard deviation from samples collected at each of two stations on each date, except for the final sampling data, when two replicate samples were collected from each of the two stations.

Mean annual zooplankton production in 2014 in Anderson Lake (36.3 g dry wt·m^{-2}·yr^{-1}) was more than double that in Seton Lake (16.3 g dry wt·m^{-2}·yr^{-1}). Cladocerans accounted for 76% of total production in Seton Lake and 83% of total production in Anderson Lake, with cyclopoids being the next most important. Zooplankton production in Seton Lake in May to October, 2014 was double that found in the transition years of

2000 to 2003 (Figure 29). In Anderson Lake, zooplankton production was 30% greater in 2014 compared to the transition period of 2000 to 2003 (Figure 29). As with primary production, these spatial and temporal comparisons are only descriptive for now. Further data from 2015 and 2016 will be needed before quantitative comparisons can be made to satisfy the BACI layout.

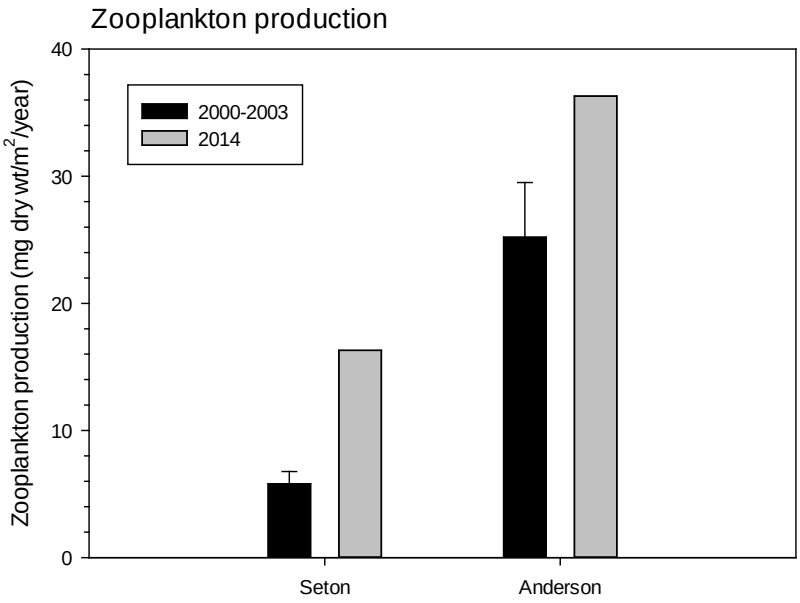


Figure 29. Annual zooplankton production in Seton Lake and Anderson Lake in the transition years 2000 to 2003 (mean and SD shown, n=4) and after year, 2014.

Rates of zooplankton production in Seton and Anderson Lakes covered a range found among other meso-oligotrophic lakes (Table 11). Zooplankton production in Seton Lake was similar to that in a couple of studies of Lake Ontario and it was at the high end of that found in oligotrophic Lake Pend Oreille in Idaho. Zooplankton production in Anderson Lake was at the high end of various measures in Lake Ontario.

Table 11. Annual rates of zooplankton production compared among lakes.

Lake or reservoir	Annual zooplankton production (g dry wt·m^{-2}·yr^{-1})	Trophic state	Reference
Seton Lake	5.8 - 16.3 range among years	meso-oligotrophic	This study
Anderson Lake	25.2 - 36.3 range among years	meso-oligotrophic	This study
Lake Ontario	15	meso-oligotrophic*	Borgmann et al. (1984)
Lake Pend Oreille	9.7 – 13.9	oligotrophic	Clarke and Bennett (2007)
Lake Ontario	15 – 33 depending on method of calculation	meso-oligotrophic*	Stockwell and Johannsson (1997)

* http://www.epa.gov/greatlakes/glindicators/water/trophicb.html

4.4 Question 3: To what extent does aquatic productivity alone limit the abundance and diversity of fish populations in Seton Lake?

4.4.1 Overview

We successfully completed summer (July 29-August 4) and fall (October 23-28) surveys on Seton and Anderson lakes in 2014 as scheduled. Hydroacoustic data for making population estimates and to examine diel vertical migration (DVM) patterns was collected. Trawl and gill net samples were taken to determine fish species composition, size, growth, age from scales or otoliths, and stock origin from DNA. Temperature, turbidity, and light profiles were collected. Laboratory processing and analysis of these samples and data sets are ongoing and preliminary results of analyses conducted to date are described below.

4.4.2 Limnetic species captured in trawls and gill nets

As is typical for Fraser system sockeye rearing lakes there were relatively few species captured in the pelagic zone of Seton and Anderson Lakes (Table 12). Various ages of *O. nerka* numerically dominated both the trawl and gill net catches in both lakes, comprising between 75 to 100% of the catch. Only three other species, all potential predators, were captured in the pelagic zone. Of these, Bull trout were the most common, followed by Northern pikeminnow and a single Rainbow trout, captured in Anderson Lake.

Table 12. Fish species captured in the summer and fall surveys.

Lake	Gear	Season	Species			
			O. nerka	Bull trout	Northern pikeminnow	Rainbow trout
Anderson	Trawl	Summer	676	13	1	
		Fall	664			
	Gill net	Summer	74	9	6	1
		Fall	52	9	2	
Seton	Trawl	Summer	358	2		
		Fall	482			
	Gill net	Summer	78	9		
		Fall	18	4	2	

4.4.3 Fish aging

The late spawning dates of Gwenish in Anderson Lake, the subsequent late fry emergence dates, and their relatively small egg size has probably created difficulties in aging these Gwenish using scales. Studies have found that the minimum fork length for scale formation by *O. nerka* was between 36 - 40 mm (Gilbert 1913, Foerster 1929). Using this criteria, many age-0 Gwenish in Anderson Lake would not have formed scales by the time of the summer survey because half of the trawl catch (54%) was <38 mm (Figure 30). There were still many fish <38 mm at the time of the fall survey (16%) and a large portion of this size group would have formed few if any circuli before the end of their first growing season. Thus the first annulus would not appear until the end of their second growing season. It seems likely, therefore, that a significant proportion of the Gwenish could be one year older than indicated by their scale age and incorrectly aged as age-0 after surviving their first winter. We suspect that these are the fish in the summer survey, between 53-63 mm, that were classified by DNA as Gwenish and were aged as age-0 (Figure 30). While we suspect they are actually age-1, further evaluation, such as a cross comparison of ages from scales and otoliths of selected fish, would be needed to be certain.

Under-aging of Gwenish in Seton Lake is likely to occur less often, as Gwenish spawning occurs somewhat earlier in Seton Lake, probably resulting in an earlier emergence date and larger fry by the time of the surveys. Seton Lake age-0 Gwenish are larger than those in Anderson Lake and only 10% of the summer trawl catch were <38mm. By the time of the fall survey only 1% were <38 mm and most fry would have formed scales and laid down at least a few circuli before the end of the growing season, allowing an annulus to be formed. The distribution of age-0 fish tends to support this supposition as only 1 fish scale aged as age-0 from the summer survey in Seton Lake appears to be disproportionately large (78 mm). Fish in this size group were classified by DNA as Gwenish.

Older Gwenish were also captured in Seton and Anderson lakes, mostly in gillnets. As gillnets are a passive fishing method and trawling is active, comparisons of relative and absolute abundance can only be made within gear types and not between gear types. Age-0 to age-4 were identified in Anderson Lake while Seton only had age-0 to age-3 Gwenish (Figures 30 and 31). This is not what would be expected if the aging error hypothesis was true but may possibly be explained by an earlier age of maturity in Seton Lake.

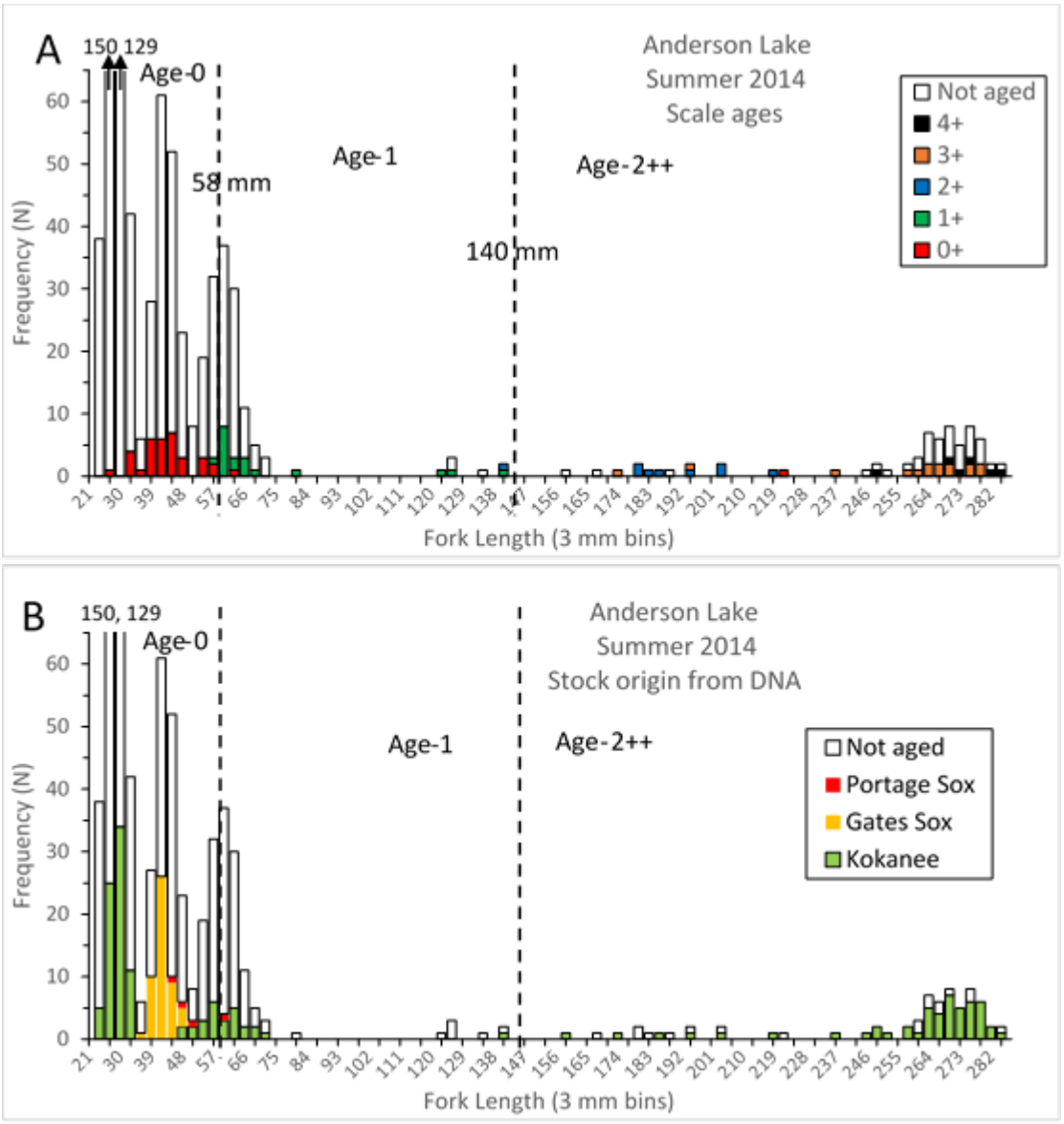


Figure 30. A) Results of scale aging of selected O. nerka (solid shading), from the summer survey, captured in midwater trawls and gillnets in Anderson Lake. Size ranges for each age class are shown (vertical dashed lines). There was considerable length overlap between age-2-4 fish and they were all categorized as age-2++. B) Results of the DNA stock determination of selected O. nerka (solid shading), from the summer survey, captured in midwater trawls and gillnets in Anderson Lake.

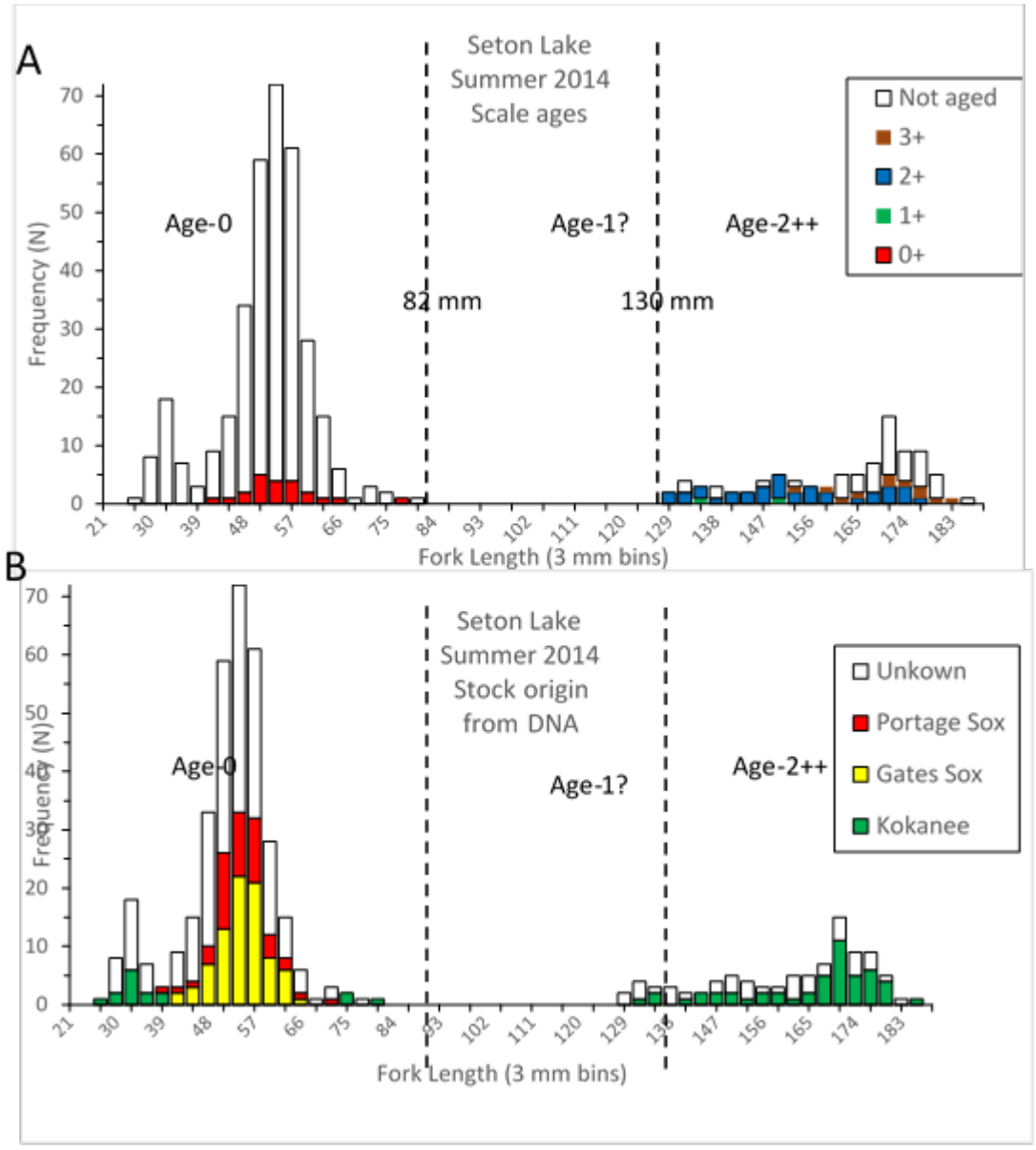


Figure 31. A) Results of scale aging of selected O. nerka (solid shading), from the summer survey, captured in midwater trawls and gillnets in Seton Lake. Size ranges for each age class are shown (vertical dashed lines). There was considerable length overlap between age-2-3 fish and they were all categorized as age-2++. B) Results of the DNA stock determination of selected O. nerka (solid shading), from the summer survey, captured in midwater trawls and gillnets in Seton Lake.

4.4.4 *O. nerka* stock origin

The Bayesian analysis of STRUCTURE, in which genotypes were clustered into one to five subpopulations (K = 1-5) indicated that both three and four subpopulations described the eight groups of Gwenish and sockeye sampled from the Seton – Anderson watershed approximately equally well. The three-subpopulation model had a likelihood value of -48934.9 and average variance among samples of 620.4, whereas the 4-subpopulation model had a likelihood value of -48842.4 and a higher average variance among samples (927). The baseline sockeye samples from Gates and Portage sockeye salmon were very distinctive and formed types (subpopulations) one and two in both models. The putative Gwenish samples from Anderson and Seton Lakes comprised the third cluster in the 3-subpopulation model and clusters 3 and 4, respectively in the four sub-population model. In the remainder of this report we use the 3-subpopulation model because it produced individual classifications with the lowest mean variance.

The putative Gwenish sampled from both lakes in the summer and fall were all identified as Gwenish, as predicted (Table 13). During both summer and fall, the remaining juvenile fish sampled from Anderson Lake were primarily a mixture of Gwenish and Gates Creek sockeye salmon, with a higher proportion of Gwenish in the fall than summer mixture. Gates Creek sockeye salmon were the most common fish identified in both the summer and fall mixture samples from Seton Lake. They comprised slightly more than half the juvenile mixture sample in summer and just less than half in the fall (Table 13). The percentage of Portage Creek sockeye in Seton Lake also decreased from summer (33.3) to fall (24.0), whereas the percentage of Gwenish increased from (10.7) to (27.5).

Table 13. Classification of juvenile *Oncorhynchus nerka* samples from Anderson and Seton Lakes in the summer and fall of 2014.

Sample	Sample Size	Gwenish		Gates Sockeye		Portage Sockeye	
		N	%	N	%	N	%
Anderson Lake - summer							
Putative Gwenish	51	51	100	0	0	0	0
Mixture	155	101	65.2	49	32.3	4	2.5
Anderson Lake - fall							
Putative Gwenish	33	33	100	0	0	0	0
Mixture	187	156	83.4	30	16.0	1	0.5
Seton Lake – summer							
Putative Gwenish	50	50	100	0	0	0	0
Mixture	150	15	10.7	83	56.0	50	33.3
Seton Lake – fall							
Putative Gwenish	23	23	100	0	0	0	0
Mixture	167	46	27.5	81	48.5	40	24.0

One of the analytical results was a probability for each individual fish that it belonged to one of the three stocks (total P=100%). Most juvenile fish (195/204) collected from the summer survey were identified to a specific stock with a >80% probability and no identified fish had less than a 56% probability of belonging to that stock. All of the *O. nerka* greater than 75mm were identified as Gwenish (Figures 30 and 31). Three fish with <50% probability of belonging to any of the three stocks were left unclassified.

4.4.5 *O. nerka* fry migration

DNA analysis of our summer trawl captures (July 29 – August 3, 2014) confirmed the migration of sockeye fry from Anderson Lake into Seton Lake. Gates Creek sockeye comprised 56% of the age-0 *O. nerka* captured in Seton Lake while Portage Creek comprised only 33% (Table 13). Gwenish comprised 11% of the age-0 population in Seton Lake.

Gwenish dominated (65%) the juvenile population in Anderson Lake. While most of the 53 fish identified as sockeye were from Gates Creek, unexpectedly, four (2.5%) were from the Portage Creek stock (Table 13). The gradient and flow velocity of the Portage Creek spawning grounds and their proximity to Anderson Lake could possibly allow some upstream migration of emerging fry but this has not been documented previously. Alternatively, these fry may be the offspring of strays from Portage Creek that have spawned in Anderson Lake or its tributaries. DFO survey crews found late run sockeye spawning on the shores of Anderson Lake in 2010 (25 fish) and 2014 (250 fish) but none were documented in 2013 (Keri Benner, Fisheries and Oceans Canada, Kamloops BC, Personal communications).

By themselves the trawl catches only allow us to determine the proportions of the multiple *O. nerka* stocks and age classes in the lakes. When combined with the acoustic population estimates they will enable us to determine the abundance of each stock and the proportion of the Gates Creek fry that migrated into Seton Lake.

4.4.6 *O. nerka* size

Overall, the summer survey found that both age-0 Gwenish and sockeye were larger in Seton Lake than in Anderson Lake (Table 14). Age-0 Gwenish in Seton Lake averaged 0.4 g (34 mm) but only 0.1 g (29 mm) in Anderson Lake. Age-0 Gates sockeye in Seton Lake averaged 1.3 g (54 mm) but only 0.7g (42 mm) in Anderson Lake. Within Seton and Anderson lakes, age-0 Gwenish were smaller than the sockeye and within Seton Lake there was no significant difference in mean size between Gates and Portage origin age-0 sockeye. There was an insufficient sample size in Seton Lake for

comparisons of the age-1 Gwenish between lakes. Contrary to the age-0 fish, both the age-2 and age-3 Gwenish were larger in Anderson Lake than in Seton Lake. There were no age-4 Gwenish captured in Seton Lake to enable a comparison of Gwenish sizes of that age between lakes.

Table 14. Length and weight of *O. nerka* captured during the summer survey in Seton and Anderson Lakes.

Lake	Age	Fish species	Length (mm)				Weight (g)			
			N	Mean	SE	±95%CI	N	Mean	SE	±95%CI
Anderson Lake	0	Gwenish	75	28.9	0.27	4.72	22	0.1	0.01	0.07
		Gates Sockeye	49	42.5	0.34	4.81	49	0.7	0.03	0.41
		Portage Sockeye	4	51.0	2.97	18.92	4	1.1	0.06	0.37
	1	Gwenish	93	67.0	1.8	33.6	93	3.1	0.5	9.30
	2	Gwenish	9	188.1	7.3	50.6	9	72.6	8.8	60.60
	3	Gwenish	15	255.9	7.98	66.32	15	190.20	16.72	138.87
	4	Gwenish	6	272.7	5.14	32.34	6	231.87	8.57	53.94
Seton Lake	0	Gwenish	13	33.6	1.0	7.8	13	0.4	0.1	0.91
		Gates Sockeye	83	54.5	0.5	9.9	83	1.3	0.0	0.77
		Portage Sockeye	49	54.4	0.8	11.6	49	1.4	0.1	0.91
	1	Gwenish	2	143.5	7.5	134.8	2	28.2	3.5	62.9
	2	Gwenish	35	152.8	2.4	28.9	35	36.8	1.8	22.1
	3	Gwenish	11	170.5	2.8	20.9	11	49.8	2.7	19.8

4.4.7 Diel vertical migration

4.4.7.1 Summer sampling

Echograms from the diel vertical migration (DVM) study showed a strong DVM of fish at the single station sampled in each lake during the summer survey (Seton SVM8, Anderson AVM5, see Figure 4 for station locations). At night, individual fish tracks were found in midwater layers in both lakes (depth range 15-25 m in Seton Lake, 20-65 m in Anderson Lake, Figure 32). During the day, individual tracks occurred in deep layers in both lakes (70-90 m in Seton Lake, 95-130 m in Anderson Lake), and small schools occurred higher in the water column (< 55 m in Seton Lake, 65-95 m in Anderson Lake). Anderson Lake also had a shallow, sparse layer of fish at 15-25 m during the day. This layer is not apparent in Figure 32 except for a couple of small targets because the image only shows 3 minutes of a 2 hour data set for ease of presentation. Alternative ways of

showing these data will be developed for the final report. Although a DVM occurred in both lakes, it was more pronounced in Anderson Lake where during the day both schools and the deep water layer of individual fish occurred deeper in the water column than in Seton Lake. Thus, as expected, the DVM was greater in the clearer water of Anderson Lake (Secchi depths at DVM stations: Seton Lake 7 m, Anderson Lake 15 m, Table 15).

Fish of a wide range of sizes were detected in both lakes during summer DVM sampling. Total ranges of TS values were -64.6 to -28.2 dB in Seton Lake (equivalent to fish lengths of 9-756 mm per Love's dorsal model (Love 1977)), and -64.9 to -28.5 dB in Anderson Lake (fish lengths 9-729 mm). These estimated fish lengths roughly corresponded to the minimum and maximum sizes of fish in the combined trawl and gill net catches from the lakes during summer sampling (Seton Lake 27 to 674 mm, Anderson Lake 23 to 600 mm).

Plots of TS versus depth in the water column for fish detected during summer DVM sampling showed that fish of a variety of sizes participated in the vertical migration. In both lakes, fish below 70 m during the day (schools and deep layers) were mainly small fish (TS < -45 dB), and the smallest fish (TS - 55 to -65 dB) were concentrated 70-90 m deep in Seton Lake and 80-130 m deep in Anderson Lake (Figure 33). The Anderson data in Figure 33 were truncated at a depth of 100 m for ease of presentation but the raw data show the concentration of smallest fish extending to 130m. During the day, bigger fish with TS > -45 dB were mainly < 60 m deep in Seton Lake and < 90 m in Anderson Lake. At night, small fish were concentrated in midwater layers (depth range 15-25 m in Seton Lake, 20-50 m in Anderson Lake), with bigger fish at much the same depth range as the small ones in Anderson Lake and slightly deeper (range 20-30 m) in Seton Lake.

4.4.7.2 Fall sampling

Echograms from the fall DVM study showed a variety of vertical migration patterns over the range of water clarity represented by the stations sampled at the time, two in Seton Lake and one in Anderson Lake (Seton west = SVM2 Secchi depth = 1 m; Seton east = SVM8 Secchi depth = 2 m, Anderson AVM5 Secchi depth = 14 m; Table 15; see Figure 4 for station locations). At night, individual fish tracks were found in loose midwater layers at all stations (approximate depth range 25-55 m at SVM2, 10-50 m at SVM8, and 25-65 m at AVM5, Figure 34). During the day at SVM2, individual fish tracks were most numerous in a sparse midwater layer (depth 30-40 m), with a few fish deeper and shallower. During the day at SVM8, fish were mainly in small schools at a depth of 10-50 m, or in a layer of tracks 70-90 m. At AVM5, fish were mainly in two layers of small schools during the day (10-30 m and 70-130 m). These patterns generally showed that the DVM became more pronounced with increasing water clarity.

As in summer, fish of a wide range of sizes were detected in both lakes during fall DVM sampling. The total range of TS was almost identical in both lakes during the fall, -64.7 to -26.8 dB in Seton Lake (fish length 9-895 mm per Love's dorsal model), and -64.9 to -26.8 dB in Anderson Lake (fish length 9-895 mm). These estimated fish lengths roughly corresponded to the minimum and maximum sizes of fish in the combined trawl and gill net catches from the lakes during fall sampling (Seton Lake 36 to 750 mm, Anderson Lake 31 to 650 mm).

Plots of TS versus depth of fish in the water column from fall DVM sampling showed that at station SVM2 (Seton west) the densest concentrations of small fish (< -45 dB) were found in the 25-40 m depth range, both day and night, with consolidation into a dense layer between 25 and 30 m at night (Figure 35). At this station bigger fish occupied similar ranges day and night (30-50 m day and 30-60 m night). At SVM8 (Seton east), small fish were densest 15-30 m at night, whereas during the day they were concentrated in the 70-90 m range (a deep water layer of individuals), with lower densities 10-50 m (mainly small schools). At this station bigger fish were mainly 25-40 m at night and 10-40 m during the day. At station AVM5 (Anderson Lake), small fish formed a dense layer 15-50 m at night, and mainly occurred in 10-30 m and 80-130 m ranges (both mainly schools) during the day. Bigger fish at this station were concentrated between 35 and 55 m at night and scattered over the water column during the day. These patterns generally showed that for both small and large fish the DVM became more pronounced with increasing water clarity.

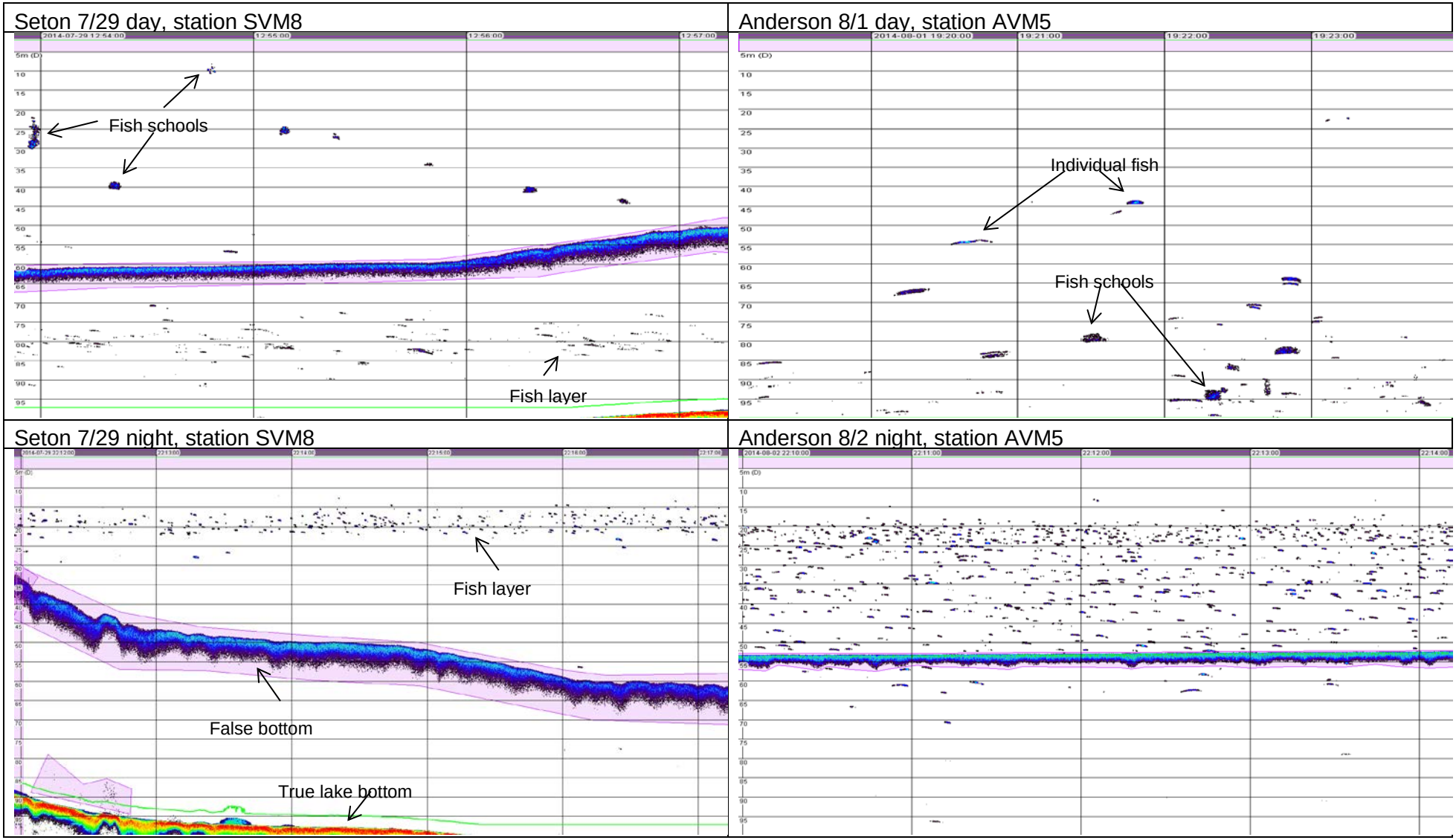
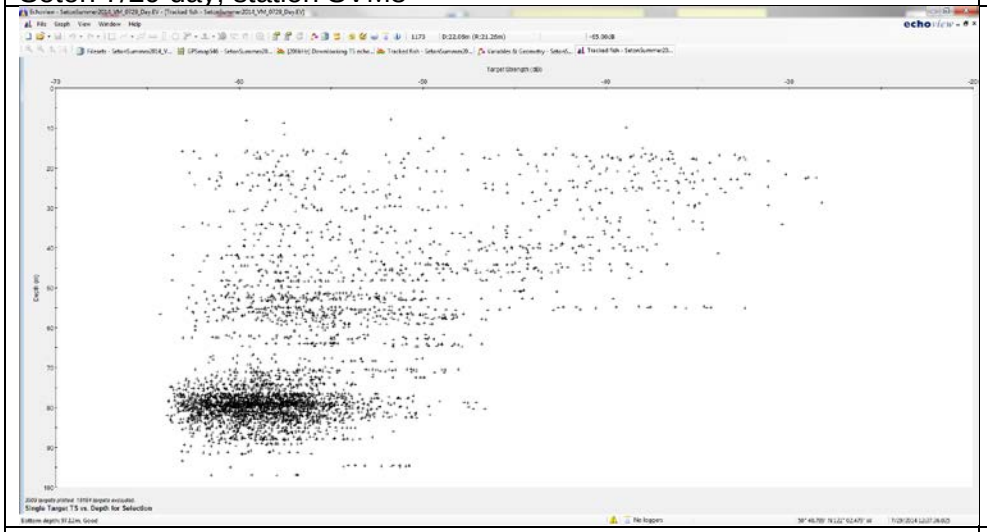
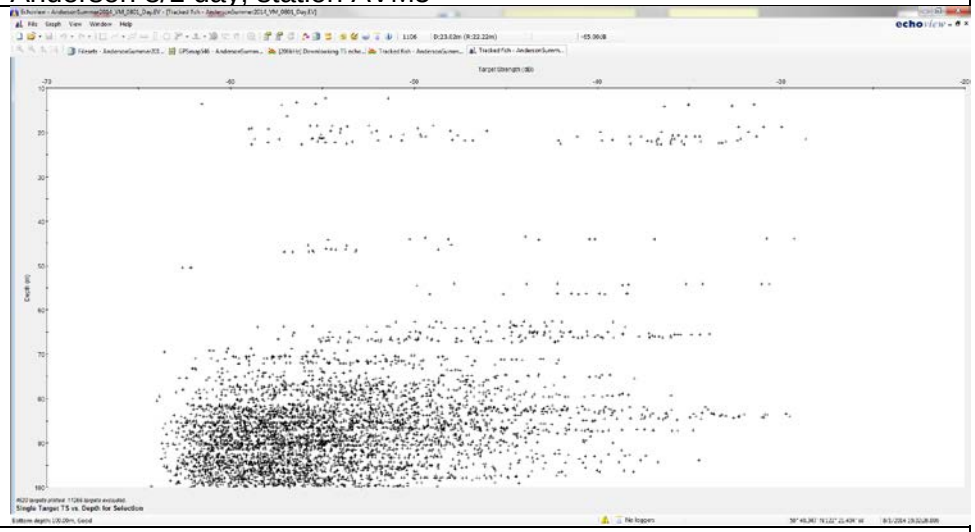


Figure 32. Three to four minute segments of up to 2 hour day and night time echograms from summer DVM sampling of Seton and Anderson Lakes (July 29 – August 2, 2014). Echogram settings (threshold = -65 dB, range compensation = 40 log R) were sensitive enough to show sockeye fry to the maximum depth displayed. Gridding is the same on all plots: Vertical, 0-100 m in 5 m intervals; Horizontal, 1 minute intervals. Purple areas are analysis exclusion zones (surface zone, false bottoms, bottom noise) for non-fish echoes; the green line above true bottom is the maximum analysis depth line. Images for Anderson Lake were truncated at a depth of 100 m for presentation with Seton Lake but fish were found deeper as discussed in the text.

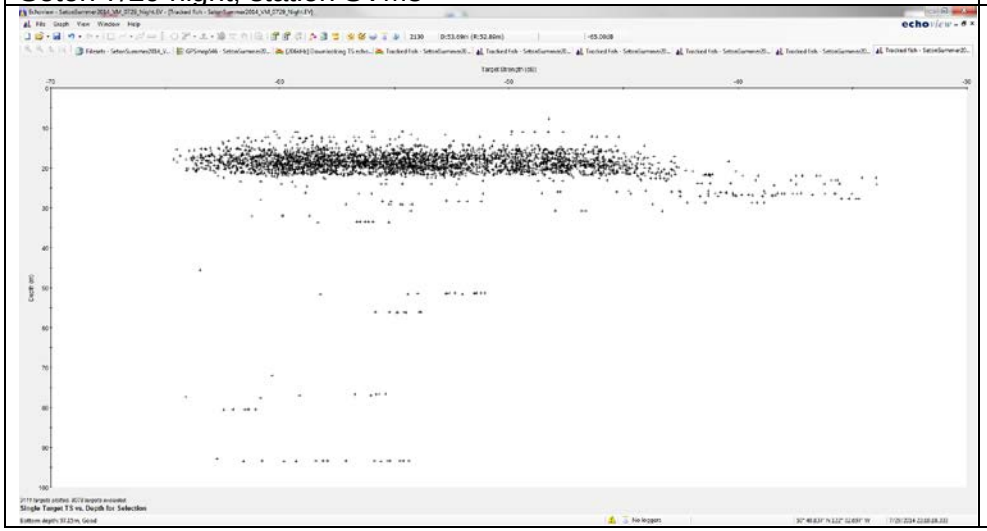
Seton 7/29 day, station SVM8



Anderson 8/1 day, station AVM5



Seton 7/29 night, station SVM8



Anderson 8/2 night, station AVM5

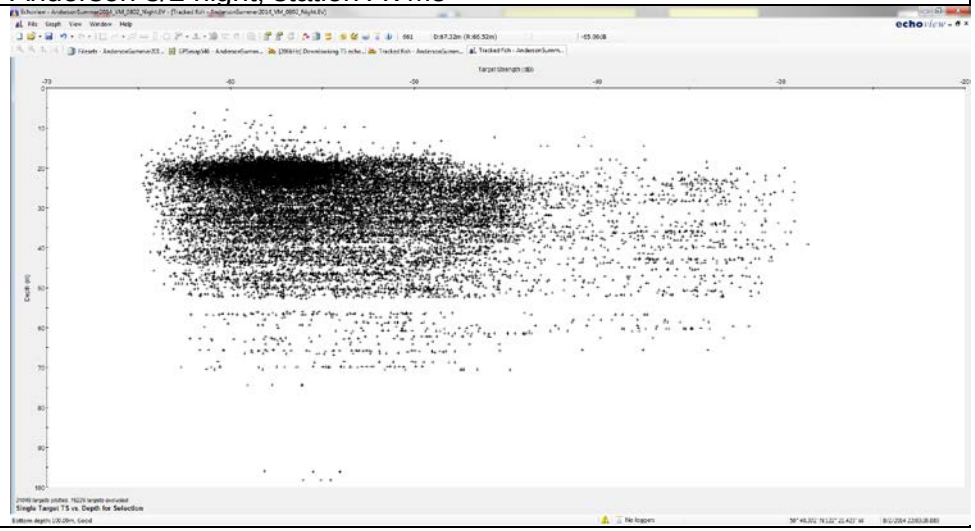


Figure 33. TS of fish (dB) versus depth (M) during summer DVM sampling of Seton and Anderson Lakes (July 29 – August 2, 2014). Data are from single echoes with threshold = -65 dB and range compensation = 40 log R. Scales are the same on all plots: Vertical, depth 0-100 m by 10 m increments, except Anderson 8/1 day 10-100 m; Horizontal, TS -70 to -20 dB by 10 dB increments, except Seton 7/29 night is -70 to -30.

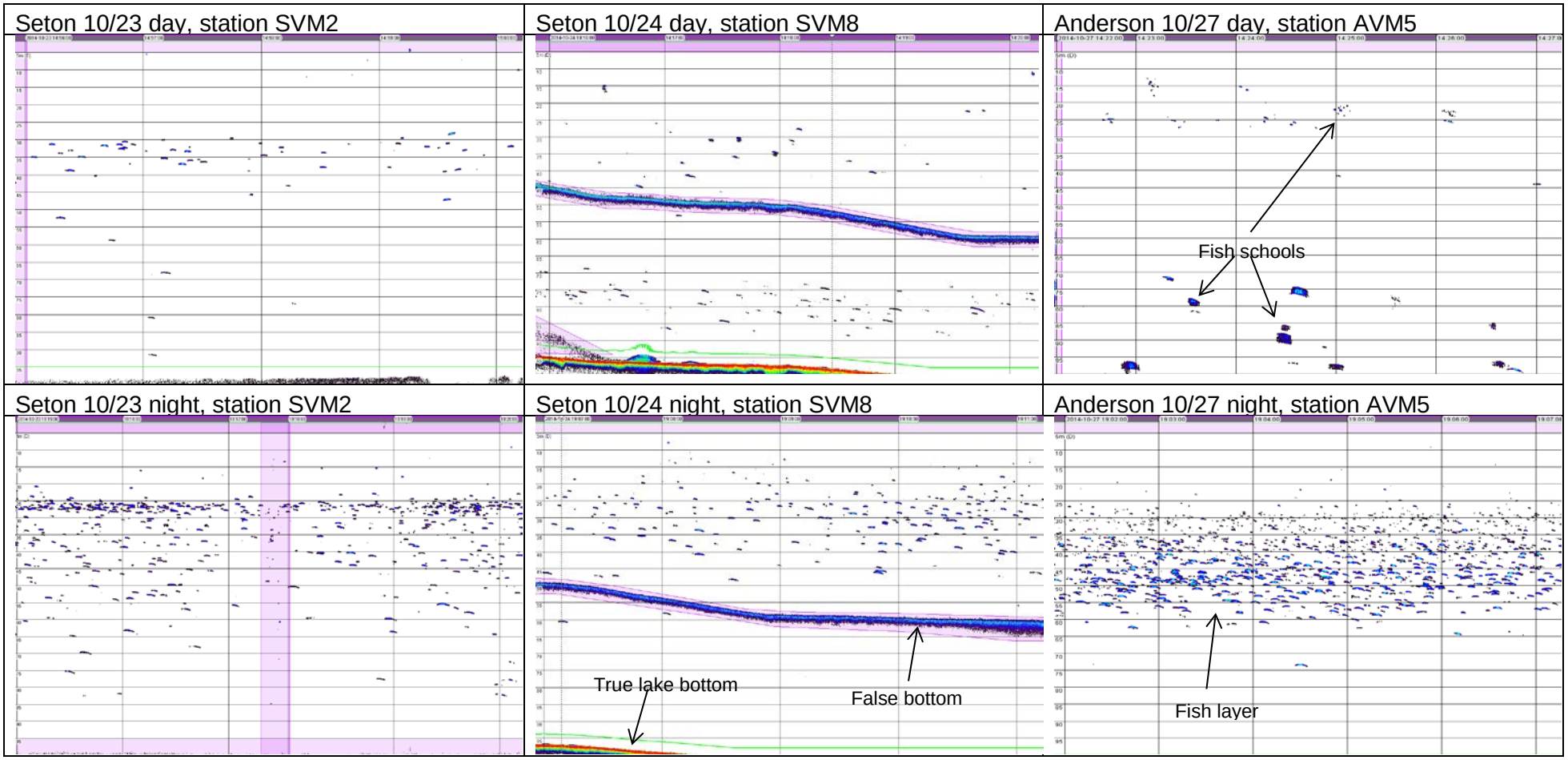


Figure 34. Several minute segments of up to 2 hour day and night time echograms from fall DVM sampling (October 23-27, 2014) of Seton and Anderson Lakes. Echogram settings (threshold = -65 dB, range compensation = 40 log R) were sensitive enough to show sockeye fry to the maximum depth displayed. Gridding is the same on all plots: Vertical, 0-100 m in 5 m intervals; Horizontal, 1 minute intervals. Purple areas are analysis exclusion zones for non-fish echoes; the green line above true bottom is the maximum analysis depth line. Images for Anderson Lake were truncated at a depth of 100 m for presentation with Seton Lake but fish were found deeper as discussed in the text.

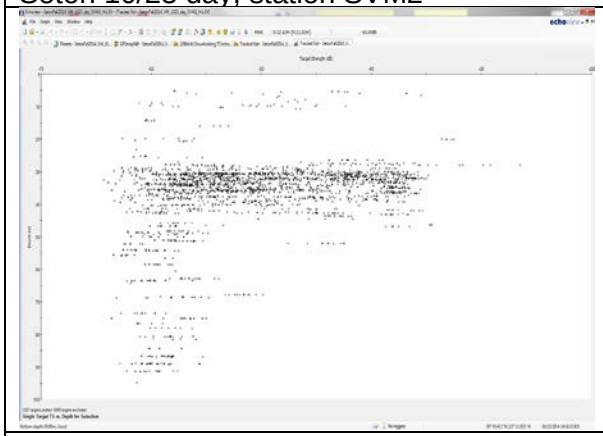
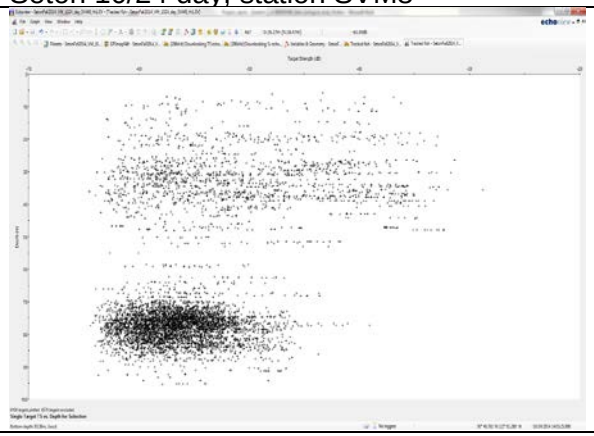
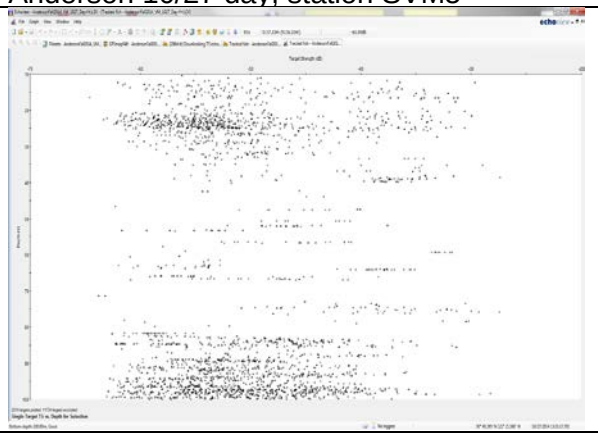
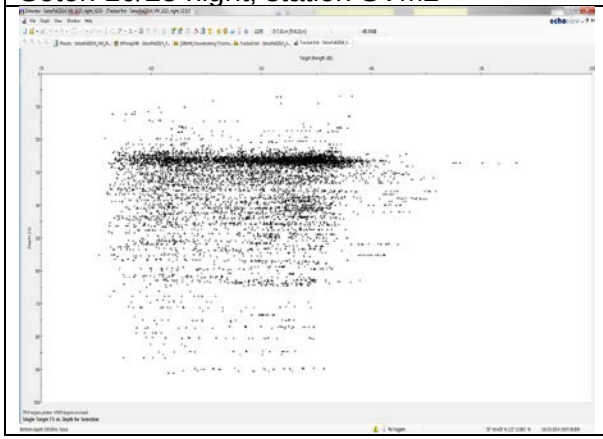
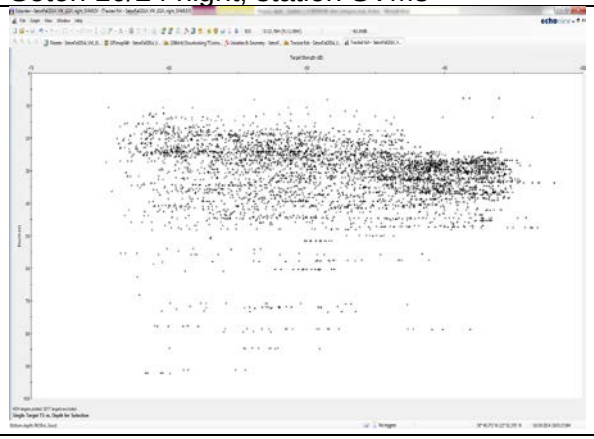
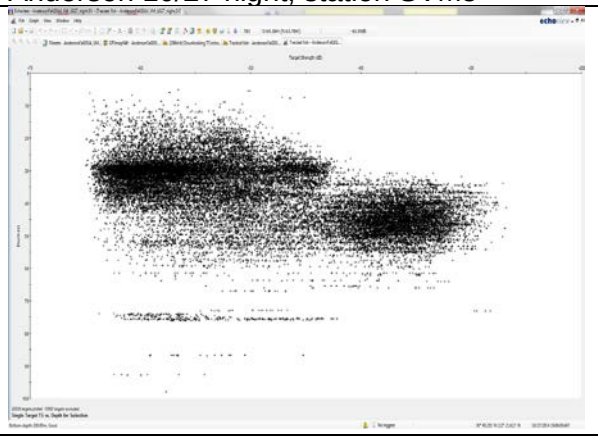
Seton 10/23 day, station SVM2	Seton 10/24 day, station SVM8	Anderson 10/27 day, station SVM5
		
Seton 10/23 night, station SVM2	**Seton 10/24 night, station SVM8**	**Anderson 10/27 night, station SVM5**
		

Figure 35. TS of fish (dB) versus depth (m) during fall DVM sampling of Seton and Anderson Lakes (October 23-27, 2014). Data are from single echoes with processing threshold = -65 dB and range compensation = 40 log R. Scales are the same on all plots: Vertical, depth 0-100 m by 10 m increments; Horizontal, TS -70 to -20 dB by 10 dB increments, except Seton 10/24 night is -70 to -30.

Table 15. Measures of water clarity in Seton and Anderson lakes during DVM sampling.

Survey period	Lake name	Limnology station	DVM transect	Date	Secchi depth (m)	Euphotic zone depth (m)*	Light attenuation coefficient
Summer	Seton	5	SVM 8	7/29/2014	7	14	0.26
"	Anderson	1	AVM 5	8/2/2014	15	22	0.16
Fall	Seton	2	SVM 2	10/23/2014	1	6	0.69
"	"	5	SVM 8	10/23/2014	2	12	0.41
"	Anderson	2	AVM 5	10/27/2014	14	24	0.16

* Depth at which 1% of incident light at lake surface remains.

5 NEXT TASKS

5.1 Question 1: What is the inter-annual variation in physical conditions in the Seton Lake caused by the diversion and did the diversion change primary and secondary production in Seton Lake?

A number of analytical tasks remain to be completed in 2015 for the Question 1 paleolimnology study. Tasks to be completed in 2015 are as follows:

- Cores from Seton Lake that are not yet aged have been prepared and are currently being counted or are in the counting queue. All counts are scheduled to be complete by the end of April 2015.
- Analysis of Cladocera from both lakes will be completed in 2015. These analyses will continue from the preliminary analyses completed in 2014.
- More slides of diatoms have been prepared for both of the Anderson Lake and Seton Lake cores. Analysis of the slides will be ongoing in 2015. Interpretations will focus on changes in nutrient status based on changes in diatom species composition.
- Sediment pigment and geochemical analysis is scheduled for completion in spring, 2015. Grain size and loss-on-ignition analyses are scheduled for summer 2015.

5.2 Question 2: Will the selected alternative (N2-2P) increase biological production in Seton Lake?

Two more years of measurements of primary and secondary production and ancillary measurements of phytoplankton and zooplankton biomass, turbidity, water chemistry, light, CTD profiles, etc. are required before analysis of the effect of N2-2P on biological production can be run. That work is scheduled for May – October, of 2015 and 2016 followed by lab work, data analysis, and reporting in 2017. This schedule will

provide the data needed to test the treatment effect using the BACI layout that is described in Section 3.1.1.

5.3 Question 3: To what extent does aquatic productivity alone limit the abundance and diversity of fish populations in Seton Lake?

A number of analytical tasks remain to be completed in 2015 for the Question 3 fish study. Tasks to be completed in 2015 are as follows:

- Finish processing and analysis of fall fish samples (DNA, aging, stomachs); finish DVM data processing and further develop the DVM model for small, medium, and large size groups of fish in relation to light levels and water clarity; test hypothesis concerning differences in the DMV between the two lakes through the use of modelling and synthesis of the data with reference to the literature on juvenile *O. nerka* particularly studies and models on DVM.

- Use irradiance profiles of the water column to examine differences in light attenuation rates between lakes and stations. Attenuation rates coupled with hourly incident solar radiation estimates will be used to model fish responses to changing light conditions within the water column during DVM sampling. Primary and secondary production that was measured as part of Question 2 studies will be used to describe differences in production of food for fish between the two lakes. All this information will be used to help interpret habitat use by fish as shown in the acoustic data and to examine links between vertical migration behaviour and habitat conditions.

- Process whole-lake acoustic survey data and from it develop fish abundance estimates for each seasonal survey. Comparison to other abundance estimates of the same brood year, including adult spawners, Gates Channel fry and smolt estimates will enable the development of freshwater survival and mortality estimates for the Seton/Anderson sockeye stocks as a whole but not necessarily for each stock.

There are several tasks outside the scope of BRGMON6 that have emerged as important to consider for future work and funding initiatives. These tasks would build on work completed to date and greatly assist with interpretation of present data. These tasks are listed as follows:

- We recommend DNA sampling for stock identification of the smolts migrating out of the Seton system in 2015. Our project will produce both absolute and proportional abundance estimates of lake rearing juvenile sockeye for two stocks (Gates and Portage). Other groups involved with sockeye salmon in the Seton watershed can or do produce similar estimates for other life history stages. The data we currently have for sockeye are number of spawning adults, potential egg

deposition, fry out of the channel (Gates only) and summer and fall lake rearing juveniles in the lakes. Smolts are currently sampled and enumerated each spring, and if DNA samples were taken as part of the sampling process we would be able to describe the complete freshwater life history and patterns of survival in the Seton watershed for the first time.

- We recommend the collection and aging of scales from sockeye smolts migrating out of Seton Lake in spring 2015 (BRGMON13). Analysis of sockeye juvenile scales that were collected in 2014 and review of aging by DFO in 2000-2003 revealed difficulty in clearly distinguishing between 0 and 1 year old fish. Age analysis of the smolts would provide more definitive evidence of true age of juveniles rearing in Anderson and Seton Lakes by stock identification from DNA analysis. These data would provide evidence of time of rearing by Gates and Portage juveniles in each lake, which is needed to resolve discrepancies of fish size between the two lakes.

- We recommend that aging comparisons be conducted using both otoliths and scales of Gwenish to determine if scale aging errors are being made (under-aging by one year). Our present analysis has raised the possibility that late fry emergence and slow growth rates of Anderson Lake Gwensih fry has resulted in the lack of an annulus formation in the scales of some fish going into their first winter (up to 16% in 2014). The lack of first winter annulus may mean that a similar percentage of age-1 and older Gwenish will be incorrectly aged through the use of scales alone. Otoliths are formed during embryonic development and will always show the winter check and can consequently be used to verify scale ages. Otoliths are more difficult and expensive to process and are not normally used for routine aging. Otoliths suitable for this analysis may be available from the 2014 trawl catch preserved in ethanol (otoliths from fish preserved in formalin cannot be read), in particular, Gwenish fry (identified by DNA analysis) that were of anomalously large size.

- We recommend that genetic analysis to distinguish Anderson and Seton Gwenish be further investigated to resolve uncertainties that emerged from earlier work by Moreira (2014) and the present study. Differences in the size, age of maturity, and morphology of Gwenish in Anderson and Seton lakes strongly suggest that the two populations are not the same demographically (Moreira 2014). Using different alleles than used in our study, Moreira also found modest but statistically significant genetic differences between the two Gwenish populations. Moreira's study was not designed to determine stock identification using current methods which would involve the collection of a larger sample of Gwenish from each lake and a subsequent analysis of the DNA. Stored tissue samples from Gwenish captured in 2014 gill netting could be used for this analysis.

5.4 Question 4: Can refinements be made to the selected alternative to improve habitat conditions or enhance fish populations in Seton Lake?

Two more years of data collection, lab work, and analysis is required before Question 4 will be answered. Multiple lines of evidence from all years of work will be used to determine if change to N2-2P will benefit fish populations as described in Section 3.4.

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7 RAW DATA APPENDICES

Raw data appendices are available via file transfer from BC Hydro.