Bridge River Project Water Use Plan

Seton Lake Aquatic Productivity Monitoring

Implementation Year 2

Reference: BRGMON-6

Study Period: 2015-2016

Limnotek Research and Development Inc. and affiliated organizations

May 16, 2016

SETON LAKE AQUATIC PRODUCTIVITY MONITORING: PROGRESS IN 2015-16

BC Hydro project number BRGMON#6

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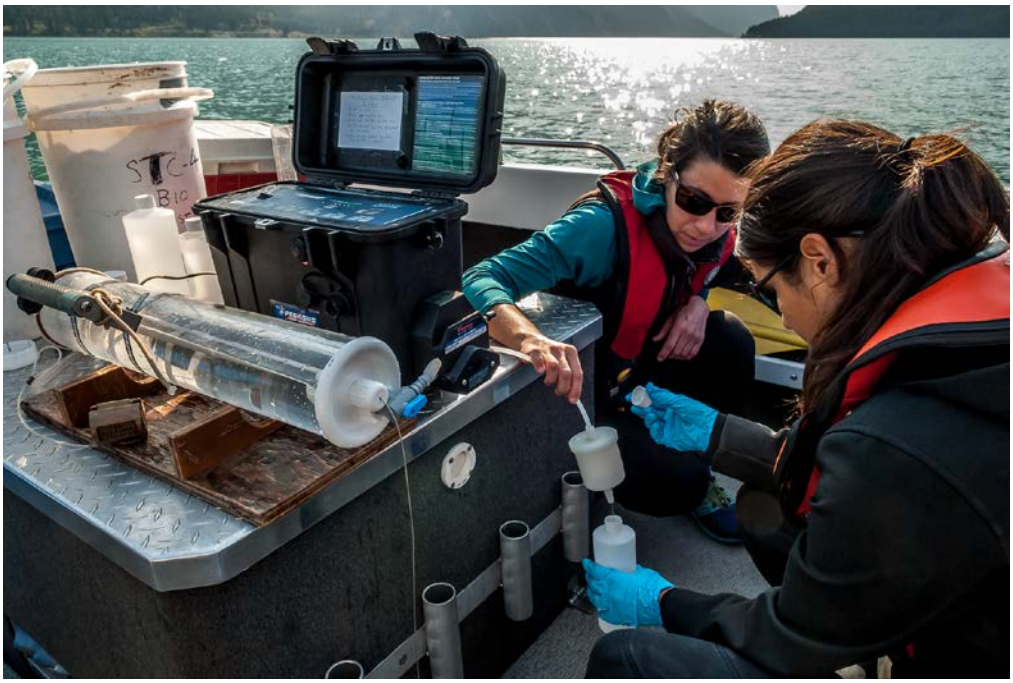

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ECO-RESOURCES

SETON LAKE AQUATIC PRODUCTIVITY MONITORING: PROGRESS IN 2015 - 2016

BC Hydro project number BRGMON#6

Submitted to

BC Hydro
Burnaby, B.C.

Prepared by

Limnotek Research and Development Inc.

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*Authors of this report are listed according to task and affiliation under Acknowledgements on page x. The study was managed by St'at'imc Eco-Resources Ltd.

Cover photo: Crew running water filtrations on board the boat on Seton Lake, June 17, 2015: C. Perrin photo.

EXECUTIVE SUMMARY

This report provides information from the first two 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 assessing 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. According to the age model to date, all three cores from Seton Lake exhibit highest rates of sedimentation near the discharge from the Bridge generating station and lower rates with increasing distance eastward and downstream from the generation station. 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.

Pigment analysis showed that algal assemblages in the cores included common diatoms, cryptophytes, and blue-greens that form the basis of the pelagic food chain in Seton and Anderson Lakes. All cores from Seton Lake showed an abrupt decline in all pigment concentrations and thus algal production coinciding with the timing of the Bridge River diversion. Time course change in diatom assemblages were consistent with this change. In contrast, pigment concentrations increased in Anderson Lake over the same time period. A decrease in concentrations of the sub-fossil cladoceran remains coincided with the timing of the Bridge River diversion in Seton Lake cores analyzed to date (one core remains to be examined) but not in Anderson Lake. Preliminary evidence shows that Seton Lake shifted from a higher meso-eutrophic state in an earlier time period to a more oligotrophic state in recent years. No shift in trophic state has been found in preliminary analysis of the Anderson Lake cores.

All findings are preliminary. Final conclusions will be developed once the cladoceran analysis of the final core is examined in late spring, 2016 and the core age model is finalized later in 2016.

Tasks to be completed in 2016 are as follows:

- Completion of the core age analysis,
- Further analysis of Cladocera assemblages over time,
- Completion of grain size analyses,
- Data analysis using the BACI design in PRIMER. In order to apply the BACI design, the age model needs be finalized for all cores, which will be done using the radioisotopic data outlined in this report.

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 used to mainly support interpretations of findings in other parts of the project (e.g. paleolimnology and the fish growth and migration patterns). Despite this focus for the limnology data, statistical tests will still be run to test the effect of N2-2P on primary and secondary production. If an effect is found after all data are collected after 2016, it will show that something other than N2-2P was producing the effect because the hydrology data show no effect of N2-2P on availability and attributes of pelagic habitat.

Stable stratification of Seton and Anderson Lakes was present in May – October, 2015 with evidence of seiche activity (east-west rocking of the thermocline). The monthly temperature profiling in 2015 showed a seiche amplitude of approximately 10m. This amount of seiche activity will produce temperature oscillations of several degrees over short periods of time in the outflow Seton River. The actual timing, frequency, and magnitude of seiche oscillation and its effect on fish habitat in the Seton River would have to be determined with more detailed measurements that were beyond the scope of the present study.

Turbid inflows to Seton Lake originate from the diversion of water from Carpenter Lake and to a small extent at certain times of the year from Whitecap Creek that flows into Portage Creek at the west end of Seton Lake. Inflow turbidity from the diversion occurs in three modes during spring through fall months. A spring influx is dispersed over much of the water column in Seton Lake and dissipates from higher turbidity at the diversion inflow end to low turbidity at the lake outflow. A fall influx of high turbidity is distributed over two distinct layers in Seton Lake: one consisting of particles that sink rapidly upon discharge from the diversion and produce turbidity along a bottom plume at the eastern end of the lake and a surface layer of very small particles that disperse along Seton Lake, producing a brilliant turquoise colour in the fall. Those surface particles must have colloidal properties to remain in suspension while the bottom particles must be relatively large to sink rapidly. Given that diversion inflow in the fall has a temperature that is similar to that of surface water in Seton Lake, the surface turbidity consisting of the very small particles, is entrained in surface water of Seton lake and travels eastward within the Seton epilimnion.

Biological production was measured among the algal (primary production) and zooplankton (secondary production) assemblages. Rates of primary production in Anderson and Seton Lakes were similar to those found in 2014 and were lower than those found in earlier measurements from 2000-2003. They were in the middle of the range of rates of primary production known among lakes and reservoirs of British Columbia. 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 Seton and Anderson Lakes are meso-oligotrophic with respect to trophic state.

One more year 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 to October of 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?

Between-lake differences between junvenile *O. nerka* ecology, biology and behaviour and physical habitat conditions (temperature, turbidity, light transmission) were examined in Anderson and Seton lakes to examine differences in growth and survival rates of *O. nerka* rearing in the lakes and to examine factors that may modify growth and abundance driven from biological production. Field work was conducted in 2014 followed by laboratory and data analyses and data interpretations in 2015 and early 2016.

Analysis of population structure, growth, and behaviour revealed complex differences among fish populations between Seton Lake and Anderson Lake. A combination of DNA analyses and acoustic sampling showed that adult Sockeye spawners in the Seton/Anderson watershed formed two distinct subpopulations of *O. nerka*, Gates Creek spawners and Portage Creek spawners. *O. nerka* caught in Anderson Lake were a mixture of Gwenish and Gates Creek Sockeye salmon. In Seton Lake, Gates Creek Sockeye salmon were the most common fish, followed by Portage Creek Sockeye salmon and then Gwenish. Spring and early summer migration of Gates Creek sockeye from Anderson Lake, through Portage Creek and into Seton Lake, that was documented in the late 1950's and early 1970's also occurred in 2014. The Gates origin fry were distributed evenly between Seton and Anderson Lakes. The Portage Creek Sockeye did not move to Anderson Lake from Seton Lake, showing a preference for Seton Lake. DNA and acoustics results showed that an estimated 1.5 million Gates origin Sockeye fry occurred in each lake in 2014, indicating that approximately half of the Gates age-0 Sockeye population left Anderson Lake for Seton Lake. The net effect of fish movements was no difference in size of the pelagic fish population between the two lakes. Growth rates of age-0 Sockeye were much higher in Seton than in Anderson. Sockeye fry in Seton were 40% longer and 300% heavier in Seton compared to Anderson. In addition, summer to fall survival rates of age-0 Sockeye in Seton were double those in Anderson. *O. nerka* in both lakes exhibit diel vertical migration (DVM) patterns but the extent of DVM was greater in Anderson Lake than in Seton Lake. The larger size of Sockeye fry in Seton Lake compared to those in Anderson Lake is not

consistent with expectations of growth related to food availability in the two lakes. Zooplankton production estimates were twice as high in Anderson compared to Seton based on standard zooplankton net hauls from a depth of 30m, however, latest analysis of hydroacoustic data in 2016 shows that most zooplankton in Seton may be deeper than 30m, and that zooplankton production estimates to date may be underestimated in that lake.

Four hypotheses were tested to explain the counterintuitive observations of high fish growth rates and larger size of *O. nerka* in Seton than in Anderson Lake but higher rates of food production in Anderson than in Seton.

H1: Sockeye fry remain in the upper water column of turbid Seton Lake throughout the day along with their zooplankton prey, and as a consequence the diel vertical migration (DVM) of Sockeye fry is reduced or absent in Seton Lake while it is extensive in relatively clear Anderson Lake.

H1 result: H1 was not confirmed. DVM occurs in both lakes but the extent of DVM is less in Seton than in Anderson at least in part due to differences in light penetration between lakes (see H2 below). The premise was that turbidity in Seton Lake would provide enough protection against predation (mainly by bull trout) that age-0 *O. nerka* would not vertically migrate, enabling them to feed continuously in the zooplankton layer during the day rather than just during brief crepuscular periods. Continuous feeding in Seton would produce larger fish than time-limited feeding in Anderson Lake. Although mean daytime depths of fish were related to light attenuation, the decrease in illumination of surface waters caused by turbidity was not enough for the majority of age-0 *O. nerka* to remain in the epilimnion at all times in Seton Lake. This conclusion was potentially affected by a strong turbidity gradient in Seton Lake that we were not aware of at the time of laying out the sampling design. Observations of individual acoustic transects showed that DVM was more strongly attenuated at places of high turbidity and less attenuated at places of low turbidity. With knowledge gained in the recent limnology analyses and this fish assessment a different sampling design with greater sample size in further study is required to determine if the effect of different amounts of turbidity on DVM patterns is significant.

H2: The daytime depth distribution of *O. nerka* fry is related to differences in light penetration in each lake and they fit Levy's (1990) model of mean depth vs the light attenuation coefficient.

H2 result: H2 was accepted. Daytime depths of *O. nerka* fry were deeper in Anderson Lake than in Seton Lake. There was a significant relationship between mean depth and the light attenuation coefficient within the study lakes and the relationship was the same as was found from Levy's (1990) model for age-0 Sockeye Salmon and Gwenish in several BC lakes.

H3: The depth distribution of *O. nerka* fry during daytime and dusk (excluding night) conforms to a well known antipredation window model (Scheuerell and Schindler's, 2002) coefficient.

H3 result: Scheuerell and Schindler's (2002) model poorly represented the multimodal (usually bimodal) daytime depth distribution of *O. nerka* fry in Anderson and Seton lakes and thus was not relevant. The Scheuerell and Schindler (2002) model is based on a single modal group of vertically migrating fish. Although juvenile *O. nerka* performed a DVM in Anderson and Seton Lakes, their day time vertical distribution was more complex than we are aware of elsewhere. Age-0 *O. nerka* were not confined to single part of the water column in either lake but were distributed bimodally during the day, sometimes with a large proportion of the fry found in well illuminated waters above the thermocline in both lakes. These shallow fish were mostly in schools, which is a predator avoidance strategy. Both the mean depth and the depth of the deepest modal group was shallower in Seton Lake than in Anderson Lake. These vertical distribution patterns suggest that fry in Anderson and Seton Lakes were using two different antipredation strategies. One was to form schools in the upper water column where illumination was sufficient for schooling, which reduces but does not eliminate visual predation risk, while allowing feeding if zooplankton were present. The other was to descend to depths where illumination was inadequate for piscivores to prey on them, but where zooplankton were mostly absent.

H4: Predator density is lower in the pelagic habitat of Seton Lake than in Anderson Lake

H4 result: H4 was accepted. The density of pelagic piscivores (mostly Bull Trout with some Northern Pikeminnow) was estimated to be 1 fish/ha in Seton Lake compared to 9 fish/ha in Anderson Lake. With larger, faster swimming age-0 Sockeye fry and fewer predators in Seton Lake, lower predation rates would be expected in Seton Lake than in Anderson Lake

An overall conclusion is that shallower daytime depths of habitat used by *O. nerka* coupled with fewer predators in Seton Lake than in Anderson Lake are factors that can explain higher growth rates and larger fish in Seton Lake compared to Anderson Lake. Further zooplankton data to be collected at depths greater than 30m in 2016 are needed to gain further insight into links between diel change in availability of food at depths covering vertical migration patterns among the pelagic fish populations in Seton Lake. Final conclusions will then be developed in 2017.

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

One more year 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 the selected N2-2P flow alternative will benefit pelagic fish populations.

A summary of the status of BRGMON6 study findings is listed in the following table:

Study objectives	**Management questions**	**Status**
Determine if the Carpenter to Seton diversion caused change in biological production in Seton Lake	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 study is on track to answering the management question with additional data in 2016-2017 using the current approach/study design
Determine if the present flow alternative has increased biological production in Seton Lake	Will the selected alternative (N2-2P) increase biological production in Seton Lake?	The study is on track to answering the management question with additional data in 2016-2017 using the current approach/study design
Determine if there are factors other than biological production that may affect fish assemblages in Seton Lake	To what extent does aquatic productivity alone limit the abundance and diversity of fish populations in Seton Lake?	The study is on track to answering the management question with additional data in 2016-2017 using the current approach/study design
Determine if changes to flow will improve habitat for pelagic fish populations in Seton Lake	Can refinements be made to the selected alternative to improve habitat conditions or enhance fish populations in Seton Lake?	The study is on track to answering the management question with additional data in 2016-2017 using the current approach/study design

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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 its land locked variety called Gwenis (also known as kokanee) (both *Onchorhynchus nerka*) 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 (Geen and Andrew (1961). 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 (Geen and Andrew 1961). Studies by the Pacific Salmon Fisheries Commission 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 (Geen and Andrew 1961). While these observations implied 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 two of three years of monitoring, sample collection, laboratory work, and analysis that is required to answer these questions.

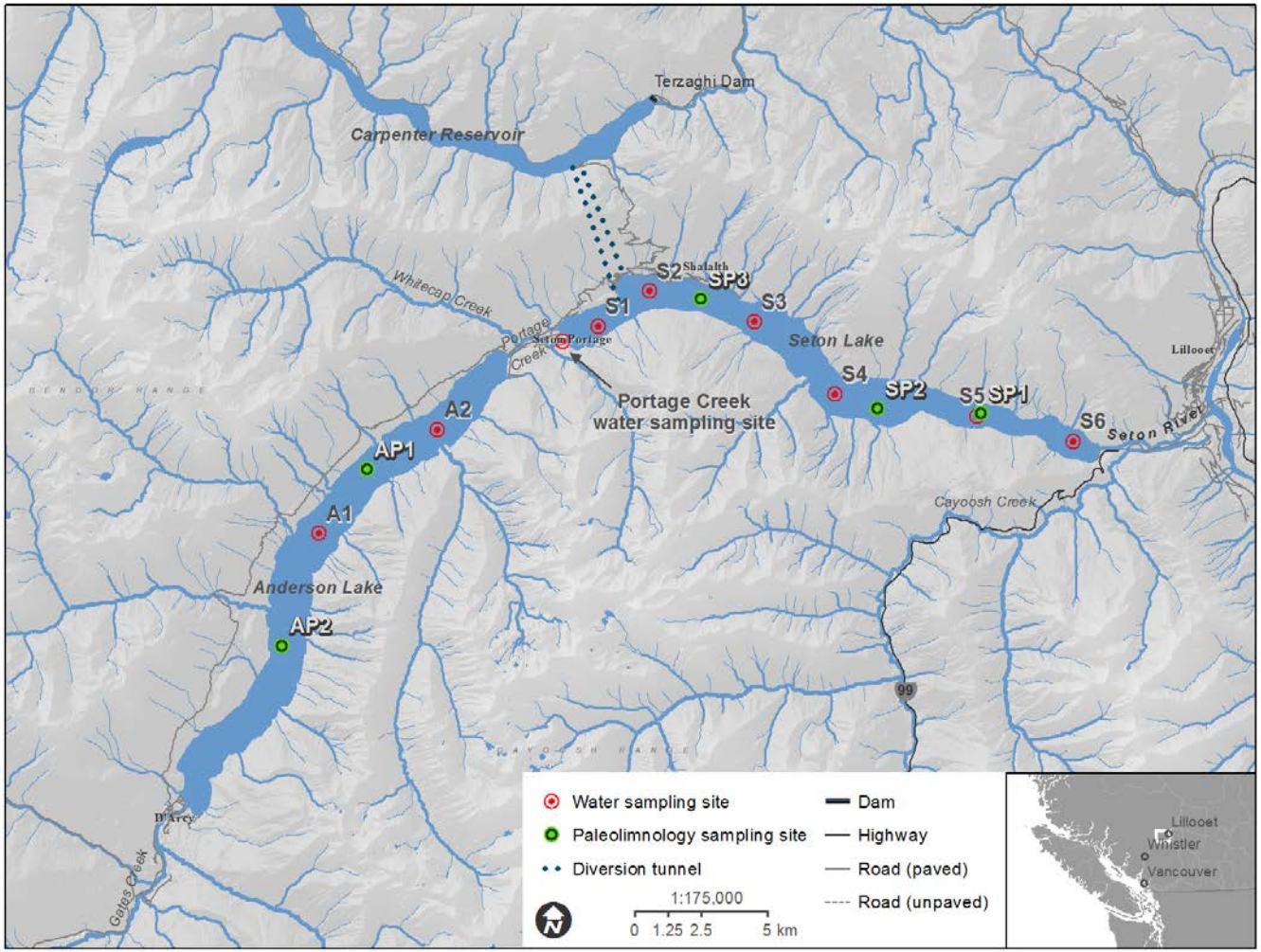


Figure 1. Study area showing Seton and Anderson Lakes, water and paleolimnology sampling sites, and the local watershed.

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. The land between Anderson and Seton Lakes is thought to have formed about 10,000 years ago when a massive slope failure separated what was previously one fjord lake into what is now Anderson and Seton Lakes (http://bivouac.com//TownPg.asp?TownId=586). As the westward water levels rose, a stream eroded the debris to form what is now known as Portage Creek, providing eastward flow of water from Anderson Lake to Seton Lake. Slopes north of Seton Portage within the Whitecap Creek drainage remain unstable with debris flows. One of these events occurred in the fall of 2015 when a debris flow blocked Portage Creek. Machines were used to excavate and clear a channel to allow migrating sockeye to pass upstream to Anderson Lake and spawning habitat in Gates Creek.

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 of 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
Average water residence time**	239 days	5.3 years

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

**Average water residence time was calculated as the 2015 average value of daily total volume divided by mean daily rate of total outflow (data from BC Hydro Power Records and Geen and Andrew (1961)).

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). With added inflows from the Carpenter diversion the mean annual total outflow from Seton Lake is 112 $m^3 \cdot s^{-1}$ (data from 2015, BC Hydro Power Records). 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 but the dominant year changed in 1997-98 (Gull et al 2014). 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 measured as the number of successfully spawned females determined by examination of egg retention in post-spawned female carcasses (Hume et al. 1996). In 2013 a total of 57,209 sockeye spawned in Gates Creek and Channel with 28,948 females and 23,004 EFS. A third of those numbers were found in 2014 and 2015. Spawning escapement to Portage Creek was much lower with only 7,509 total spawners with 4,406 females and 4,181 EFS in 2013. Numbers more than doubled in 2014 but collapsed in 2015 to only 36 total spawners and 17 EFS.

Sockeye fry emerge from the spawning gravel of Gates and Portage Creeks 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).

A number of other Pacific salmon and resident fish species share the Seton-Anderson system with Sockeye Salmon and Gwenish. Coho salmon (*O. kisutch*) and Chinook salmon (*O. tshawytscha*) spawn in Portage and Gates creeks and the Seton River. Pink salmon (*O. gorbuscha*) spawn in Portage and Cayoosh creeks and the Seton River (Geen and Andrews 1961). Gwenish are a unique variant of landlocked sockeye (*Oncorhynchus nerka*), 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). Redside Shiner (*Richardsonius balteatus*), Northern Pikeminnow (*Ptychocheilus oregonensis*), Prickly Sculpin (*Cottus asper*), Coastrange Sculpin (*C. aleuticus*), Longnose Sucker (*Catostomus catostomus*),

Rainbow Trout (*O. mykiss*), Mountain Whitefish (*Prosopium williamsoni*), Lake Whitefish (*Coregonus clupeiformis*), Bull Trout (*Salvelinus confluentus*), Sturgeon (*Acipenser* sp.), Bridgelip Sucker (*Catostomus columbianus*), Peamouth (*Mylocheilus caurinus*), and Dolly Varden (*Salvelinus malma*) have also been reported from the Seton-Anderson system (BC MOE Fish Inventory Data Query system, http://a100.gov.bc.ca/pub/fidq/infoSingleWaterbody.do, accessed February 1, 2016). Many of these fish species other than Sockeye Salmon and Gwenish are primarily riverine or littoral species that are seldom found in the pelagic zone of lakes.

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 seasonal mean 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.

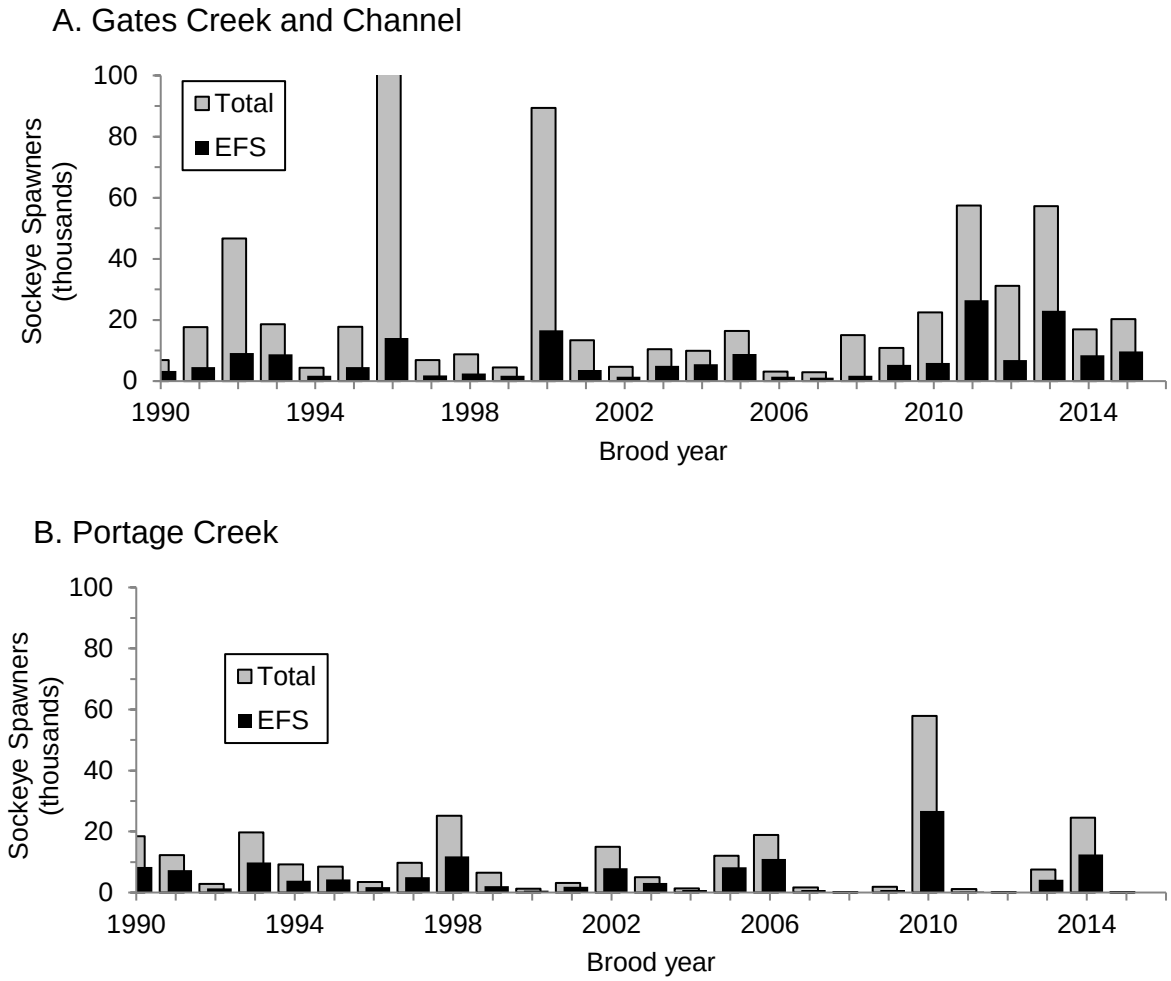


Figure 2. Number of sockeye salmon spawners and number of effective female spawners in Gates Creek and Portage Creek in brood years 1990 through 2013. Data are from Grant et al. (2011) and Personal Communication with Keri Benner, Fraser Sockeye Stock Assessment, Fisheries and Oceans Canada, Kamloops, B.C.

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) is being 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 can 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 are using 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 are being used in conjunction with physical and isotopic variables to interpret changes in nutrient source and production.

Sockeye salmon carcasses contain enriched levels of $\delta^{15}N$ (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 salmon-derived nutrients (SDN) can be a major source of nutrient subsidies to freshwater ecosystems that are linked to populations of migrating salmon (Chen et al. 2011a, 2011b). The deposition of SDN during the fall can enhance the productivity of nursery lakes at each trophic level, enhancing primary and secondary production for the subsequent year (Schindler et al. 2003, Selbie et al. 2009, Chen et al. 2011a,b). However, flushing rates of salmon nursery lakes can mediate overall food web attenuation of SDN (Holtham et al. 2004). The nutrient enrichment from SDN 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 SDN (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 (Selbie et al. 2007, 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, Selbie et al. 2007).

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 appeared 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 spring 2016 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 122°03.081W	110	74.5	41.5
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 spectrometers at Queen's University, a number of radioactive (i.e. unstable) isotopes were measured from selected intervals from each core, to provide a means of estimating the age of the sediment intervals. From each core, between 14 and 30 sediment samples were analyzed for the activities of ^{210}Pb (lead), ^{137}Cs (Cesium), ^{214}Pb and ^{214}Bi (Bismuth) following the procedures outlined by Schelske et al. (1994). The isotope of ^{210}Pb has a half live of approximately 22.3 years, therefore decay of the ^{210}Pb activity allows the dating of sediments up to ~150 years old. 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, which marks the peak fallout of atmospheric nuclear testing. The activities of ^{214}Pb and ^{214}Bi provide 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 can provide a reasonable and accurate chronology for lakes with variability in sedimentation rates (Appleby and Oldfield 1978). For Anderson Lake, every centimeter from 0 to 10 cm and at 12 cm, 15 cm, 20 cm and 25 cm 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 Loss-On Ignition

Loss-On-Ignition (LOI) provides estimates of the percent of carbonate and organic content in sediments. LOI has been completed following the methods outlined in Heiri et al. (2001). Clean crucibles were placed in a muffle furnace at ~105°C during 30 min to dry them out completely and then transferred into desiccators to cool down. The empty crucibles were weighed and, ~0.1 g of freeze-dried sediment was added to the crucibles. The samples were heated to 550°C in a muffle furnace for four hours. The crucibles were transferred into desiccators to cool down and the dry weight (crucible + sediment) after 550ºC (DW550) was recorded for each crucible. The fraction of weight loss between 105 and 550°C is an estimate of the percent organic matter in the sediments. The crucibles were then returned to the muffle furnace and heated to 950°C for 2 hours. The crucibles were then weighed to obtain the dry weight after 950ºC (DW950). The weight loss between 550°C and 950°C is an estimate of the carbonate fraction of the sediments.

3.1.5 Pigment Analysis

Algal pigments were used as indicators of environmental time course changes in Seton and Anderson Lakes (Table 3). Pigment analysis was completed using the High Pressure Liquid Chromatography (HPLC) at the Institute for Environmental Change and Society at the University of Regina. In order to preserve the pigment in the sediment, the sediment bags were kept frozen prior to analysis. The sediment sample bags were then freeze-dried. The first step of the pigment analysis consists of pigment extraction from the sediment matrix: 50mg of freeze-dried sediment for each sample was transferred to 4-dram vials and 5ml of extraction solvent was added. After 5 minutes, if no distinct colour change (light yellow) was observed, more freeze-dried sediment was added in the solution (in the case of Seton and Anderson up to 2 g of sediment was added to get this colour change). Following the addition of sediment, the vials were capped and the extracting sediments were then placed in a freezer for 24 hours. In the second step, the solvent containing the extracted pigments was filtered to eliminate the sediment matrix. The solvent was transferred from the 4-dram vial into a 50ml beaker. The 4-dram vial was gently rinsed using HPLC grade acetone and decanted into the beaker. The solvent was then drawn up into a syringe and a 0.22 µm filter was attached at the tip of the syringe. The solvent was filtered into a new 4-dram vial. The vials were capped and kept in the freezer overnight. In the third step, the samples were dried using nitrogen gas loaded into the vials. When samples dried, a yellow residue coated the bottom of the vial. To avoid oxidation of the pigments, the vials were capped. In the fourth step, depending on the amount of pigment contained in each of the vials, from 500 to up to 2000µl of injection solution was added in each 4-dram vial. When pigments were completely dissolved into the injection solution, the solution was transferred into HPLC autosampler vials; three standard HPLC vials were filled with a SUDAN solution, an injection solution, and Geranium chlorophyll. The samples were then run in

the HPLC. The resulting chromatography for each sample was then analysed and the pigment concentrations were expressed in ng/g organic matter.

Table 3 Pigments as indicators in lake sediment cores (modified from Leavitt and Hodgson 2001).

Pigment	Affinity	Additional information
Diatoxanthin	Dinophyta, Bacillariophyta, Chrysophyta	None
Fucoxanthin	Dinophyta, Bacillariophyta, Chrysophyta	None
Alloxanthin	Cryptophyta	None
Pheophytin B	Chl-b derivate	None
Lutein	Chlorophyta, Euglenophyta, Plantae	Because of their similar retention time the separation of Lutein and Zeaxanthin is often unclear on the chromatogram. It is for this reason that they are plotted together.
Zeaxanthin	Cyanobacteria	
Canthaxanthin	Colonial cyanobacteria, herbivore tissues	None
Echinenone	Cyanobacteria	None
B-carotene	Plantae, Algae, some phototrophic bacteria	None
Pheophytin A	Chl-a derivate	None
Chlorophyll a	Plantae, Algae	None

Indicators	Information
Chlorophyll a : pheophytin a (Chla:pheoa)	Indicator of pigment conservation
UV index	Sed C/(Alloxanthin+Diatoxanthin+Lutein)*100 Sed A, B and C are all scytonemin derivatives which arise from cyanobacteria. when exposed to damaging levels of high energy irradiance (UV), cyanobacteria produce an extracellular 'black' compound called Syctonemin.

3.1.6 Stable Isotopes

Isotope analysis was completed at Idaho State University following the methods outlined in Talbot et al. (2001) and Wolf et al. (2001). A subset of samples were HCl acid washed to

assess the influence of carbonate on ^{13}C isotopes, however, comparison between acid washed and raw samples revealed no significant difference in carbon isotopes, and raw samples were retained for analysis. Homogenized sediment samples were freeze-dried and analysed for C (%), N (%), $\delta^{13}C_{org}$ and, $\delta^{15}N_{org}$ by continuous flow — isotope ratio mass spectrometry (CF-IRMS). $\delta^{15}N$ has commonly been used as a proxy of salmon population dynamics, and $\delta^{13}C$ and C:N ratio as a proxy of inputs from the catchment relative to autochthonous sources. C(%) and N(%) were transformed into atomic weight. The correlation between $\delta^{15}N$ and C:N was determined using Pearson correlation coefficient.

3.1.7 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.

3.1.8 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 (20 individuals for samples at very low concentration) were enumerated (Kurek et al. 2010). Standard identification keys were used to identify the remains of cladocera (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). The size of *Bosmina sp.* and *Daphnia sp.* provided insight into predation pressure on the cladocerans (Korosi et al. 2013).

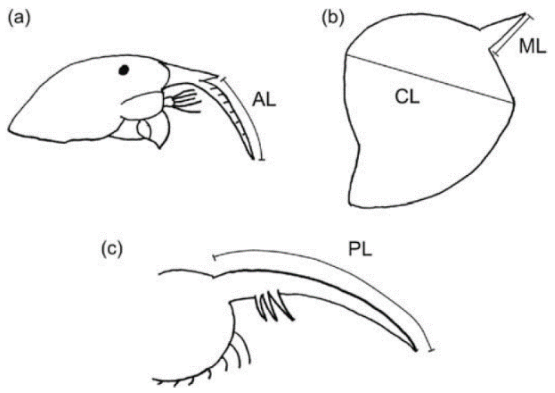


Figure 3. Schematic view of the headshield (a) and carapace (b) of *Bosmina* and the post-abdominal claw of *Daphnia* (c) from which length measurements were made. Abbreviations: AL, antennule length; CL, carapace length; ML, mucro length; PL, post abdominal claw length). The figure is reprinted from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

Cladocera were counted at a coarser 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.

3.1.9 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.

3.1.10 Grain Size Analysis

The processing of the samples for grain size analysis was completed at PEARL (Queen's University) following the procedure of the Limnological Research Center at the University of Minnesota. The analysis of the samples was completed using the Malvern Mastersizer2000 of the Chemical Engineering Department at Queen's University, assuming a reflecting index of 1.54, which is common for lake sediments.

In a 50mL centrifuge tube, 100 to 150 mg of wet sediment were suspended into ~30 mL of H_2O_2 and left for an extended period of time to slowly react and remove the organic matter. Once the reaction has stopped (no bubbles observed in the tube), the samples were centrifuged at 3500 rpm for 30 min to 1 h (until the supernatant was transparent), the supernatant was then aspirated. The sediment was re-suspended in a 40mL of deionized water using a vortex agitator. Each sample was centrifuged, aspirated and rinsed three times. To remove any excess of organic matter, 2 mL of nitric acid was added for 10 minutes. The samples were centrifuged, aspirated and rinsed with DI water, three times. The samples were then treated in 40 mL of 1M of sodium hydroxide to remove the biogenic silica that could be detected as sediment particles of the size range of the silt. The samples were placed in a water bath with loosened caps. After 10 min, the samples were shaken and put back in the water bath for an additional 10 min. The samples were then centrifuged for at least 40 min and aspirated three times, after ~8 mL of 0.5N HCl was added and tubes were filled up with 35 mL of DI water. The samples were then centrifuged and aspirated three times.

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 a long water residence time in Seton Lake (average estimated to be 238 days: calculated from raw data supplied by BC Hydro Power Records and total lake volume reported by Geen and Andrew (1961)), 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 Andrews (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 and 2015 that will be repeated in 2016 matched those used in 2000 – 2003 to facilitate a balanced design. Primary production sampling in 2014 and 2015 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 in 2014 and May through September in 2015.

3.2.3 Habitat attributes

Measurements were made monthly at both stations in each lake in 2014 and 2015 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:

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

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). In 2014, 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 of 2014 and in all of 2015, 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 2014, 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 2014 and in May through October 2015. 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). Dissolved inorganic nitrogen (DIN) was considered the sum of NO_3-N and NH_4-N concentrations. 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. In 2014, 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. In 2015, the particle size distribution in water discharged from BR1 (flowing into Seton Lake) was sampled again along with water flowing into Downton Reservoir (Upper Bridge River) and water flowing into Carpenter Reservoir (Middle Bridge River) in May, June, September and October. 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 were 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.

In 2014, 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 –20ºC 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 duplicate vertical hauls of a 153 µm mesh Wisconsin net having a 30 cm intake opening at each of the two 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:

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

where P is annual zooplankton production (mass$\cdot m^{-2} \cdot yr^{-1}$), B is mean annual dry weight biomass (mass$\cdot m^{-2}$) of the population of interest, and y is a rate in units of yr^{-1} (Benke and Huryn 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):

$$[P{:}B]_{daily} = 10^{(\alpha_{taxon}+\beta T_{daily})}$$ 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 2015 and for existing data that were collected by DFO in 2000-2003. The same will be done for zooplankton data to be collected in 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

It is well known that turbidity in aquatic ecosystems can reduce the predation risk from visual predators (Gregory 1993; Gregory and Levings 1998; Hansen et al 2013). Turbidity increases light attenuation in the water column, thereby reducing the depth range that has enough light to allow visual predation (e.g., Beauchamp et al 1999). Even at light levels sufficient for visual predation, turbidity can reduce visibility of prey by decreasing the contrast between the prey and their background, thereby reducing the effective search volume of predators, especially piscivores (e.g., De Robertis et al 2003).

Juvenile Sockeye Salmon (*Oncorhynchus nerka*) no doubt exploit both these effects of turbidity to reduce their risk of predation, but the first one appears especially important to this species. Sockeye fry rearing in the pelagic zone of lakes typically reside in deeper water during the day than at night, performing a twice-daily vertical movement in the crepuscular periods known as a diel vertical migration (DVM). Their mean daytime depth is negatively correlated to the light attenuation coefficient which in turn is negatively related to increasing turbidity (they dive deeper in clearer lakes), and the depth occupied during dusk and dawn vertical migrations

appears to minimize the ratio of predation risk to feeding rate (Clark and Levy 1988, Scheuerell and Schindler 2003). The twilight period when light is adequate for foraging by Sockeye but minimal for their predators is known as the antipredation window and is typically very short (Clark and Levy 1988).

Since the time of the Bridge River diversion from glacially fed Carpenter Reservoir, Seton Lake has become more turbid, resulting in decreased light transmission and visibility (Geen and Andrew 1961; Woodey 1972; Shortreed et al 2001). Based on observations from previously unpublished data from juvenile Sockeye surveys of Seton and Anderson Lakes between 2001 and 2003 (DFO, Lakes Research Program, Cultus lake Salmon Research Laboratory, Data on file), we hypothesized that Sockeye fry have exploited this change and that elevated turbidity in Seton Lake has: 1) increased growth and survival rates of juvenile Sockeye rearing in Seton Lake; and; 2) promoted an unusual early migration of many Sockeye fry from Anderson Lake to Seton Lake where they rear for much of their lacustrine phase. More specifically, we theorized that increased turbidity in Seton Lake hides the fry from predators while they remain in the upper water column with their zooplankton prey for much of the day, increasing their survival rate and feeding opportunities. We further theorized that this reduced DVM and enlarged antipredation window allows Sockeye fry to grow faster in turbid Seton Lake than in the clear waters of Anderson Lake, despite higher zooplankton densities in Anderson Lake (Shortreed et al 2001, Question 2 studies of zooplankton in this report). These conjectures may also apply to kokanee, the lake resident form of *O. nerka*, which are known as Gwenish (Moreira, 2014) in these lakes.

To test these ideas we examined the relationship between the biological and physical habitat attributes of Anderson Lake (clear water) and Seton Lake (turbid water) and the behavioural adaptations of their respective Sockeye fry and Gwenish populations using two data sets: 1) mostly unpublished pelagic fish data for Sockeye fry population estimates (coordinated acoustic and trawl surveys) and limnological data that was collected by DFO in 2000 – 2003; and 2) similar data from studies we performed in the summer and fall of 2014 that also included:

- Pelagic gill net sampling to assess larger age-1 and older Gwenish and large pelagic piscivores,
- genetic identification of *O. nerka* fry to determine growth and survival rates of subpopulations and their migration patterns between lakes,
- diet analysis of various size classes of *O. nerka* and large pelagic piscivores;
- Acoustic monitoring to determine horizontal (lake-wide) and vertical (DVM) patterns of *O. nerka* and large pelagic piscivores.

The objectives of this report were to analyze the 2014 and earlier data to determine if increased turbidity in Seton Lake has benefited juvenile Sockeye Salmon that rear there. To this

end the following specific hypotheses (H_n) were tested statistically or examined in some less rigorous but appropriate fashion:

- H_1: Sockeye fry remain in the upper water column of Seton Lake throughout the day along with their zooplankton prey and, as a consequence, the diel vertical migration of Sockeye fry is extensive in relatively clear Anderson Lake and reduced or absent in turbid Seton Lake;
- H_2: The daytime depth distribution of *O. nerka* fry is related to differences in light penetration in each lake and fits Levy's (1990) model of mean depth vs the light attenuation coefficient;
- H_3: The depth distribution of Sockeye fry during daytime and dusk (excluding night) conforms to Scheuerell and Schindler's (2002) antipredation window model;
- H_4: Sockeye fry migrated from Anderson Lake to rear in Seton Lake in their first summer of life, but not vice versa;
- H_5: Age-0 *O. nerka* growth and survival is higher in Seton Lake than in Anderson Lake;
- H_6: Predator density is lower in the pelagic habitat of Seton Lake than in Anderson Lake;
- H_7: Juvenile Sockeye losses to predation are lower in Seton Lake than in Anderson Lake.

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. Trawl and gill net catch data were used to apportion the acoustic population estimates among fish species and stocks of *O. nerka* (from DNA). Two surveys were conducted on each lake in 2014, a summer survey from July 28 to August 4 and a fall survey October 22-29. Each survey of each lake was composed of two parts: a whole-lake survey for developing an abundance estimate of pelagic fish, and a daily vertical migration (DVM) study to quantify diel vertical migration patterns of fish in the pelagic zone. Acoustic data were also used to qualitatively assess diel changes in the depth distribution of zooplankton.

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, see Table 4 for additional settings). 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).

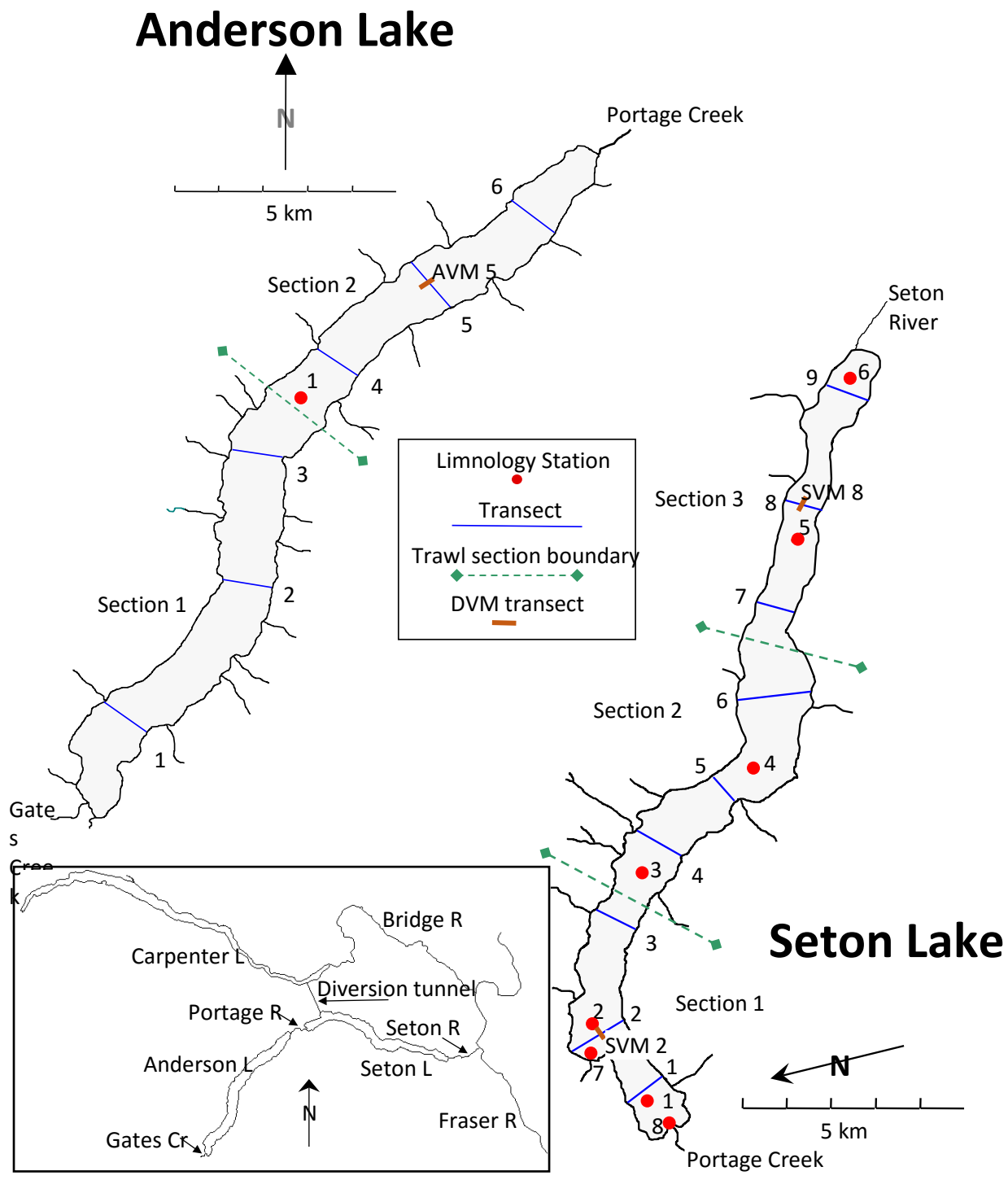


Figure 4. Maps of Anderson and Seton lakes showing hydroacoustic transects and mid water trawling sections. Limnology sampling stations from Figure 1 are shown for reference.

Table 4. Equipment specifications and settings for collection and processing of acoustic data collected from Seton and Anderson Lakes in summer and fall 2014 for whole-lake and DVM surveys.

Project Phase	Category	Parameter	Value
Data collection	Transducer	Type[1]	Split-beam
"	"	Sound frequency (kHz)	206
"	"	Nominal (full) beam angle	6.7°
"	"	Depth below lake surface	1.0 m
"	Settings	Pulse width	0.4 ms
"	"	Transmit power	0.0 dB (high power)
"	"	Collection threshold	-100 dB
"	"	Minimum data range[2]	1.0 m
"	"	Ping rate per transducer	4-5 pps for whole-lake surveys 3-6 pps for DVM study
"	GPS	Type[3]	Differential
"	"	Datum	NAD83
"	Other	Transecting speed	1.5-2.0 m/s (5.4-7.2 km/h)
Data Analysis	General	Calibration offset (dB)	0.0
"	"	Range processed [2]	2-80 m for whole-lake surveys 2-100 m for Seton DVM 2-125 m for Anderson DVM
"	"	Time varied gain	20 log R for echo integration, 40 log R for TS and tracking
"	"	Minimum threshold (dB)[4]	-65 dB for fish -87 dB zooplankton
"	"	Maximum threshold (dB)[4]	none
"	"	Time varied threshold	None for fish -120 dB @ 1m for zooplankton
"	TS of echoes	Beam pattern threshold (dB)	-6
"	"	Beam full angle	6.7°
"	"	Single target filters	0.5-1.5 @ -6 dB
"	Fish tracking for whole- lake surveys [5]	method	4 dimensional
"	"	Minimum # echoes/fish track	1
"	"	Max ping gap/track	2
"	"	Max range change/echo	0.2 m
"	"	Alpha: major, minor, range	1, 1, 3
"	"	Beta: major, minor, range	1, 1, 3
"	"	Exclusion: major, minor, range	4, 4, 0.2
"	"	Expansion %:major, minor, range	50, 50, 50
"	Fish tracking for DVM surveys	method	2 dimensional
"	"	Minimum # echoes/fish track	1
"	"	Max ping gap/track	3
"	"	Max range change/echo	0.2 m

[1] Biosonics DT-X split-beam.
[2] range from transducer.
[3] WAAS differential GPS.
[4] Processing threshold after application of calibration offset.
[5] Same tracking parameters used by DFO for processing earlier year's data from these lakes

3.3.3 Data processing and analysis

Echo integration scaled by in-situ split-beam target strength (TS, the acoustic size of fish) measurements of tracked fish was used to estimate fish abundance from the whole-lake acoustic surveys because overlapping echoes (multiple targets) were numerous in high density Sockeye fry layers at night. Echoview© software (v4.8) was used for all acoustic processing. Key data collection and processing settings used in 2014 were the same as those used in the past by DFO (e.g., collection threshold of -100 dB, pulse width 0.4 ms, and processing threshold -65 dB; see Table 4 for additional settings). A -65 dB processing threshold is low enough to detect Sockeye fry but high enough to exclude system noise and unwanted targets such as plankton. Passive data were collected during surveys to describe background noise levels (Parker-Stetter et al. 2009). In-situ field calibration test results were all within 1 dB of the -39.5 dB expected value, so no calibration correction was necessary during data processing in Echoview.

Acoustic data from each transect were analyzed separately. Depth intervals for acoustic data analysis were 2 m thick; i.e., 0-2 m, 2-4 m, 4-6 m, and so forth to a maximum depth of 80 m. Acoustic data were also divided horizontally into 100 m long intervals to form a grid of cells encompassing the length and depth of each transect. The primary analysis outputs from Echoview (mean volume backscattering strength [Sv] of the detected targets, the TS of single targets and tracked fish, and counts of tracked targets) were summarized within each 2 x 100 m spatial cell and then within each transect to produce total fish density (fish/m^3) and mean TS estimates. Fish density within a cell was calculated as mean Sv / mean TS using data from that cell.

Total fish abundance of each layer-transect cell was calculated by multiplying its mean fish density by its volume. The volume of a cell (obtained from DFO records) was its surface area at mid-depth multiplied by a stratum thickness of 2 m (MacLellan and Hume 2010). Total fish abundance estimates were apportioned among three size-groups of fish using stratum specific TS data TS from tracked targets (small fish: TS > -65 dB and ≤ -45 dB; medium fish: TS > -45 dB and ≤ -34 dB; large fish: TS > -34 dB). Size-groups breakpoints were chosen based on examination of TS frequency distributions and comparison with length frequency distributions from trawling and gill netting converted to dB via Love's (1977) dorsal aspect model of TS versus fish length:

$$\text{Length (mm)} = 10 \cdot 10^{(TS + 0.9 \log (kHz) + 62)/19.1}, \text{ where}$$

TS = target strength (dB)

kHz = echo sounder operating frequency in kilohertz = 206 kHz

According to this model the ranges of fish lengths represented by these size groups were 9-100 mm for small fish, 101-376 mm for medium fish, and > 376 mm for large fish. The -45 dB breakpoint between small and larger fish is the same as was used by DFO in earlier studies of these lakes, in which small fish were considered to represent age-0 Sockeye and

Gwenish, while larger fish included both older Gwenish (age 1-4) and large pelagic piscivores (DFO only used two size-groups, MacLellan and Hume 2010). In the 2014 study, the medium size class mainly represented age 1-4 Gwenish plus some smaller individuals of the pelagic piscivores found in the lakes, while the large size class represented pelagic piscivores only (see Results section for additional information).

For modeling fish detectability versus acoustic system noise and depth in the water column we estimated TS from fork length used Love's (1977) dorsal aspect equation (above) with depth compensation according to Bolye's Law (Mukai and Iida1996):

$$TS = 19.1 \log (L/10) – 0.9 \log (kHz) – 62 – (6.67 \log (1 + (d/10))), \text{ where}$$

TS = target strength (dB)

L = fish length in mm

kHz = echo sounder operating frequency in kilohertz = 206 kHz

d = fish depth in the water column in m

Species composition of the trawl and catch was used to apportion abundance estimates of the small size class. The proportion of each species in the trawl catch (fish ≤ 100 mm only) was applied to the acoustic estimate of small fish to derive an estimate of juvenile Sockeye and other small fish in each spatial cell. The various estimates for each cell were summed to provide an abundance estimate of juvenile *O. nerka* and other fish species for each transect. These estimates were divided by the surface area represented by each transect to estimate fish densities (fish/ha) for each transect. We did not apportion acoustic estimates of the medium size class (101-376 mm in length) or large size class (> 376 mm) because gill net catches were insufficient for reliable estimation of their species composition, especially for large fish (see gill netting and trawling results).

Following DFO 2000-2003 analysis methods, and based on the stratified systematic design, fish density results from each transect in a lake section were averaged to provide an estimate of density relative to surface area (fish/ha) for the section for each category of fish described above. The mean density was multiplied by the surface area of the section to provide an abundance estimate for the section. The section abundance estimates were summed to provide a total abundance estimate for the lake. The mean whole-lake density was calculated by dividing the lake population estimate by the total surface area. The variance of each section estimate was calculated as the product of the variance of transect densities weighted by (i.e., multiplied by) the square of the section area. The sum of the weighted section variances was divided by the square of the whole lake area to provide a variance for the whole-lake population estimate. Following common practice, the 95% confidence intervals (95% CI) of the total abundance estimate was computed as for a stratified random sample (Cochran 1977).

Assessing statistical significance of comparisons of fish abundance among lakes and seasons was based on overlap of 95% confidence intervals of the stratified abundance estimates, using the rule that non-overlapping CI indicated a significant difference, whereas overlapping CI indicated that any observed difference may not be significant (Sokal and Rohlf 1981 p 248). Further statistical testing will be undertaken at a later date using a method (e.g., ANOVA or a nonparametric test) compatible with the study's stratified sampling design and the characteristics of the data (log-normal distribution, unequal sample sizes and variances among test groups).

3.3.4 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 and data collection settings (see Table 4) 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, Figure 5). 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 5). 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 (turbidity deceased from west to east), on transects SVM2 and SVM8 in the fall.

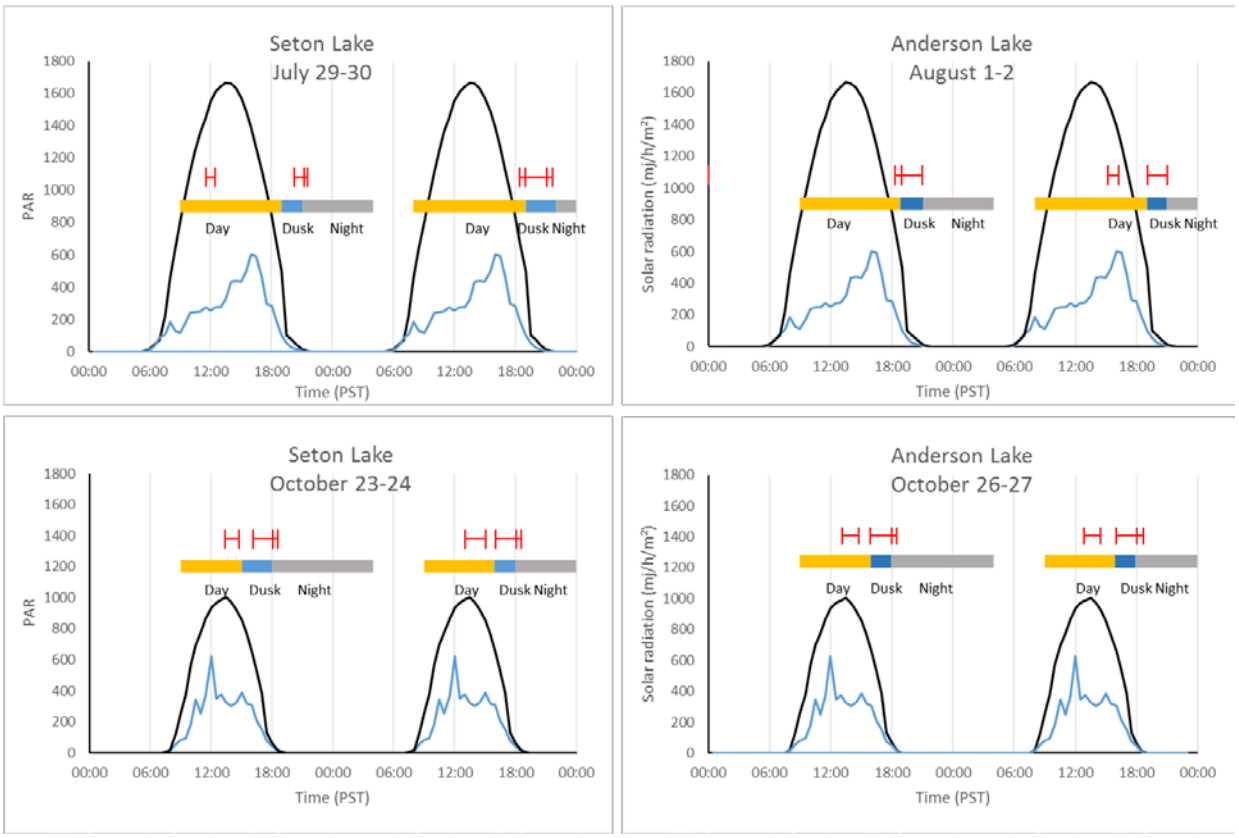


Figure 5. The DVM sampling schedule in 2014, showing the approximate maximum and minimum solar radiation on survey dates (black and blue lines), sampling windows for the three diel periods of interest (coloured bars for day, dusk, and night), and intervals when acoustic sampling actually took place (red lines). Actual light data for the study site at the time of acoustic sampling were not available, so solar radiation curves show light levels (PAR, µmoles/m²/s) on a clear day (black line) and a cloudy day (blue line) at Seton Portage from the same time period in 2015.

3.3.5 Data processing and analysis for fish

Acoustic data from DVM sampling was processed in Echoview by automatically counting tracked fish on TS echograms (40 log R amplification) according to standard echo-trace counting methods (Thorne 1983, Simmonds and MacLennan 2005). Compared to echo integration, trace counting provided a more quantitative estimate of fish density by depth over the widely varied acoustic conditions that we encountered during the daily cycle of fish behavior (individual targets in deep water with schools shallower during the day, versus layers of targets in relatively shallow water at night). The main deciding factor was that during the day a large fraction of the fish in Anderson Lake descended too deep to be visible on Sv echograms used for echo integration. For trace counting, data files were processed in Echoview to extract fish traces, measure their TS, and define the acoustic beam angle for beam volume calculations. Fish traces were recognized on echograms by their shape, cohesiveness, TS, and number of

echoes. Occasional bubbles rising through the water column, recognized by their characteristic slope and tendency to form columns, were excluded from fish counts. Echoview processing methods and settings were the same for the DVM study as for the 2014 whole-lake surveys (see Table 4).

For DVM analysis, fish densities (fish/m^3) were calculated for 5 m deep (2-5 m, 5-10 m, 10-15 m, and so forth to 100 m) by 1-minute long data cells. For each cell, fish density was computed as the total number of fish counted divided by the volume sampled, using the trace model (Keiser and Mulligan 1984) to correct the acoustic beam volume at shallow depths. Fish densities were calculated for the same fish size groups and TS breakpoints as were used for the 2014 whole-lake surveys. The primary statistic used for comparing depth distributions of fish among DVM periods and lakes was weighted mean fish depth (W) in the water column, computed for each fish size group for each minute of sampling as:

$$W_s = \Sigma(F_{si} \cdot D_i) / \Sigma F_{si}$$

for I = 1 to n, where:

W = weighted mean fish depth (meters)

F = fish density (fish/m^3)

D = layer midpoint depth (meters)

s = fish size group (small, medium, large)

i = depth layer number

The weighted mean depths of small fish from 2014 DVM sampling were fitted to Levy's (1990) daytime depth vs light attenuation coefficient model and Scheuerell and Schindler's (2003) anti-predation window model to test for agreement. Weighted mean depths of medium and large fish were examined graphically to describe the DVM behavior of these size groups under the conditions of the study.

3.3.6 Data processing and analysis for zooplankton

We analyzed echograms from the DVM study to estimate the vertical extent of zooplankton layers during day and night diel periods, data that were not provided by Wisconsin net sampling of zooplankton described in Section 2 of this report (limnology studies). Our method for this analysis was based on findings that zooplankton in BC Sockeye lakes can be quantified using a 420 kHz echo sounder (Morton and MacLellan 1992), the theoretical capabilities of a 206 kHz echo sounder such as we used in this study (Simmonds and MacLennan 2005), and our own observations that layers of acoustic targets appearing to be plankton (smaller than fish) are often visible on echograms from a 206 kHz sounder. In this study, Sv echograms (20 log R amplification) with a -87 dB flat threshold and a time varied threshold (TVT) of -120 dB @ 1m were visually spot-checked for "scattering layers" of small

acoustic targets in the water column that might be zooplankton. Within each station/survey combination, echograms from the three diel periods (day, dusk, night) were compared for diel vertical differences in these layers that would indicate a DVM by zooplankton. These comparisons among diel periods were used to help distinguish zooplankton from other marks on echograms from non-biological sources, such as system noise, false bottom, or glacial sediment that would not perform a DVM. Data were acquired by observing and recording the minimum and maximum depth of zooplankton layers at several times during each diel sampling period at each DVM station. We did not attempt to further quantify zooplankton distributions due to the presence of fish targets and noise from various sources (e.g., false bottom) in the zooplankton layer during some sampling periods.

3.3.7 Trawling and Gill Netting

3.3.7.1 Sampling

Pelagic fish were sampled with a midwater trawl and gill nets. At least one midwater trawl tow was made in each lake section (Figure 4) on each survey (e.g., Seton, summer) to apportion the corresponding acoustic estimate of small fish (< ~ 100 mm in length) by species and to collect biological samples from fish (e.g., length, weight, scales, DNA, diet). The purpose of trawl sampling was to catch a representative sample of the fish observed with acoustics (not to randomly sample the fish population), so trawling targeted depths and locations where acoustics found high densities of fish. Locations to fish were radioed to the trawl boat following acoustic sampling of an area.

All trawl sampling occurred during hours of darkness, conducted by a crew of three in an open, 8 m long welded aluminum work skiff equipped with a boom and hydraulic winches capable of towing the net to a maximum depth of about 47 m as operated on the surveys (Figure 6a). The net was an 18 m long beam trawl with a 3 m wide by 7 m deep mouth opening, constructed with a graded series of meshes, decreasing in size from the mouth (10.2 cm stretch mesh) to the codend of 6 mm stretched mesh knotless nylon (Figure 6b, Enzenhofer and Hume 1989). A 75 mm diameter plastic PVC tube with a threaded cap and perforated with 3 mm holes was used to collect fish from the codend. The mouth of the trawl was kept open by top and bottom spreader bars, with a 22.7 kg lead ball on each end of the bottom bar. The net was operated using a separate hydraulic winch and 4.8 mm (3/16”) wire tow cable for each bar. This system allowed the net to be opened and closed by varying the upper cable length, enabling sampling at discrete depths without contaminating the catch with fish from shallower depths (Enzenhofer and Hume 1989). Standard practice was to deploy and retrieve the net closed, only opening it while at the fishing depth. Prior to the surveys, the trawl system was calibrated with a depth recorder attached to the lower spreader bar to establish the relationship between net depth and cable length and angle at towing speed (0.6 m/s). Towing speed was determined from the GPS and engine revolutions per minute. During the survey the depth fished by each tow was initially estimated from tow cable length, then cross checked while under way using tow cable angle, and also measured with a depth recorder to determine the actual depth fished.

Gill netting was performed by the same boat and crew used for trawling. The boat hydraulics were used to operate a winch and capstan for deploying anchors in deep water (110-215 m). Gillnets were set in pelagic habitat to sample large fish including predators of juvenile *O. nerka*. All nets were standard RISC 91.2 x 2.4 m floating or sinking variable mesh gillnets (RISC 1997) consisting of 6 panels, each of a different mesh size (25, 89, 51, 76, 38, and 64 mm stretched mesh). These nets mainly catch salmonids > 100 mm in length, with 100-400 mm the optimal size range (RISC 1997). Nets were fished horizontally in midwater 15-40 m below the lake surface, targeting depths where fish > 100 mm long (TS > -45 dB) were most abundant on night time echograms. Nets were set in gangs of three per lake section (each gang fished three different depths), and each lake section was sampled during each survey (e.g., Seton, summer). All nets were set in late afternoon and pulled the next morning, usually on the night after the whole-lake acoustic survey and trawl sampling. A single “set” was defined as one net fished overnight.

A

B

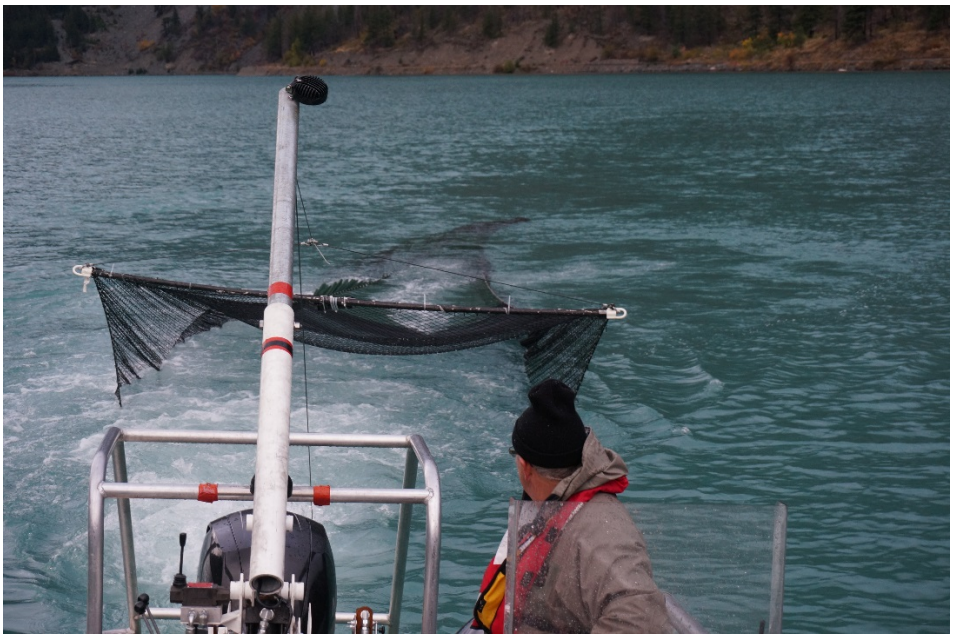


Figure 6. (A) The trawl boat and crew, showing the capstan used for gill netting on the forward davit and the trawl boom at the rear; (B) The trawl boat with the net partially above water during practice (actual sampling was only at night).

3.3.7.2 Sample processing and analysis

Initial processing of the trawl catch took place on the sampling boat. The catch of small fish from each tow was bottled and preserved right from the net to await further processing. Fish were anaesthetized with a lethal dose of clove oil to prevent regurgitation of stomach contents prior to preservation in either 10% formalin (for lengths measurements) or 85% ethanol (for DNA samples). Trawl caught fish too large for easy storage were identified to species, measured to the nearest mm in length (but not weighed), and released alive without anaesthetization. The bottled small fish were stored for ≥ 30 days before processing to allow body size to stabilize in the preservative (Shields and Carlson 1996). They were then identified to species, measured to the nearest mm, and weighed to the nearest 0.1 g. Smears of scales were removed from preferred body areas (along or near the lateral line immediately posterior to the dorsal fin) of all measured fish and stored in plastic paper sleeves in labeled envelopes for later aging by reading annuli under magnification. A fin clip from each *O. nerka* specimen was stored in an individual vial of 85% ethanol for DNA analysis for subsequent DNA analysis to determine stock origin. Stomach samples were excised from a sample of fish of each species and preserved in 70% isopropyl alcohol for later examination. Scale and stomach samples from *O. nerka* fry were chosen to cover the complete size range collected in the trawl nets. Organisms from the stomachs were identified to the lowest reliable taxon (usually family) and counted.

All fish captured in gill nets were processed on shore on the day nets were retrieved to determine species, fork length to the nearest mm, weight to the nearest gram, stage of sexual maturity, and to obtain aging structures, stomachs, and tissue samples for DNA analysis. Aging structures were only taken from salmonids (scales from trout and Gwenish, otoliths from bull trout), tissue samples were only taken from *O. nerka*, and stomachs were taken from a subsample of fish representing the range of sizes in the gill net catch. Aging and stomach samples from gill netting were processed as for the trawl catch.

The stock and age structure of the *O. nerka* in the two lakes are complex and overlapping. We used length frequency histograms, ageing data from scales, and stock identification using DNA to attempt to separate *O. nerka* into five groups (age-0, age-1, and ages 2-4 Gwenish; age-0 Gates Creek Sockeye Salmon and age-0 Portage Creek Sockeye Salmon). Our main interest was in comparing the growth of the two Sockeye Salmon stocks in the two lakes, and due to the difficulty in separating the taxa, we confined our analysis of growth to these two stocks and only used specimens identified with DNA for best accuracy. Fish size data was used to calculate mean (and 95% CI) length and weight. Growth from summer to fall (the difference between summer and fall survey means) was also calculated.

3.3.7.3 Stock identification from DNA

Tissue samples (fin clips) for DNA analysis were taken from a sample of small and large *O. nerka* caught in the trawl and gillnets in 2014. Samples from the trawls were selected to represent all lake sections and size classes. Samples were stored individually in glass vials filled with non-denatured ethanol until analysis. Adult Sockeye for baseline data were collected from the spawning grounds on Portage and Gates Creeks by DFO field crews (Withler et al

2000), and *O. nerka* >120 mm from gill netting were a priori assumed to be Gwenish (putative Gwenish) and used for the Gwenish baseline data.

All fish included in the genetic analysis were genotyped using the polymerase chain reaction (PCR) for DNA amplification at 14 microsatellite loci. These were Ots2, Ots3 (Banks et al. 1999); Ots100, Ots103, Ots107, and Ots108 (Beacham et al. 1998; Nelson and Beacham 1999); Oki1 (two loci), Oki6, Oki10,Oki16, and Oki29 (Smith et al. 1998 and unpublished); One8 (Scribner et al. 1996); and Omy77 (Morris et al. 1996). Alleles (amplified DNA fragments) were size fractionated on denaturing polyacrylamide gels, and allele sizes were determined with the ABI 3730 automated DNA sequencer. Baseline samples of adult Sockeye Salmon were sampled between 1986 and 2000 in each of Portage (N=119) and Gates (N=138) creeks, and juvenile fish were sampled in the summer and fall 2014 from Anderson (summer = 206, fall = 220) and Seton lakes (summer = 200, fall = 190). The identification of juvenile fish from Seton and Anderson lakes was carried out with the STRUCTURE program (V2.3.3, Pritchard et al. 2000). In both the summer and fall samples, fish from both lakes considered too large to be juvenile Sockeye (> 120 mm) were tentatively identified as Gwenish prior to genetic analysis and included in the analysis to test their classification and potentially increase the number of Gwenish for genetic identification. Posterior probabilities for the existence of up to five Sockeye/Gwenish subpopulations were evaluated with independent runs using the STRUCTURE program with K = 1–5. All runs were performed with a burn-in of 100,000 cycles and 10 iterations of 250,000 MCMC repetitions with sample type/location/time (eight combinations) used as prior information. Allele frequencies were assumed to be correlated and admixture was allowed.

The eight samples included in the STRUCTURE analysis consisted of 1073 genotypes from 1) baseline Gate Sockeye 2) baseline Portage Creek Sockeye 3) Seton Lake summer and fall putative Gwenish 4) Anderson Lake summer and fall putative Gwenish 5) Anderson Lake summer mixed Sockeye and Gwenish 6) Seton Lake summer mixed Sockeye and Gwenish 7) Anderson Lake fall mixed Sockeye and Gwenish and 8) Seton Lake fall mixed Sockeye and Gwenish. Individual juveniles were identified with probabilities to life history type (Sockeye or Gwenish) and, for Sockeye, river of origin (Gates or Portage creeks).

3.3.8 Supplementary environmental data collection

During the seasonal fish surveys in 2014 we collected turbidity, temperature, and irradiance (light) profiles of the water column at or near the DVM stations from the gill net or acoustics boat. We used a YSI sonde or Seabird CTD to measure turbidity and temperature from the lake surface to a depth of 60 m (YSI) or the lake bottom (100-200 m, Seabird). We determined irradiance profiles by measuring light at 1 m depth intervals down to a maximum depth of 50 m using a Li-Cor irradiance meter equipped with a PAR sensor. The visibility of a Secchi disk was also determined using a weighted white 20cm disk. We used these data and those from monthly limnological surveys (Question 2 studies) to help interpret differences in Sockeye behaviour and growth between the lakes.

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 collected and analysed in 2015 were added to that from 2014 to contribute to answering the management questions by the end of the three years of study. Field and laboratory work from 2015 and the completed fish population and DVM analyses done in 2015 contributed 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) were analyzed in the first year of the project (Figure 7 and Figure 8). The three cores from Seton Lake (S1, S2 and, S3) were analyzed in 2015 (Figure 9). Locations from where the cores were taken are shown in Figure 1. In Anderson, the total ^{210}PB activity of the two cores followed an exponential decay that was modeled by a first-order exponential decay (r^2 ranged from 0.87 to 0.9). This response provided

evidence of undisturbed sediment cores that were suitable for modeling a depth-time relationship. 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 (Figure 7a). In that same core, the cesium peak was distinguishable and reached its maximum in measured samples at 8 cm (Figure 7a). 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 Seton, the total ^{210}Pb activity of the three cores followed an exponential decay that was modeled by a first-order exponential decay (r^2 ranged from 0.78 to 0.9). S3, the core located closest to the diversion discharge displayed a noisy distribution of the total ^{210}Pb activity, with the lowest r^2 value of 0.78. The interval at which the background (or supported) levels of ^{210}Pb (where total ^{210}Pb activity and total ^{214}Bi activity intersect) were reached is not as clear as for Anderson Lake, but is approximately situated around ~26cm, ~40cm and, ~38cm for cores S1, S2 and, S3, respectively. The cesium peak is distinguishable in all three cores, albeit of lower magnitude in S3, and reached its maximum in measured samples at 20cm, 24cm and, 38cm in cores S1, S2 and, S3, respectively. The cesium peak also occurs in the Anderson lake cores at depths of 8 and 9 cm, respectively, in A1 and A2.

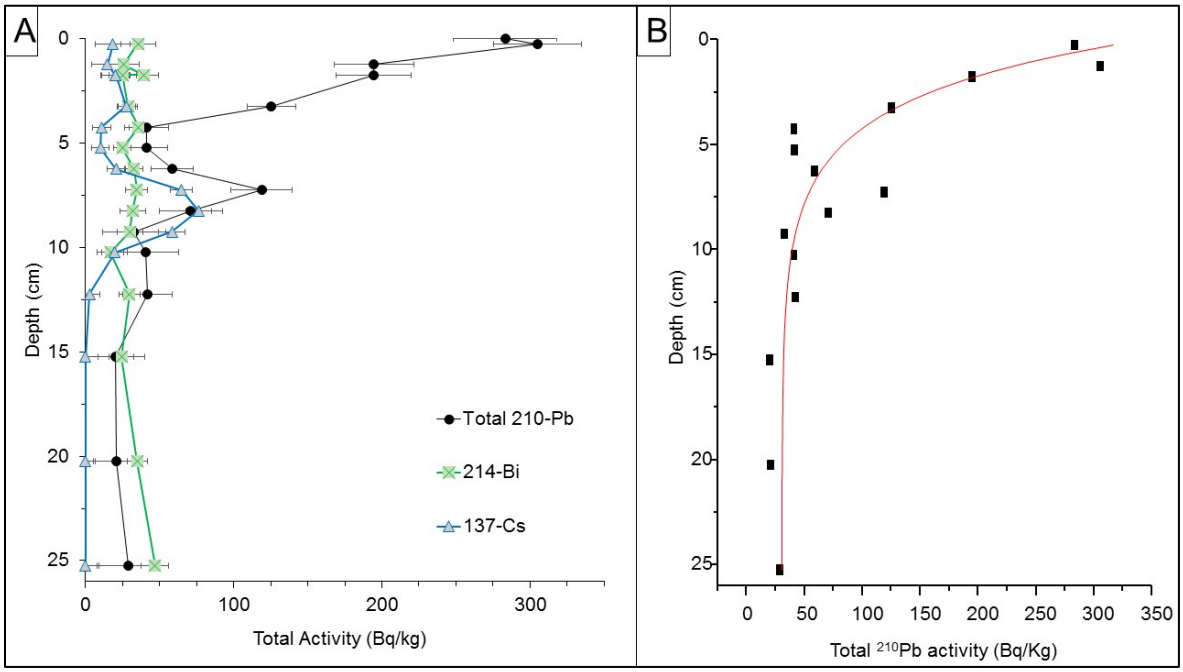


Figure 7. Measured activities of ^{210}Pb, ^{137}Cs and ^{214}Bi in Core AP1 from Anderson Lake (A) and exponential decay of ^{210}Pb activity with core depth (r^2= 0.87) (B).

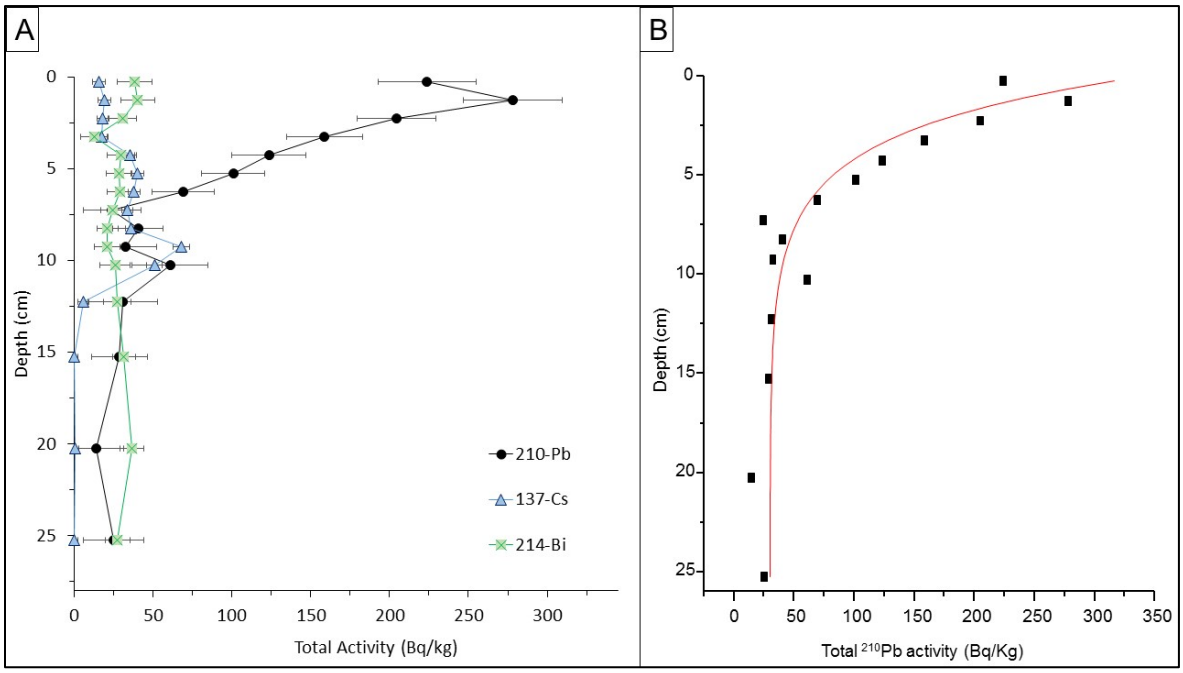


Figure 8. Measured activities of ^{210}Pb, ^{137}Cs and ^{214}Bi in Core AP2 from Anderson Lake (A) and exponential decay of ^{210}Pb activity with core depth (r^2= 0.90) (B).

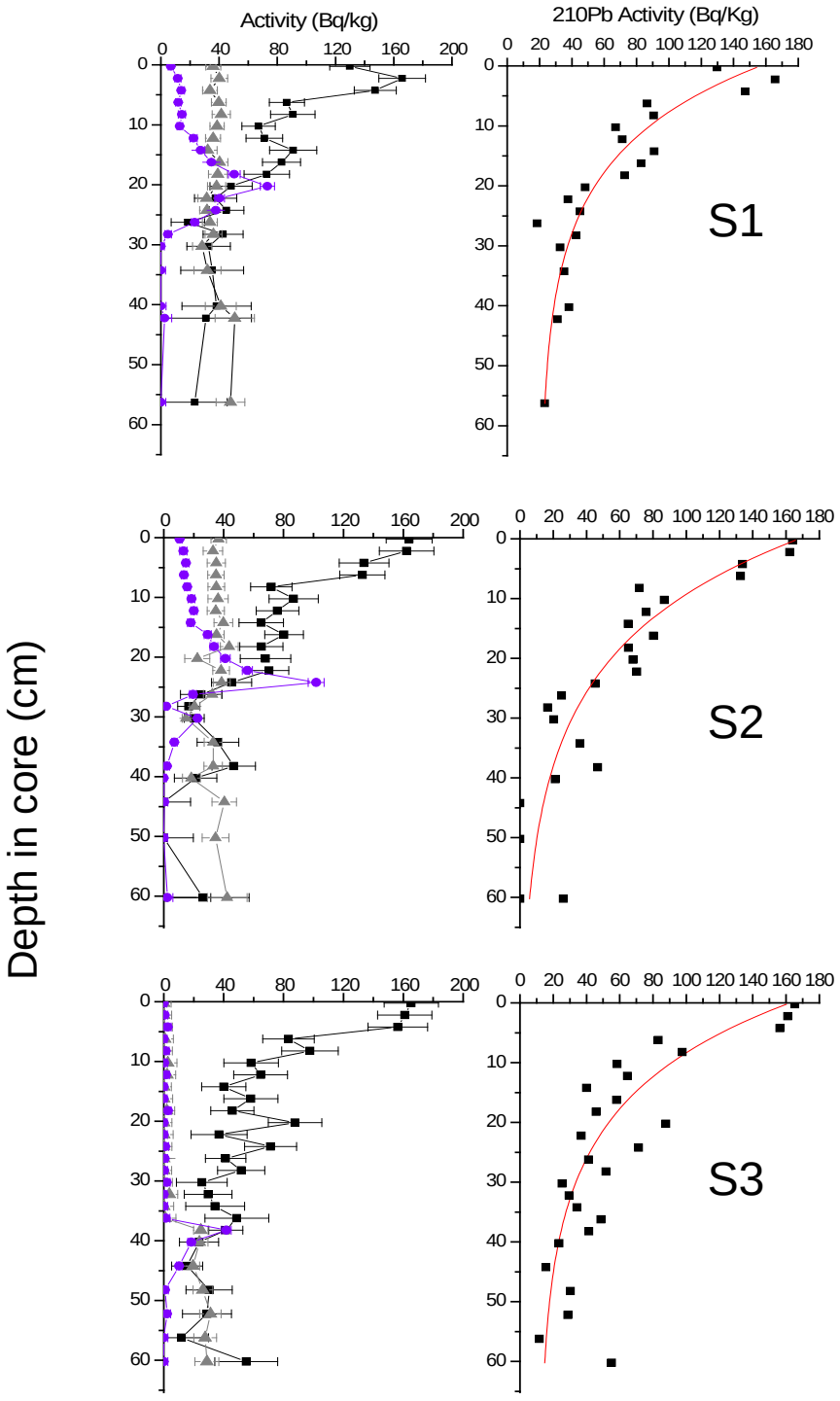


Figure 9. Measured activities of ^{210}Pb (black square), ^{137}Cs (purple circle) and ^{214}Bi (grey triangle) in Core S1, S2 and, S3 (top to bottom respectively) (left column). Exponential decay of ^{210}Pb activity with core depth (top to bottom: S1, percent variance explained by 1st-order exponential decay (shown in red) were high for all cores (S1, r2= 0.85, S2, r2= 0.90 and, S3, r2= 0.78) (right column).

Comparison of modelled age versus depth of all sediments cores indicates different sedimentation rates between the cores taken from Seton and Anderson lakes (Figure 10). The two cores from Anderson Lake display similar age-depth profiles with little variation in sedimentation rates. Cores from Seton Lake exhibit variability of the sedimentation rate both between and within the cores. Core S3 has a much higher sedimentation rate, reaching background deeper in the core compared to cores S1 and S2, which exhibit similar profiles. All three cores from Seton Lake indicated a higher sedimentation rate after the beginning of the 1930's which corresponded to ~35cm in core S3 and, ~20cm in core S1 and S2.

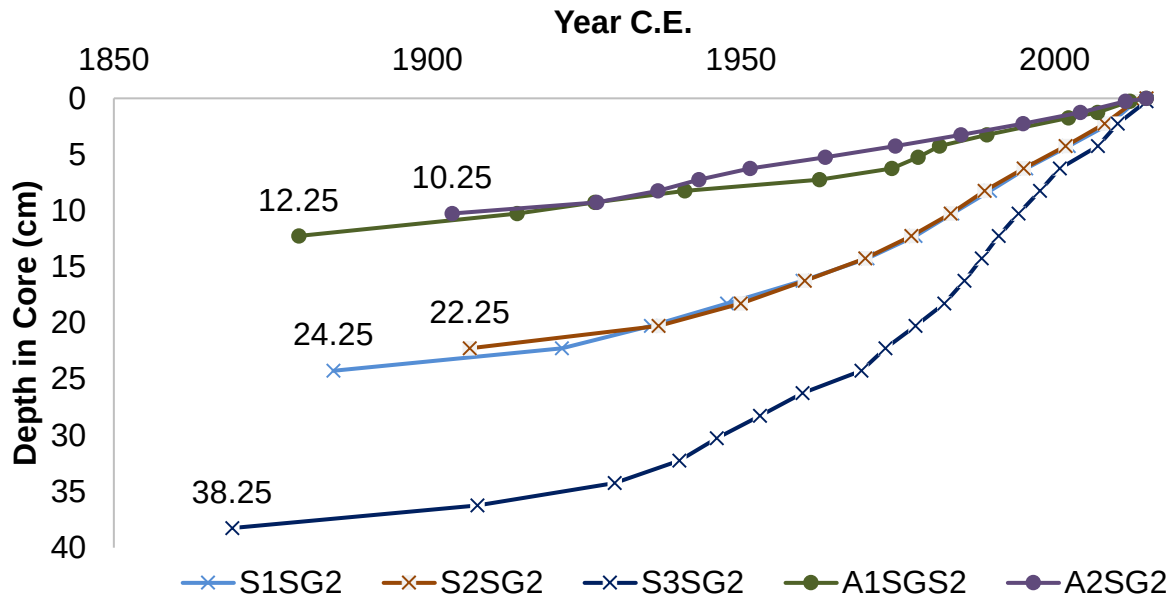


Figure 10. Year corresponding to core depth according to the Constant Rate of Supply (CRS) model for the three cores of Seton Lake and the two cores of Anderson Lake. The labels correspond to the interval where background or supported ^{210}Pb activities were reached.

The sediment cores from all lakes show the expected exponential decay of supported ^{210}Pb with core depth. However, in the Seton Lake cores, the unsupported ^{210}Pb activities are relatively low, and show substantial variation to the fitted exponential curve. This finding in combination with the presence of very distinct peaks in ^{137}Cs concentrations that are much older than the ^{210}Pb estimated dates of the CRS model, lead us to put more faith in using maximums of ^{137}Cs activities in the cores as the most accurate measures of age in the cores. However, the comparison of the sedimentation rates over time from the CRS model, does result in the same interpretation as using the ^{137}Cs maximums, in that sedimentation rates in the Seton Lake cores are greater than in the reference Anderson Lake cores, and that the core most proximal to the diversion had the highest sedimentation rate.

The presence of an undisturbed ^{137}Cs peak deeper in core S3 compared to S1 and S2, the lower magnitude of the ^{137}Cs peak in S3 likely as a result of dilution, and the year versus depth profiles is consistent with the higher sedimentation rate in core S3. The variability of the sedimentation rate within Seton Lake (differences between S1, S2 and, S3) is consistent with the input of riverine material, including fine-grained silt-sized particles after the establishment of the Bridge River Diversion. The difference in sedimentation rates of Seton Lake and Anderson Lake is likely associated with the increase in sedimentation rates in Seton Lake as a result of the diversion. The best estimate of sedimentation rates for Seton lake cores prior to the diversion will be likely best based on the sedimentation rates based on the cores from Anderson Lake.

4.2.2 Loss-on Ignition

All cores have low organic composition with an average fluctuating between 4% and 8% (Figure 11). The percent organic matter in the Anderson Lake cores fluctuate with slightly higher amplitudes compared to the Seton Lake cores. The cores from Anderson Lake display a general increase of the percent organic matter around 5cm, while the percent organic matter in the Seton Lake cores slightly decrease, with the exception of increases in S3 and smaller increases in S1 in the uppermost samples. Although the organic percent of cores S1 and S2 are relatively low, they display a decrease of 50% the percent organic matter (from an average of ~7% to ~4%) around 40 and 55cm respectively, which stabilized around 30 and 35cm respectively. The carbonate in all cores was constant throughout the entire core.

The slower sedimentation rates in the cores from Anderson Lake in comparison to the cores from Seton Lake indicate that the Bridge River Diversion did result in changing the sedimentation rate in Seton Lake. This finding is further supported in sedimentation rates of Seton cores that varies with distance from the diversion. The higher organic matter found in the top of the cores from Anderson Lake is likely the effect of diagenesis processes with the degradation of the organic matter deeper in the core. These recent increases in organic matter, observed in Anderson Lake, are not apparent in the cores from Seton Lake, likely as a result of continued inputs of inorganic material from the diversion. This dilution effect is further support by the fact that the organic matter in Seton Lake tends to decrease in the top half of cores S1 and S2.

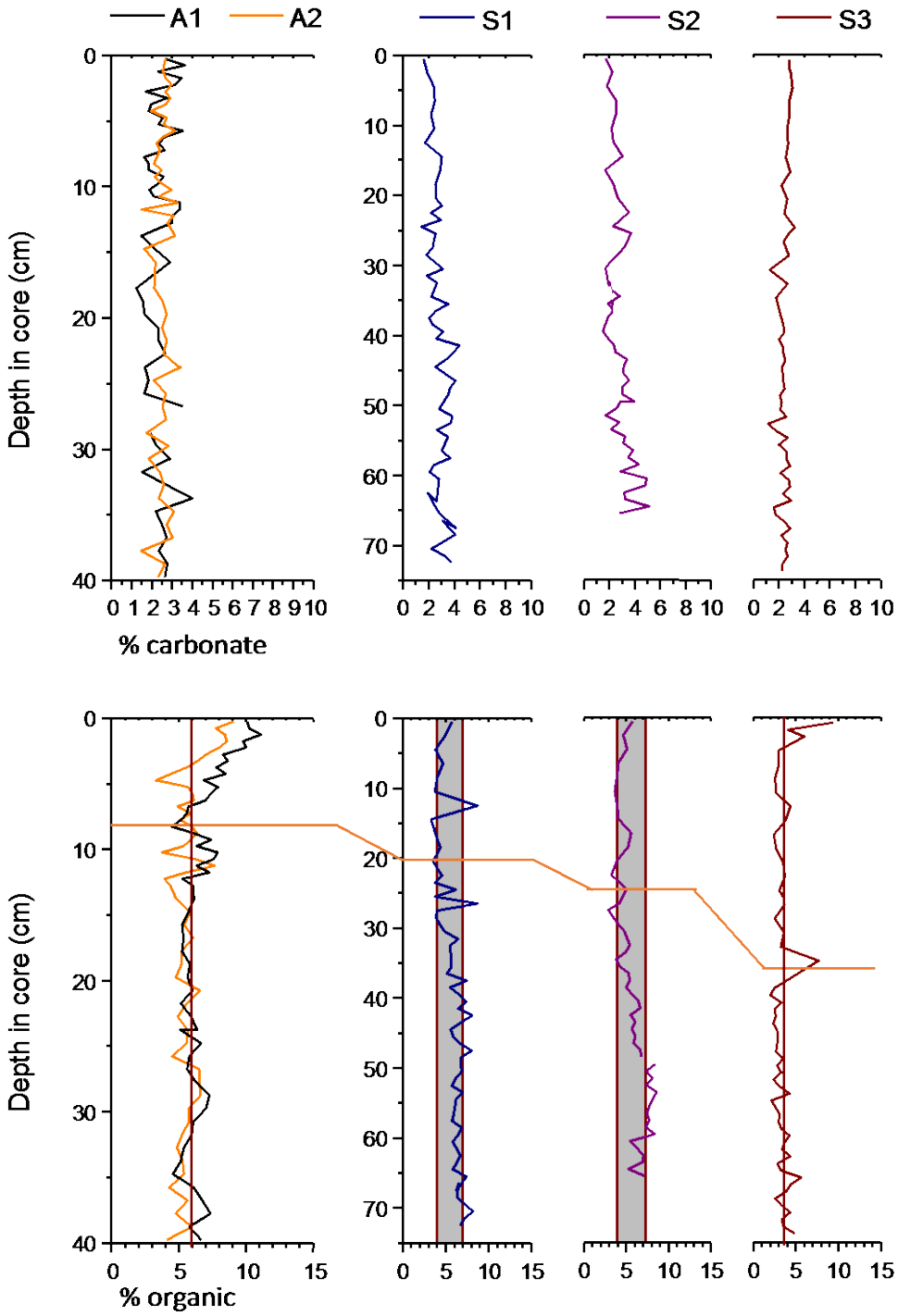


Figure 11 Changes in the percent organic matter in the cores from Anderson Lake (A1, A2) and Seton Lake (S1, S2, S3). The orange line represents the interval at which the ^{137}Cs peak reaches a maximum in each core (~1963), thereby serving as a temporal horizon across the cores. The red lines on the bottom graphs represents averages of percent organic for each cores, the grey box represents the difference of average between the bottom and top of cores S1 and S2.

4.2.3 Pigment Analysis

The ratio of chlorophyll-a to phaeophytin-a varied little throughout the cores, indicating that differential degradation of pigments was not an issue of concern for interpreting trends in pigment concentrations in the cores (i.e. higher variation of ratios of chlorophyll-a to phaeophytin-a would show incomplete degradation, and therefore issues related to degradation and/or preservation). The fluctuation of chlorophyll-a in the Anderson Lake cores (Figure 12), a pigment associated with overall primary production, varied between 50 and 100 ng/g throughout the cores, with some periods of higher concentration. Pigment profiles indicated that algal assemblages were mainly diatoms (indicated by fucoxanthin and diatoxanthin), cryptophytes (indicated by alloxanthin) and, blue-green algae (indicated by Lutein-zeaxanthin, canthaxanthin and, echinone).

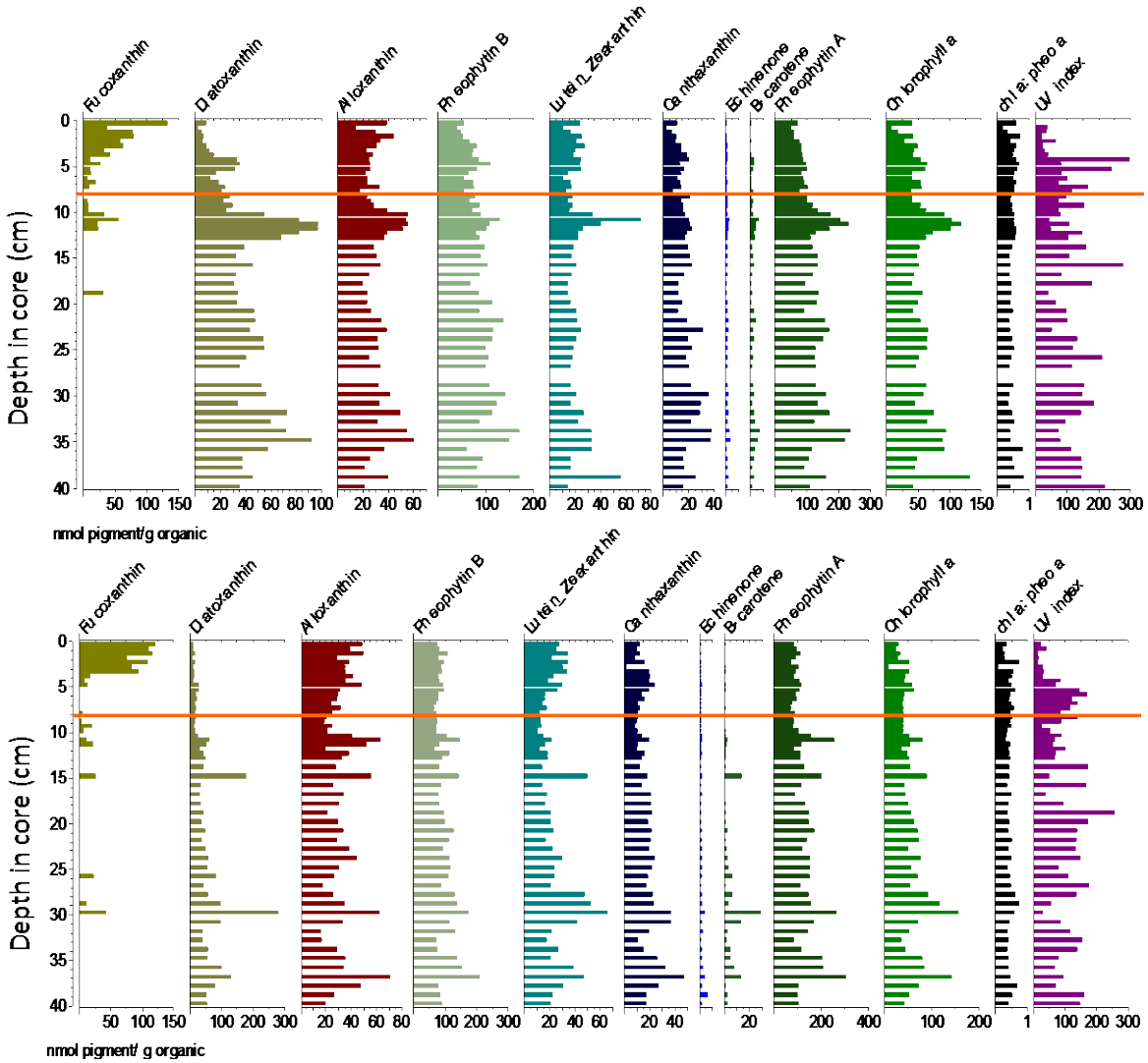


Figure 12 Pigment concentration (ng pigment/g organic) in Anderson Lake cores (top: A1, bottom: A2). Orange line indicates the interval at which ^{137}Cs reaches its maximum concentration. Refer to Table 3 for pigments and indicator details.

All three cores from Seton Lake exhibited an abrupt decrease of all pigment concentration at 40cm, 45cm and 48cm in S1, S2 and S3 respectively, while the chl(a):pheo(a) remained relatively constant throughout the cores. S2 is characterized by the highest pigment concentrations compared to the other cores from Seton Lake, whereas S3 presents the lowest concentrations (Figure 13). The dominant pigments present prior to the abrupt decline in concentrations was similar to the Anderson Lake cores, but concentrations in the Seton Lake cores were much reduced in comparison, typically < 10 ng/g.

The large decrease and timing of the changes in pigment concentrations in the cores from Seton Lake, in comparison to the cores in Anderson Lake, are consistent with a large change in algal production associated with the Bridge River Diversion. Smaller scale fluctuations are present in the cores from Anderson Lake, but these cores encompass a much greater period of time in comparison to the cores from Seton Lake. The abrupt increase of fucoxanthin in the Anderson Lake cores matches with the increase of *Cyclotella comensis.* Although fucoxanthin is generally associated with the overall diatoms biomass (Table 3, Leavitt and Hodgson 2001), the relationship between the increase of fucoxanthin and *Cyclotella comensis* in the top 5cm of the cores from Anderson Lake are likely driven by the regional changes, they are however not fully understood.

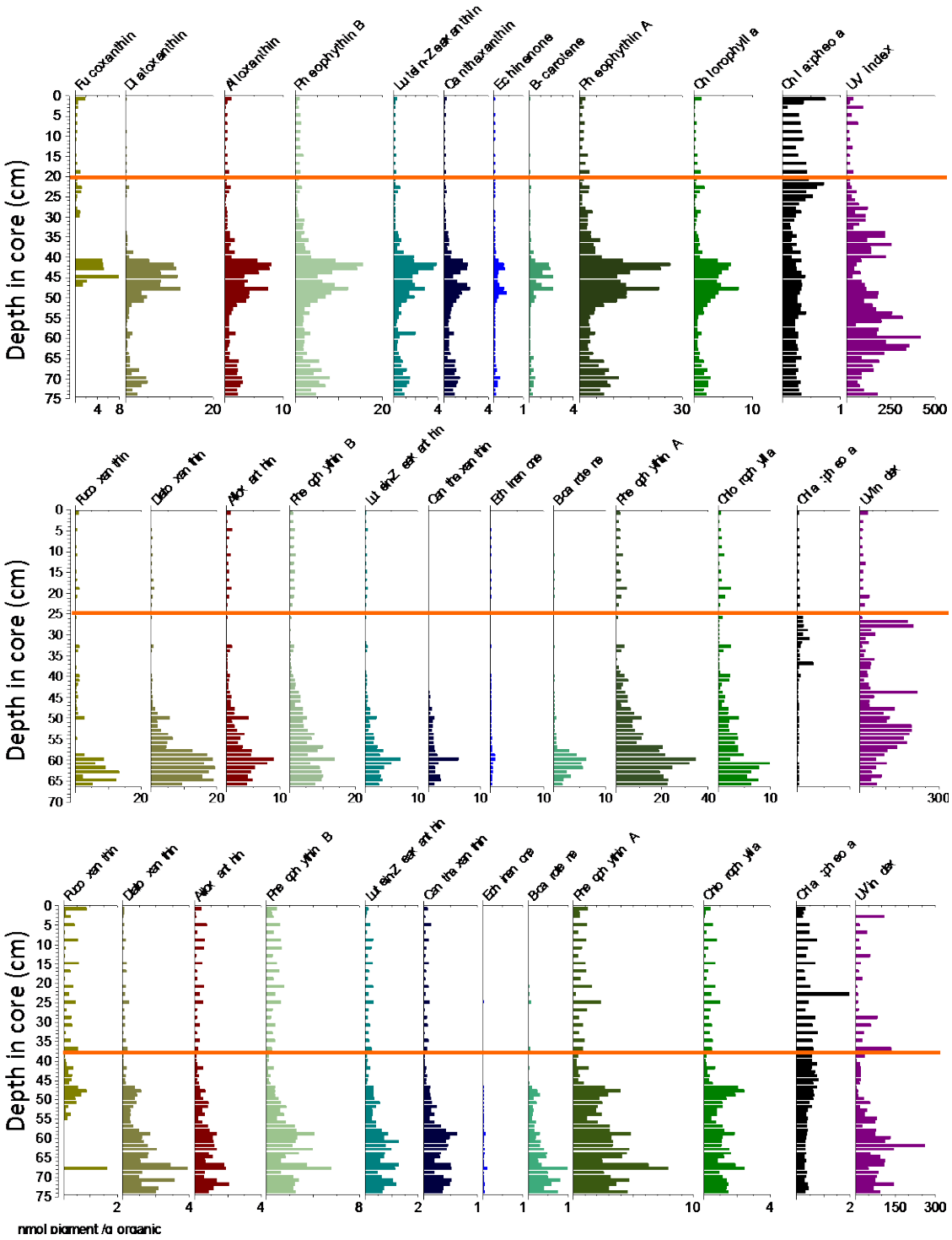


Figure 13 Pigment concentration (ng pigment/g organic) in Seton cores (top to bottom – further to closer from the diversion: S1, S2 and S3). Refer to Leavitt and Hodgson (2001) for pigments and indicator details. Orange line indicates the interval at which 137Cs reaches its maximum concentration.

4.2.4 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 14a) within the top 20cm of core depth. Dating of cores from SP1 and SP3 is not complete but given the known dating from SP2 (Limnotek 2015), 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 14b), 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.

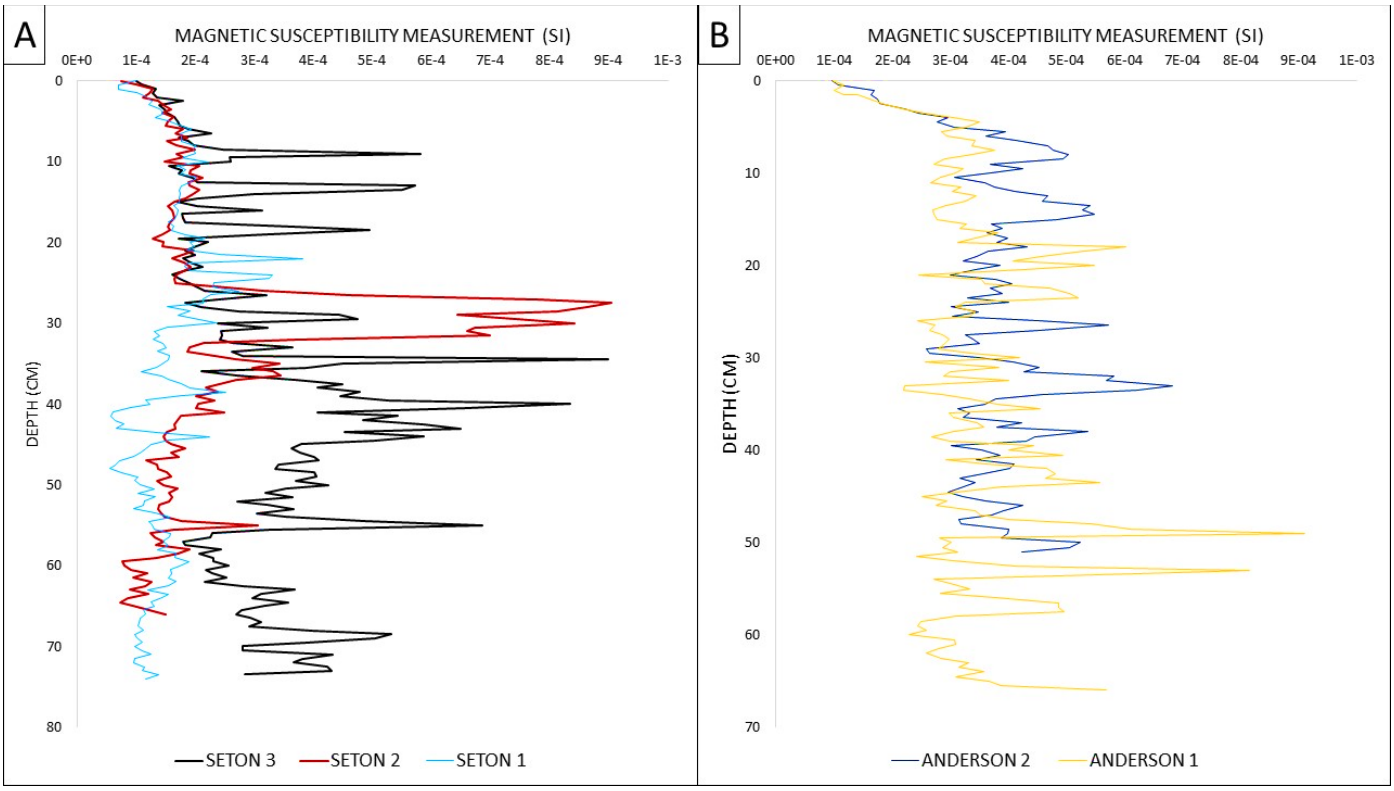


Figure 14. Comparison of the magnetic susceptibility in each core from (A) Seton Lake (cores SP1, SP2, and SP3), and (B) the cores from Anderson Lake (AP1and AP2).

4.2.5 Stable Isotopes

The two cores from Anderson Lake and core S3 from Seton Lake show lower amplitudes and variation of $\delta^{15}N$, $\delta^{13}C$ and C:N ratio compared to cores S1 and S2 from Seton Lake, with few exceptions (e.g., the C:N ratio in Core A1 from Anderson Lake has higher values at the bottom of the core, an older time period not present in the cores from Seton Lake), (Figure 15). The top 18cm of core S1 and 27cm of S2 were characterized by low variation in $\delta^{15}N$ values around an average of 2.5‰ compared to the bottom of the cores where the $\delta^{15}N$ values were enriched, fluctuating around an average of ~7.5‰ (bottom 22 and 13cm of cores S1 and S2, respectively). The $\delta^{15}N$ in the Anderson Lake cores (A1 and A2) show greater variation at the bottom 20cm of the core reaching a maximum value of ~4.5‰ while at the top 20cm, the $\delta^{15}N$ stabilized around an average of 2.5‰. Core S3 from Seton Lake showed low variability with an average of 2.5‰, similar to the top of cores from S1 and S2. Between 28 and 30 cm, core S2 was characterized by an abrupt increase of the δ13C and C:N ratio (Figure 15). In the Anderson Lake cores, $\delta^{13}C$ varied between -30 to -25 and C:N varied between ~11 and ~17. In Seton Lake, $\delta^{13}C$ varied between -30 to -23 and C:N was between ~7 and ~15.

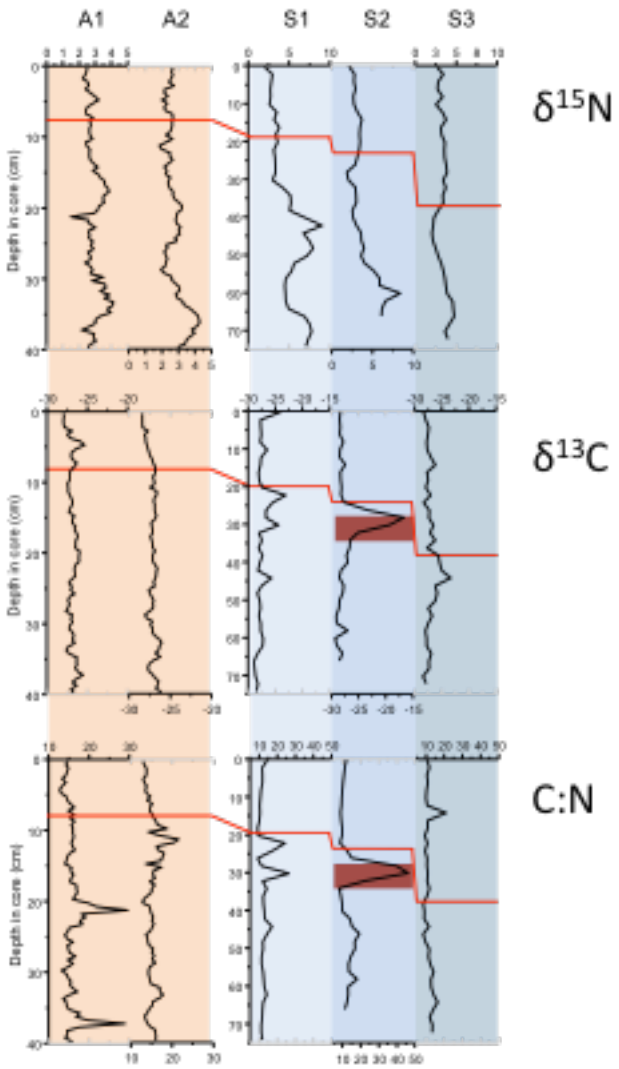


Figure 15 $\delta^{15}N$ (‰), $\delta^{13}C$ (‰) and C(atomic mass):N(atomic mass) ratio by depth in two cores from Anderson Lake (A1 and A2) and three cores from Seton Lake (S3, S2 and S1). The red line represents c.e1963 according to the maximum concentrations of Cesium. Each core has been plotted on the same scale for each measurement to facilitate the comparison between the cores.

Measurement of $\delta^{15}N$ in sediment cores can be a useful technique to track salmon population dynamics in spawning lakes (Gregory-Eaves et al. 2009; Selbie et al. 2009). Considering that all salmon have an average $\delta^{15}N$ of ~11‰ (Satterfied and Finney 2002), the relatively low $\delta^{15}N$ values found in Seton Lake sediment prior to the diversion and Anderson Lake sediments likely reflects low SDN. The relatively low values of $\delta^{13}C$ and C:N in all cores suggest primarily autochthonous contributions of organic matter from the lake as well as the catchment (Meyer et al. 1993). The decrease in $\delta^{15}N$ in cores S2 and S1 and the relatively low values of $\delta^{15}N$ could be due to changes in sedimentation rates, water residence times associated with the diversion, and/or decreases in abundance of salmon.

4.2.6 Cladoceran analysis

In both lakes, *Daphnia longispina complex* and *Bosmina sp.* were the dominant cladoceran species throughout the cores. Alona and Chydorus species were found in relatively low abundance (<2% in all samples) and were not further considered.

Both cores from Anderson Lake were characterized by an increase of *Daphnia longispina* complex relative to *Bosmina sp.* above a core depth of 17cm in A1 and 14cm in A2. Cladoceran concentrations fluctuated greatly between ~1000 and ~8000 individuals/g dry sediment, in both cores. A2 was marked by an increase in total concentration of animals (up to ~10,000 individuals/g dry sediment), mainly due to increased concentrations of *Daphnia sp.* in the top 8cm of the core (Figure 16).

Analysis of cladoceran remains for core S3 in Seton Lake was completed during Year 1 of the project (Figure 16), and analysis of core S2 was completed in 2015. Analysis of core S1 is nearing completion. Core S3 was characterized by a stable and marked dominance of *Bosmina sp.* which decreased in relative abundance between 35 cm and 50 cm in the core compared to relative abundance at depths greater than 50cm. In the top 50cm, the dominance between *Bosmina sp.* and *Daphnia longispina complex* varied, which may in part be exacerbated by the low counts of cladoceran remains. An increase of *Daphnia* ephippia, the resting stage of *Daphnia*, was observed at 38cm (200 individuals/g dry sediment) while they were not present or present in low abundances (<35 individuals/g dry sediment; found only in 4 other intervals) in the rest of the core. Cladoceran concentrations were low throughout the core (<100 individuals/g dry sediment) with the exception of the bottom 23cm where the concentration was relatively high. In core S2, *Bosmina sp.* relative abundance was constant and it dominated along the entire core. *Daphnia longispina complex* was not found at 10 and 14cm intervals. The overall concentration of cladoceran sub-fossils in S2 was low at the top of the core (<400 individuals/ g dry sediment) down to 40cm. Deeper in the core the concentration was higher, reaching 4000 individuals/g dry sediment.

The increase in the concentration of cladoceran remains at the top of Core A2 from Anderson Lake could be due to an increase in the productivity of the lake or a decrease of the grazing pressure by juvenile salmon. Further analysis in year 3 of the project needs to be done to fully understand this trend.

The recent decrease in concentration of cladoceran remains in the S2 core from Seton Lake (top 40 cm of the core) may be due to a temporal decrease in secondary production and/or changes in the sedimentation rate in Seton Lake affecting zooplankton abundance. According to the ^{137}Cs profiles the earliest drop in zooplankton fossil concentration corresponds with the beginning of the diversion in the mid-1930's. The high and relatively constant concentrations of cladoceran remains throughout the entire cores from Anderson Lake, stands in contrast to the much lower concentrations see in the cores from Seton Lake. Such changes are consistent with both higher sedimentation rates and potentially lower overall secondary production in Seton Lake at S2 after the diversion. This same trend was not observed at S3. Consequently, analysis

of S1 data needs to be completed in 2016 before conclusions about causal processes can be made.

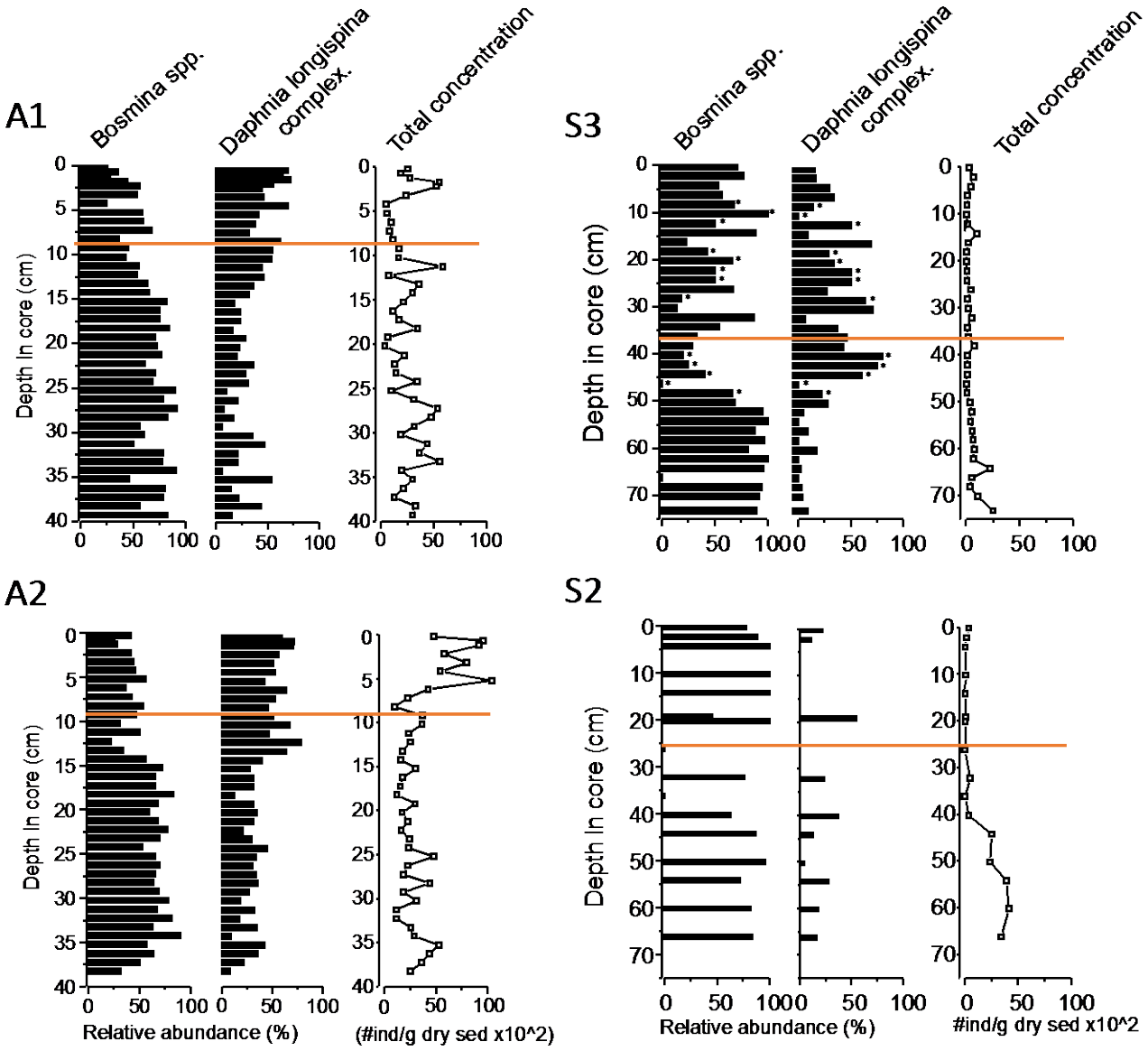


Figure 16. Relative abundance (%) of the dominant cladocera taxa and cladocera concentration (#individuals/g dry sediment x10^2, open squares) for cores A1 and A2 (left column) from Anderson Lake, and S3 and S2 from Seton Lake. The orange line represents 1963 according to the ^{137}Cs radioisotope independent dating. The stars on S3 stratigraphy correspond to the intervals where the minimum count was not reached due to low concentrations (<10 individuals counted).

4.2.7 Diatoms analysis

In the following discussion, various diatom species are linked to trophic state. This association is based on what is known as the TP model developed by Cumming et al. (2015) that was applied to the Seton and Anderson diatom fossil assemblages to describe trophic condition based on the diatom assemblages.

Diatom assemblages in the Anderson Lake cores were generally stable with small fluctuations in oligotrophic (i.e. *Cyclotella ocellata* , *Cyclotella stelligera*) and eutrophic planktonic taxa (i.e. *Aulacoseira subarctica*, *Stephanodiscus minutulus*) (Figure 17). In the recent sediments (Zone A), a striking increase in the oligotrophic planktonic *Cyclotella comensis* (includes small amounts of *Cyclotella gordonensis*, a similar taxa) were observed. Pennate planktonic taxa, typically more mesotrophic (i.e. *Asterionella formosa*, *Fragilaria crotonensis*) also increase slightly in this zone, and *Cyclotella ocellata* and *Stephanodiscus minutulus* decline. Diatom concentrations remain relatively stable, with a small increase in Anderson A1 in the top sediments (upper portion Zone A).

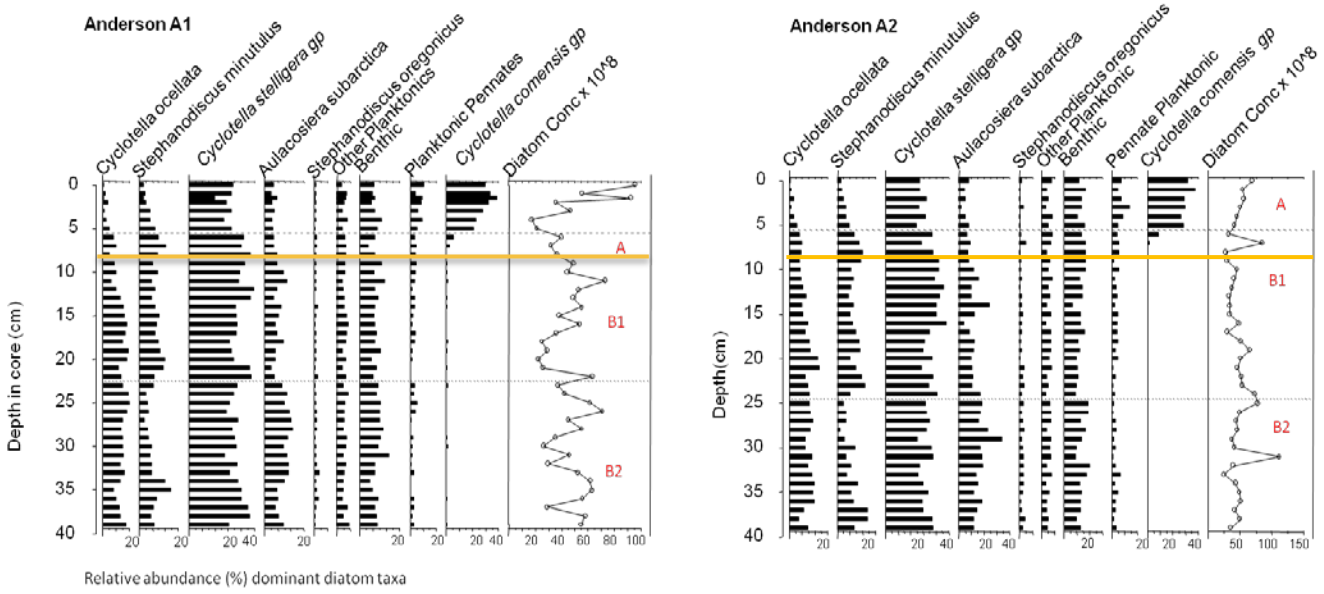


Figure 17 Relative abundance (%) of the dominant diatom taxa and total diatom concentration (#valves/g dry sediment x10^8, open circles) for Anderson Lake cores. Zones (A, B1, B2) are based on a constrained cluster analysis of the dominant taxa.

Comparing across all of the Seton Lake cores (S3 is the closest to the diversion), there is a striking similarity of the diatom assemblage changes (Figure 18). For example, *Cyclotella ocellata* is most abundant in the bottom portions of all cores (Zone B2, and also B3 in Seton 1),

and *Cyclotella comensis* becomes more abundant in the topmost sediments of all cores (Zone A - similar to the increase seen in the Anderson Lake cores, but of lower magnitude). There is a distinct decrease in total diatom concentration in all cores, which corresponds to an increase in the percent abundance of the meso-eutrophic planktonic *Fragilaria* and *Asterionella formosa* in all cores, as well as an increase in total benthic taxa, which is most pronounced in Seton core 3.

The longer temporal length of Seton 1, indicates there have been distinct fluctuations in the eutrophic planktonic, *Aulacosiera subarctica,* and oligotrophic planktonic taxa such as *Cyclotella ocellata* and *Cyclotella stelligera* (Zone B3).

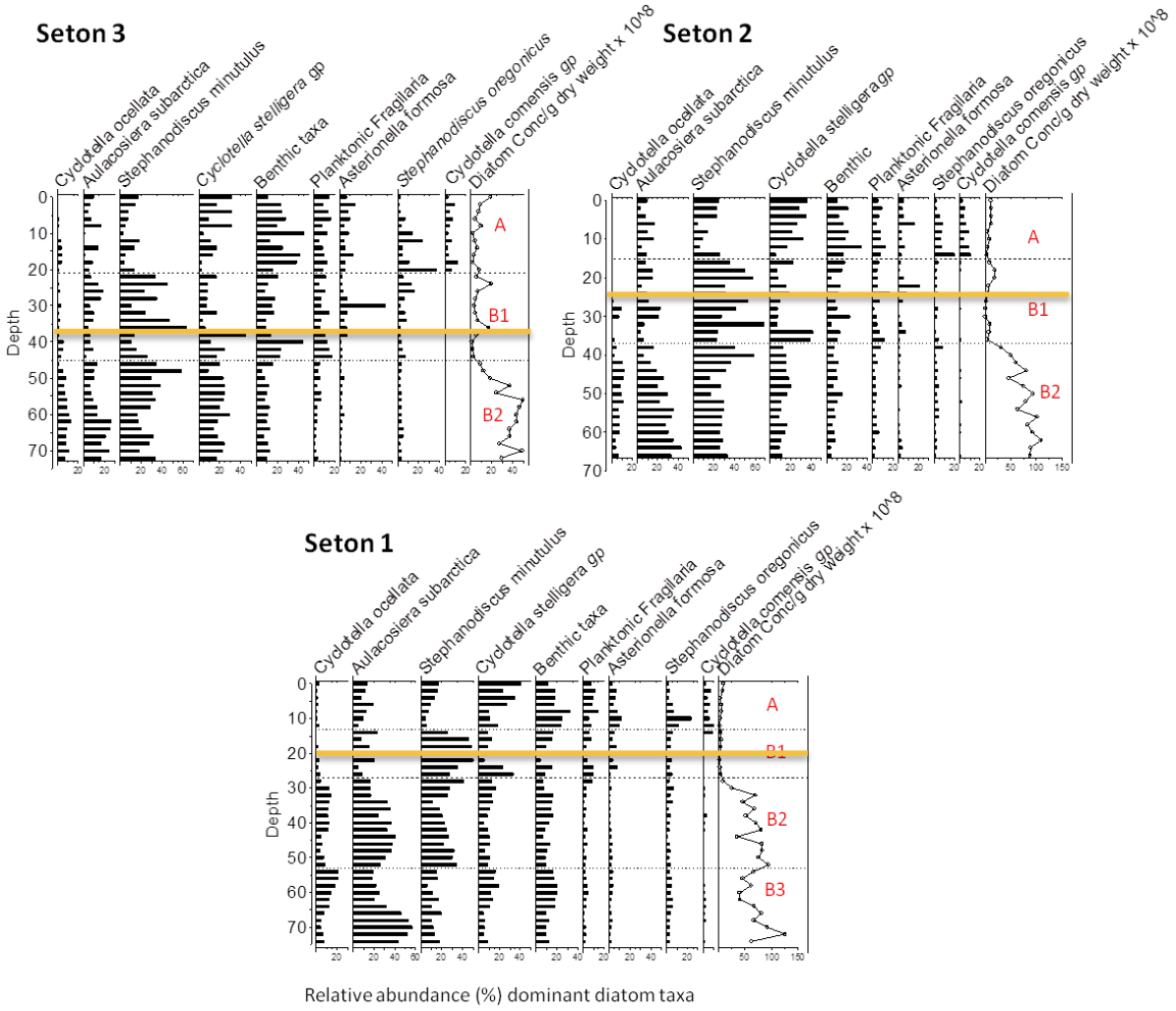


Figure 18 Relative abundance (%) of the dominant diatom taxa and total diatom concentration (#valves/g dry sediment x10^8, open circles) for Seton Lake cores. Zones (A, B1, B2 and B3 in Seton 1) are based on a constrained cluster analysis of the dominant taxa.

Diatom concentrations (Figure 19) are relatively stable in both cores of Anderson Lake, varying around a mean of 46.7 valves/g dry weight x 10^8 and 48.4 valves/g dry weight x 10^8 in cores A1 and A2 from Anderson Lake. In all Seton cores there is a rapid decline in diatom concentrations at depths of ~ 50 cm in S3, at ~ 37 cm in S2, and at ~ 30 cm in S1. These abrupt shifts are consistent with the dating and therefore represent the same event across all of the cores and thus illustrates the increasing sedimentation rate moving from S1 to S3 cores. Prior to this decline, diatom concentrations in cores Seton 1 and Seton 2 varied around means of 66.4 valves/g dry weight x 10^8 and 75.7 valves/g dry weight x 10^8, respectively (slightly higher than in the Anderson Lake cores). The mean of concentration in the S3 core from Seton Lake, prior to the decline, was lower at 35 valves/g dry weight x 10^8. The decrease of the total diatom concentrations in the Seton cores may indicate a decrease of the primary production in Seton Lake, and/or in addition be the result of increased sedimentation rates. Based on the chronology from the ^{137}Cs peaks in the Seton Lake cores, the decrease in diatom concentration is coincident with the beginning of the establishment of the diversion.

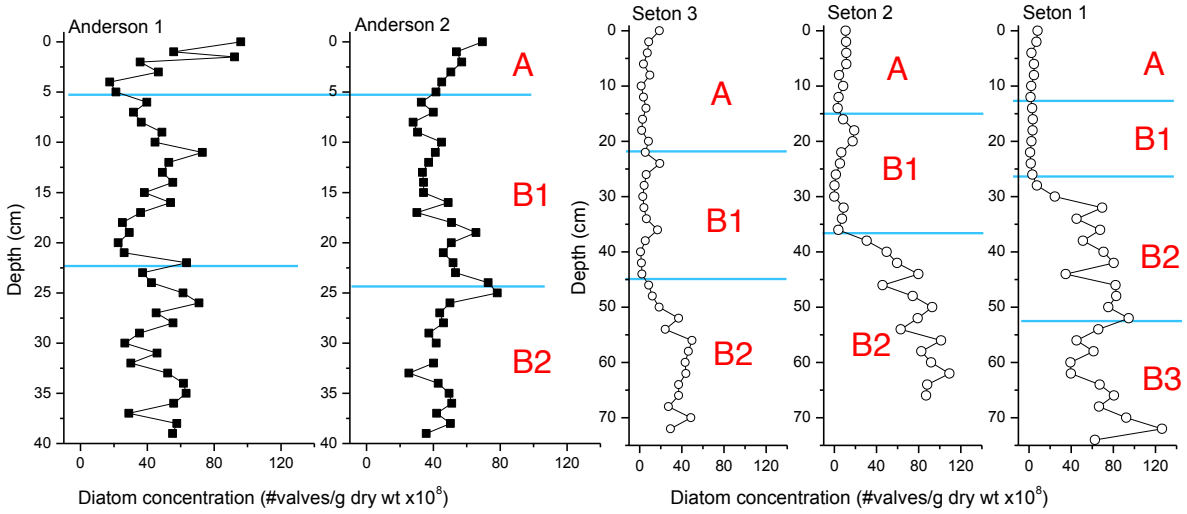


Figure 19 Total diatom concentration (#valves/g dry sediment x10^8, open circles) for Anderson and Seton lake cores. Blue lines correspond to the break down of the cores in different Zones (A, B1, B2 and B3 in Seton 1), based on the constrained cluster analysis of the dominant diatom taxa

The diatom assemblages of the Anderson cores were dominated by oligotrophic taxa in the genus *Cyclotella*, with *Cyclotella stelligera* (now *Discostella*) being the most common (Figure 20). In the Seton cores, diatom assemblages were dominated by more meso-eutrophic taxa, particularly *Stephanodiscus minutulus* and *Aulacoseira subarctica*. All cores from both Anderson and Seton lakes indicate an increase in the percentage of oligotrophic taxa in the top sediments being driven by the increase in *Cyclotella comensis*. This increase is likely being driven by a similar forcing factor, such as changes in climate that can influence physical factors (e.g. water temperature, mixing depth), which in turn could influence seasonal dynamics of the diatom taxa.

The decline in meso-eutrophic taxa in Seton Lake may also be linked to changes in nutrient conditions. This finding needs to be further explored in the 2016 data analyses.

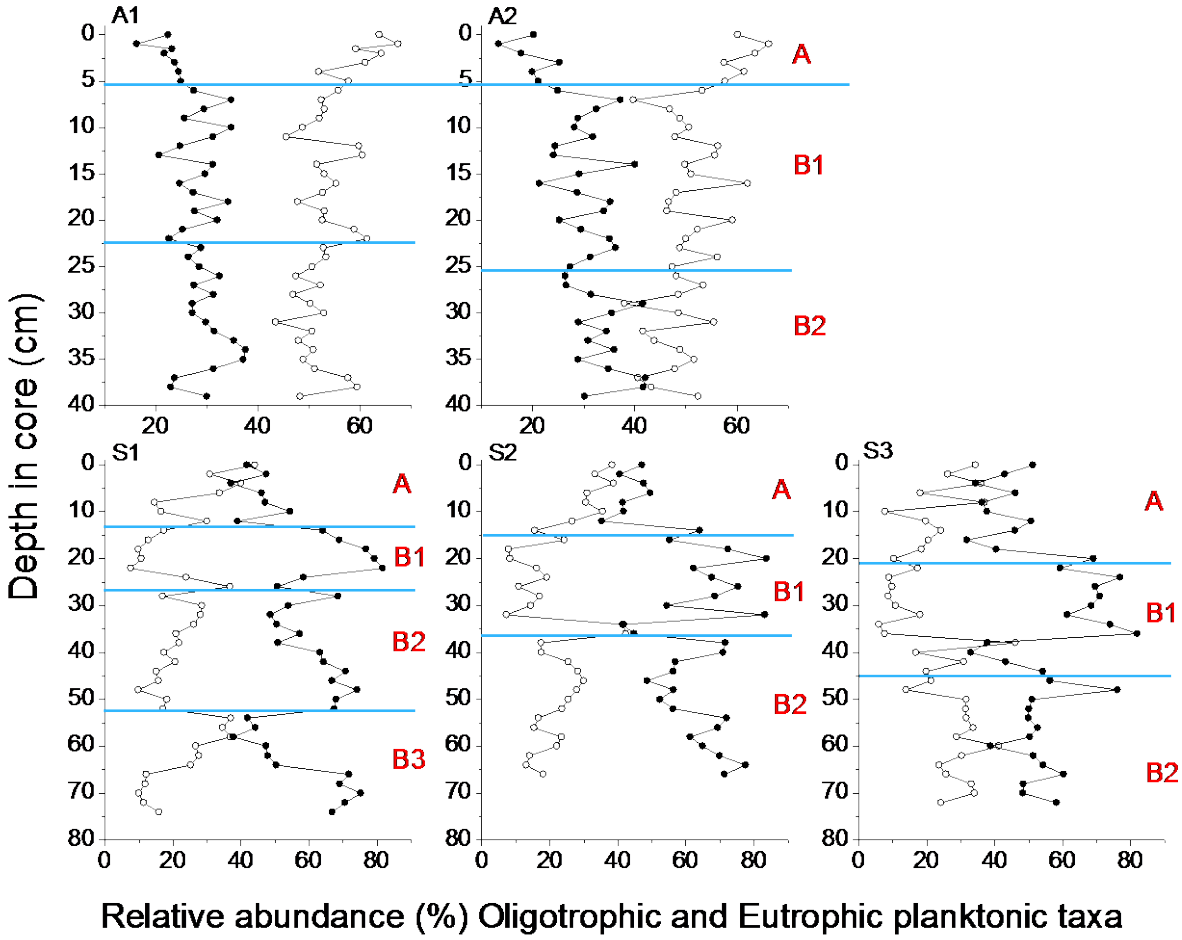


Figure 20 Relative abundance (%) of total oligotrophic planktonic taxa (*Cyclotella*, open circles) and total meso-eutrophic planktonic taxa (*Stephanodiscus*, *Aulacoseira*, *Fragilaria*, *Asterionella*, closed circles). (Cumming et al. 2015)

4.2.8 Grain Size Analysis

Only 8 samples have been analysed thus far. Samples in the uppermost portion of Seton Lake indicate a broad number of particle types, whereas sediments in the lower portions of Seton Lake generally have a more even distribution of larger particles. All samples will be analysed for the distribution of grain size by the end of May 2016. This information will be used in conjunction with the ^{137}Cs activities to determine the initial timing of the Bridge River diversion in the sediment cores.

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

Similar to findings in 2014, both lakes were thermally stratified during the period of measurement in 2015 (Figure 21 and Figure 22). In May of 2014, the thermoclines were broad, showing weak resistance to mixing and establishment of a well-defined thermocline occurred in early summer. In 2015, the thermocline was well defined by late spring over a depth range of 20-25m in both lakes. Surface temperatures in Seton and Anderson Lakes in 2015 were within 1 ºC in July and August, with a maximum surface temperature of 21 ºC in Anderson on the July casts, and 20 ºC in Seton on the August casts. Surface temperature in Seton Lake peaked earlier in 2015 (July) compared to 2014 (August) and overall, the maximum surface temperatures recorded on the casts in 2015 were slightly lower than the maximum surface temperature of 22ºC recorded on the August casts in 2014 in both lakes. The thermocline remained well defined into the fall, despite surface temperatures cooling to 12-14 ºC. This pattern was observed in both 2014 and 2015, where resistance to mixing remained high despite cooling surface temperatures, and a distinct epilimnion and hypolimnion remained intact in both lakes at the end of sampling in late October. Hypolimnetic temperatures were 4-5ºC in both lakes in October 2015, the temperature at which water has highest density.

Thermal patterns observed in Seton Lake in 2015 were similar to those in 2014. Structuring of the Seton Lake thermal data over the distance from S1 to S6 in August of each year showed no disturbance of the thermal structure in Seton Lake from the inflow of Carpenter Lake water (Figure 22). 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. In 2014, 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. In 2015, the presence of a seiche was again detected in July, although this time there was a downward tilt of the thermocline west to east (Figure 22). 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 due to seiche activity was 5m in August 2014 and closer to 10m in July 2015, but it could be more or less at other times in relation to the pattern of oscillation. 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 23) in 2014 and 2015.

In 2014 and 2015, water temperatures in the tailrace at BR1 were 8ºC in May, which increased to 16ºC in late August and cooled to 11 ºC in late October (Figure 24). The episodic 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 observed in 2014 were far less frequent in 2015. In both years, the baseline of the curve in Figure 24a 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.

In both 2014 and 2015, Portage Creek temperature was approximately 10ºC in late May, it increased to 20ºC by mid-July or early August, and then declined to 13ºC by late October (Figure 24b). The peak temperature was approximately one month earlier than in diversion water from Carpenter Reservoir (BR1) (Figure 24a) 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.

In both 2014 and 2015, episodic changes in water temperature were observed in Portage Creek (Figure 24b) that were not typical of its source in Anderson Lake where surface water temperature changed gradually over time (Figure 21b). 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.

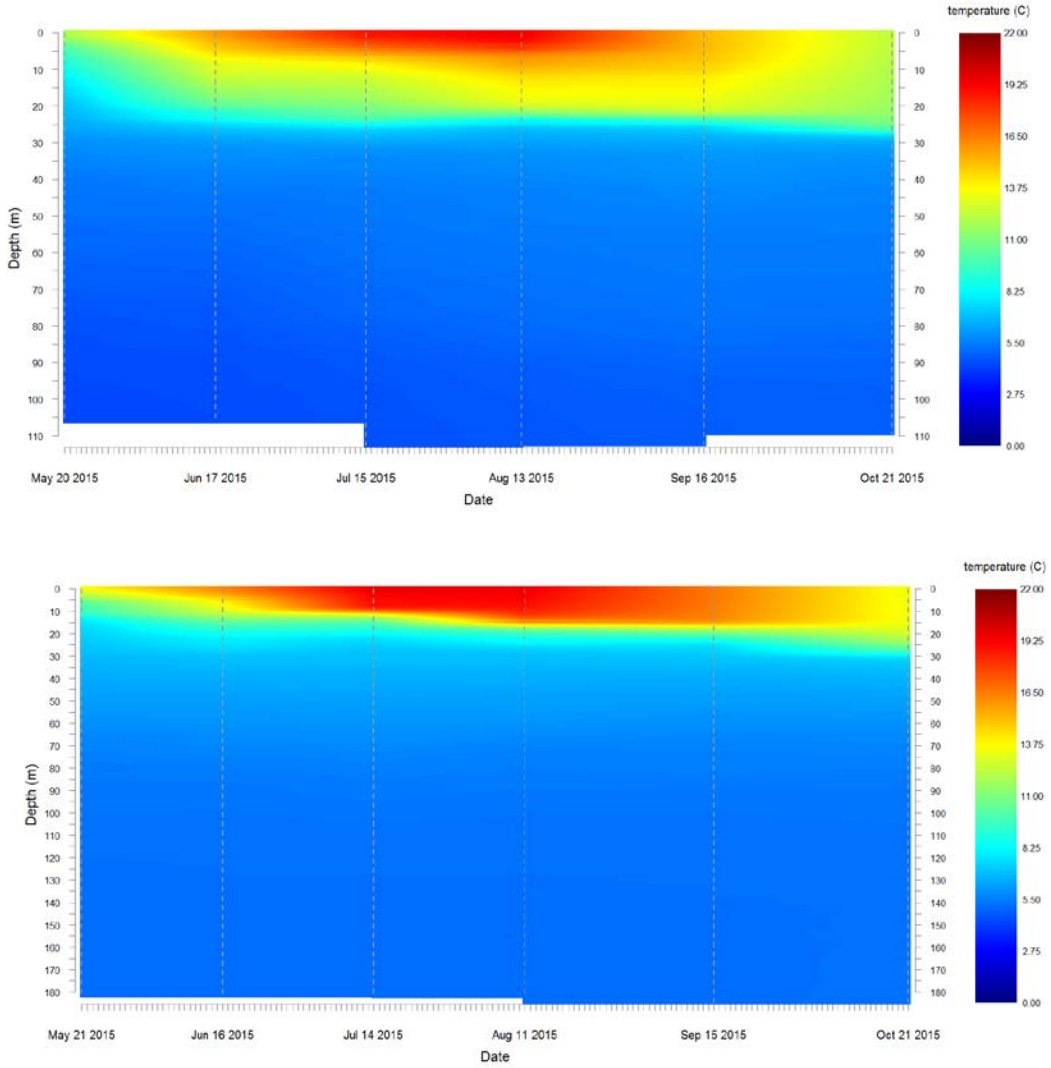


Figure 21. Water temperature during May-October, 2015 in Seton Lake at S4 (a) and Anderson Lake at A1 (b). The profiles extended to the lake bottom in all months using a SeaBird CTD. The vertical dotted lines indicate dates of measurement. Data between those dates were linearly interpolated.

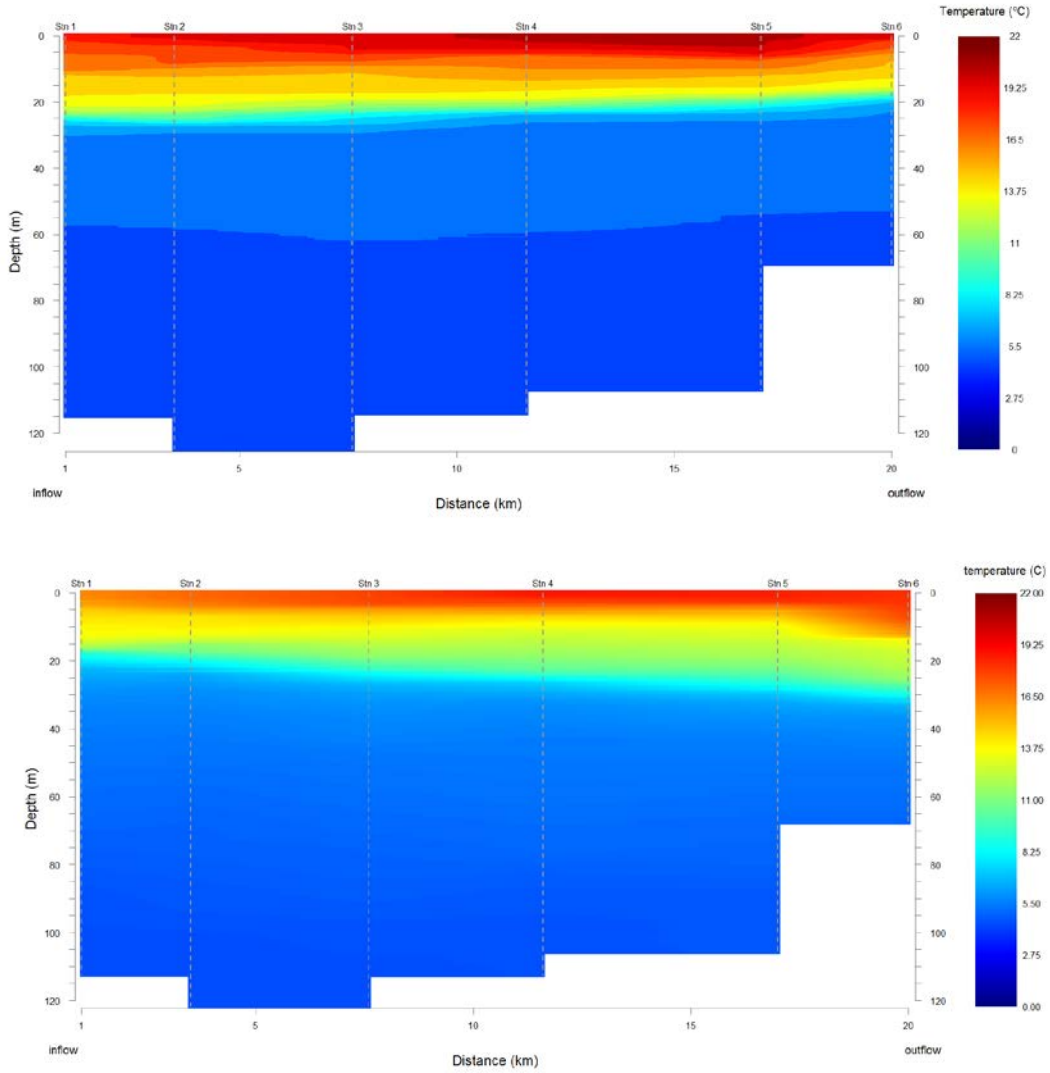


Figure 22. Water temperature profiles integrated between all six stations on August 2014 (top) and July 2015 (bottom) in Seton Lake. S1 is west of Shalalth and S6 is close to the outflow at the east end. Vertical dotted lines indicate stations of measurement. Data between those stations were linearly interpolated.

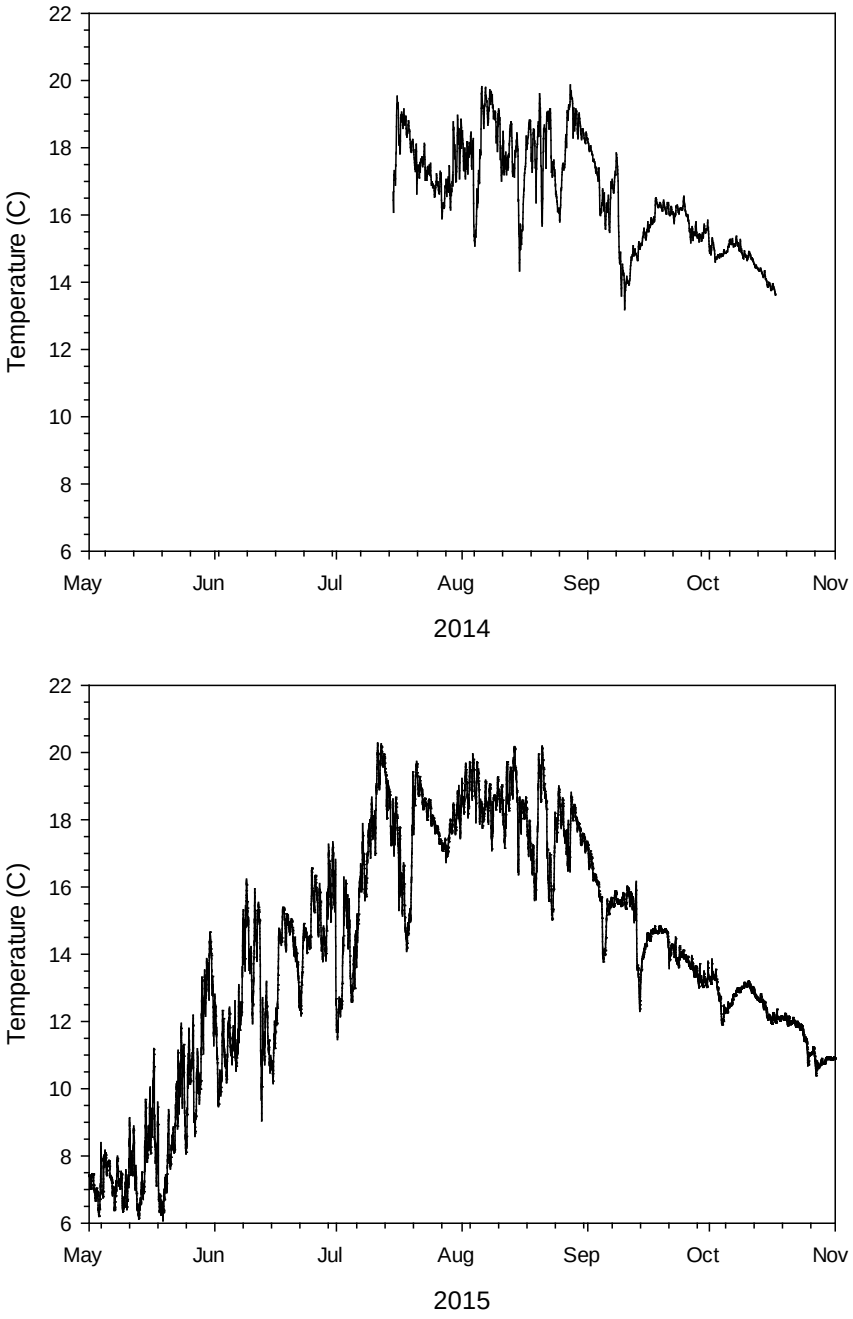


Figure 23. Hourly temperature in the Seton River 200m downstream of the Seton Dam (1 km downstream of the outflow of Seton Lake) in 2014 (top) and 2015 (bottom).

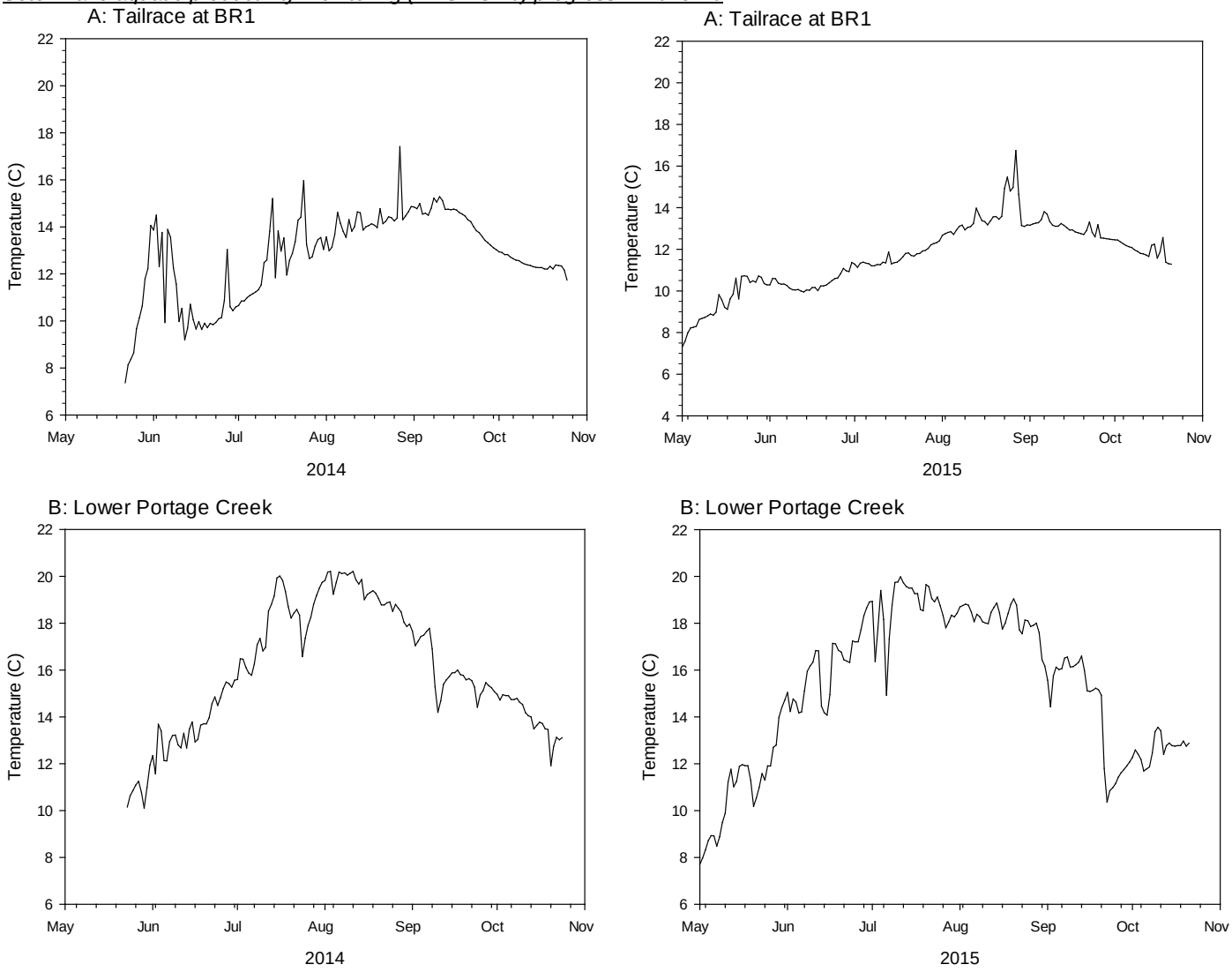


Figure 24. Mean daily temperature in the tailrace at BR1 (a) and Lower Portage Creek (b) in 2014 and 2015.

4.3.2.2 Turbidity

In 2015, turbidity of the Portage Creek inflow to Seton Lake was highest in May and was lower over the remainder of the sampling period (Figure 25). The spike of turbidity in Portage Creek in the fall of 2014 did not occur in 2015 (Figure 25). Similar to findings in 2014, the turbidity in Anderson Lake was <1 NTU for the entire sampling period in 2015 (Figure 26) which shows that the higher turbidity values observed in Portage Creek in May originated as outwash of glacial fines from Whitecap Creek. Turbidity in the diversion from Carpenter Lake was highest in June (26.7 NTU) and September (29.3 NTU) with values ranging from 10 to 18 NTU in the other months (Figure 25). With the exception of July, the monthly turbidity was 12 to 27 NTU higher in the Seton Lake inflow than the outflow. In July, there was a spike in turbidity in the Seton Lake outflow to 15.8 NTU.

Seton Lake turbidity in 2015 changed spatially as it did in 2014 (Limnotek, 2015) (Figure 27). In May, turbidity was <15 NTU throughout the water column near the inflow end of the lake. At station 4 at the same time, turbidity was <6 NTU in the bottom 50m but higher turbidity of 10-15 NTU persisted as interflow in the middle of the water column and within a 5-15m band. At the far eastern end of the lake turbidity throughout the water column was <6 NTU in May. In October two distinct turbidity bands were found. One was in the top 30m of the water column having 10-15NTU that extended from S1 at the inflow end to S5 eastward. This turbidity declined to <6NTU further east at S6. The second band was within the bottom 30m of the water column with turbidity reaching 32NTU at S1 but increasing to 44NTU immediately downstream of the BRG1 discharge. This bottom turbidity dissipated between S3 and S4. These two turbidity layers show that much of the turbid Carpenter water plunged to the bottom of Seton Lake in October but another fraction remained in the 30m eplimnetic surface layer (Figure 21). That surface layer likely contained very small particles entrained with flow of BRG discharge water having similar density to that in Seton Lake. The bottom layer likely contained larger particles having relatively high settling velocities that resulted in rapid deposition to bottom sediments.

Seton Lake turbidity also changed temporally in 2015 as it did in 2014 (Limnotek 2015) (Figure 28). At S2, located immediately downstream of the BRG discharge, turbidity was 10-12 NTU throughout the water column in May, 2015. In the summer it declined to <6 NTU but in the fall the two layers of high turbidity discussed above (a surface layer within the eplimnion and a bottom 30m layer) were prevalent. At S5 in the spring, turbidity of 10-12 NTU was found with highest turbidity occurring in an interlayer of 30-60m water depth. Again turbidity was relatively low in summer over the whole water column and in the fall, the two layers of turbidity were again detected but at much lower turbidity (<12NTU) than was found at the inflow end of the lake. This difference in turbidity between the inflow and outflow ends of the lake shows settlement and retention of particles contributing to turbidity.

Mean particle size contributing to turbidity increased from spring through fall, but it was always <4.3 µm in 2015 (Figure 29). In 2014 it was always <3.6 µm. Mean particle size was larger in the Upper Bridge River (inflow to Downton Reservoir) and the Middle Bridge River (inflow to Carpenter Lake), especially during the spring months. It was not unexpected for mean

particle size to be highest in May during a period of higher flows due to spring snow melt. However, the decreasing mean particle size from Upper Bridge to Middle Bridge River to the tailrace waters in samples collected each month showed settlement and retention of largest particles over the upstream to downstream gradient. The smaller particles were typical of clay (Ashworth et al. 2001) and would have settled at approximately 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. Given the Seton water residence time of 239 days (Table 1), most of those fine particles will not settle out of the water column before being discharged downstream, thus producing the turquoise colour of Seton Lake.

Differences in turbidity between the BRG tailrace inflow to Seton Lake and the Seton River outflow (Figure 27) showed that settlement of particles did occur in the lake within the water residence time. This finding means that data in Figure 29 do not show the full range of particle sizes loading Seton Lake. Particles having settling velocities greater than clay must have been present in the BRG1 discharge but were not captured in the monthly grab samples from BRG1. More frequent sampling would be needed to detect the full range of particle size transport but that level of effort is beyond the scope of present study.

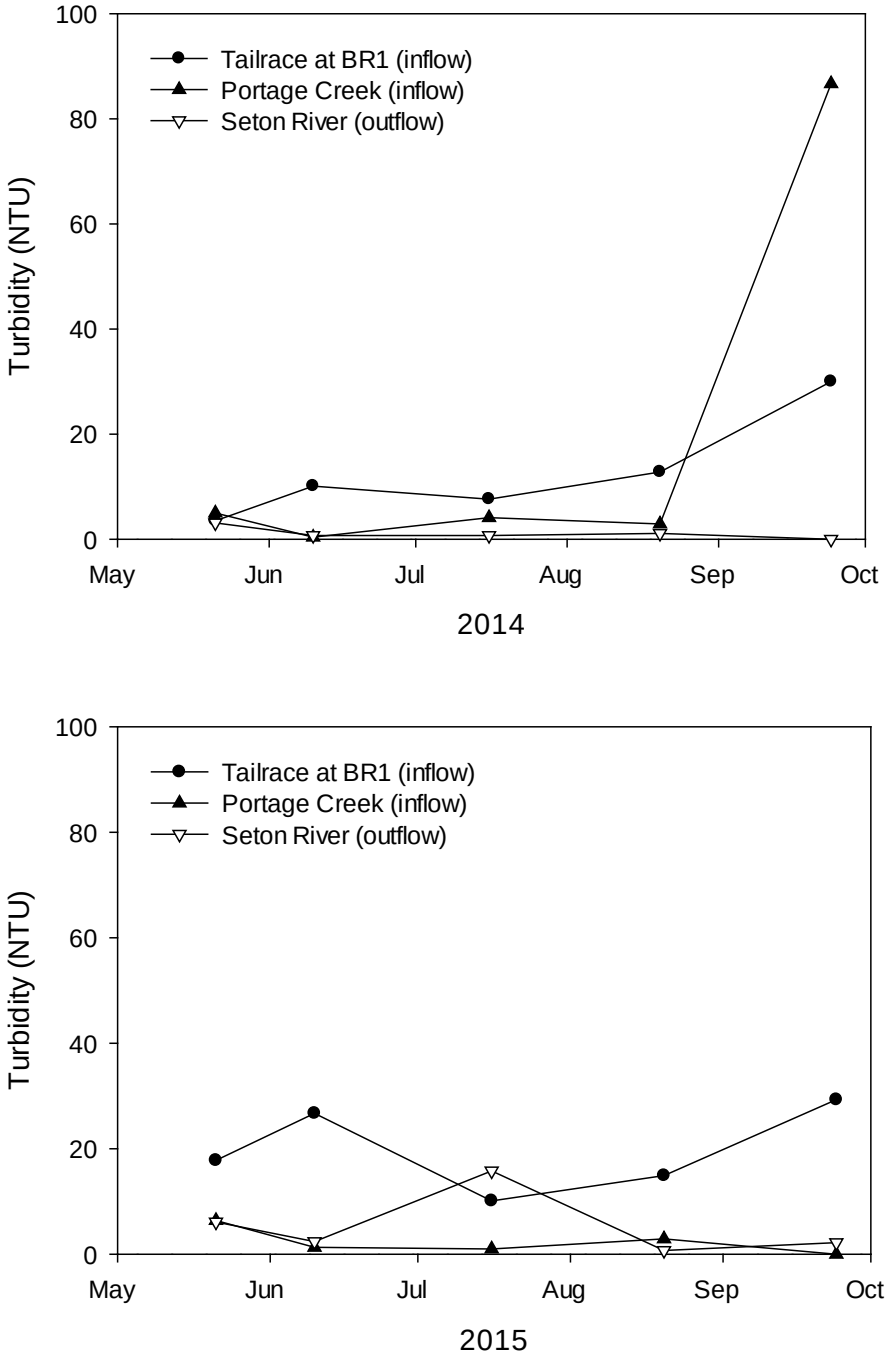


Figure 25 Monthly turbidity of Seton Lake inflow at BR1 and Portage Creek and outflow in the Seton River in 2014 (top) and 2015 (bottom).

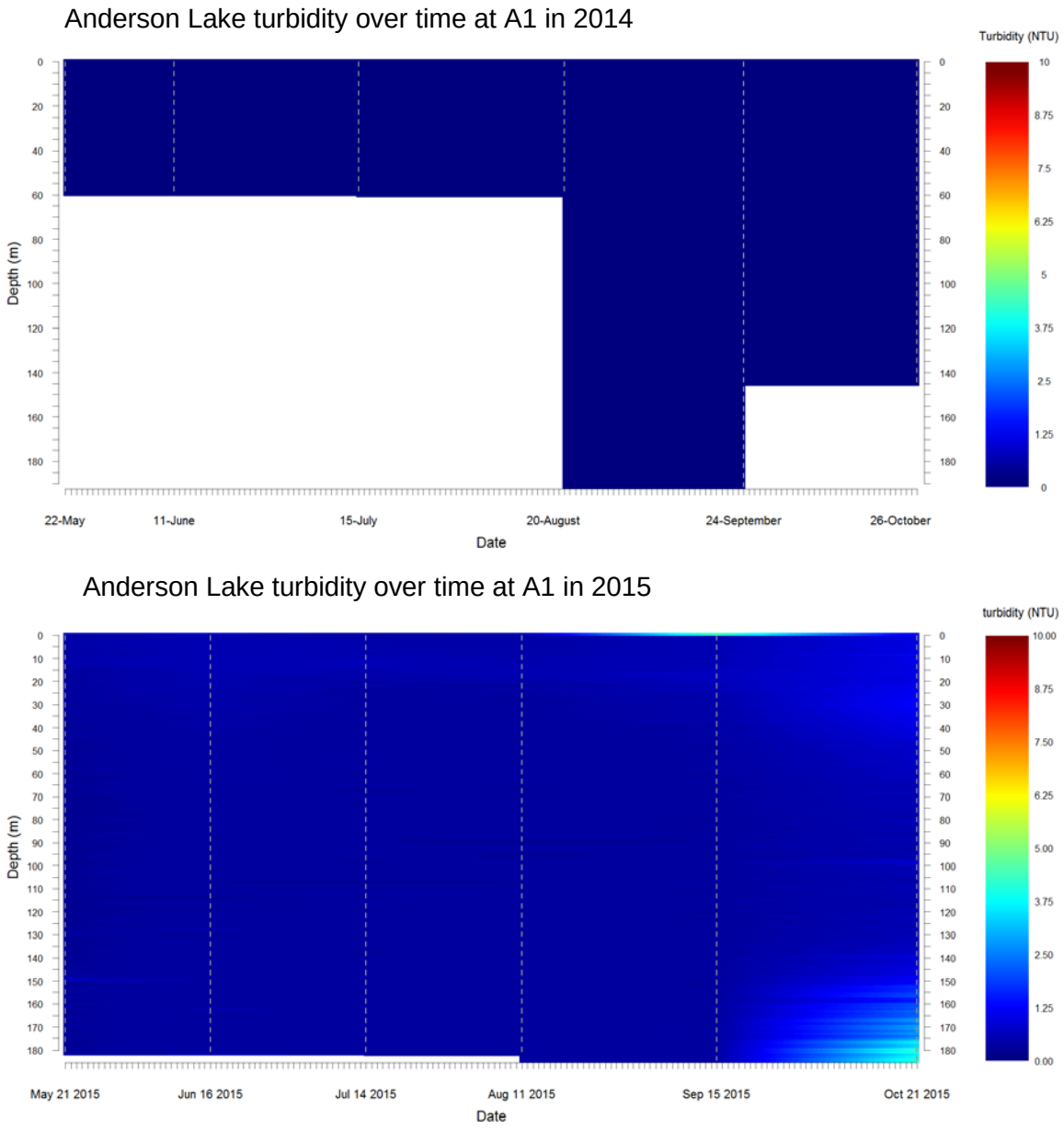


Figure 26. Turbidity in Anderson Lake at A1 during May through October in 2014 (top) and 2015 (bottom).

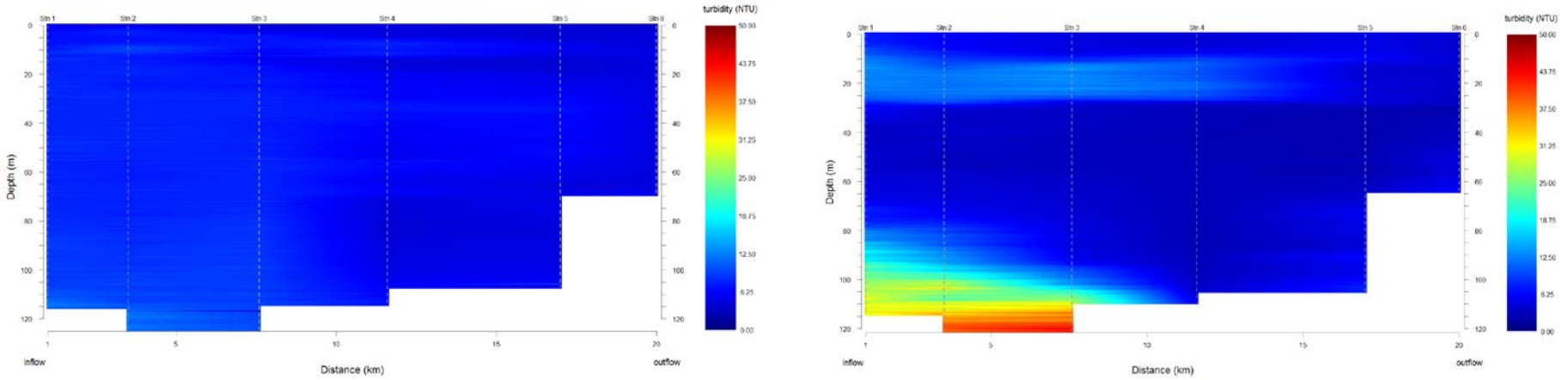


Figure 27. Turbidity in Seton Lake in May 2015 (left) and October 2015 (right). Vertical dotted lines indicate stations of measurement. Data between the lines were linearly interpolated.

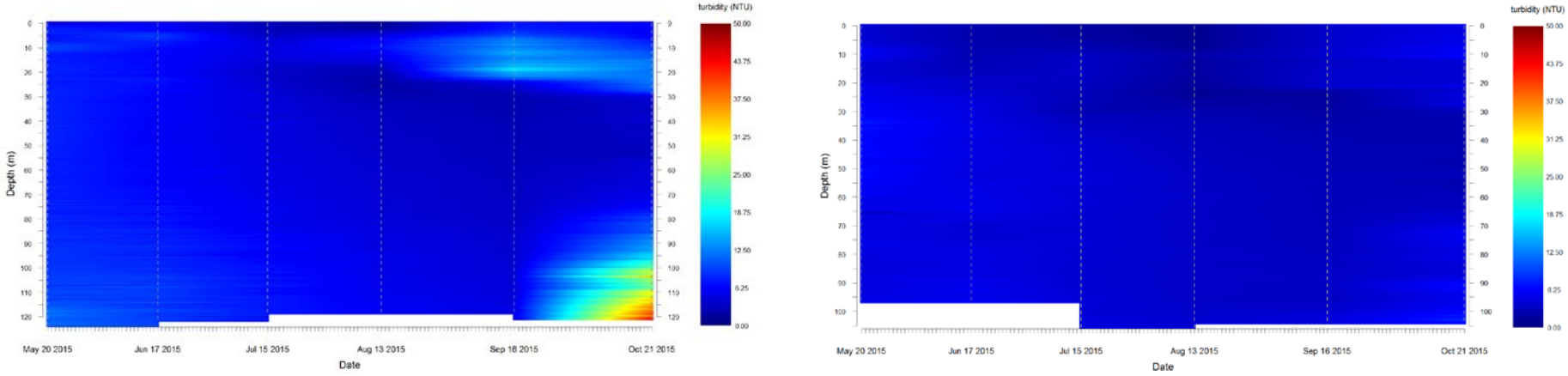


Figure 28. Turbidity in Seton Lake at S2 (left) and S5 (right) during May through October 2015. Vertical dotted lines indicate dates of measurement. Data between the lines were linearly interpolated.

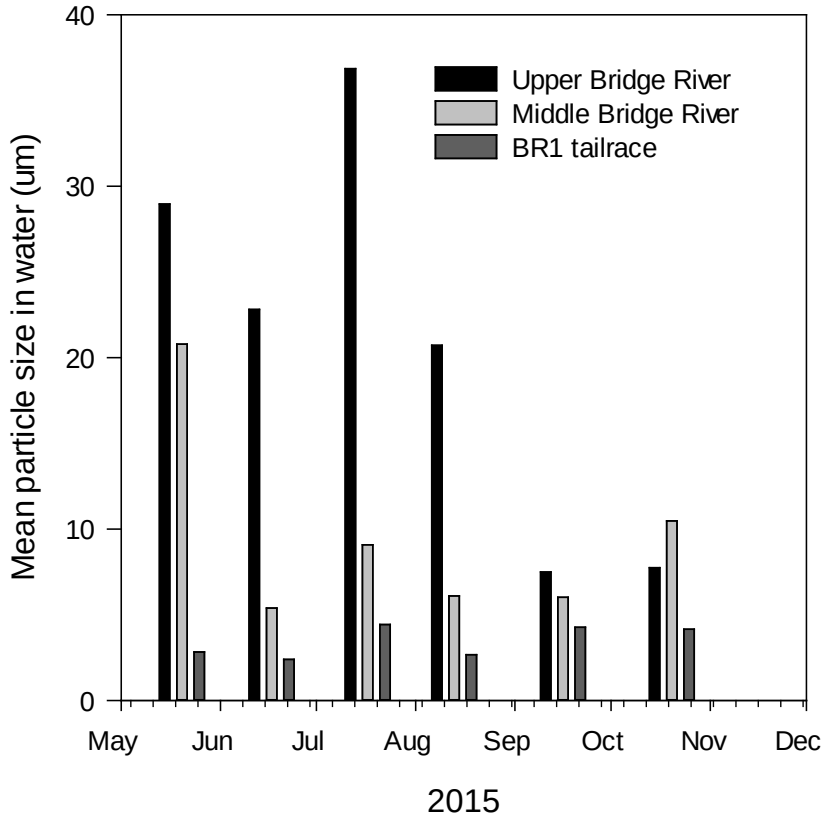


Figure 29. Arithmetic mean size of detectable particles in water flowing into Downton Reservoir (Upper Bridge River), water flowing into Carpenter Reservoir (Middle Bridge Reservoir) and in tailrace water at BR1 when all turbines were running and no backwatering from Seton Lake was occurring. Mean sizes are overestimated because the lower half of the particle size distribution (<1.3 µm) was not detectable in all samples.

4.3.2.3 Light

In 2014 and 2015 in both lakes, Secchi depth was lowest in May and September with greater Secchi depths in June through August. Greatest transparency in 2014 and 2015 occurred in July and August respectively, consistent with the time of lowest turbidity (Figure 26 and Figure 27). Light attenuation was generally 2 to 3 times less in Anderson Lake than in Seton Lake (Figure 30, Table 6) due to the differences in turbidity. In the general absence of turbidity in Anderson, time course change in Secchi depth may be related to change in plankton density that can influence light attenuation.

Table 6. Mean euphotic zone depth, Secchi depth and light extinction coefficient over the growing season in 2014 and 2015. 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 2014	Seton Lake 2015	Anderson Lake 2014	Anderson Lake 2015
Euphotic zone depth (m)	13.4 ± 3.6	9.7 ± 2.4	28.8 ± 2.5	24.5 ± 2.4
Secchi depth (m)	4.1 ± 1.6	3.5 ± 2.2	12.5 ± 4.0	11.7 ± 2.7
Light extinction coefficient	0.373 ± 0.087	0.302 ± 0.085	0.162 ± 0.016	0.190 ± 0.004

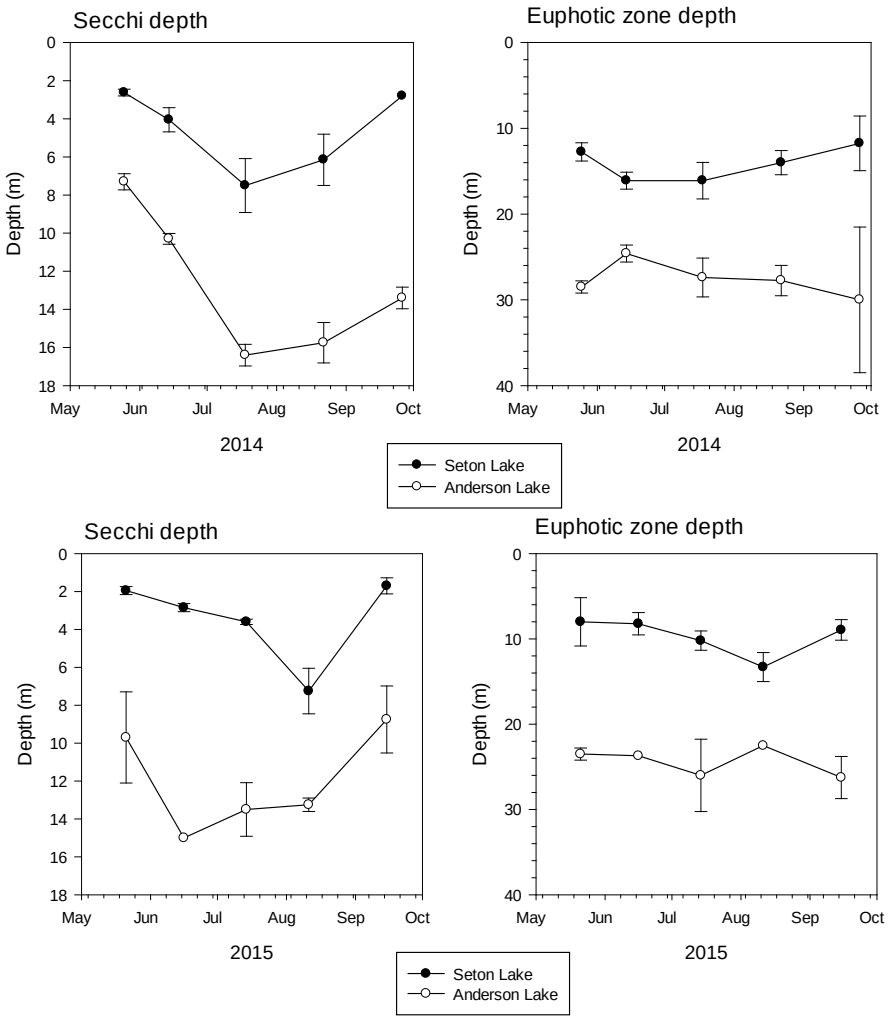


Figure 30. Mean Secchi depth (left) and euphotic zone depth (right) (±sd) in Seton and Anderson Lakes in 2014 (top) and 2015 (bottom). Euphotic zone depth was the depth at which PAR was 1% of surface irradiance as measured using a LiCor LI250A irradiance meter equipped with a spherical quantum sensor. Values are from one measurement at each of 2 stations on each lake by date.

4.3.2.4 Chemistry

In 2015, the water chemistry in Seton and Anderson Lakes was consistent with oligotrophic conditions found in 2014 and in earlier studies (e.g. Shortreed et al. 2001) (Table 7). The pH was slightly alkaline in both lakes and total suspended sediment concentration was <1 $mg \cdot L^{-1}$ in 2014 and <3 $mg \cdot L^{-1}$ in 2015 at all times. 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.

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.

In both 2014 and 2015, 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 (Table 7). Even concentrations of TN that contain inorganic and complex organic fractions of N that are not bioavailable were near or below the detection limit in the epilimnion of both lakes in both years. 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 Lakes in 2014 (n=10) and 2015 (n=10). Epilimnion and hypolimnion data are from water depths of 2m and 70m respectively.

Analyte	Seton Lake 2014		Seton Lake 2015		Anderson Lake 2014		Anderson Lake 2015	
	Epilimnion	Hypolimnion	Epilimnion	Hypolimnion	Epilimnion	Hypolimnion	Epilimnion	Hypolimnion
pH	7.9 ± 0.04	7.8 ± 0.02	7.8 ± 0.09	7.8 ± 0.02	8.0 ± 0.13	8.0 ±0.05	8.0 ± 0.13	8.0 ±0.05
TSS ($mg \cdot L^{-1}$)	<1*	<1*	<3*	<3	<1	<1	<3	<3
NH_4-N ($\mu g \cdot L^{-1}$)	< 5	< 5*	< 5	< 5	< 5	< 5	< 5	< 5
NO_3-N ($\mu g \cdot L^{-1}$)	< 5**	41.4 ± 6.7	<5**	42.3 ± 13.4	20.7 ± 18.8	69 ± 8.1	14.0 ± 13.8**	64.0 ± 21.3
TN ($\mu g \cdot L^{-1}$)	<50*	58 ± 23	38.8 ± 9.5	53.5 ± 11.2	<50**	89 ±13	46.7 ± 13.5	84.7 ± 18.1
SRP ($\mu g \cdot L^{-1}$)	< 1	<1	<1**	<1**	<1**	<1	<1	<1
TDP ($\mu g \cdot L^{-1}$)	< 2**	<2**	<2**	<2**	<2**	<2**	<2	<2
TP ($\mu g \cdot L^{-1}$)	2.3 ± 1.2	3 ± 2.5	2.7 ± 0.8	2.9 ± 0.7	<2**	<2**	<2	<2

* 1 value greater than MDL

** at least half of values <MDL

4.3.3 Phytoplankton

Phytoplankton biomass, measured as the growing season average chlorophyll-a concentration, and mean growing season rates of primary production were lower in Seton Lake than in Anderson Lake, and lower in 2014 and 2015 in both lakes than during the transition period in 2000-2003 (Figure 31 and Figure 32). These temporal comparisons are only descriptive at this point in the study because samples to be collected in 2016 are required before statistics can be run to test the before after control impact comparisons.

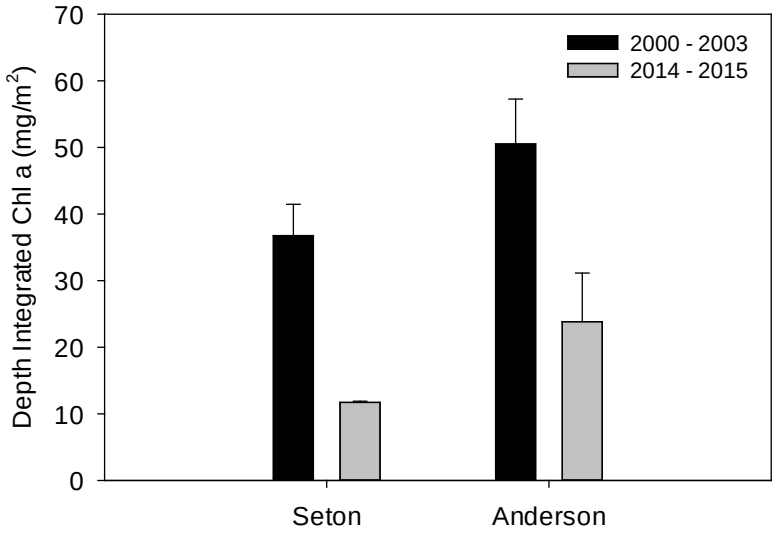


Figure 31 Mean sampling period (May to Sept) depth-integrated areal phytoplankton chlorophyll-a concentrations in Seton Lake and Anderson Lake during the transition period (2000 – 2003, n=4) and the after period (2014 and 2015, n=2).

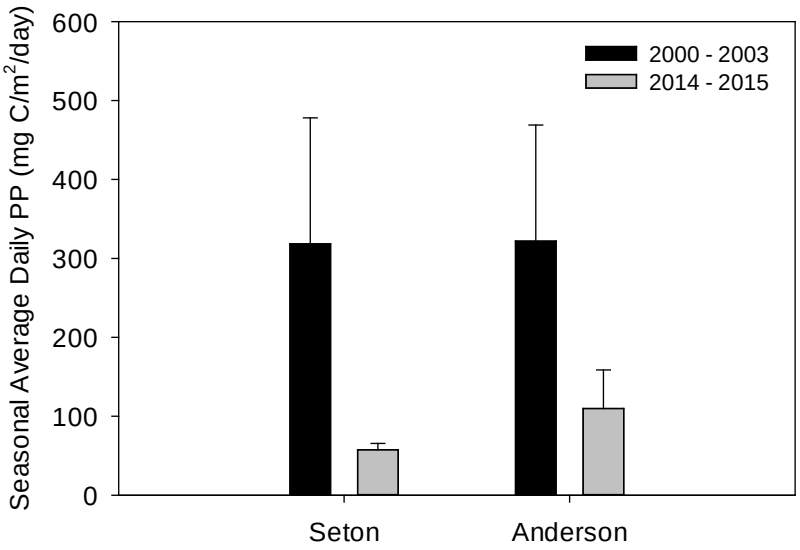


Figure 32 Seasonal average daily primary productivity (PP) in Seton Lake and Anderson Lake during the transition (2000 to 2003, n=4) and after (2014 and 2015, n=2) periods.

Mean growing season daily primary production in Seton Lake and Anderson Lake in 2014 and 2015 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 2016 will be needed to determine if an actual change has occurred or whether the lower rates in 2014 and 2015 or the DFO rates in 2000 – 2003 were anomalous. The mean rate of primary production in Seton Lake in 2014 and 2015 (57 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 will be investigated in 2016.

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

Lake or Reservoir	Growing season average primary production (mg C m^{-2} d^{-1})	Fertilized or not	Reference
Seton Lake mean from 2014 and 2015	57	No	This report
Seton Lake mean from 2000-2003	318	No	This report
Anderson Lake mean from 2014 and 2015	110	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)
Alastair Lake	209	no	Stockner and Shortreed (1979)
Bear Lake	144	No	Stockner and Shortreed (1979)
Johanson Lake	66	No	Shortreed et al. (1998)
Kisumkalum Lake	33	No	Shortreed et al. (1998)
Kitwanga Lake	265	No	Shortreed et al. (1998)
Lakelse Lake	74	Affected by recreational shoreline development	Shortreed et al. (1998)
Morice Lake	65	No	Stockner and Shortreed (1979)
Morrison Lake	108	No	Shortreed et al. (1998)
Sustut Lake	88	No	Shortreed et al. (1998)
Swan Lake	93	No	Shortreed et al. (1998)

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 and 2015 to support later statistical tests in the BACI layout. Results showed that mean 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. Polycarbonate filters are also known to pass particles larger than the nominal pore size, introducing possible error (Stockner et al. 1990). One more year of data including both the 0.2 µm polycarbonate filters and the 0.8 µm glass fibre filters will be collected before running a statistical test to examine filter effects on rates of primary production.

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. Values shown are the mean seasonal values for 2014 and 2015.

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	39.3 ± 19.2	57.4 ± 8.2
Anderson	105.1 ± 17.7	109.7 ± 48.9

Phytoplankton in Seton and Anderson Lakes were similar between 2014 and 2015 (Figure 33). The community included diatoms (Bacillariophyceae), green algae (Chlorophyceae), flagellates (Chrysophyceae and Cryptophyceae), blue green algae (Cyanobacteria), and dinoflagellates (Dinophyceae). Low biovolumes of Euglenoids (Euglenophyceae) were present in both lakes in 2015 but only Seton Lake in 2014. Other algae included Prymnesiophyceae, Eustigmatophyceae and Bicosecophyceae which were present in both lakes in both years. In Seton Lake, the average phytoplankton biovolume increased from 292 $mm^3 \cdot mL^{-1}$ in 2014 to 453 $mm^3 \cdot mL^{-1}$ in 2015. In Anderson lake, the average phytoplankton biovolume decreased from 324 $mm^3 \cdot mL^{-1}$ in 2014 to 253 $mm^3 \cdot mL^{-1}$ in 2015. Similar to 2014, the single largest phytoplankton assemblage in both lakes in 2015 was the flagellated chryso-cryptophytes. Thirteen species of flagellates were present in both lakes, accounting for 46% of the average biovolume in Seton Lake and 44% of the average biovolume in Anderson Lake compared to 53% and 38% in 2015 in Seton and Anderson respectively. Green algae (Chlorophyceae) were the second largest division in both lakes in 2014 and the second largest division in Anderson Lake in 2015, with 14 species accounting for 32% of total biovolume in Anderson Lake in 2014, 14 species accounting for 28% of total biovolume in Anderson Lake in 2015, and 10 species accounting for 15% in Seton Lake in 2014. In Seton Lake, diatoms were the second largest division in 2015. Six species of diatoms accounted for 25% of total biovolume in Seton Lake which was an increase

from 2014 when four species of diatoms accounted for 3% of phytoplankton biovolume. There was an increase in the biovolume of diatoms in Anderson lake between 2014 and 2015 as well, from 8% to 15% respectively. The total biovolume of blue green algae decreased from 3% to 1% in Seton Lake between 2014 and 2015, and from 12% to 2% in Anderson Lake. In 2015, 4 species of blue green algae were present in Anderson Lake, and 3 species in Seton lake. Similar to findings in 2014, one species of each yellow green algae (Xanthophyta), Haptophyta and Bicosecophyceae (all shown as other algae in Figure 33) accounted for 6% and 8% of total phytoplankton biovolume in Seton Lake and Anderson Lake respectively in 2015. In 2015, there were five species of dinoflagellates in Seton Lake and 3 species of dinoflagellates in Anderson Lake, accounting for 8% and 4% of phytoplankton biovolume in Seton and Anderson respectively.

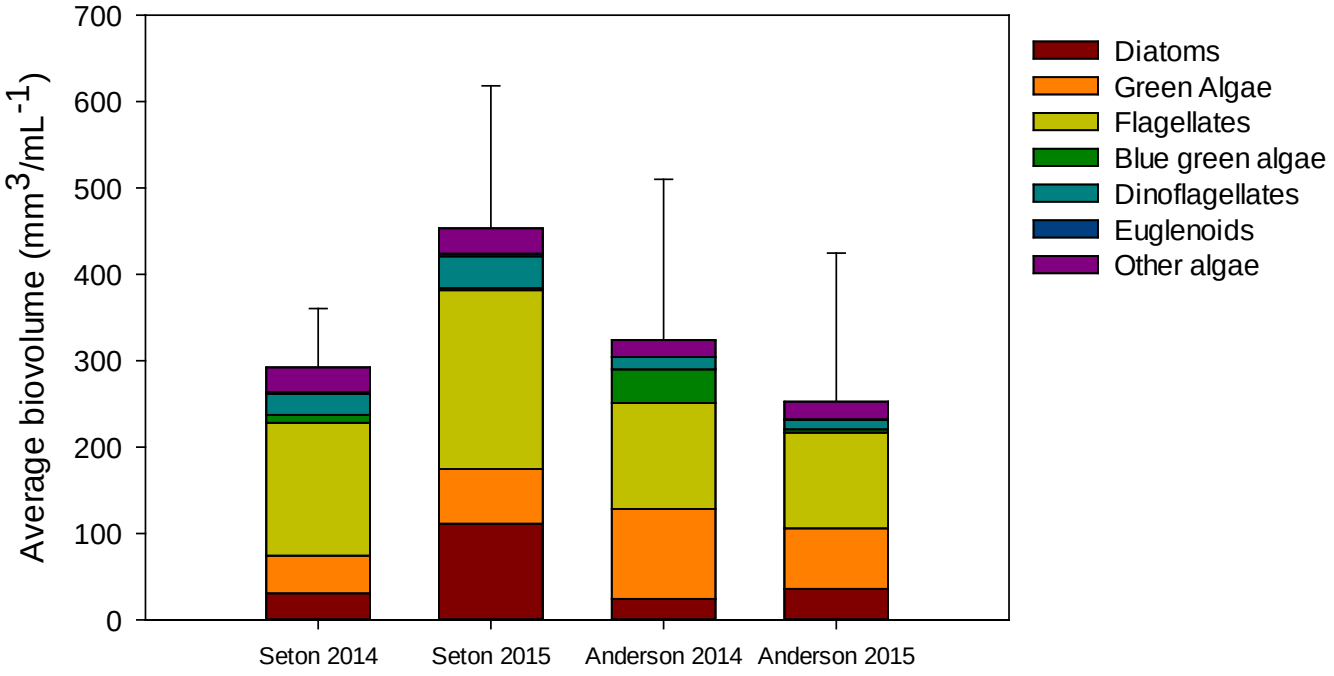


Figure 33 Mean biovolume of phytoplankton by division over the growing season in 2014 and 2015 (± standard deviation) (n=10 in 2014, 2 stations for each lake sampled monthly from May to September; n=20 in 2015, 2 replicates collected at 2 stations for each lake sampled monthly from May to September).

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 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 from the 2014 and 2015 sampling periods. 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	52	ultraoligotrophic	71	ultraoligotrophic
	range	<1 – 250	307 - 1630	361 - 1387	393 - 6100	<30 – 106		<30 - 109	
Chl-*a* (μg/L)	mean		1.7	4.7	14.3	0.9	oligotrophic	0.9	oligotrophic
	range	0.01 – 0.5	0.3 – 4.5	3 - 11	3 - 78	0.02 – 1.9		0.04 – 3.0	
Secchi depth (m)	mean		9.9	4.2	2.5	3.8	Not relevant***	12.2	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					57 in 2014-15, 318 in 2000-2003	Oligotrophic to mesotrophic	110 in 2014-15 322 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

Ten species of zooplankton were found in Seton Lake and seven species in Anderson Lake in 2014 and 2015. Cladocerans common in both lakes included *Eubosmina longispina, Daphnia ambigua and Chydorous sphaericus. Leptodora kindtii*, and *Daphnia pulicaria* were also present in Seton Lake. Two Cyclopoida were present in both lakes: *Cyclops scutifer* and *Cyclops sp*., and a third *Cyclops bicospidatus tomasi* was present in Seton Lake only. Two calanoid copepods were present including *Epischura nevadensis* and *Acanthodiaptomus denticornus*. Overall in 2014 and 2015, peak zooplankton biomass was 1,423 mg dry wt·m^{-2} in August in Seton and 3,321 mg dry wt·m^{-2} in June in Anderson. Cladoceran biomass accounted for 36% to 88% of total biomass in Anderson Lake (Figure 34). In Seton Lake, cladoceran biomass accounted for 4% to 78% of total biomass, with the lowest cladoceran biomass occurring in May (4%). Biomass of calanoid copepods was ≤ 4% in all months in Anderson lake and ≤8% in all months in Seton Lake.

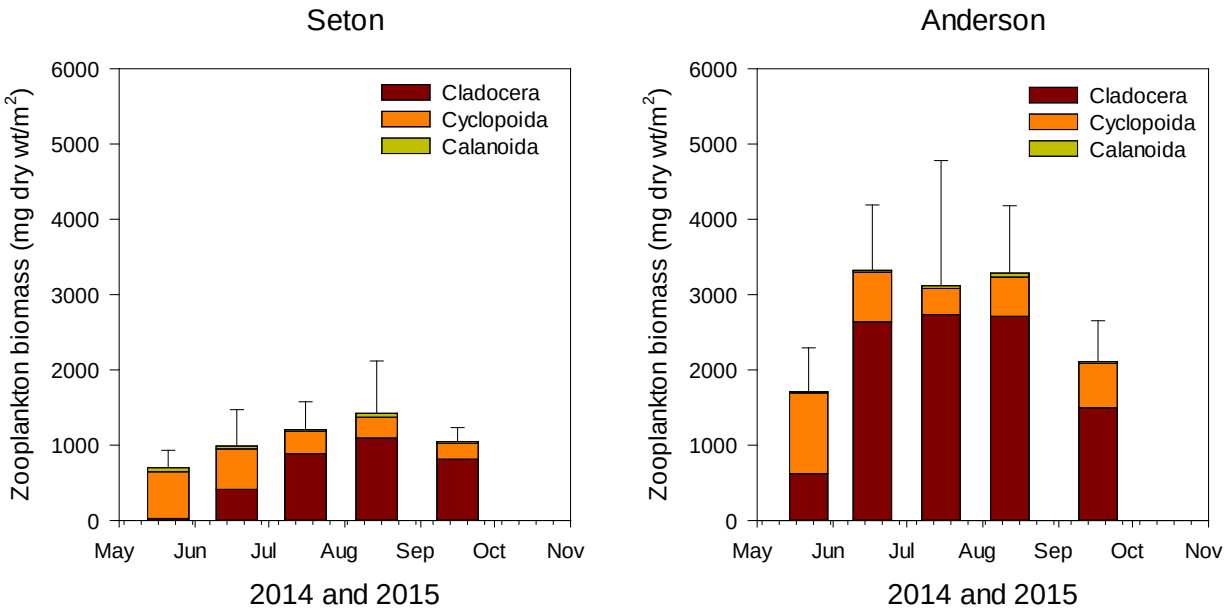


Figure 34 Zooplankton dry weight biomass in Seton Lake (left) and Anderson Lake (right) in 2014 and 2015. Data are shown for all three orders of zooplankton (Cladocera (suborder of Diplostraca), Cyclopoida and Calanoida). Data are shown as a mean and standard deviation from duplicate samples collected at each of two stations on each date.

Mean annual zooplankton production in 2014 and 2015 in Anderson Lake (28.5 g dry wt·m^{-2}·yr^{-1}) was more than double that in Seton Lake (12.7 g dry wt·m^{-2}·yr^{-1}) (Figure 35). Cladocerans accounted for 67% of total production in Seton Lake and 75% of total production in Anderson Lake, with cyclopoids being the next most important. Mean zooplankton production in

Seton Lake in May to September of 2014 and 2015 was double that found in the transition years of 2000 to 2003 (Figure 35). In Anderson Lake, a 13% increase in zooplankton production in 2014 and 2015 was observed in comparison to zooplankton production in the transition period of 2000 to 2003 (Figure 35). As with primary production, these spatial and temporal comparisons are only descriptive for now. Further data from 2016 will be needed before quantitative comparisons can be made to satisfy the BACI layout.

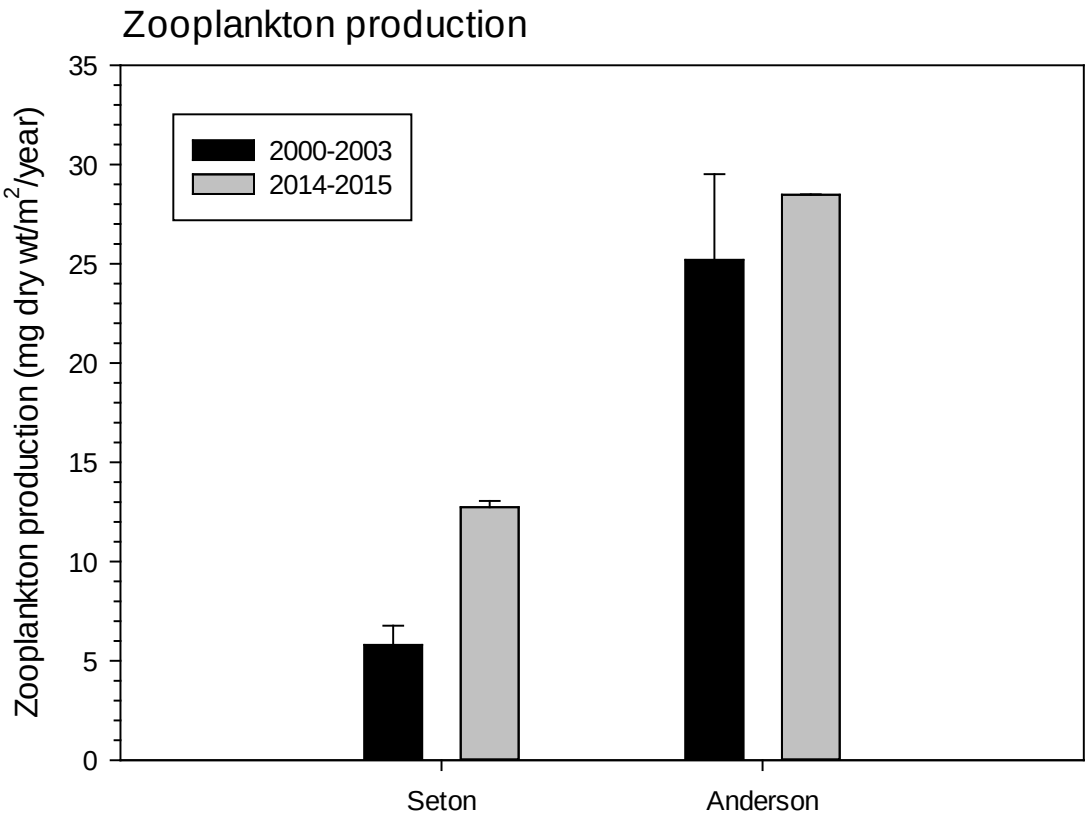


Figure 35. Annual zooplankton production in Seton Lake and Anderson Lake in the transition years 2000 to 2003 (mean and SD shown, n=4) and after years 2014 and 2015 (mean and SD shown, n=2).

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 were 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.

4.4.2 Pelagic fish

In all years sampled, a total of five species were captured in the trawl and gillnets in the two lakes (Table 12). By far, the most common species in all four years and both lakes were *O. Nerka,* mainly of the small size class (≤100 mm in fork length), comprised of Sockeye Salmon and Gwenish. Next in abundance in all years were older and larger *O. Nerka* of the medium size class (101-376 mm), all that were identified by DNA analysis in 2014 were Gwenish (Table 12). The large size class (>376 mm), the least numerous size group in the catch, was mostly composed of large piscivores, with the exception of a few adult Sockeye Salmon in Seton Lake in fall 2014 (Table 12). The major piscivorous fish captured in both lakes were Bull Trout, followed by Northern Pikeminnow, and fewer piscivores were captured in Seton Lake than in Anderson Lake. Some fish of both piscivore species fell into the medium size class.

A few Coastrange Sculpin (small size class) were captured in a number of years in Seton Lake but only one in Anderson Lake. One Rainbow Trout (medium size class) was caught in Anderson Lake in 2014.

In 2014, the small size class was entirely juvenile *O. Nerka*, all of which were caught in the trawl (Table 13). Most of the medium size class were also *O. Nerka* and about the same number were captured in Anderson (127) and in Seton (119). Gillnet caught *O. Nerka* in Anderson Lake were 55% larger than in Seton Lake. The large size class was dominated by Bull Trout in both lakes. Most were caught by the gillnets (38) rather than in the trawl (5) and twice as many were captured in Anderson Lake (29) as in Seton Lake (14). Although the Bull Trout had a similar size range in both lakes they averaged 22% larger in Seton Lake.

4.4.2.1 O. Nerka stock origin

The Bayesian analysis of the DNA data using the STRUCTURE program, 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 -48935 and average among samples variance of 620, whereas the 4-subpopulation model had a similar likelihood value of -48842 but a higher average among samples variance of 927. The baseline adult Sockeye spawner samples from Gates and Portage creeks 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.

As predicted, the putative Gwenish in the medium size class (all were >120 mm) sampled from both lakes during the summer and fall of 2014 were all identified as Gwenish (Table 14). They were the only group identified in the medium size class from either the summer or fall samples in both lakes. During both summer and fall, the trawl caught samples of small *O. Nerka* taken from Anderson Lake were primarily a mixture of Gwenish and Gates Creek Sockeye Salmon, with a higher proportion of Gwenish in the fall (83%) than in the summer mixture (65%, Figure 36). Sockeye from Portage Creek comprised ≤ 3% of the small *O. Nerka* mixture in Anderson Lake in both seasons.

In Seton Lake, Gates Creek Sockeye Salmon were the most common fish identified in both the summer (56%) and fall (49%) small *O. Nerka* trawl caught mixture samples (Table 14, Figure 36). The proportion of Portage Creek Sockeye in Seton Lake decreased from summer (33%) to fall (24%), whereas the percentage of Gwenish increased from (11%) to (27%).

Considering Sockeye only, Gates Creek Sockeye formed the largest proportion of the small size class in all samples taken, comprising ≥92% of them in Anderson Lake, 63-67% of

them in Seton Lake, and 70% of them in the spring 2015 Sockeye smolt run (Figure 37). Part of the smolt run (7%) was of the medium size class, 85% of which were from Gates Creek. While these fish were in the medium size class they were still much smaller than the putative Gwenish used in the DNA analysis. All smolts were ≤115 mm and most (10/13) were ≤105 mm. In comparison, all of the putative Gwenish in the medium size class were ≥ 122 mm. We therefore consider all of the smolts in the DNA analysis to be age-1 fish from a single (2013) brood year.

Table 12 Trawl and gillnet catch of fish by size class in Seton and Anderson lakes.

		Small (≤100 mm)		Medium (101-376 mm)				Large (>376 mm)		
Season	Year	*O. nerka*	Coast-range sculpin	*O. nerka*	Bull Trout	Northern Pike-minnow	Rainbow Trout	Bull Trout	Northern Pike-minnow	Sockeye adult
Seton Lake										
Trawl										
Summer	2001	205	2	4						
	2002	744	1							
	2003	378	2	5						
	2014	344		14				2		
Fall	2000	40		17						
	2001	146	1	3						
	2002	215		21						
	2003	109		5						
	2014	473		9						5
Gill net										
Summer	2003	3		1						
	2014			78	2			7		
Fall	2014			18		1		4	1	
Anderson Lake										
Trawl										
Summer	2001	383								
	2002	168								
	2003	374		2				1		
	2014	675		1	6	1		7		
Fall	2000	523	1	1	1					
	2001	361								
	2002	99		1	1					
	2003	184		2						
	2014	664								
Gill net										
Summer	2003			4						
	2014			74	3	6	1	15		
Fall	2014			52	3	2		6		

Table 13 Catch and length (mm) of fish caught by trawl and gillnet in Anderson and Seton lakes in 2014, organized by TS size classes.

			Size class													
			Small (≤100 mm)					Medium (101 – 376 mm)					Large (>376 mm)				
Gear	Lake	Taxa	N	Mean	95% CI	Min	Max	N	Mean	95% CI	Min	Max	N	Mean	95% CI	Min	Max
Trawl	Anderson	*O. nerka*	1337	43.7	0.7	23	84	1	275.0		275	275					
		Bull Trout						1	300.0		300	300	3	453.3	136.8	400	510
	Seton	*O. nerka*	817	61.6	1.1	27	98	23	153.2	9.3	109	179					
		Bull Trout											2	442.5	730.6	385	500
		Adult Sockeye											5	618.0	20.4	600	640
Gillnet	Anderson	*O. nerka*						126	254.3	9.0	122	316					
		Bull Trout						10	326.2	25.9	250	360	26	452.7	31.4	370	650
		N. pikeminnow						9	334.2	16.4	300	352					
		Rainbow Trout						1	160.0		160	160					
	Seton	*O. nerka*						96	164.3	3.4	129	199	1				
		Bull Trout						1	294.0		294	294	2	583.1	67.6	372	750
		N. pikeminnow						1	211.0		211	211	1	392.0		392	392

Table 14 Classification using DNA of juvenile O. nerka samples from Seton and Anderson lakes in the summer and fall of 2014 and from the Seton River smolt run in the spring of 2015. The medium Gwenish samples were a priori thought to be Gwenish (putative Gwenish) and comprised the 3rd cluster in the 3 population model.

		Small (≤100 mm)				Medium (101 - 365 mm)		
Location	Season	Gwenish	Gates	Portage	Total	Gates	Portage	Gwenish
Seton L.	Summer	16	83	49	148	-	-	50
		(11%)	(56%)	(33%)	(100%)			(100%)
	Fall	45	81	40	166	-	-	24
		(27%)	(49%)	(24%)	(100%)			(100%)
	Total	61	164	89	314	-	-	74
		(19%)	(52%)	(28%)	(100%)			(100%)
Anderson L.	Summer	101	49	4	154	-	-	51
		(66%)	(32%)	(3%)	(100%)			(100%)
	Fall	155	30	1	186	-	-	33
		(83%)	(16%)	(1%)	(100%)			(100%)
	Total	256	79	5	340	-	-	84
		(75%)	(23%)	(1%)	(100%)			(100%)
Seton River	Smolts	-	123	53	176	11	2	-
			(70%)	(30%)	(100%)	(85%)	(15%)	

O.nerka classified as Sockeye only

		Small (≤100 mm)		
Location	Season	Gates	Portage	Total
Seton L.	Summer	83	49	132
		(63%)	(37%)	(100%)
	Fall	81	40	121
		(67%)	(33%)	(100%)
	Total	164	89	253
		(65%)	(35%)	(100%)
Anderson L.	Summer	49	4	53
		(92%)	(8%)	(100%)
	Fall	30	1	31
		(97%)	(3%)	(100%)
	Total	79	5	84
		(94%)	(6%)	(100%)
Seton River	Smolts	123	53	176
		(70%)	(30%)	(100%)

Medium (101 - 365 mm)	
Gates	Portage
11	2
(85%)	(15%)

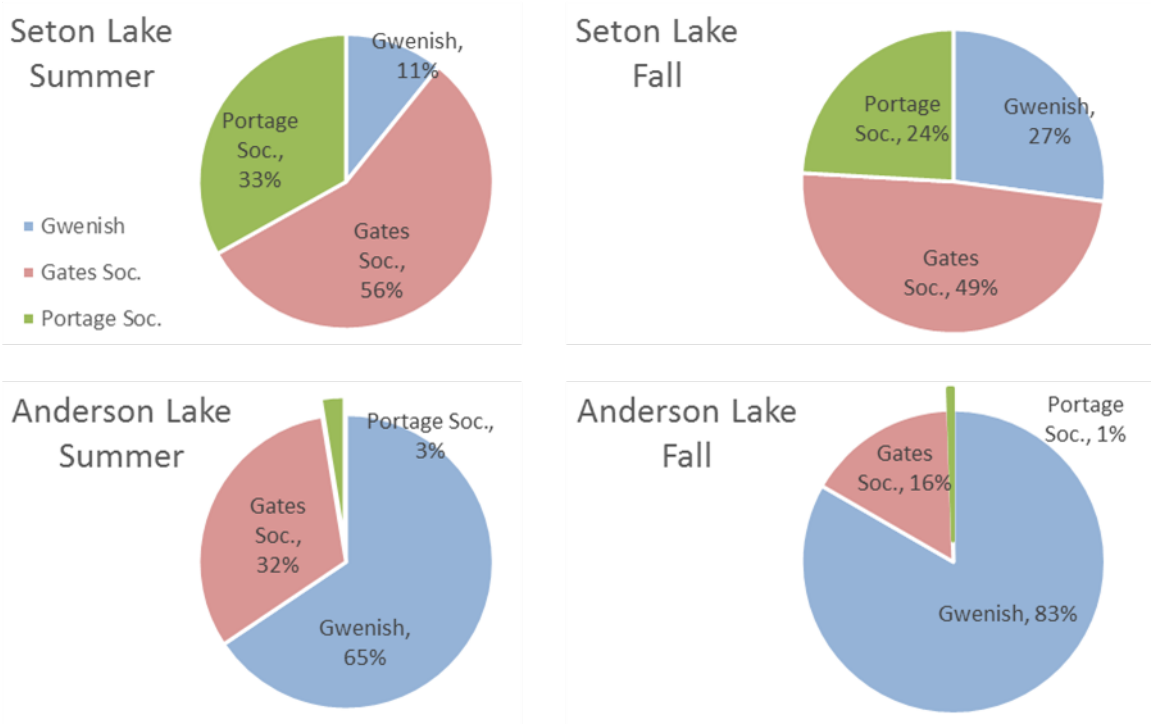


Figure 36. Classification using DNA of O. nerka samples from the small size class in Seton and Anderson lakes in the summer and fall of 2014.

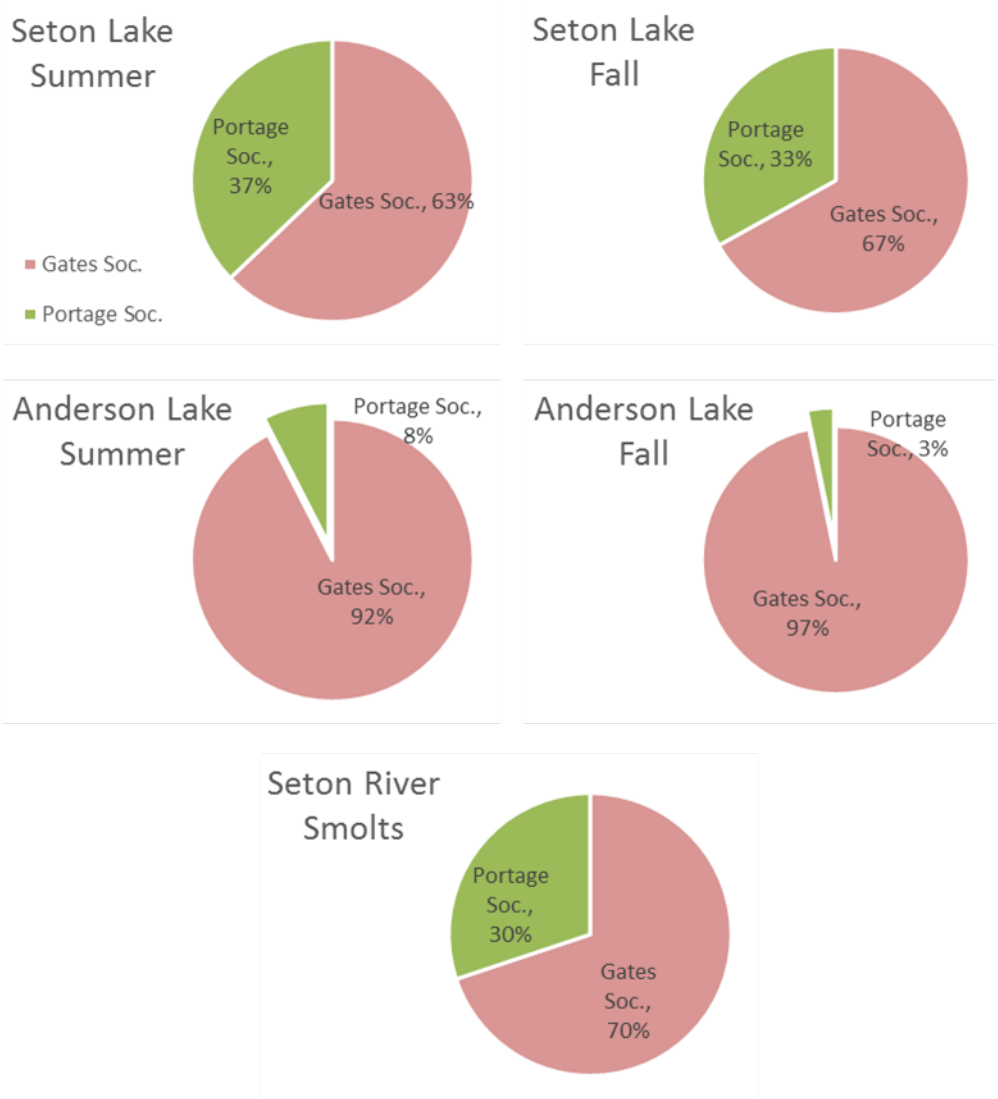


Figure 37 Classification using DNA of juvenile Sockeye samples only from Seton and Anderson lakes in the summer and fall of 2014 and from the Seton River smolt run in the spring of 2015.

4.4.3 Age Determination of Sockeye and Gwenish

In all years of sampling, scale readings indicated that in both lakes most *O. Nerka* of the small size class were age-0 or age-1, however, the age composition of Gwenish was somewhat confounded by their unusual life history pattern in both lakes, especially in Anderson Lake. Uncertainty about ages of Gwenish was likely due to the relatively small size of the Gwenish spawners as compared to the Sockeye and their probable smaller egg size; the late spawning date of the Gwenish, especially in Anderson Lake, and their probable subsequent late fry emergence dates. These factors result in relatively small Gwenish in their first two years of life and cause difficulties in determining age from scales. Uncertainty about age did not appear to be a problem for Sockeye Salmon.

In both the summer and fall surveys of Anderson Lake there is some overlap in the size of *O. Nerka* identified as age-0 or age-1 (Figure 38). The minimum fork length for scale formation by O. nerka is between 36 - 40 mm (Gilbert 1913, Foerster 1929, Clutter and Whitesel 1956). Using this criteria, many age-0 Gwenish in Anderson Lake would not have formed scales by the time of the fall surveys because on average 43% of the *O. Nerka* trawl catch (range from 2000-2003 and 2014 = 23-58%) were ≤38 mm (Table 15). A large portion of this size group would have formed few if any circuli before the end of their first growing season, so their first annulus would not appear until the end of their second growing season, and they would be one year older than indicated by their scale age (incorrectly aged as age-0 when actually age-1). In 2014, all fish ≤38 mm during the summer and fall surveys were Gwenish (Figure 39, Figure 40), and, although Sockeye and Gwenish were only distinguished in that year, it seems reasonable to assume that fish of this size were also Gwenish in other years. It seems likely, therefore, that a significant proportion of the Gwenish in Anderson Lake could be one year older than indicated by their scale age, and we suspect that the 60-69 mm long fish in the 2014 summer survey that were aged as age-0 were actually age-1 (Figure 38). Further evaluation that would be needed to verify this speculation, such as a cross comparison of ages from scales and otoliths of selected fish, was beyond the scope of this study.

Under-aging of Gwenish in Seton Lake is likely to occur less often, as Gwenish spawning occurs somewhat earlier there, probably resulting in an earlier emergence date and larger fry by the time of the surveys (Figure 38). In 2014, Seton Lake age-0 Gwenish were larger than those in Anderson Lake and on average only 25% (range =1 to 81%) of the summer trawl catch were ≤38mm. By the time of the fall survey <1% were ≤38 mm in any of the four years examined. Most fry in Seton Lake 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.

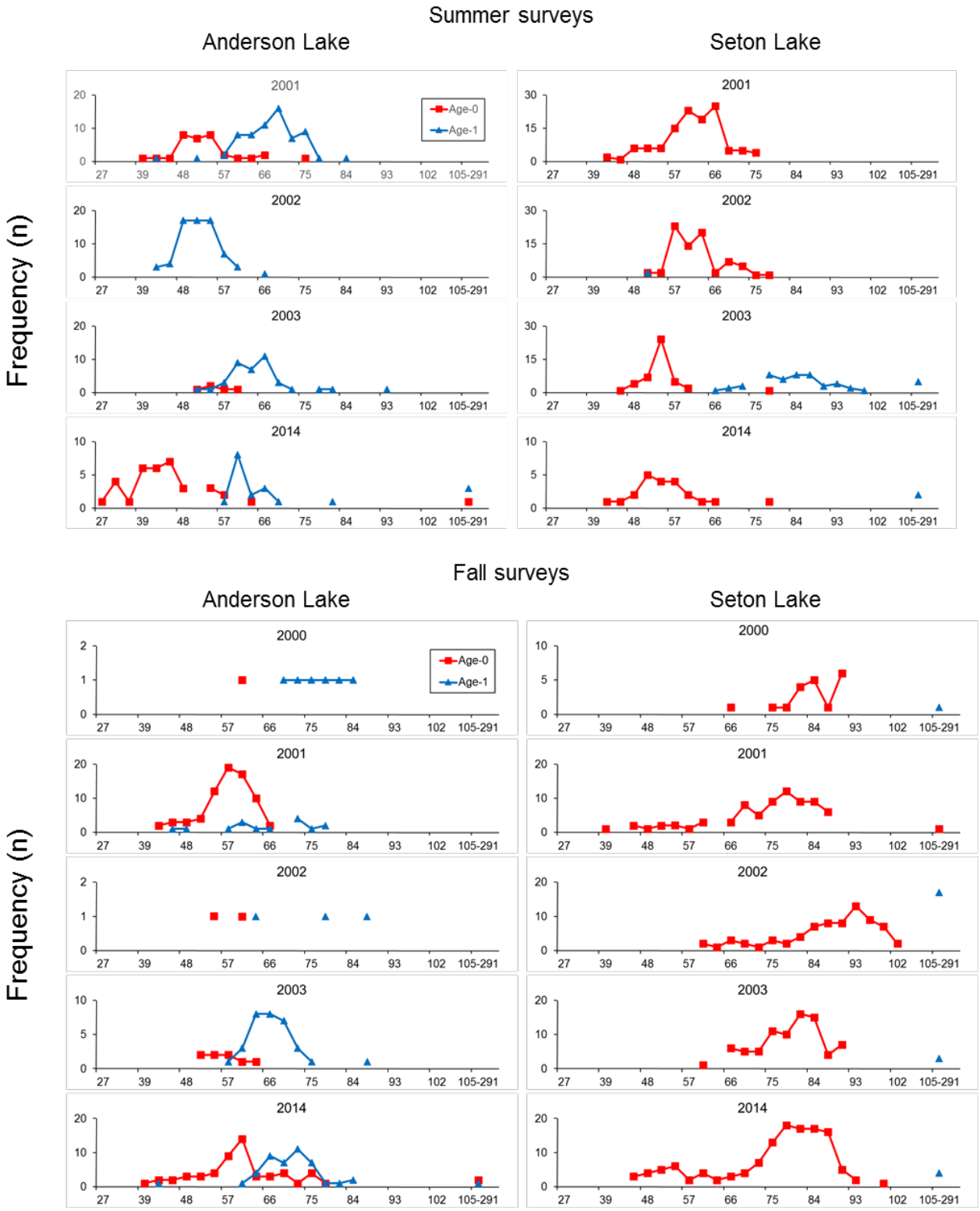


Figure 38 Comparison of the size of O. nerka aged from scales as age-0 and age-1 from 2000 to 2014.

Table 15. Catch of small O nerka unlikely to have formed scales (≤ 38 mm) at the time of capture in the summer and fall surveys of Seton and Anderson lakes.

	Seton Lake				
Sample Year	2000	2001	2002	2003	2014
Summer Surveys					
Fork length ≤ 38 mm		15	599	2	35
Fork length ≥ 39 mm		190	145	379	309
Proportion ≤ 38 mm		7%	81%	1%	10%
Fall Surveys					
Fork length ≤ 38 mm	0	1	1	0	4
Fork length ≥ 39 mm	40	145	214	109	469
Proportion ≤ 38 mm	0%	1%	0%	0%	1%

	Anderson Lake				
Sample Year	2000	2001	2002	2003	2014
Summer Surveys					
Fork length ≤ 38 mm		194	78	138	370
Fork length ≥ 39 mm		189	90	236	305
Proportion ≤ 38 mm		51%	46%	37%	55%
Fall Surveys					
Fork length ≤ 38 mm	253	125	48	107	152
Fork length ≥ 39 mm	270	236	51	77	510
Proportion ≤ 38 mm	48%	35%	48%	58%	23%

4.4.4 O. Nerka Stock ID and age interactions

In the 2014 summer survey of Anderson Lake, there were three clear size groups of small *O. Nerka* with modes at 27, 42, and 60 mm (Figure 39). DNA analysis identified members of the smallest modal group (range 23 – 34 mm) as Gwenish. Members of the second modal group (36 to 48 mm) were classified as Gates Sockeye, In addition the three largest fish classified as Gates Sockeye (48 – 59 mm) were a component of the third modal group. Most members of the third modal group (51 to 72 mm) were also classified as Gwenish. All *O. Nerka* from the first two modal groups that were aged from scales (27-48 mm) were classified as age-0. Most members (71%) of the third modal group that were aged were classified as age-1 but six fish (<63 mm) were classified as age-0. Age-1 fish this small or smaller were identified in all previous summer and fall surveys of Anderson Lake but were rarely observed in Seton Lake (Figure 38).

Three groups were still observable in the fall 2014 samples from Anderson Lake but they were somewhat larger, with modes at 39, 60, and 69 mm (Figure 40) indicating apparent

growth. The smallest group (31 - 52 mm) were identified by DNA and scale reading as age-0 Gwenish. The middle group overlapped with the large group and most of the smallest fish (51-63 mm) in the combined range were identified as age-0 Gates Sockeye (67%) but there were also some age-0 and age-1 Gwenish in this range. The larger fish (60-84 mm) in this combined group was a combination of age-0 and age-1 (59%) Gwenish.

In the summer survey of Seton Lake during 2014 there were only 2 distinct groups of small *O. Nerka* with modes at 33 mm (range 27- 39 mm) and 54 mm (40- 81 mm, Figure 41). All sampled fish were classified from scales as age-0. The smallest group were classified as Gwenish. The larger group was primarily classified as a mix of Gates (61%) and Portage (36%) Sockeye, plus 2% Gwenish that were some of the larger fish in this group (75 &81 mm). No *O. Nerka* between 81 and 129 mm were captured. All of the *O. Nerka* ≥ 130 mm that were sampled were classified as Gwenish. Most ranged from age 2 to 4; two age-1 fish that were ≥135 mm may have been mis-aged.

There were still two distinct groups in the 2014 fall sample from Seton Lake with modes at 48 mm (range 36 – 60 mm) and 81 mm (63 – 99 mm) corresponding to the summer modes but with some growth (Figure 42) again all sampled fish <100 mm were classified as age-0. In the smaller group 88% of the fish were classified as Gwenish but a few of the larger ones were Portage Sockeye. Most of the larger group were Sockeye from Gates Creek (79%) and Portage Creek (29%) with the remainder being Gwenish. All of the *O. Nerka* over 121 mm were Gwenish aged 2-4.

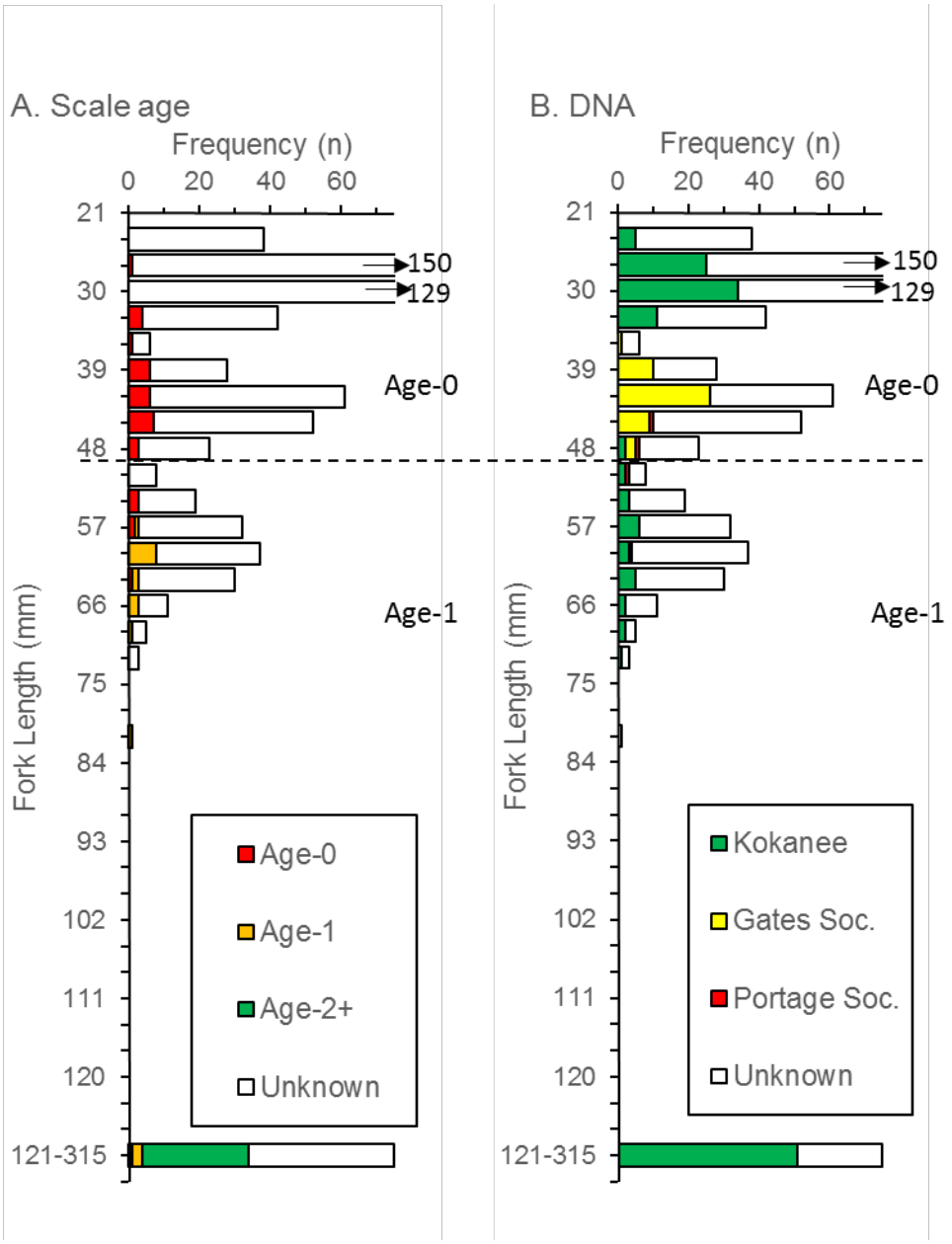


Figure 39 Results of scale ageing and DNA stock determination of selected O. nerka (solid shading) captured in midwater trawls and gillnets during the 2014 summer survey of Anderson Lake. While there is overlap between the age groups the approximate separation point between age-0 and age-1 fish is shown for reference (dotted line).

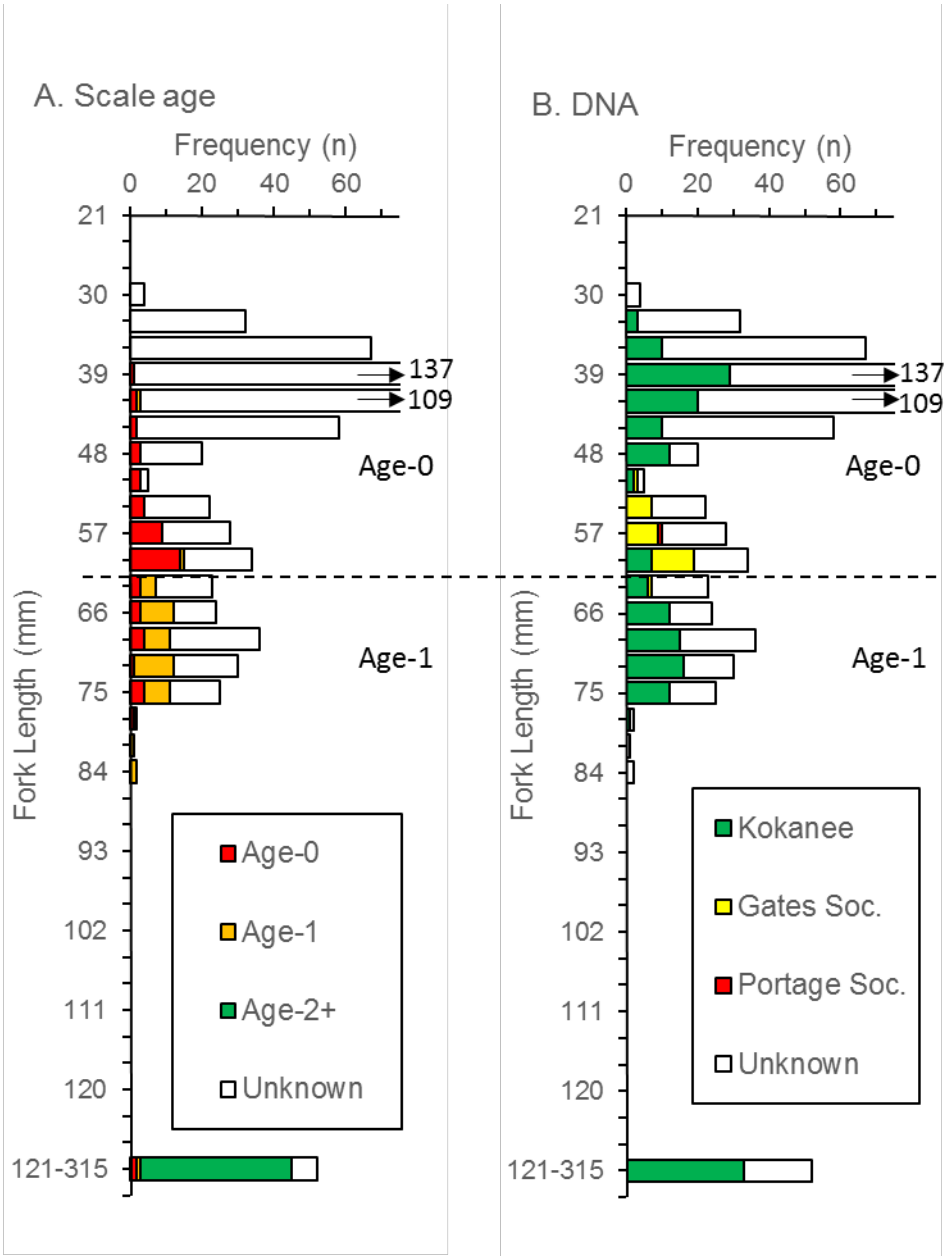


Figure 40 Results of scale ageing and DNA stock determination of selected O. nerka (solid shading) from Anderson Lake, captured in midwater trawls and gillnets during the 2014 fall survey. While there is overlap between the age groups the approximate separation point between age-0 and age-1 fish is shown for reference (dotted line).

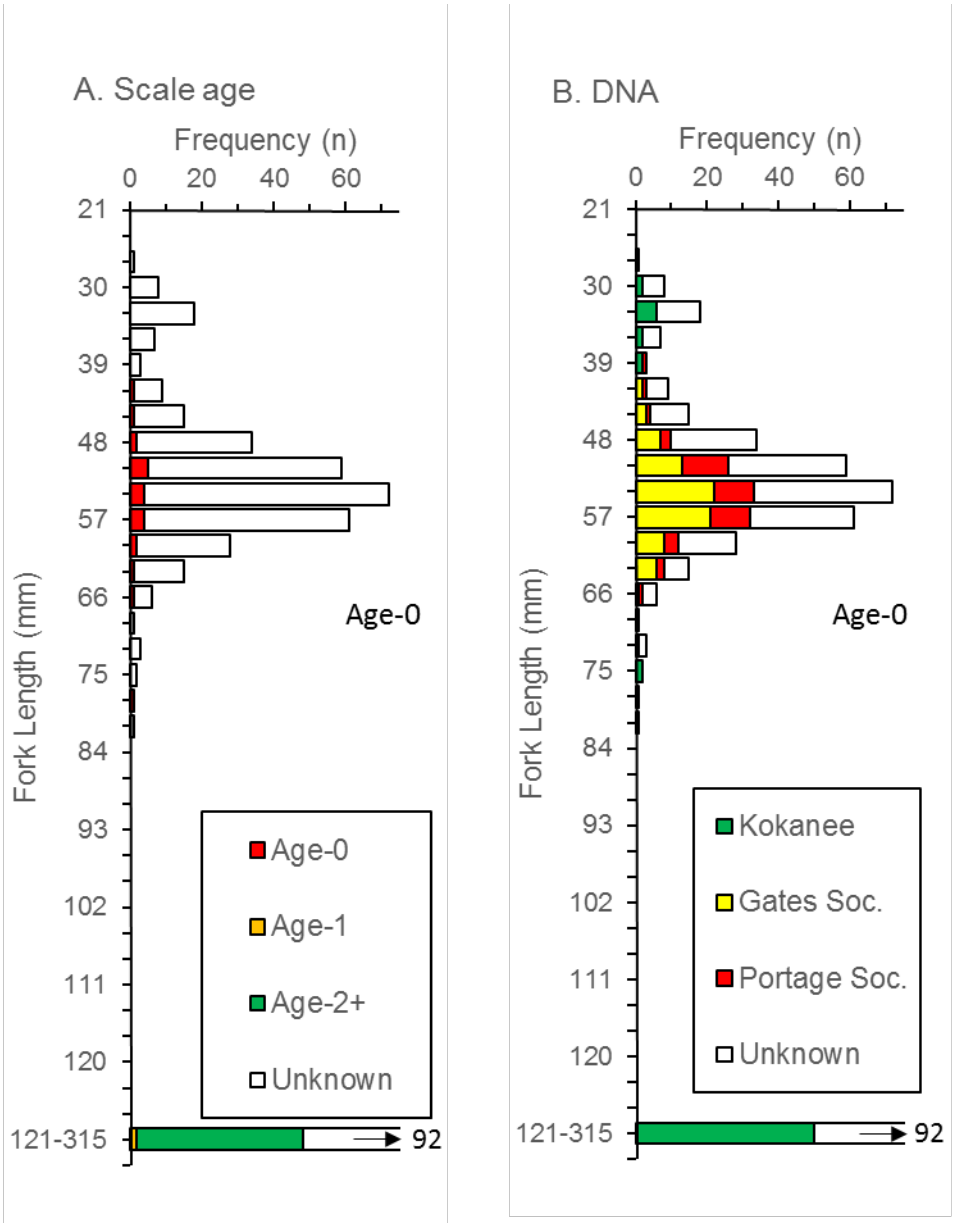


Figure 41 Results of scale ageing and DNA stock determination of selected O. nerka (solid shading) from Seton Lake, captured in midwater trawls and gillnets during the 2014 summer survey.

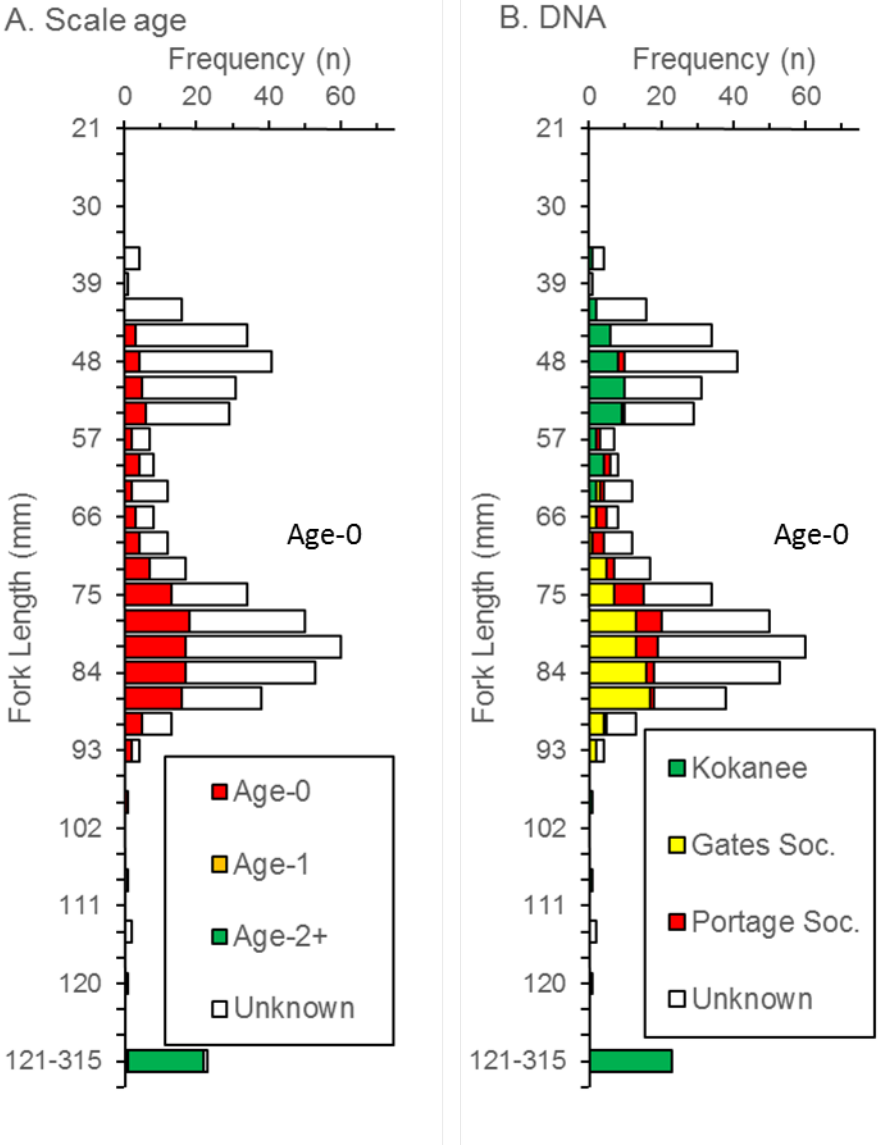


Figure 42 Results of scale ageing and DNA stock determination of selected O. nerka (solid shading) from Seton Lake, captured in midwater trawls and gillnets during the 2014 fall survey.

To test the hypothesis that there is a growth advantage to rearing in Seton Lake we compared the length and weight of the Sockeye rearing in the two lakes in 2014. For this comparison we only used fish that were identified by DNA analysis as Sockeye because considerable overlap in size and age between the various stocks of *O. Nerka* made it difficult to reliably classify the many fish that were not aged or identified (none of the fish that were aged were also identified using DNA). Overall, Sockeye captured in the summer and fall in Seton Lake were significantly longer and heavier than those rearing in Anderson Lake, averaging 40% longer and over 3 times heavier in the fall (Table 16, T-test, P<0.001).

We conducted two tests to determine if the differences were due to different growth potentials of the two stocks. We first compared the size of Gates Creek Sockeye rearing in Seton Lake to those from Portage Creek also rearing in Seton Lake. We found no significant differences in the summer samples; in the fall the Gates Creek Sockeye were significantly longer (11%, P<0.001) and heavier (36%, P<0.001, Table 17) than the Portage Creek Sockeye. Eliminating the Portage fish (the much less numerous stock in both lakes) from the comparison showed that Gates Creek Sockeye grew significantly bigger in Seton Lake than in Anderson Lake (Table 18, P<0.001). In the fall samples, they were longer by 42% (83 vs. 57 mm) and were heavier by an astounding 360% (4.8 vs. 1.3 g). Much of this difference occurred between the summer and fall surveys when length increased by 49% in Seton Lake as compared to 35% in Anderson Lake while weight increased by 263% in Seton Lake vs. 101% in Anderson Lake (Table 18).

Table 16 Comparison of the size of all DNA identified juvenile Sockeye (Portage and Seton stocks combined) between lakes.

		Length (mm)		Weight (g)	
Season	**Descriptor**	**Anderson**	**Seton**	**Anderson**	**Seton**
Summer	Mean	43.2	54.5	0.69	1.33
"	N	53	132	53	132
"	SE	0.48	0.46	0.03	0.04
"	T-test	16.95		13.29	
"	P	<0.001		<0.001	
Fall	Mean	57.3	78.5	1.32	4.36
"	N	31	121	31	121
"	SE	0.51	0.78	0.04	0.13
"	T-test	22.75		22.96	
"	P	<0.001		<0.001	

Table 17 Comparison of the size of Gates and Portage creek Sockeye in Seton Lake only.

		Length (mm) by Sockeye stock		Weight (g) by Sockeye stock	
Season	**Descriptor**	**Gates**	**Portage**	**Gates**	**Portage**
Summer	Mean	54.5	54.4	1.31	1.36
"	N	83	49	83	49
"	SE	0.55	0.83	0.04	0.06
"	T-test	0.14		0.55	
"	P	0.89		0.58	
Fall	Mean	81.3	72.9	4.78	3.52
"	N	81	40	81	40
"	SE	0.69	1.57	0.13	0.23
"	T-test	4.88		4.85	
"	P	<0.001		<0.001	

Table 18 Comparison between lakes of the size of Gates Creek Sockeye only.

		Length (mm) by lake		Weight (g) by lake	
Season	**Descriptor**	**Anderson**	**Seton**	**Anderson**	**Seton**
Summer	Mean	42.5	54.5	0.66	1.31
"	N	49	83	49	83
"	SE	0.34	0.55	0.03	0.04
"	T-test	18.58		12.66	
"	P	<0.001		<0.001	
Fall	Mean	57.4	81.31	1.33	4.78
"	N	30	81	30	81
"	SE	0.52	0.69	0.04	0.13
"	T-test	27.62		25.3	
"	P	<0.001		<0.001	

4.4.5 Smolt run Stock ID

All fish sampled from the spring 2015 smolt run for DNA identification were classified as either Gates Creek Sockeye (70%) or Portage Creek Sockeye (30%, Table 14, Figure 37). None of the smolts were aged but they are probably all age-1 as: a) no age-1 Sockeye were detected in the lake in 2014; b) as discussed earlier, all of the DNA identified smolts were likely too small to be age-2 and; c) scales from returning adult between 1968 – 2006 (39 years) indicate that on average age-2 smolts comprised 0.3% of the Gates returns and 1.4% of the Portage returns. It is therefore likely that all the smolts were age-1.

Gates smolts averaged 91.2 mm (± 95%CI = 1.3 mm, Table 19) and 5.8g (±0.22 g) while the Portage smolts were not significantly different at 89.1 mm (±1.8 mm) and 6.0 g (±0.32 g, T-tests, P>0.05, Figure 43). The combined means were 89.6 mm and 5.76 g. The Portage Creek smolts, which almost all reared in Seton Lake, were 2.5 g (71%) larger than in the Seton Lake fall samples, whereas Gates smolts, which reared in both lakes more evenly, averaged 1.0 g (21%) and 4.5 g (336%) larger than weights in the respective fall fry samples from Seton and Anderson Lakes. It is probable that the size differential between sockeye stocks in the Seton Lake fall sample (Gates fish were significantly larger) was not reflected in the smolt sample because Gates smolts were a mix of large fish from Seton Lake and smaller fish from Anderson Lake. Unfortunately, we had no means of determining the rearing lake of individual Sockeye smolts. While there was considerable overlap in the migration timing of Gates and Portage creek smolts, the early part of the run (<02-May) was mostly Gates Creek Sockeye (Figure 43).

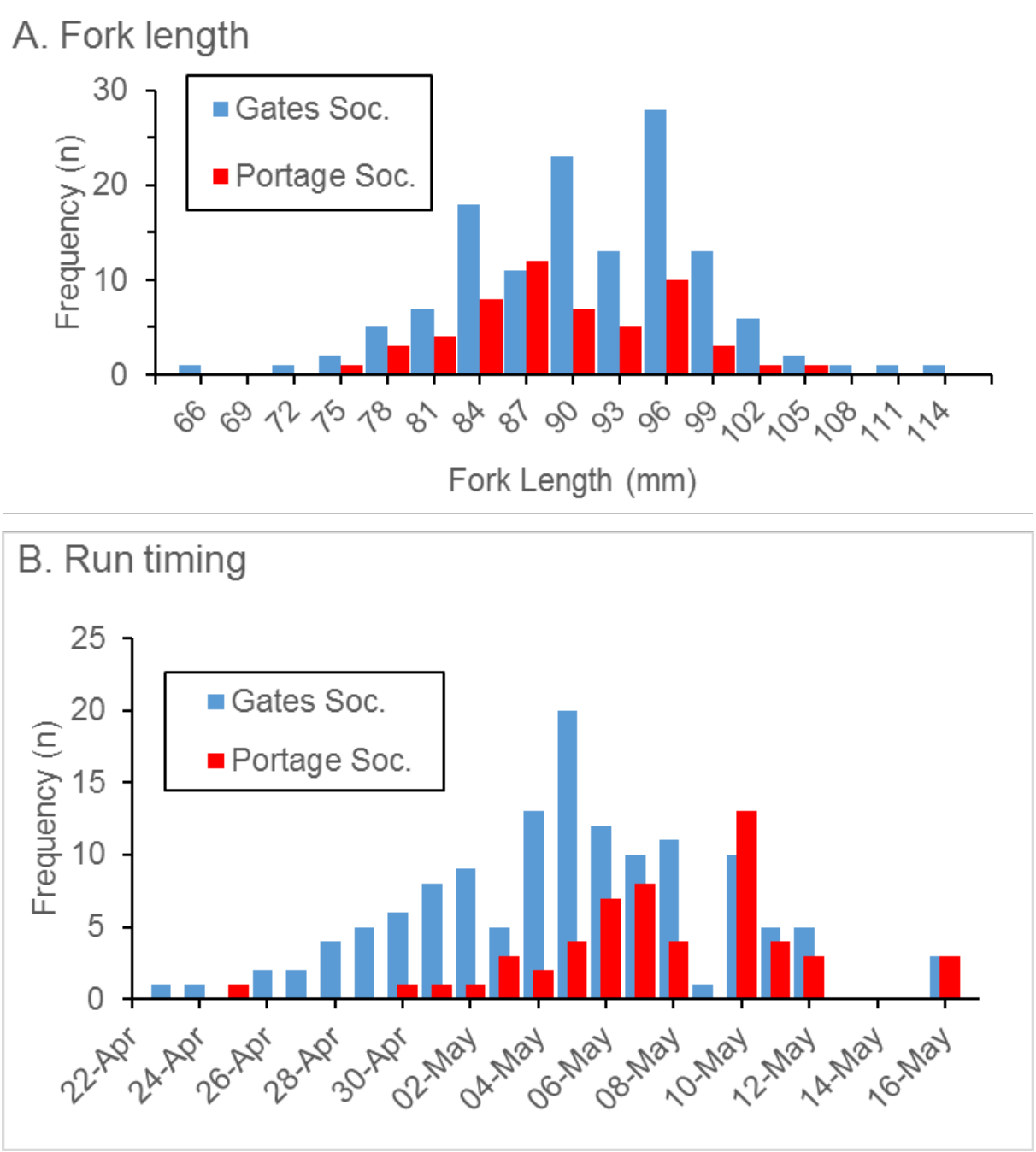


Figure 43 Size and run timing of 2015 smolts identified by DNA analysis.

Table 19 Comparison of the size of Gates and Portage creeks origin Sockeye smolts.

	Length (mm) by lake		Weight (g) by lake	
Descriptor	Gates	Portage	Gates	Portage
Mean	91.2	89.2	5.8	6.0
N	134	55	131	54
SE	0.67	0.88	0.11	0.16
T-test	1.84		0.8	
P	0.07		0.43	

4.4.6 O. Nerka Diet

Macrozooplankton was nearly the only prey type in the diet of all size classes of *O. nerka* examined in this study and in the surveys done in previous years (Figure 45). The macrozooplankton in the diet was mainly comprised of four taxa; the cladocerans *Daphnia* sp. and the smaller *Bosmina* sp., as well as Calanoid copepods, and the smaller Cyclopoid copepod, *Diacyclops* sp. Other organisms such as terrestrial insects and Chironomid larvae comprised from 0 to 1.6% of the stomach contents in any of the five years examined.

In the summer of 2014, the diet of *O. nerka* of the small size class (≤100 mm) was very similar in the two lakes, with the cladoceran *Daphnia* comprising 95% of the stomach contents in both lakes (Figure 45a, Table 20). Many fish in both lakes also had small numbers (≤2.5%) of the smaller cladoceran, *Bosmina*. The small copepod, *Diacyclops*, occurred in low numbers in Seton's age-0 *O. nerka*, while *Diacyclops* and the larger Calanoid copepods were found in very low numbers in about one third of small size class *O. nerka* stomachs in Anderson Lake.

In the fall of 2014, *Daphnia* were not as predominant in the diet of small *O. nerka* in either lake (Figure 45b). Although almost all stomachs (97%) contained *Daphnia,* they comprised only 57% of the stomach contents in Seton Lake small *O. nerka* and 44% in Anderson lake fish (Table 20). Diacyclops were also common in Seton Lake stomachs, comprising 42% of the diet, while in Anderson Lake, *Bosmina* were the predominate prey items (53%) in the stomachs of small *O. nerka*. No *Bosmina* were found in small *O. nerka* captured in Seton Lake, while low numbers of Calanoids were found in many Anderson Lake fish. A few Chironomids were found in the stomachs of age-0 *O. nerka* from both lakes.

The fall diet of older (age-1-3) Gwenish of the medium size class (>100 mm) was very similar to that of small size class O. nerka found in the same lake (Figure 45, Table 20). *Daphnia* comprised 56% of stomach contents of medium size class Gwenish in Seton Lake and 64% in Anderson Lake. *Diacyclops* were common in these fish in Seton Lake, comprising 42% of the diet, while no *Bosmina* were found in their stomachs. *Bosmina* were common in the diet

of Anderson Lake Gwenish (33% of contents) while a few Calanoid copepods (2%) were found in most of their stomachs.

To investigate the variability in the diet of *O. nerka* of the small size class we compared the 2014 results with those of stomach samples taken in the fall of 2000 and in the summer and fall of 2001 to 2003 (DFO, Lakes Research Program, Cultus lake Salmon Research Laboratory, Data on file). There were notable differences between Seton and Anderson lakes in the annual variation of prey items found in the stomachs of small *O. Nerka* in both species composition and in the mean number of prey items per stomach (Figure 44, Table 20). The diet of small *O. Nerka* in Seton Lake, in particular, showed considerable annual variation in species composition. For example, the most common prey item in Seton Lake in the summer was *Diacyclops* in 2002 and 2003 but was *Daphnia* in 2001 and 2014. *Daphnia* comprised only 4-17% of the diet in 2002 and 2003, versus ≥ 90% of the 2001 and 2014. In Seton Lake fall samples, *Bosmina* comprised 90% of the 2000 diet but were less than 5% of the diet in the other four years, which were dominated by *Daphnia* and Cyclopoids. There was relatively little variation in the Anderson Lake summer samples, where *Daphnia* was consistently the most common prey item, ranging from 51% in 2002 to 95% in 2014. In contrast, in Anderson Lake fall samples the most common prey item in 2000 and 2001 was *Daphnia* (91% and 89%), in 2002 and 2003 it was Cyclopoids (46% and 52%), and in 2014 if was *Bosmina* (53%).

In Seton Lake, the mean number of prey items/stomach for O. nerka of the small size class varied annually from around 250 to over 1,000 in both the summer and fall (Figure 46 A and B). There was less annual variation in mean prey/stomach in Anderson Lake, where mean prey numbers varied from 190 to 350 in the summer and from 200 to 530 in the fall. The mean number of prey items per fish was larger in Seton Lake than in Anderson Lake in all four years in the summer survey and four of five years in the fall surveys (Figure 46 A and B). Combining all years, there were no significant differences between the number of prey per fish in Seton Lake during the summer ($N=4$, $T=2.07$, $P= 0.06$) or fall ($N=5$, $T=1.20$, $P= 0.15$) but when seasons and years were combined, Seton averaged significantly more prey per fish (Figure 46C, Seton mean = 545, Anderson mean = 331, $N=9$,$T=2.31$, $P=0.02$). A similar analysis conducted on the major prey categories separately (*Daphnia*, *Bosmina*, Cyclopoids, and Calanoids) found no significant differences between lakes either within each season or within both seasons combined ($T<1.5$, $P>0.10$).

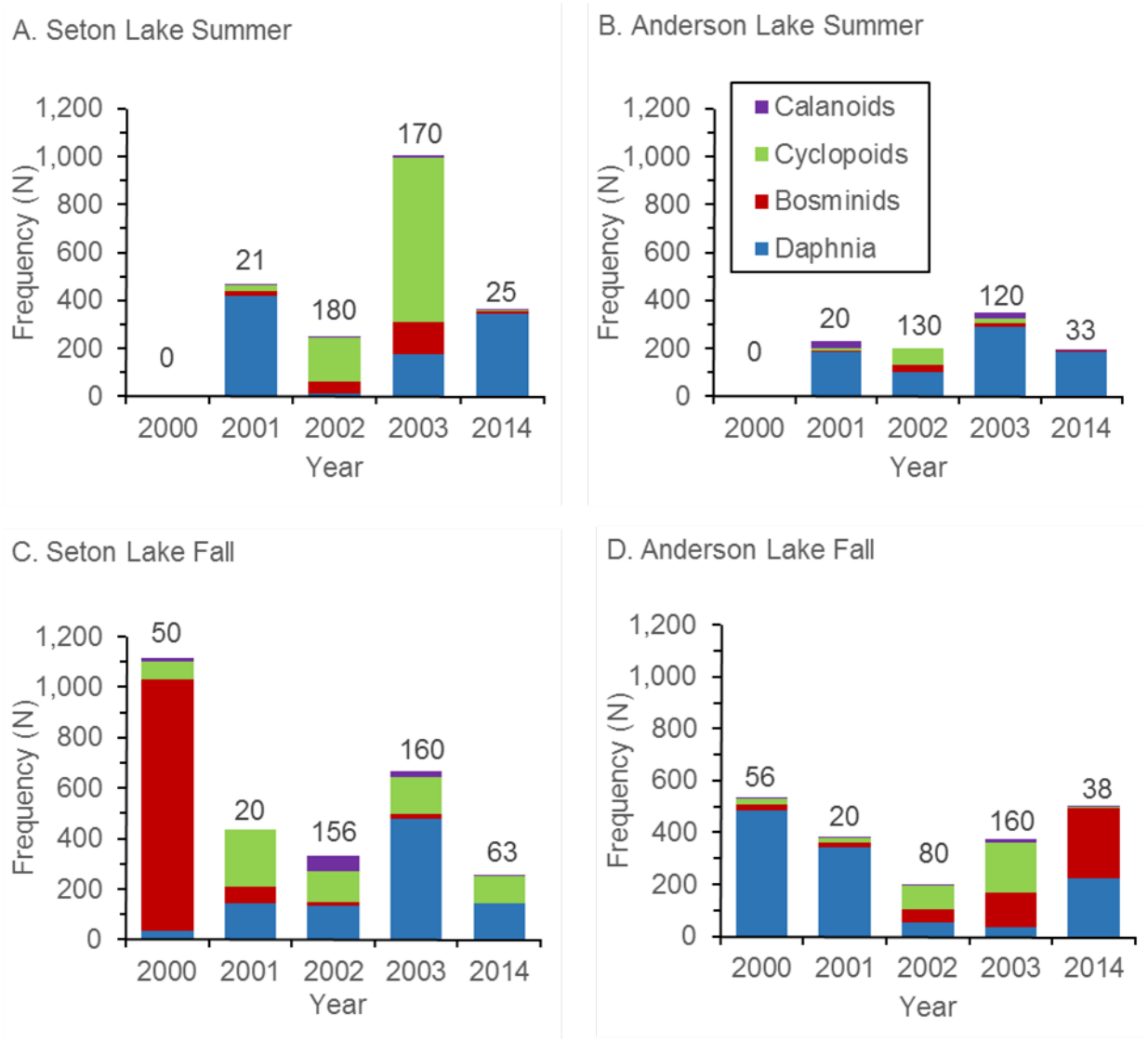


Figure 44 Prey abundance in the diet of small O. nerka (≤ 100 mm in length) from Anderson and Seton lakes during summer and fall surveys in all years sampled. Non-zooplankton (terrestrial insects and chironomids) and other zooplankters (*Scapholoberis* and *Diaphanosoma*) were rare and comprised less than 1.5% of the diet in all years. Sample size (stomachs) is shown atop each bar.

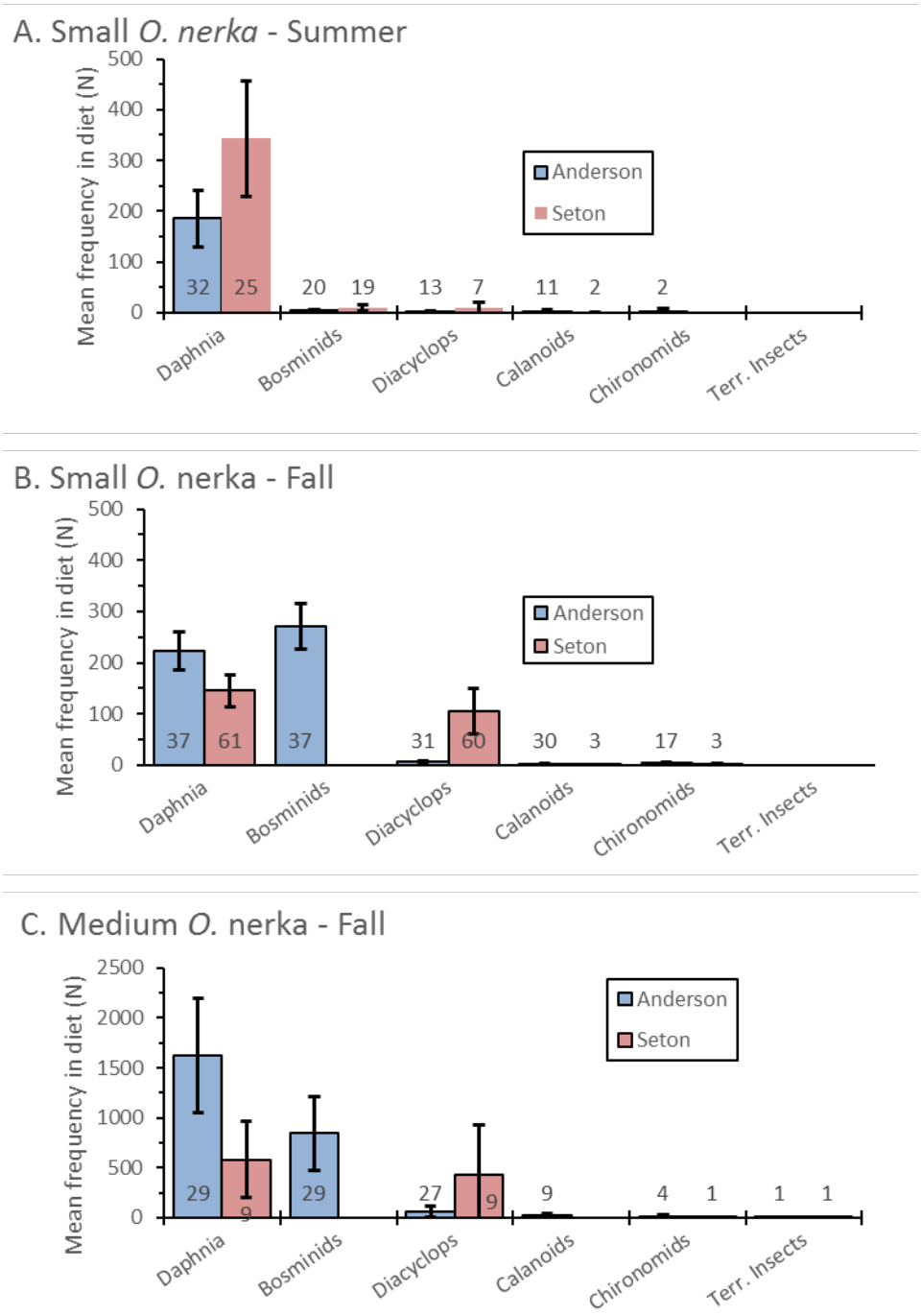


Figure 45 Diet of small (≤100 mm, A, B) and medium (>100 mm, C) size class *O. nerka* captured in the summer and fall 2014 surveys of Anderson and Seton lakes. ±95% CI (vertical lines) and numbers of stomachs containing each prey category are shown on bars. The diet of the medium size class was only sampled during fall surveys.

Table 20 Diet of small (≤100 mm) and medium (>100 mm) size class O. nerka captured in 2014 summer and fall surveys of Anderson and Seton lakes.

			Anderson Lake			Seton Lake		
Size Class	Season	Prey Taxa	N stomachs	Mean # of prey	±95% CI	N stomachs	Mean # of prey	±95% CI
Small	Summer	Daphnia	32	186.0	56.2	25	343.0	114.0
"	"	Bosminids	20	3.3	1.3	19	9.2	5.6
"	"	Diacyclops	13	2.0	0.5	7	9.1	10.5
"	"	Calanoids	11	2.6	1.6	2	1.0	0.0
"	"	Chironomids	2	1.5	6.4	0		
"	"	Terr. Insects	0			0		
"	"	Total[a,b]	33	195.0		25	362.0	
"	Fall	Daphnia	37	222.0	37.7	61	145.0	30.9
"	"	Bosminids	37	270.0	44.4	0		
"	"	Diacyclops	31	6.5	1.7	60	106.0	44.9
"	"	Calanoids	30	2.7	0.7	3	1.0	0.0
"	"	Chironomids	17	3.9	2.0	3	1.3	1.4
"	"	Terr. Insects	0			0		
"	"	Total[a,b]	38	505.0		63	253.0	
Medium	Fall	Daphnia	29	1,626.0	575.0	9	580.0	383.0
"	"	Bosminids	29	844.0	372.0	0		
"	"	Diacyclops	27	58.9	52.1	9	435.0	495.0
"	"	Calanoids	9	21.3	16.9	0		
"	"	Chironomids	4	7.5	19.6	1	9.0	
"	"	Terr. Insects	1	1.0		1	8.0	
"	"	Total[a,b]	29	2,558.0		9	1,032.0	

[a]Total fish examined.
[b]Sum of mean # of prey items.

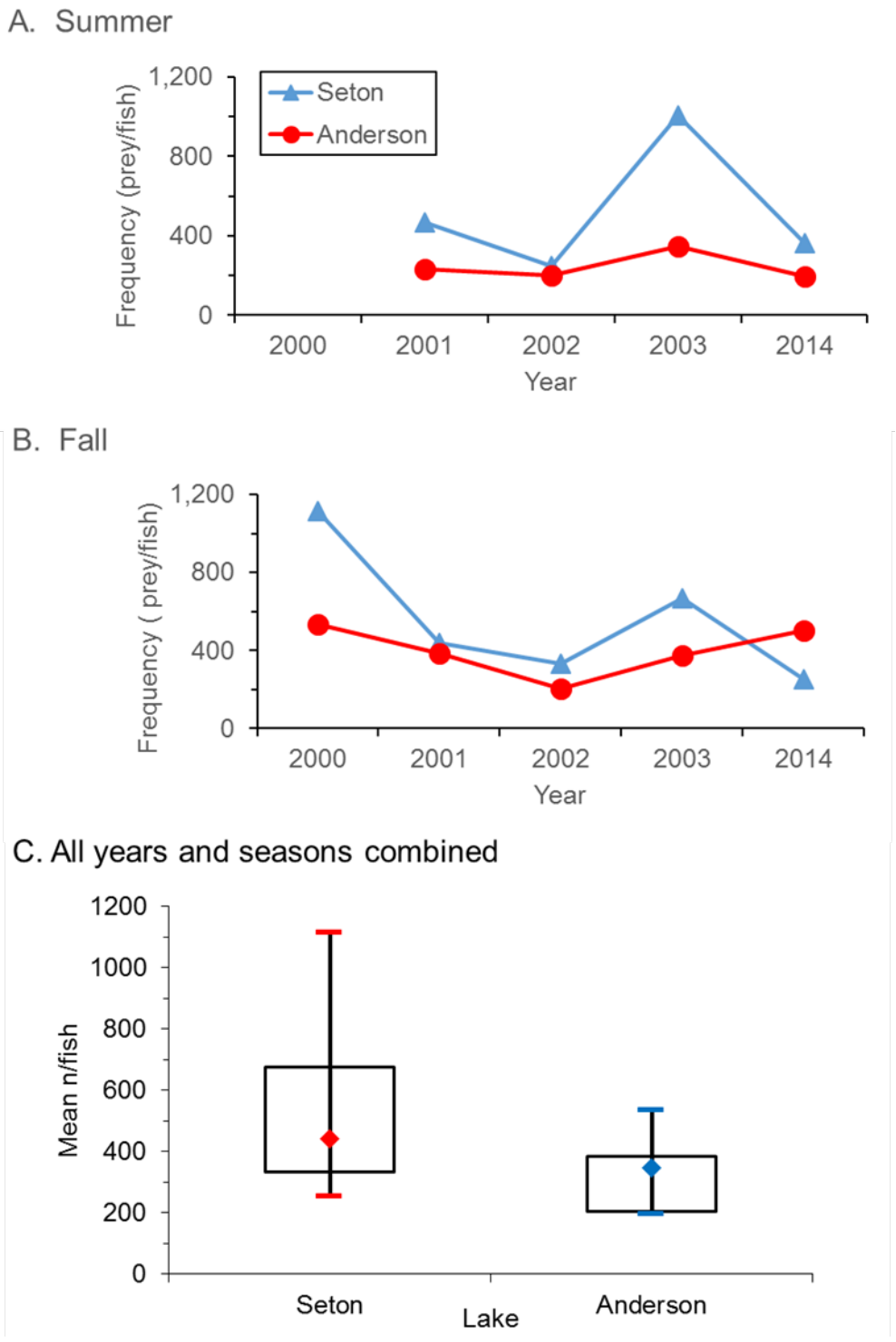


Figure 46 Annual variation in the mean number of prey items per stomach in the diet of small *O. Nerka* (≤ 100 mm in length) in the (A) summer and (B) fall diet samples, and (C) box plots of all years and seasons combined.

4.4.7 Piscivore Catch and Diet

We captured two piscivorous fish species - Bull Trout and Northern Pikeminnow – in each lake during summer and fall 2014 pelagic trawling and gill netting. Sample sizes were small, with a total of 35 stomachs examined from potential piscivores from Anderson Lake and only 15 from Seton Lake (Table 21). Bull Trout size range was similar in the two lakes in both seasons, ranging from 285 to 750 mm overall (Table 21). Rainbow Trout are also found in the lakes but they appear to not be significant piscivores in the pelagic zone, as only one was caught in our pelagic gill nets and it had not fed on fish. The only other large fish captured in the lakes were four adult pre-spawn Sockeye Salmon which typically do not feed after re-entering freshwater and are not piscivorous in any case.

Table 21 Fork length (mm) of piscivorous fish captured in Seton and Anderson Lakes in summer and fall 2014.

		Anderson Lake					Seton Lake				
Species	**Season**	**N**	**Mean**	**±95%CI**	**Min**	**Max**	**N**	**Mean**	**±95%CI**	**Min**	**Max**
Bull Trout	Summer	18	421.3	31	306	559	9	516.8	98.9	294	674
"	Fall	9	438.8	95.2	285	650	4	660	101.5	600	750
N. pikeminnow	Summer	6	334.3	20.5	300	352					
"	Fall	2	326	330.4	300	352	2	301.5	1149.9	211	392

Fish were found in 93% of the stomachs of Bull Trout from Anderson Lake and in 88% of those from Seton Lake (Table 22). The sample size of Northern Pikeminnow was small, but 88% of the Northern Pikeminnow captured in Anderson Lake had fish in their stomachs while only one of the two Northern Pikeminnow captured in Seton Lake contained fish. Northern Pikeminnow lack an esophageal sphincter and often regurgitate fish on capture, potentially resulting in an underestimate of their piscivoury (Brown and Moyle 1981). Because of digestion, not all of the 96 fish in the piscivores stomachs could be identified to species. Of the 46 that could, all but one (a sculpin) was an *O. Nerka* and as all 50 of the unidentified fish were readily determined to be salmoniform (i.e. not sculpins) they were also assumed to be *O. Nerka*.

Table 22 Number of potential predator stomachs examined and the number of stomachs empty of fish.

		Anderson		Seton	
Predator	Season	Total stomachs	Containing fish	Total stomachs	Containing fish
Bull Trout	Summer	18	16	9	7
	Fall	9	9	4	4
	Combined	27	25	13	11
N. pikeminnow	Summer	6	6	0	0
	Fall	2	1	2	1
	Combined	8	7	2	1
Total	Summer	24	22	9	7
	Fall	11	10	6	5

In Anderson Lake in both the summer and fall, small *O. Nerka* were the most common prey item in the stomachs of Bull Trout (Table 23). They were found in 78% of the stomachs in the summer and 89% of the fall stomachs, averaging 1.4 fish/stomach (f/s) in the summer and 1.7 f/s in the fall. Medium sized *O. Nerka* were found in only 11% of Bull Trout stomachs in the summer and fall. They were preyed on at a much lower rate in the summer (0.11 f/s) than were small sized *O. Nerka* (1.4 f/s), and the difference in these rates was significant (T=3.358, P=0.003). In the fall medium fish were preyed on at lower rate (0.22 f/s) than small fish (1.7 f/s), but the difference was insignificant (T=1.78, P=0.11).

Table 23. Diet of Bull Trout, Northern Pikeminnow, and Rainbow Trout captured in the 2014 summer and fall surveys of Anderson and Seton lakes.

			Anderson			Seton			Both Lakes			Both Lakes		
Species	Season	Prey	N stomachs	Mean # of prey	±95% CI	N stomachs	Mean # of prey	±95% CI	T	P		N stomachs	Mean # of prey	±95% CI
Bull Trout	Summer	Small *O. nerka*	14	1.44	0.82	3	0.67	0.43	1.604	0.12	ns	27	1.19	0.73
		Medium *O. nerka*	2	0.11	0.16	6	0.78	0.33	2.84	0.02	*	27	0.33	0.28
	Fall	Small *O. nerka*	8	1.67	1.84	1	0.25	0.38	1.69	0.12	s	13	1.23	1.60
		Medium *O. nerka*	1	0.22	0.34	4	6.75	2.69	3.7	0.03	*	13	2.23	2.77
		Sculpin	1	0.11								1	1.00	
N. pikeminnow	Summer	Small *O. nerka*	1	1.17	0.43	-						6	1.17	0.43
		Medium *O. nerka*	-	-		-						6	0.00	
		Terrestrial insects	1	0.33		-						1	2.00	
	Fall	Small *O. nerka*	1	0.50	6.35	1	0.50	6.35				4	0.50	5.19
		Medium *O. nerka*	-	-		-	0.00					4	0.00	
Rainbbow trout	Summer	Small *O. nerka*	-	-		-						1	0.00	
		Medium *O. nerka*	-	-		-						1	0.00	
		Terrestrial insects	1	50.00		-						1	50.00	

In Seton Lake, small *O. Nerka* were found in only 33% of Bull Trout stomachs in the summer and in 25% in the fall, while medium *O. Nerka* were found in 66% of summer stomachs and 100% of the fall stomachs (Table 23). There was no significant difference between the

summer predation rates on small (0.67f/s) and medium (0.78f/s) *O. Nerka* (T=0.305, P=0.76), however, in the fall, medium *O. Nerka* were preyed on at a much higher rate (6.75 f/s) than were small *O. Nerka* (0.25 f/s), and this difference was significant (T=3.678, P=0.03).

Comparing between lakes, predation rates on small *O. Nerka* were not significantly different in either the summer (T=1.60, P=0.12) or fall (T=1.69, P=0.12), while Seton Bull Trout had significantly higher predation rates on medium sized *O. Nerka* in both the summer (T=2.84, P=0.02) and fall (T=3.70, P=0.03). Too few stomachs were examined from Northern Pikeminnow to allow valid inter-lake or inter-season comparisons for this species.

All of the unidentified fish contained in piscivore stomachs were assumed to be small or medium *O. Nerka* (see explanation above) and 30 of them were intact enough to be measured and included in regression analysis. This analysis showed no evidence of any relationship between predator size and prey number (R^2 <0.29, P >0.05). However, prey size was positively related to predator size (Figure 47, R^2 = 0.73, P <0.01) and all of the small sized *O. Nerka* (≤100 mm) were eaten by the smaller Bull Trout (≤550 mm).

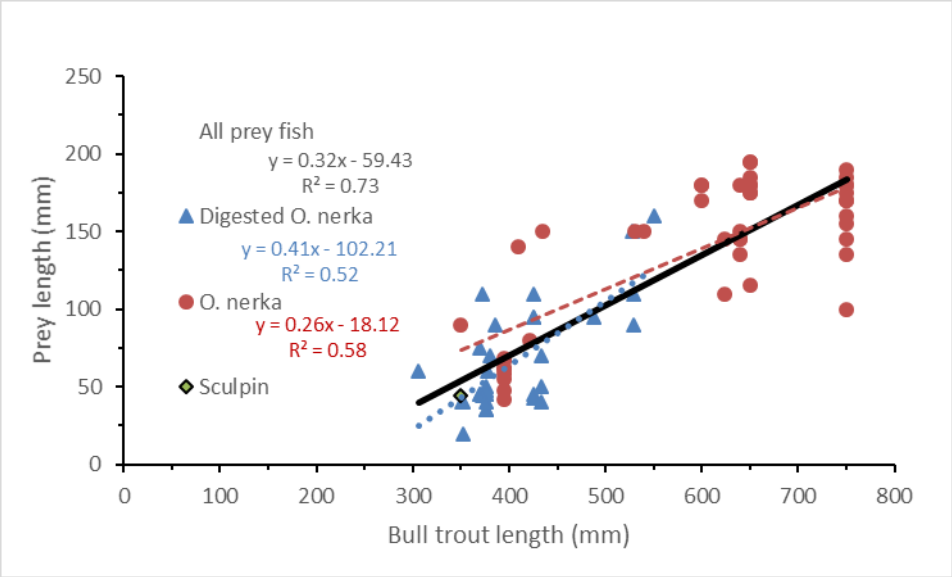


Figure 47 Relationship between length of prey fish in the stomach of Bull trout and the predator fish in Anderson and Seton lakes.

4.4.8 Whole-lake acoustic surveys

4.4.8.1 General spatial distribution patterns of fish

In both lakes, most pelagic fish occurred in midwater layers during the night time summer and fall whole-lake surveys, which is typical of Sockeye fry and Gwenish in lakes (Figure 48). The vertical extent of fish layers was greater (they were wider) in Anderson Lake than in Seton Lake during both surveys. In the summer, most fish were detected between 25 m and 50 m in Anderson Lake versus 10-30 m in Seton Lake. In the fall, most fish were 20-50 m in Anderson Lake compared to 20-40 m in Seton Lake. Areal densities of pelagic fish on individual transects (fish/ha, all sizes and species combined) tended to be lower in the western 1/3 of Anderson Lake in the summer (transects 1 & 2, lake section 1), whereas there was no clear longitudinal trend in the fall (Figure 49). In Seton Lake, areal densities were highest by far in the east end of the lake in the summer (transects 8 & 9, Lake Section 3), whereas densities were similar on all transects in the fall (Figure 49).

4.4.8.2 Total fish abundance estimates

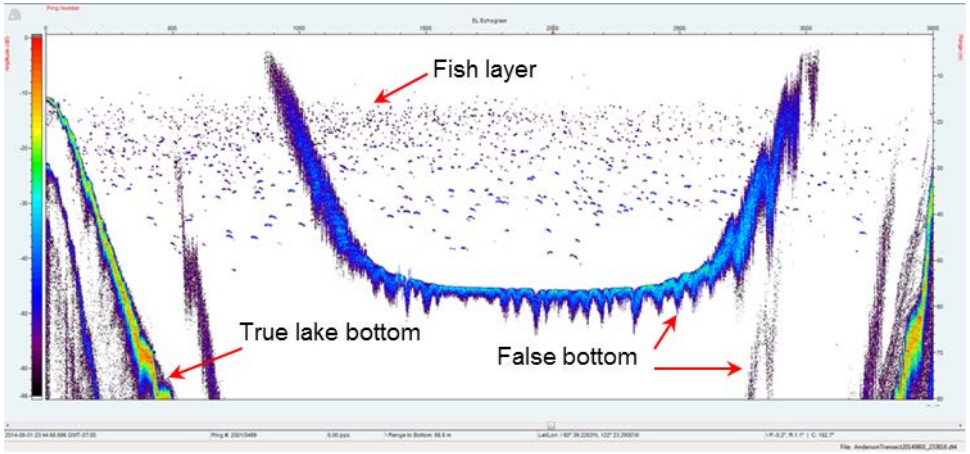
Total pelagic fish abundance (all sizes and species combined) was higher in Anderson Lake than in Seton Lake during summer and fall surveys, although the 95% confidence intervals of all surveys overlapped, suggesting that differences were not statistically significant (Table 24, Figure 50). Total pelagic fish abundance estimates for Anderson Lake were 5.4 million and 3.8 million (summer and fall), compared to 3.2 million and 2.6 million for Seton Lake (Table 24). The bound of error of these estimates (95% confidence intervals) ranged from ±14% to ±44% (Table 24).

4.4.8.3 Fish abundance estimates by size class

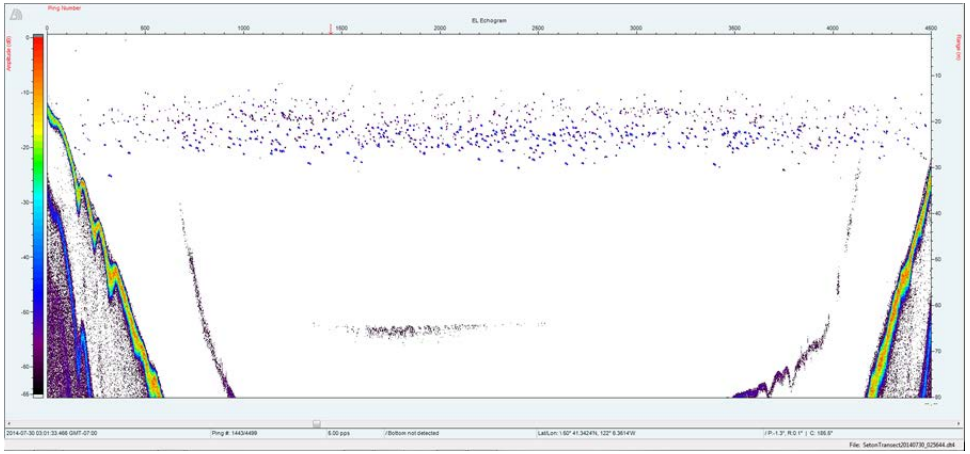
Target strength (TS) measurements that were obtained from 4,391 to 5,426 individual fish tracks per survey indicated that fish of a wide range of sizes were present on each survey. In Anderson Lake, the range of TS was -64.6 dB to -30.5 dB in the summer and -64.6 dB to -29.3 dB in the fall, compared to -64.6 dB to -30.5 dB in the summer and -64.6 dB to -29.3 dB in the fall at Seton Lake. According to Love's (1977) dorsal aspect model, summer and fall TS ranges corresponded to fish lengths of 9-572 mm and 9-664 mm in Anderson Lake, and 9-474 mm and 9-504 mm in Seton Lake. The modeled size ranges from TS for Anderson Lake agreed closely with minimum and maximum sizes from trawl and gill net sampling (23-600 mm summer, 27-674 mm fall). Those from Seton Lake agreed less closely with trawl and gill net results (27-674 mm summer, 36-750 mm fall), with modeled results indicating a smaller maximum size, however, small sample sizes (a low number of large fish captured in gill nets and detected per survey with acoustics) plus the high inherent variability of TS measurements (Simmonds and MacLennan 2005) may explain this discrepancy.

Summer

Anderson Lake Transect 4

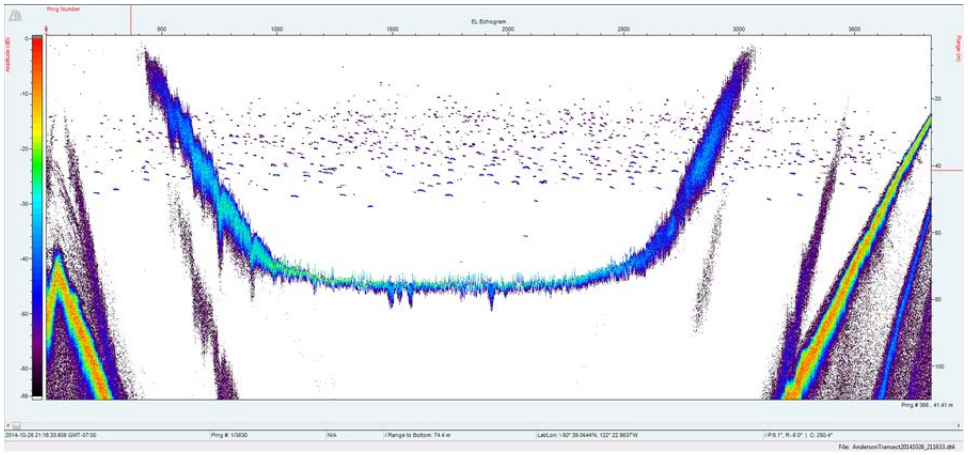


Seton Lake Transect 6



Fall

Anderson Lake Transect 4



Seton Lake Transect 6

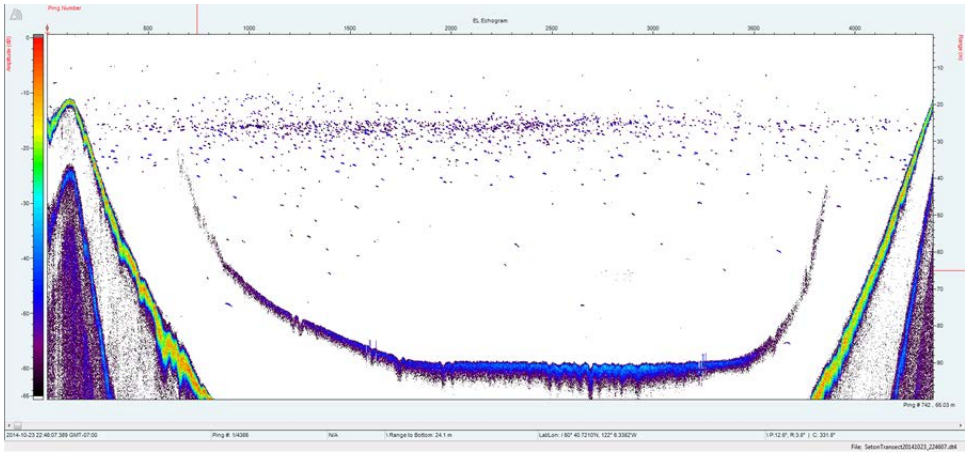


Figure 48 Typical echograms from summer and fall 2014 night time whole-lake acoustic surveys of Seton and Anderson Lakes. All echograms have 40 log R amplification and a display threshold of -65 dB.

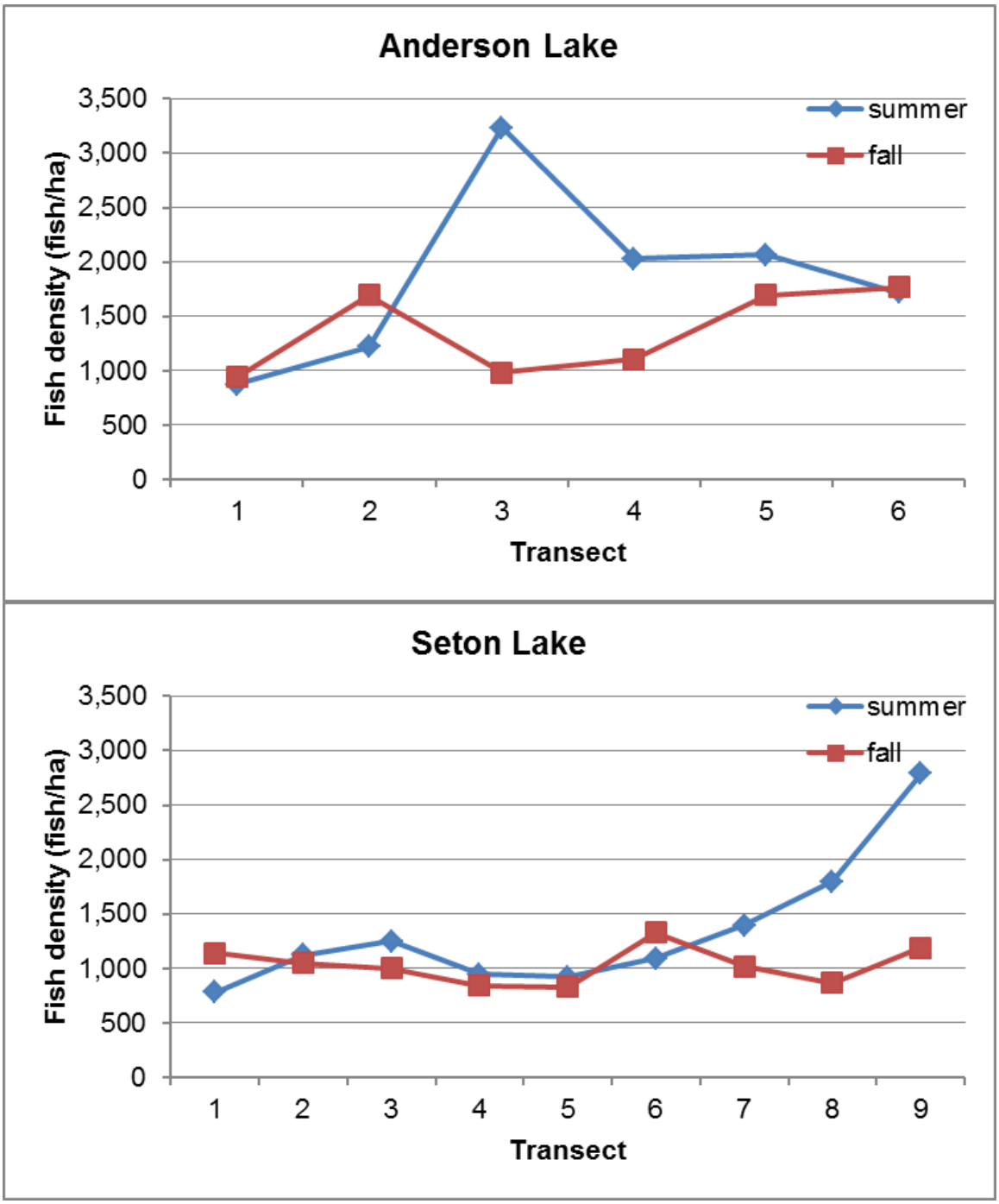


Figure 49 Mean areal fish density on individual transects (fish/ha, all sizes and species combined) during summer and fall acoustic surveys of Seton and Anderson Lakes. Transect numbers increase from west to east in both lakes (from inlet to outlet ends).

Table 24 Fish abundance, density (fish/ha), and 95% confidence limits (CL) by taxa and O. nerka stock from coordinated hydroacoustics and fish capture sampling (trawling and gill netting) of Anderson and Seton Lakes during summer and fall 2014. Stock identification was from DNA analysis. Na = no data available.

			Anderson						Seton					
			Summer			Fall			Summer			Fall		
Size-group	Taxa	Stock	Abundance	Density (#/ha)	CL (± %)	Abundance	Density (#/ha)	CL (± %)	Abundance	Density (#/ha)	CL (± %)	Abundance	Density (#/ha)	CL (± %)
Small	Gwenish	na	3,194,020	1,123	65%	2,958,444	1,040	27%	253,043	100	37%	507,787	202	16%
	Sockeye	Gates	1,534,133	539	14%	544,526	191	23%	1,541,864	612	21%	1,131,493	449	14%
		Portage	141,569	50	12%	19,527	7	45%	830,240	330	32%	475,745	189	13%
		Total Sockeye	1,675,702	589	13%	564,054	198	23%	2,372,104	942	25%	1,607,238	638	13%
		Total small	4,869,722	1,712	46%	3,522,498	1,239	26%	2,625,147	1,042	26%	2,115,025	840	13%
Medium	na	na	362,344	127	38%	277,248	97	26%	550,083	218	33%	470,282	187	27%
Large	na	na	25,572	9	50%	28,843	10	85%	12,400	5	86%	1,858	1	121%
Combined			5,257,625	1,849	44%	3,828,589	1,346	25%	3,187,630	1,265	19%	2,587,375	1,027	14%

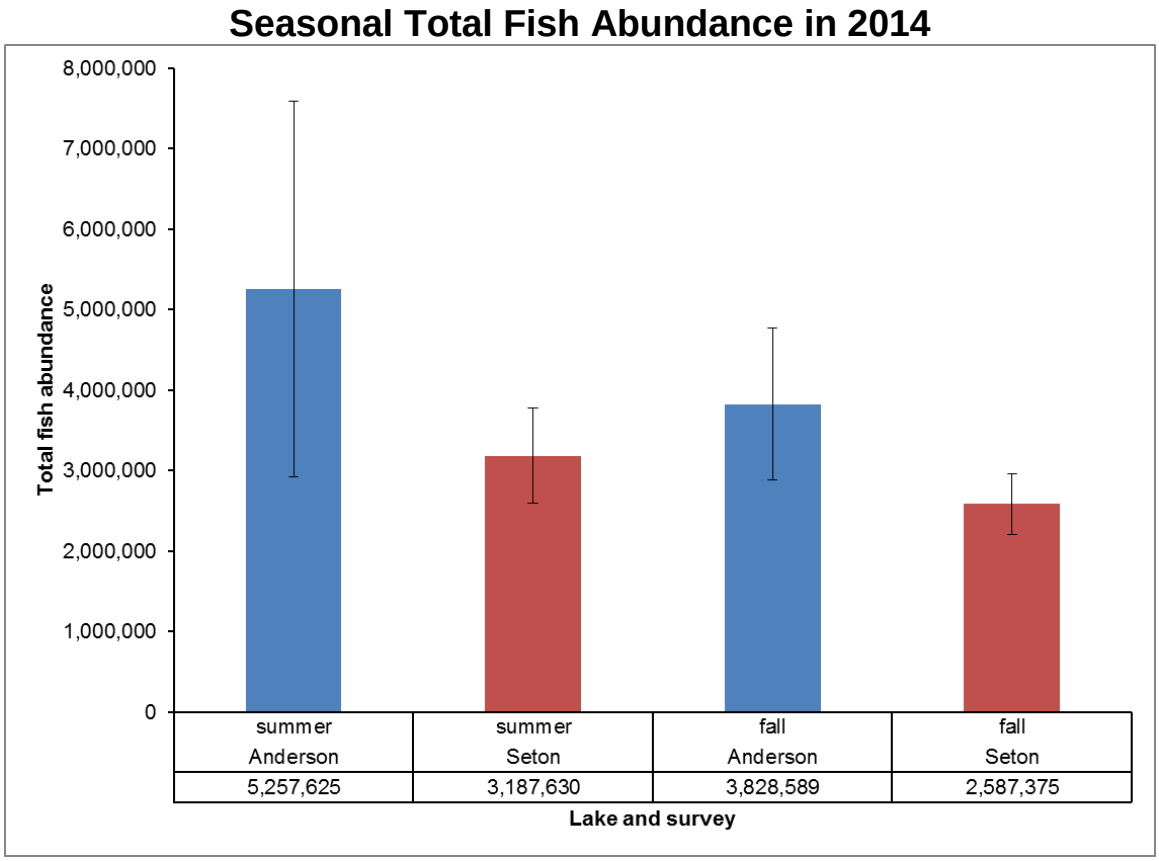


Figure 50 Total pelagic fish abundance (all size-groups and species combined) during summer and fall 2014 acoustic surveys of Anderson and Seton Lakes. Error bars are 95% confidence intervals.

Target strength (TS) frequency distributions (Figure 51) show that in both lakes during both seasons the small size class (≤100 mm in length) greatly predominated in terms of relative abundance (>80% of individual survey totals), followed by the medium size class (101-376 mm, <20%), and the large size class (>376 mm, <1%). In both seasons the medium size class was more prevalent in Seton Lake than in Anderson Lake, whereas the large size class was always more prevalent in Anderson Lake, especially in the fall. In each survey small fish appeared to have two primary modes, whereas modes were less distinct for the medium size class, and large fish were too scarce for identification of modes. The multi-modality of the TS frequency distributions corresponds to the complex mix of fish taxa, stocks, and ages that analysis of the trawl and gill netting catch showed them to represent. This complexity masked expression of seasonal increases in fish body size through shifting of TS modes to larger values from summer to fall. Although some shifts in mode position clearly indicate fish growth (e.g., a clear positive shift of the major mode for small fish in Anderson Lake) others are more ambiguous (e.g., modes for small fish in Seton Lake).

Total fish abundance estimates partitioned by size-class using TS data showed that during both surveys the small and large size-classes were more numerous in Anderson Lake than in Seton Lake, whereas medium sized fish were more abundant in Seton Lake (Table 24, Figure 52). In both lakes, the

abundance of all size-classes decreased from summer to fall, with one exception: the number of large fish in Anderson Lake increased slightly (Table 24, Figure 52). Only two of these comparisons were clearly statistically significant (95% confidence intervals non-overlapping): in the fall, small and large fish were more abundant in Anderson Lake than in Seton Lake (Figure 52). The size composition of fish within lakes (% by size group) varied little from summer to fall, changing <1% for any size group (Figure 53).

4.4.8.4 Sensitivity analysis of size class breakpoints

Reliable estimates of Sockeye fry and large piscivore abundance are necessary for testing the hypotheses concerning relative abundance of predators and prey in the two lakes, but the relative abundance of size-classes is affected by the choice of TS breakpoints defining them. In our data set, peaks within each TS frequency distribution (within each survey) overlapped extensively and were indistinct from each other, and the chosen breakpoints fell on the slopes of peaks rather than at minima between them (Figure 51). In this situation, a small shift in a breakpoint can cause a large change in the abundance of size classes. A sensitivity analysis showed that small fish abundance changed no more than 14.5% when the small-medium size class breakpoint varied ± 2dB from the -45 dB value used for abundance estimates in Table 24, and the change was <10% in all but two cases (Table 25; a shift of ± 2 dB encompasses a reasonable range of uncertainty for either breakpoint considering the size of the fish of interest in the trawl and gill net catch). Therefore, the estimate of Sockeye fry abundance, a major component of the small size class, appears robust with respect to choice of breakpoint. The large size class, representing large pelagic piscivores, was affected much more severely by changes, especially decreases, in the medium-large size class breakpoint of -34 dB. A decrease of 2 dB (to -36 dB) caused an 88-664% increase in the large fish abundance estimate, while a 2 dB increase (to -32 dB) caused a 64-100% decrease in it (Table 25). Just a 1 dB change in this breakpoint caused a 31-236% change in estimated abundance. Even so, when actual population estimates were computed using a -35 dB breakpoint, statistical conclusions about between-lake and seasonal comparisons based on overlap of 95% confidence intervals did not differ from those reached using a -34 dB threshold.

A major factor in our choice of the medium-large size class breakpoint was the relative size of Gwenish and piscivores. Inclusion of even a small fraction of the relatively abundant Gwenish in the large size class could have considerably inflated abundance estimates of piscivores (the large size class). As a rule of thumb, the mean lengths of two groups of fish should differ by a factor ≥2 to reliably distinguish them using TS measurements (Crockett et al. 2006). In Anderson Lake, gill netting showed that piscivores (Bull Trout and Northern Pikeminnow of medium and large size groups combined) and large Gwenish (those > 250 mm in length) differed in mean length by a factor of 1.4-1.5 (fall and summer), and Gwenish 250-316 mm in length were numerous during both seasons (Figure 54). We therefore chose a fairly high (-34 dB) breakpoint dividing the medium and large size groups to minimize the error of classifying Gwenish as piscivores (i.e., putting them in the large size class) while risking classification of some Bull Trout and Northern Pikeminnow in the medium size class, and thereby not counting them as piscivores. Considering potential classification of Gwenish as large fish, the expected dorsal aspect TS of the largest Gwenish captured from either lake was -35.4 dB, or 1.4 dB smaller than the breakpoint. This left a minimal margin for error considering the highly variable stochastic nature of

TS measurements and the possibility that the lakes contained Gwenish larger than those we captured (Crockett et al. 2006). The second type of error, the risk of assigning piscivores to the medium size class, appeared greater in Anderson Lake where an average (seasons combined) of 40.9% of piscivores in the combined gill net and trawl catches were in medium size group (length <376 mm corresponding to dorsal aspect TS -34 dB, Love 1977), compared to 11.8% in that group in Seton Lake (Figure 54, Table 26). This suggests that piscivore abundance in Anderson Lake was appreciably higher than indicated by the large fish estimate alone, whereas large fish abundance more accurately reflected piscivore abundance in Seton Lake. Estimates of piscivore abundance for both lakes would have been improved if the medium size class abundance estimates could have been apportioned by species, however, reliable species composition estimates would have required a much more extensive gill net sampling program than we were able to perform (Crockett et al. 2006, Beauchamp et al. 2009, Stables and Perrin 2016).

In summary, the estimate of Sockeye fry abundance, appears robust with respect to choice of small-medium breakpoint. Although the large size class abundance estimates were greatly affected by small changes in the choice of medium-large breakpoint, statistical conclusions about between-lake and seasonal comparisons of large piscivores were not affected. Due to the relatively small difference in average size between the medium and large size classes and the high abundance of Gwenish in the middle size class we chose a fairly high (-34 dB) breakpoint in order to minimize the error of classifying Gwenish as piscivores. This decision does increase the chance of underestimating the abundance of piscivores especially in Anderson Lake where gillnet catches showed many piscivores where in the medium size class (<37 mm)/

4.4.8.5 Size class, taxa, and stock specific estimates of fish abundance

Abundance estimates of the three size classes were apportioned among fish taxa to the level required to meet study objectives. The large and medium size classes were not subdivided quantitatively among taxa. The medium size class simply represented mainly age 1 and older Gwenish, plus a small fraction of Northern Pikeminnow and Bull Trout. The large size class represented mainly Bull Trout with a small fraction of Northern Pikeminnow. The small size class was subdivided quantitatively among taxa. Gwenish, Portage Sockeye, and Gates Sockeye (all O. nerka) were the only fish <100 mm in length caught in the trawl in either lake, and therefore constituted the entire estimate of small fish. Abundance and the relative proportions of O. nerka types differed considerably among the lakes. Gwenish greatly predominated in both surveys of Anderson Lake (2.9-3.2 million fish, 65.6-84.0% of all small fish, Table 24 and Figure 55), with Gates Sockeye fry the next most abundant group (0.5-1.5 million fish, 15.5-31.5%). In Seton Lake, Gates Sockeye were the most abundant group in both surveys (1.1-1.5 million fish, 53.5-58.7%), followed by Portage Sockeye (0.5-0.8 million fish, 22.5-31.6%). Considering Sockeye alone, the Gates stock was predominant in both lakes, constituting 65.0-70.4% and 91.6-96.5% of Sockeye fry in Seton and Anderson Lakes, respectively (Figure 56). The stock composition of the fall 2014 Seton Lake survey (70.4% Gates, 29.6% Portage) was very similar to the composition of the spring 2015 Sockeye smolt outmigration in the Seton River (70% Gates, 30% Portage, Figure 56). Considering the total population of O. nerka of the small size class in both lakes together, patterns were similar in summer and fall: Anderson Lake supported a larger fraction of the

total population (62-65%) and of Gwenish alone (85-93%), while Seton Lake supported a larger proportion of total Sockeye (59-74%, Figure 57). Both individual Sockeye stocks were more abundant in Seton Lake (68-96% of individual stock total), except during the summer when Gates stock abundance was the same in both lakes (Figure 57).

4.4.8.6 Spatial distribution patterns of fish size classes, taxa, and stocks

The areal density (fish/ha) of size classes and taxa varied from transect to transect, and some longitudinal patterns were apparent within the lakes. Within the small size class, densities of Sockeye as a whole and of the more abundant Gates stock tended to be highest in the east end (the outlet end) of Anderson Lake during both seasons (Figure 58). The much less abundant Portage stock was denser in the east end of Anderson Lake during the summer and in the west end during the fall. No longitudinal trends were apparent for small Gwenish in Anderson Lake in either season. In Seton Lake, densities of all small O. nerka types were highest in the east end (the outlet end) during the summer (Figure 58). In the fall, Portage Sockeye and Gwenish were densest in the east end of Seton Lake, whereas densities of the more abundant Gates stock and Sockeye as a whole were lowest there. Densities of the medium size class varied erratically among transects with no clear longitudinal trends in either lake (Figure 59). Densities of the large size class also varied erratically among transects in both lakes, for the most part, except they were relatively high in the eastern half of Anderson Lake, especially on transect 6 (nearest the outlet), during the fall (Figure 59). Scatterplots of these data showed a significantly positive relationship between mean transect densities (fish/ha) of medium and large size classes at Seton Lake in the summer (R^2=0.72, P<0.05) and in Anderson Lake in the fall (R^2=0.74, P<0.05). There was no relationship between densities of any other size class pair.

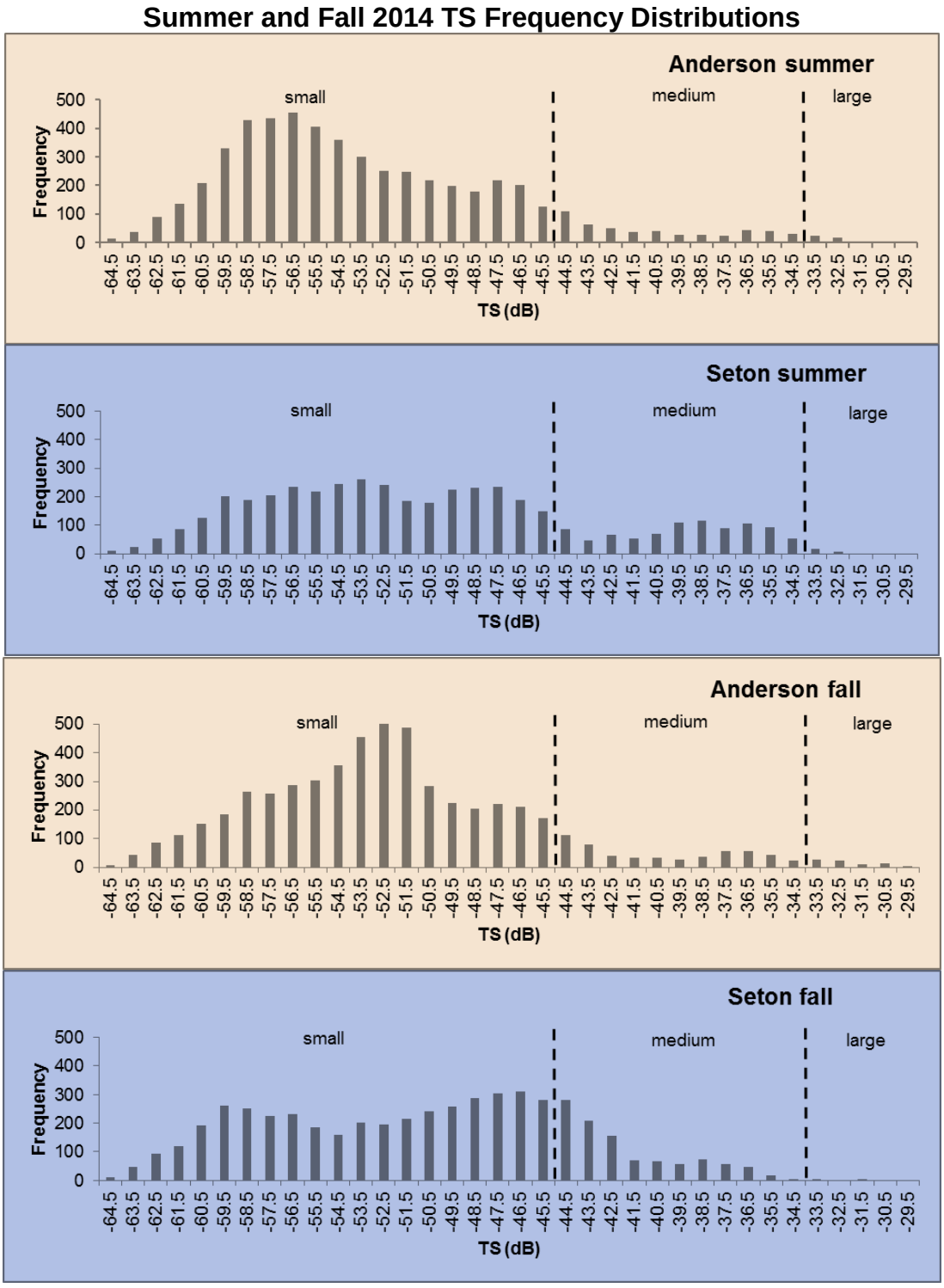


Figure 51. Frequency distributions of tracked fish from summer and fall 2014 acoustic surveys of Seton and Anderson Lakes. Dashed lines show breakpoints between TS size-groups used for analysis. Fish length ranges represented by the size-groups are: small 9-100 mm, medium = 101-376 mm, large > 376 mm)

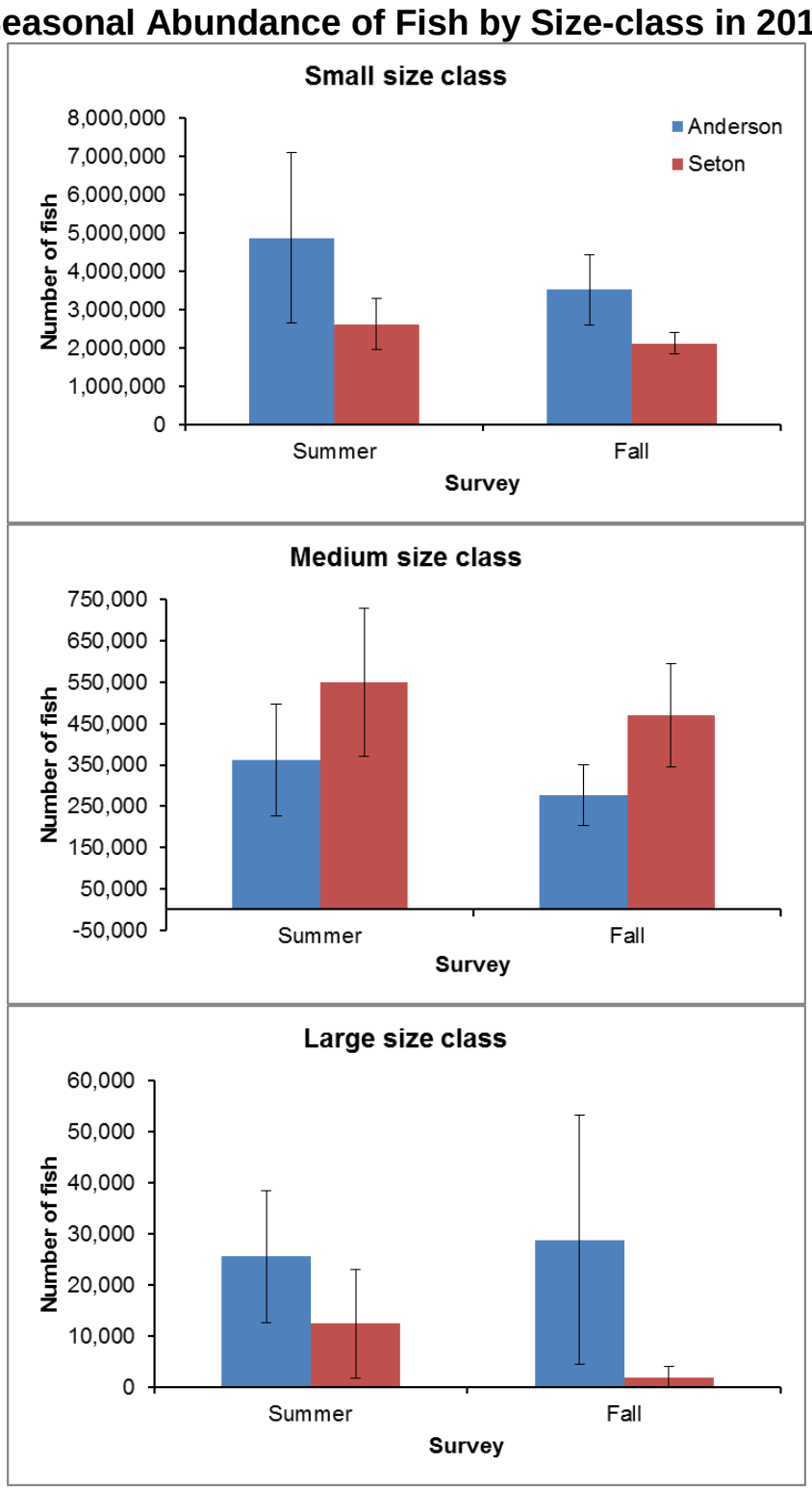


Figure 52 Abundance of each fish size-class in Anderson and Seton Lakes during summer and fall 2014 acoustic surveys. Error bars are 95% confidence intervals.

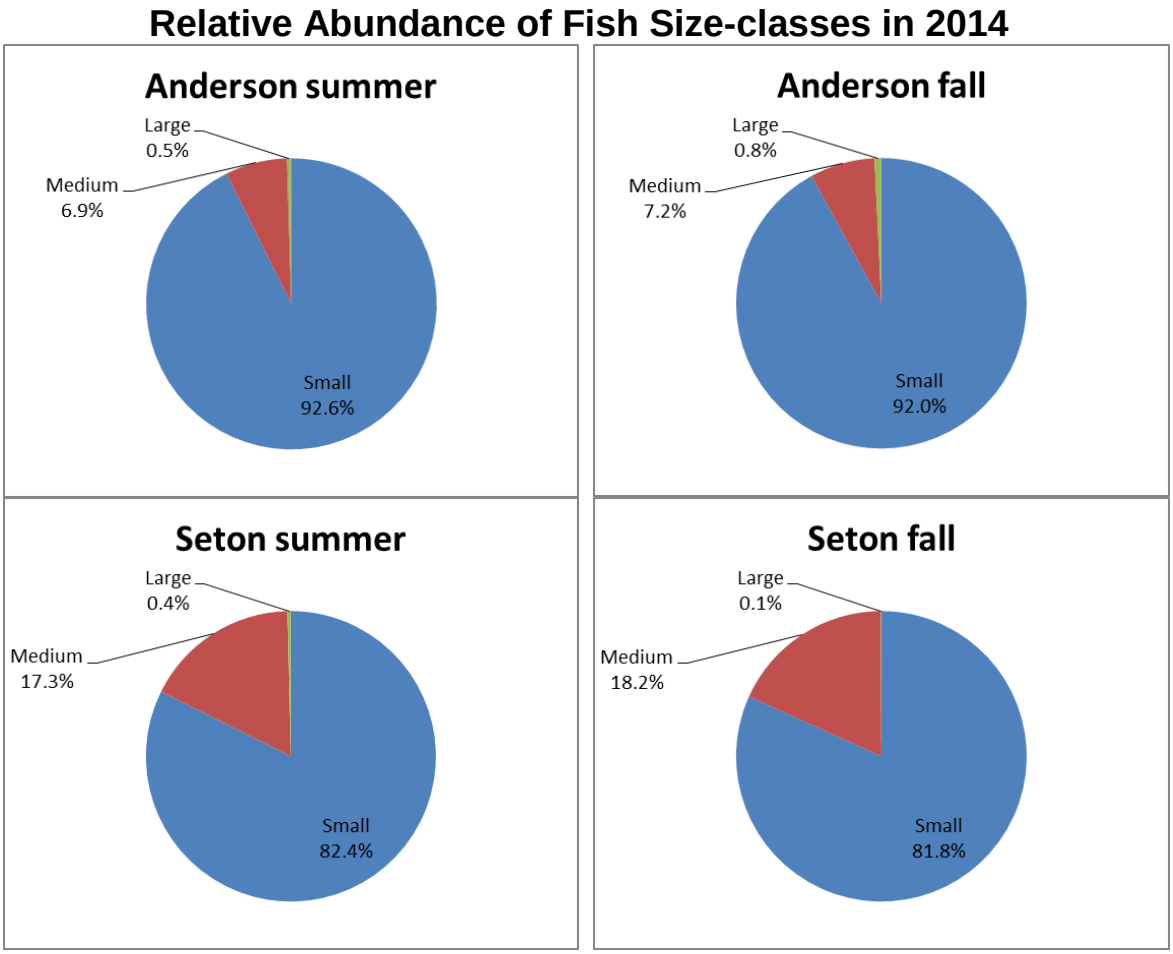


Figure 53 Relative abundance of small, medium, and large size classes of fish (% of total) in Anderson and Seton Lakes during summer and fall 2014 acoustic surveys. Data are from lake section and depth stratified abundance estimates.

Table 25 Sensitivity analysis to determine the effect of size-class breakpoints on size composition of fish abundance estimates from acoustics. Values for breakpoints used for actual population estimates in this report (-45 dB and -34 dB) are shaded gray. This analysis was not stratified by lake section or depth stratum, hence percentages differ slightly from those in Figure 53.

Small fish size class

		Percentage of fish classified as Small relative to breakpoint					Change (%) in Small fish abundance relative to a -45 dB breakpoint				
		Small-Medium size class breakpoint (dB)					Small-Medium size class breakpoint				
Lake	Season	-47	-46	-45	-44	-43	-47	-46	-45	-44	-43
Anderson	Summer	84.0%	87.7%	90.1%	92.1%	93.3%	-6.8%	-2.6%	0.0%	2.3%	3.6%
Seton	"	71.6%	75.9%	79.2%	81.2%	82.3%	-9.7%	-4.3%	0.0%	2.5%	3.8%
Anderson	Fall	81.6%	85.5%	88.7%	90.8%	92.2%	-8.0%	-3.6%	0.0%	2.4%	4.0%
Seton	"	68.1%	74.2%	79.7%	85.2%	89.3%	-14.5%	-6.9%	0.0%	6.9%	12.1%

Large fish size class

		Percentage of fish classified as Large relative to breakpoint					Change (%) in Large fish abundance relative to a -34 dB breakpoint				
		Medium-Large size class breakpoint (dB)					Medium-Large size class breakpoint				
Lake	Season	-36	-35	-34	-33	-32	-36	-35	-34	-33	-32
Anderson	Summer	2.2%	1.4%	0.8%	0.4%	0.1%	157.8%	66.7%	0.0%	-48.9%	-86.7%
Seton	"	3.8%	1.7%	0.5%	0.1%	0.0%	663.6%	236.4%	0.0%	-72.7%	-100.0%
Anderson	Fall	2.6%	1.8%	1.4%	0.9%	0.5%	88.0%	30.7%	0.0%	-34.7%	-64.0%
Seton	"	0.5%	0.1%	0.1%	0.0%	0.0%	500.0%	50.0%	0.0%	-75.0%	-75.0%

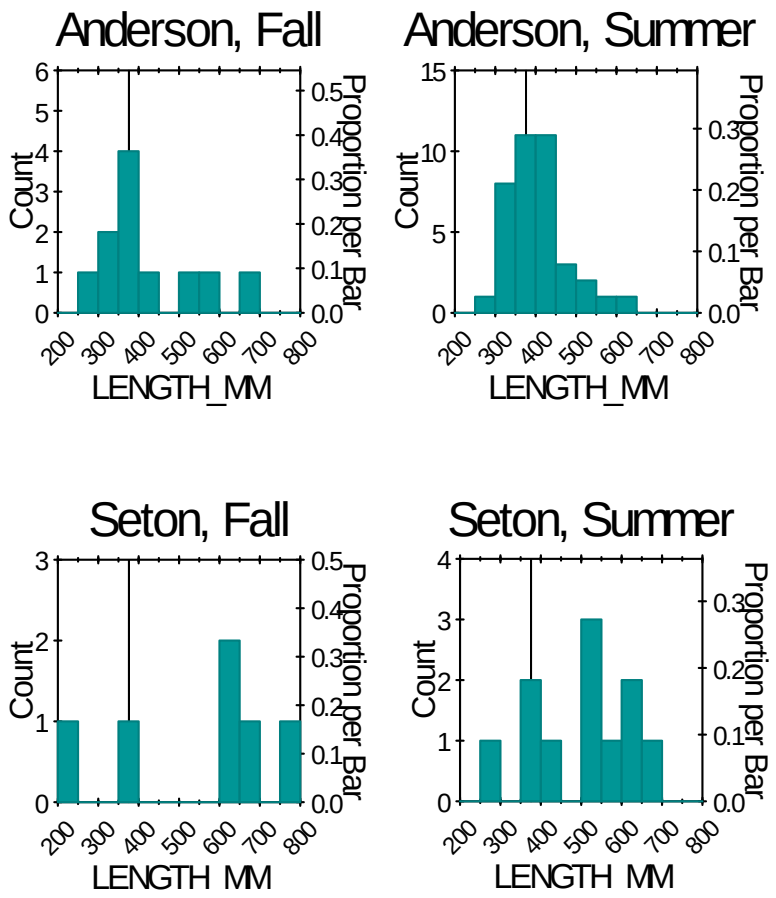


Figure 54 Length frequency distributions of piscivores (Bull Trout and Northern Pikeminnow) in the combined gill net and trawl catch from Anderson and Seton Lake in 2014. Vertical dashed lines at 376 mm indicate the breakpoint dividing medium and large size classes.

Table 26 Percentage of the combined gill net and trawl catch of piscivores (Bull Trout and Northern Pikeminnow) assigned to the medium and large size classes, based on a breakpoint of 376 mm between classes. Data are from Seton and Anderson Lakes in 2014.

		% of total by size class		
Lake	**Season**	**Medium**	**Large**	**Total**
Anderson	Summer	39.5%	60.5%	100.0%
	Fall	45.5%	54.5%	100.0%
	Combined	40.8%	59.2%	100.0%
Seton	Summer	9.1%	90.9%	100.0%
	Fall	16.7%	83.3%	100.0%
	Combined	11.8%	88.2%	100.0%
	All combined	33.3%	66.7%	100.0%

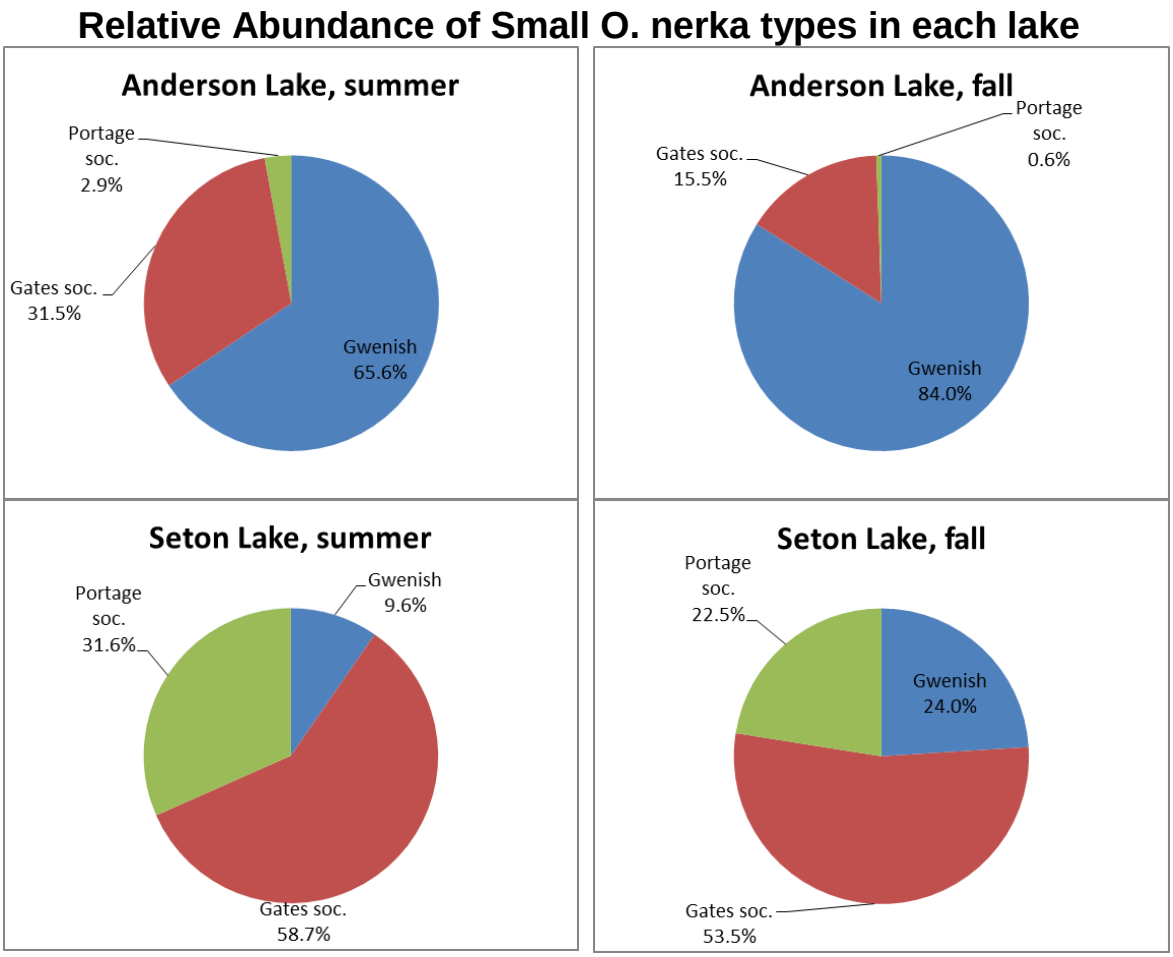


Figure 55 Relative abundance of Gwenish, Gates Sockeye, and Portage Sockeye in summer and fall 2014 acoustic estimates of small O. nerka (< 100 mm in length) abundance in Anderson and Seton Lakes.

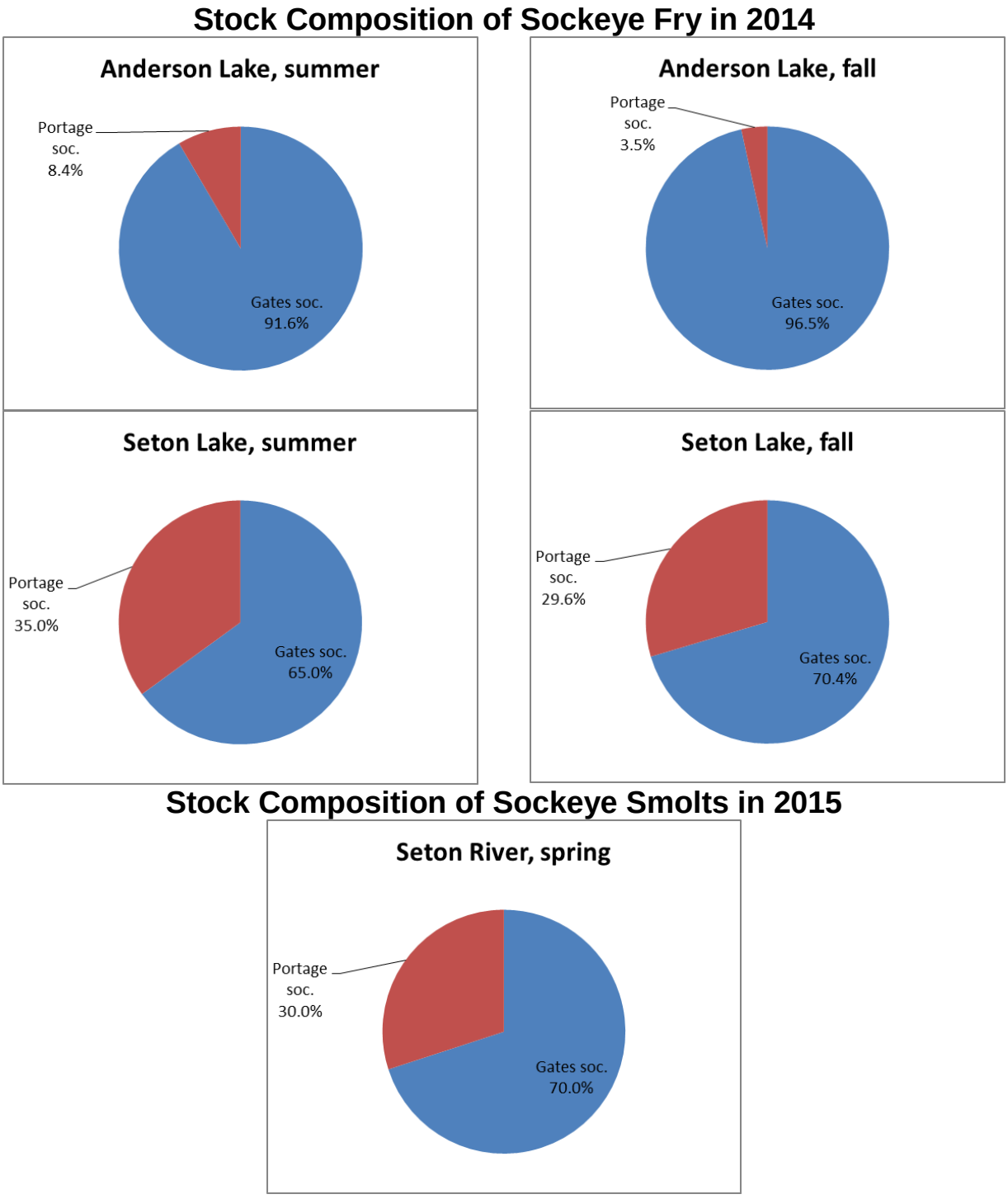


Figure 56 Relative abundance of Gates and Portage Creek stocks in Seton and Anderson lakes and in the subsequent smolt run. The lake data was determined by applying the DNA results to the summer and fall 2014 acoustic estimates. As the 2015 Seton River smolt abundance estimate was unreliable, relative abundance is base on DNA and catch only.

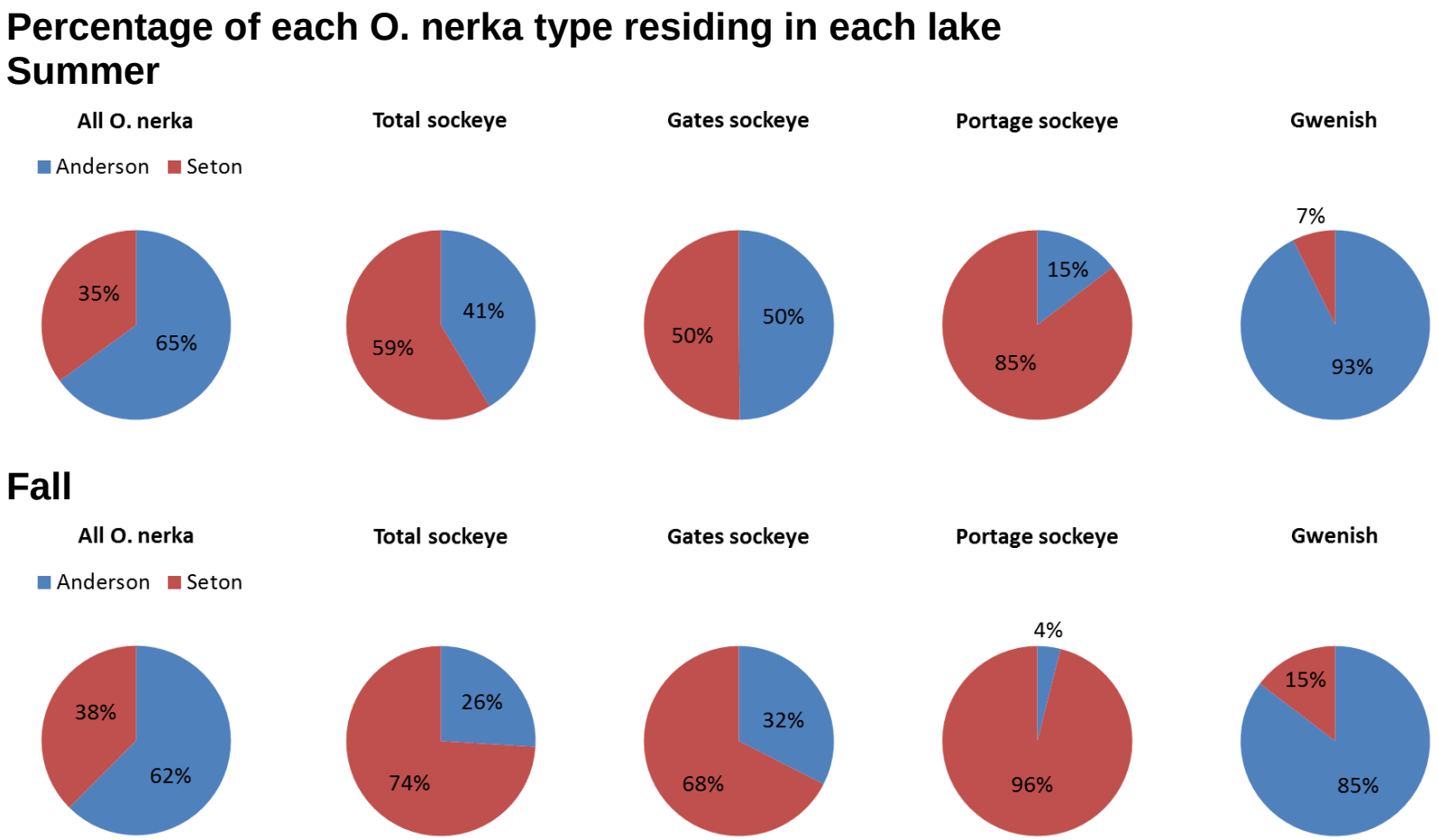


Figure 57 Percentage of small (<100 mm long) O. nerka of each taxa or stock residing in each lake during summer and fall 2014 surveys of Anderson and Seton Lakes.

Small fish abundance by transect

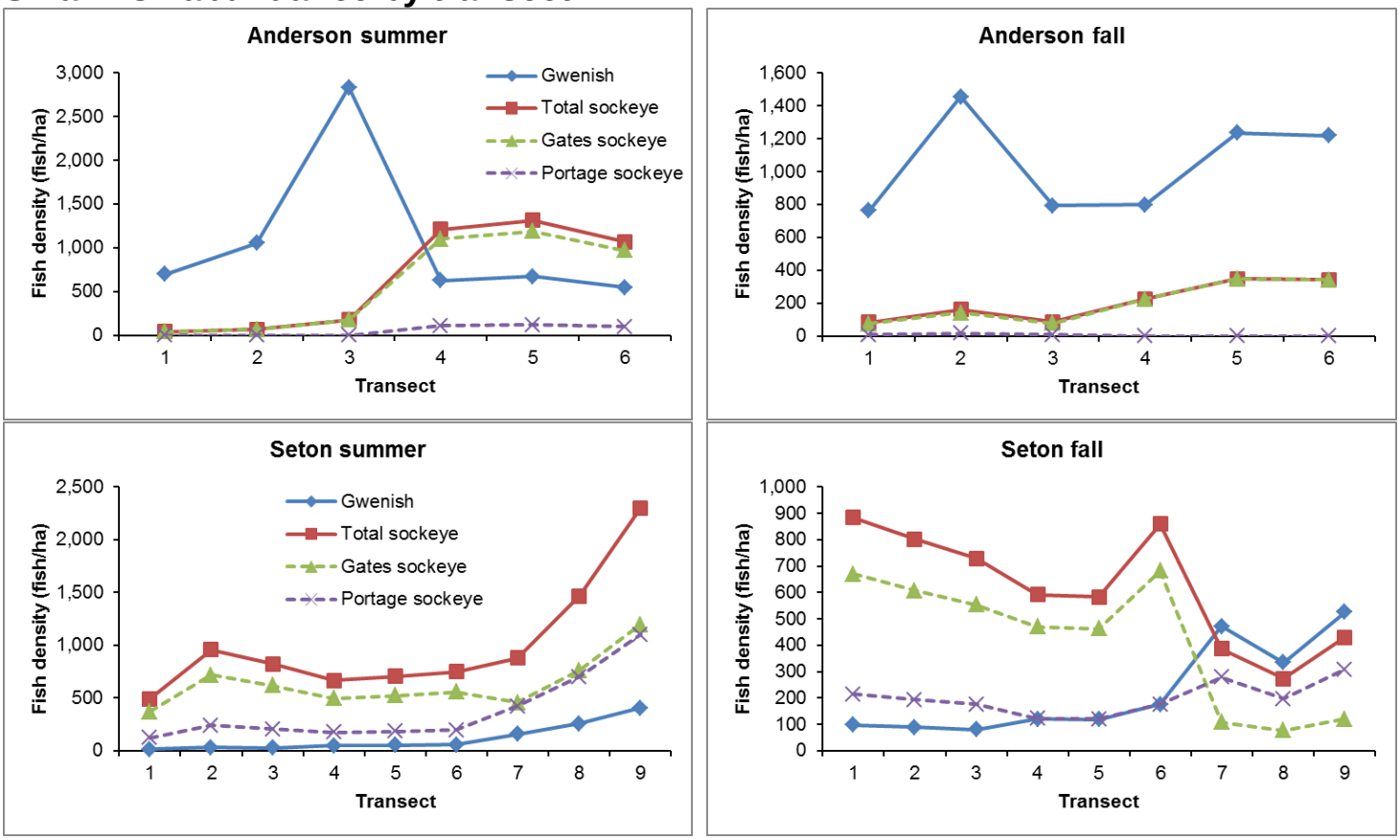


Figure 58 Fish density by transect (fish/ha) of taxa in the small fish size class during summer and fall 2014 acoustic surveys of Seton and Anderson Lakes. In both lakes transect numbering increases from the inlet end to the outlet end (going from west to east).

Medium and Large fish abundance by transect

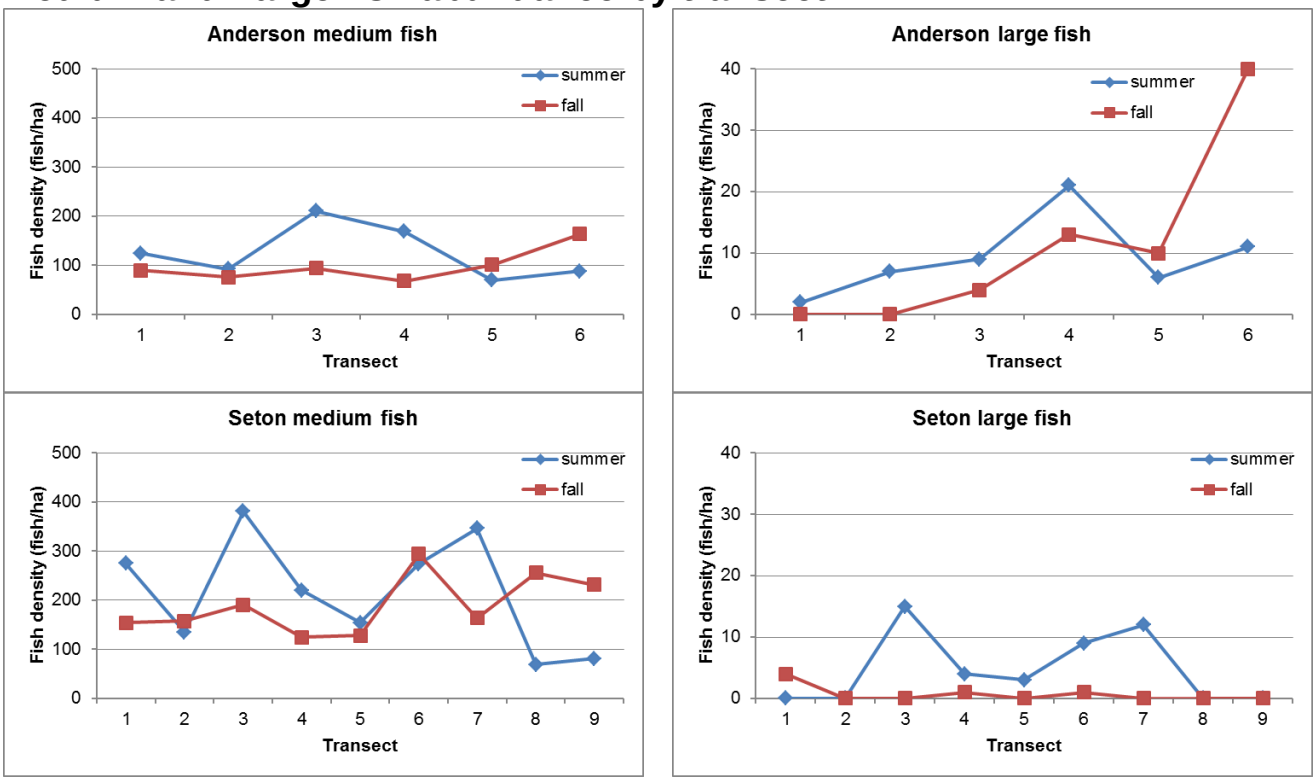


Figure 59 Fish density by transect (fish/ha) of medium and large fish size classes in summer and fall 2014 acoustic surveys of Seton and Anderson Lakes. In both lakes transect numbering increases from the inlet end to the outlet end (going from west to east).

4.4.9 Potential causes of seasonal changes in fish abundance

Abundance within each lake decreased from summer to fall 2014 for all categories of fish (size groups, taxa, and stocks), except for large fish in Anderson Lake and small Gwenish in Seton Lake (Table 27). These seasonal changes may have been due to numerous factors, of which the following seem most likely:

- natural mortality from summer to fall,
- migration from or between the lakes,
- a change in the spatial distribution of fish (e.g., a shift from littoral to pelagic zone, or vice versa),
- growth resulting in a shift from one size class to another,
- sampling error (e.g., random error where fish abundance estimates were imprecise due to a patchy spatial distribution of fish or small sample size, or systematic error where abundance estimates were inaccurate due to bias in sampling methods).

From the data we have it is impossible to know for certain which factor or factors were responsible for abundance changes in each category of fish, however, the evidence is fairly clear in some cases.

Seasonal decreases in the abundance of small size class Gates and Portage Sockeye, two of the key fish groups in this study, were greater in Anderson Lake than in Seton Lake, and these decreases were statistically significant (summer and fall 95% CL did not overlap), except for the Gates stock in Seton Lake (Table 27, Figure 60). These declines appear to be mainly due to natural mortality. Late summer and fall spawning Sockeye stocks in British Columbia watersheds are typically fully pelagic and accessible to acoustic sampling from late July to smolting the following spring (McDonald and Hume 1984; Morton and Williams 1990; Hume et al 1996). Thus, considering the spawning time of these stocks, their fry should have been fully pelagic and accessible to the acoustic gear in both lakes before the summer surveys started in late July, and they would have remained so until they emigrated from the lakes as smolts the next spring. Gates Sockeye fry do migrate from Anderson Lake to Seton Lake, and as we did not sample Portage Creek in 2014 we cannot entirely rule out an inter-lake fry migration between our summer and fall acoustic surveys. However, in prior studies the bulk of the inter-lake fry migration took place in April and May, and it had nearly ceased by the end of July, the time of our summer surveys (Geen and Andrews 1961, Woodey 1975). Summer and fall 2014 trawl data indicate that during both seasons Sockeye fry in both lakes were smaller than the 100 mm (-45 dB) breakpoint for the small size class, so none grew out of the small size class. Acoustic and trawl survey methods, coverage, and sample sizes met recognized standards (MacLellan and Hume 2010), making appreciable sampling error unlikely.

If decreases in numbers of small Sockeye are taken to represent summer to fall survival rates, then Sockeye fry of both stocks survived much better in Seton Lake than in Anderson Lake (Gates: 73% in Seton vs 35% in Anderson; Portage: 57% vs 14%; Stocks combined: 68% vs 34%), and the differences in rates were statistically significant. Even when survival rates were "maximized" for Anderson Lake (using the lower 95% CL for summer abundance and the upper 95% CL for fall abundance), survival rates in Seton Lake remained significantly higher (Pearson Chi-Square, df=1, $p<0.01$).

Causes of seasonal changes in abundance of the large size class are not entirely clear. This group was also especially important to our study because it represented piscivores that were presumably the major cause of mortality in juvenile *O. Nerka* . In Anderson Lake, the slight increase (13%) in large fish abundance from summer to fall was not statistically significant (Figure 52, summer and fall 95% CL broadly overlapped), suggesting stable population numbers from survey to survey. During the same period the estimated number of large fish in Seton Lake decreased markedly (85%, Figure 52, summer and fall 95% CL barely overlapped). All large fish abundance estimates from both lakes had a high bound of error (50-121% of the estimate), indicating that random sampling error was high for this size class. The low density of large fish (fish/ha) coupled with the small number of acoustic transects and gill net samples all contributed to this situation. The only other obvious factor that may have played a part in the seasonal decrease of large fish in Seton Lake is a fall migration of bull trout to spawning streams, which might have been expected to affect bull trout abundance in Anderson Lake similarly. For the objectives of this study, the most important conclusions about large fish from this analysis are that despite seasonal changes in their abundance 1) both estimates for Anderson Lake were higher than those for Seton Lake, and 2) even with high variability, fall abundance in Anderson Lake was significantly higher than in Seton Lake.

Although study objectives did not require discussion of seasonal changes in small Gwenish abundance, our data offer some explanations for the surprisingly high fall abundance and "apparent survival rates" of this taxon in both lakes (Table 27). Contrary to what would be expected based on body size (Gwenish fry were smaller), in both lakes the apparent survival rate of small Gwenish was higher than that of small Sockeye. It appears that in both lakes Gwenish fry recruited to the pelagic zone after the summer survey, thereby increasing fall abundance and apparent survival rates. Both the acoustic and trawl data support this conclusion: the Seton Lake fall TS frequency distribution shows a high abundance of very small fish (mode at -59.5 dB, Figure 51) and genetic analysis showed that the very small fish in the fall trawl catch were Gwenish (Figure 40). The fall TS frequency distribution for Anderson Lake also shows that very small fish were numerous, although no mode is obvious, and genetic analysis again showed that the very small fish in the trawl catch were Gwenish (Figure 42). Late recruitment to the pelagic zone is plausible considering the late fall though winter spawning time of Seton and Anderson Lake Gwenish, much later than Sockeye in this system. Migration of

small Gwenish from Anderson Lake to Seton Lake between our surveys is another possible explanation for the large fall increase in abundance in Seton Lake, but no such midsummer migration has been documented.

Table 27 Apparent survival rates of pelagic fish in Seton and Anderson Lakes from summer to fall 2014 (apparent survival rate = fall fish density / summer fish density). Na = no data available.

			Anderson			Seton		
			No. of fish		Apparent survival	No. of fish		Apparent survival
Size-group	Taxa	Stock	Summer	Fall	rate (%)	Summer	Fall	rate (%)
Small	Gwenish	na	3,194,020	2,958,444	93%	253,043	507,787	201%
"	Sockeye	Gates	1,534,133	544,526	35%	1,541,864	1,131,493	73%
"	"	Portage	141,569	19,527	14%	830,240	475,745	57%
"	"	Total Sockeye	1,675,702	564,054	34%	2,372,104	1,607,238	68%
"		Total small	4,869,722	3,522,498	72%	2,625,147	2,115,025	81%
Medium	na	na	362,344	277,248	77%	550,083	470,282	85%
Large	na	na	25,572	28,843	113%	12,400	1,858	15%
Combined			5,257,625	3,828,589	73%	3,187,630	2,587,375	81%

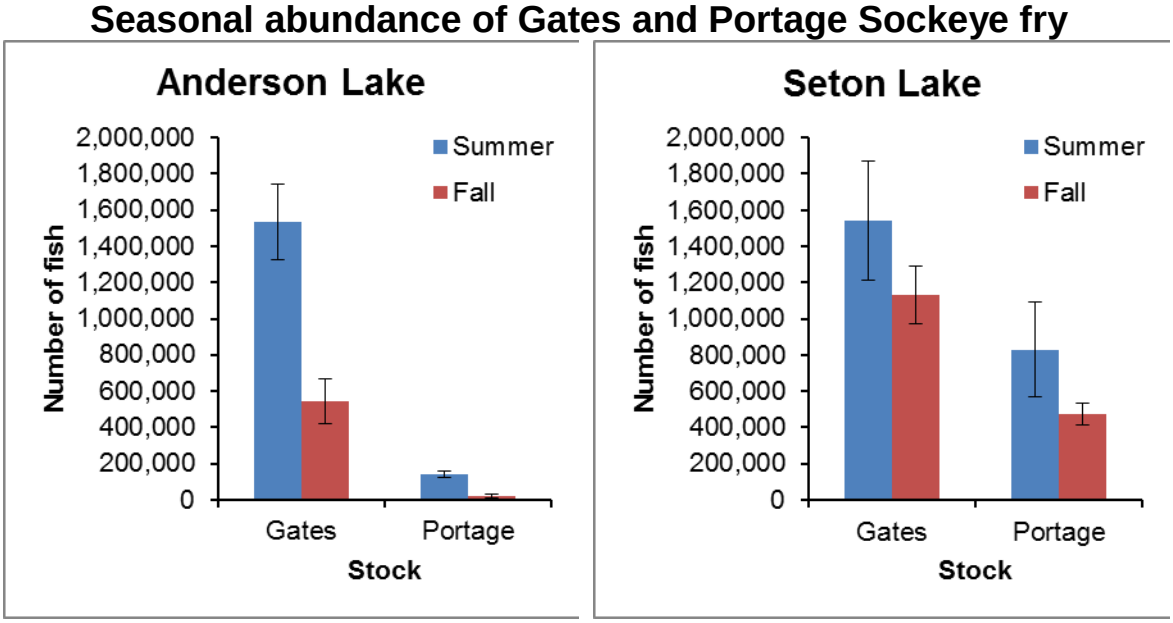


Figure 60 Abundance of Gates and Portage stock Sockeye fry during summer and fall 2014 surveys of Anderson and Seton Lakes. Error bars are 95% confidence intervals.

4.4.10 Diel Vertical Migration (DVM) study

4.4.10.1 Limnology at DVM sites

The information in this section documents the physical and biological variables to which fish distributions were compared in the DVM study. Although some of it is similar to measurements presented in Section 4.3 of this report (limnology studies), the data presented here were collected *during* the DVM sampling trips to best describe conditions experienced by the fish at the time of acoustic sampling.

Thermal structure at the DVM sampling sites was similar in the two lakes, with surface temperatures around 21°C in the summer and 13°C in the fall (Figure 61). The summer metalimnion depth ranged from 10-29 m in Anderson Lake and 20-30 m in Seton Lake (Table 28). Fall thermoclines were more sharply defined with the Anderson Lake metalimnion ranging from 22-33 m and the Seton Lake metalimnion ranging from 23-29 m (SVM-2) and 21-33 m (SVM-8).

Turbidity was very low in Anderson Lake as is shown by Secchi depths >10 m on all sampling occasions (Table 28). Anderson Lake turbidity was only measured once in the fall of 2014 (0.4 NTU). Turbidity was much higher in Seton Lake, averaging from 0.8-11.1 NTU in the top 30 m during the summer and fall. As a result of the higher turbidities, light attenuation rates were much higher in Seton Lake (k = -0.26 to -0.69) than in Anderson Lake (k = -0.15 to -0.19), while Secchi depths and euphotic zone depths were much shallower in Seton Lake (Table 28).

Acoustic processing found distinct zooplankton layers in both lakes, and they performed a diel vertical migration (DVM) in Seton Lake at station SVM 8 near the east end of the lake (Figure 62 and Figure 63). At that station, zooplankton were between 20 m and 45 m during the day and much shallower, between 2 m and 15 m, at night in both the summer and fall surveys (Figure 62 and Figure 63, Table 28). Seton Lake station SVM 2 (near the water diversion outfall at Shalalth and only sampled in the fall) had a much narrower, sparser zooplankton layer (between 2 m and 8 m) that remained near the lake surface day and night (no DVM). At the single station sampled in Anderson Lake (AVM 5), zooplankton did not migrate vertically, remaining between 2 m and 25 m during day and night in summer and fall (Figure 62 and Figure 63, Table 28). Particulates that cause turbidity in Seton Lake did not interfere with these observations, as turbidity layers were not visible on echograms. It is noteworthy that the daytime depth of zooplankton layers in Seton Lake (20-45 m) was mostly below the 30 m maximum depth of daytime net sampling for zooplankton. Thus, acoustic observations suggest that densities of zooplankton (the food supply for *O. nerka*) in Seton Lake may have been substantially higher than suggested by results of plankton hauls, at least in some parts of the lake. Because acoustic observations of zooplankton were only made at two stations and DVM only occurred at one of them, the spatial extent of this behavior throughout the lake is unclear from our data.

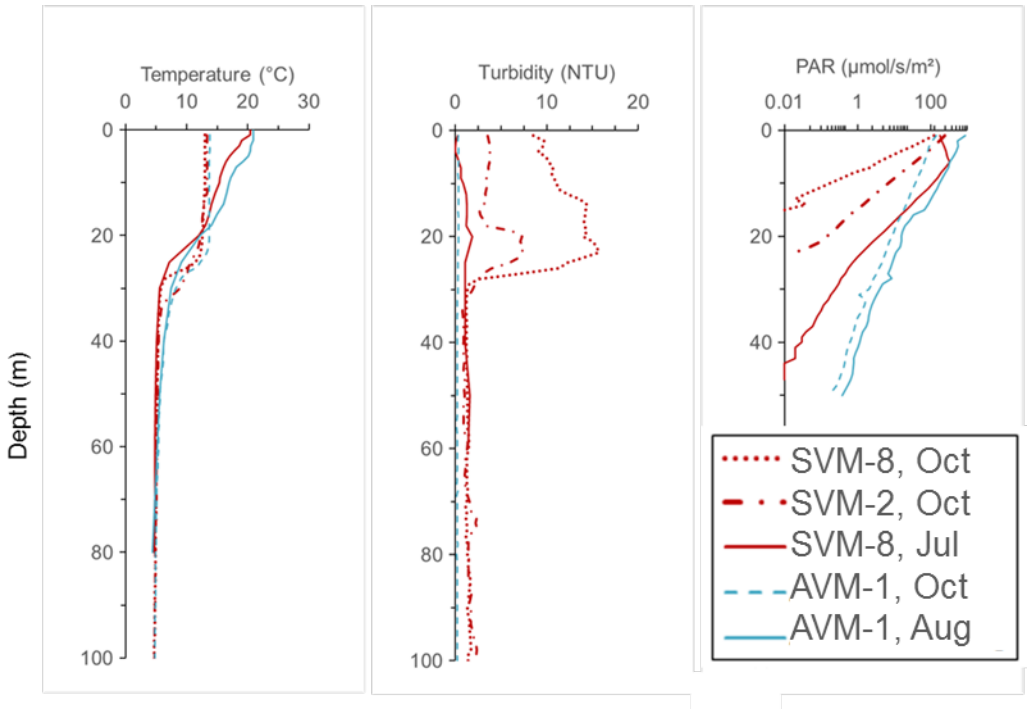


Figure 61 Temperature, turbidity, and light profiles at the DVM stations in Anderson and Seton lakes. Turbidity was not sampled in Anderson Lake in August.

Table 28 Physical and biological limnological variables during DVM sampling periods.

Year	Date	DVM transect	Turbidity (mean <30 m, NTU)	Cloud cover (%)	Light attenuation coefficient (k)	Secchi depth (m)	Euphotic zone depth (m)	Metalimnion depths (m)	Zooplankton depth range (m)	
									Day	Night
Seton Lake										
2003	Jul 26	Tr 6	1.4		-0.48	3.4	8.8	10 - 32		
2014	Jul 29	SVM 8	0.8	0	-0.26	6.9	14.0	20 - 30	25-45[a]	2-15
2014	Jul 30	SVM 8	0.8	0	-0.26	6.9	14.0	20 - 30	25-45[a]	2-15
2003	Oct 24	Tr 6	2.2		-0.42	3.0	10.4	28 - 32		
2014	Oct 23	SVM 2	11.1	100	-0.69	1.0	6.4	23 - 29	2-5[b]	2-8[b]
2014	Oct 24	SVM 8	4.1	30 - 70	-0.41	1.5	11.6	21 - 33	2-8[a]	5-15
Mean			3.4		-0.42	3.8	10.9			
Anderson Lake										
2003	Aug 07	Tr 3			-0.19	11.0	17.7	14 - 29		
2014	Aug 01	AVM 5		<10	-0.16	15.0	22.0	10 - 29	3-25	3-25
2014	Aug 02	AVM 5		<30	-0.16	15.0	22.0	10 - 29	3-25	3-25
2003	Oct 15	Tr 3			-0.15	10.4	21.7	23 - 27		
2014	Oct 26	AVM 5	0.37	70	-0.16	14.0	24.0	22 - 33	5-25	5-25
2014	Oct 27	AVM 5	0.37	60-90	-0.16	14.0	24.0	22 - 33	5-25	5-25
Mean			0.3		-0.16	13.2	21.9			

[a]The zooplankton coincided with a layer of fish schools and single targets.
[b]The zooplankton layer had very low density.

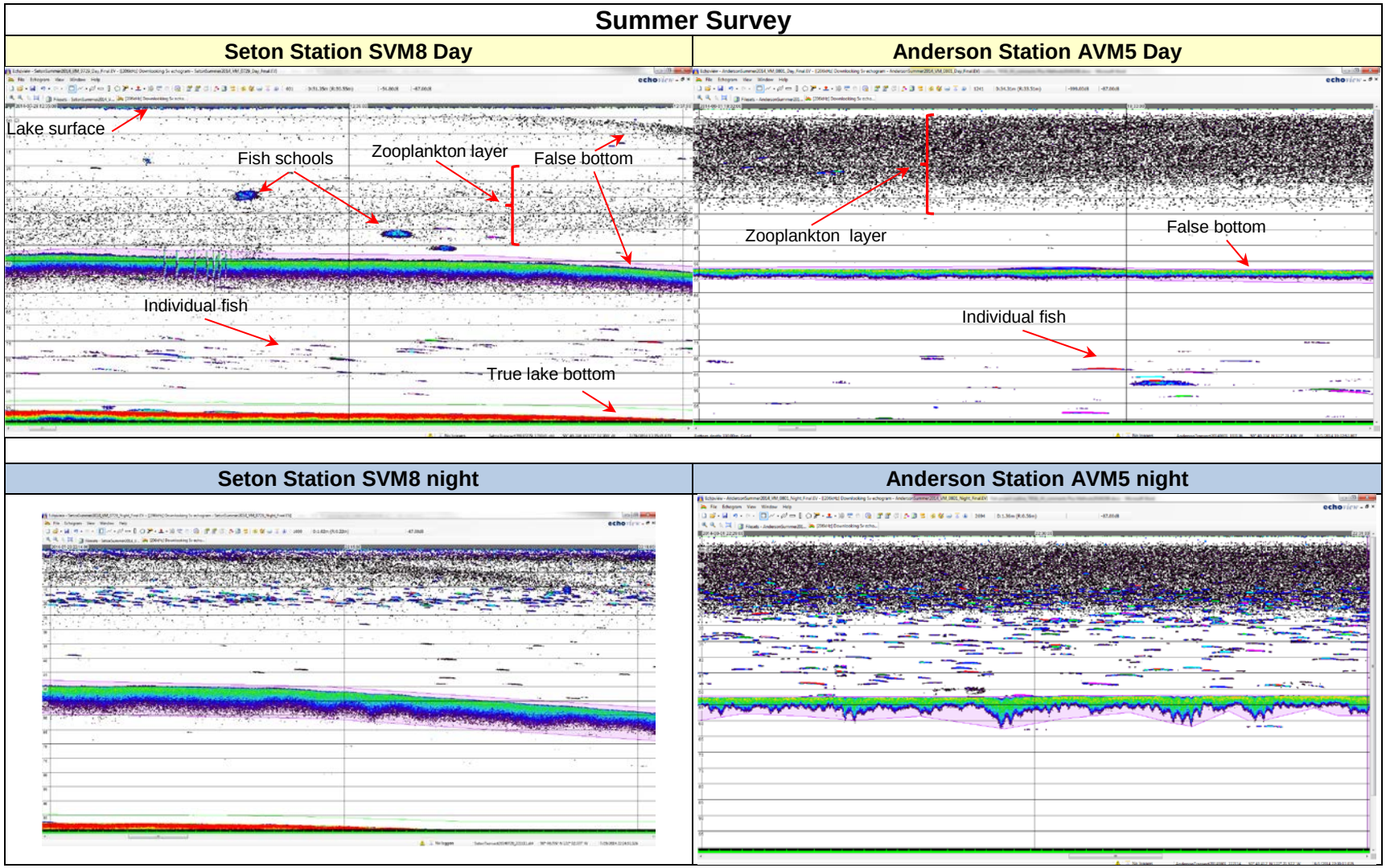


Figure 62 Day and night echograms from summer DVM surveys showing zooplankton and fish in the water column. A low display threshold (-87 dB, 20 log R amplification, and time varied threshold -120 dB @ 1 m) was used to make zooplankton visible. Each cell shows an approximately two minute transect segment with a 2-100 m vertical range gridded by 5 m layers.

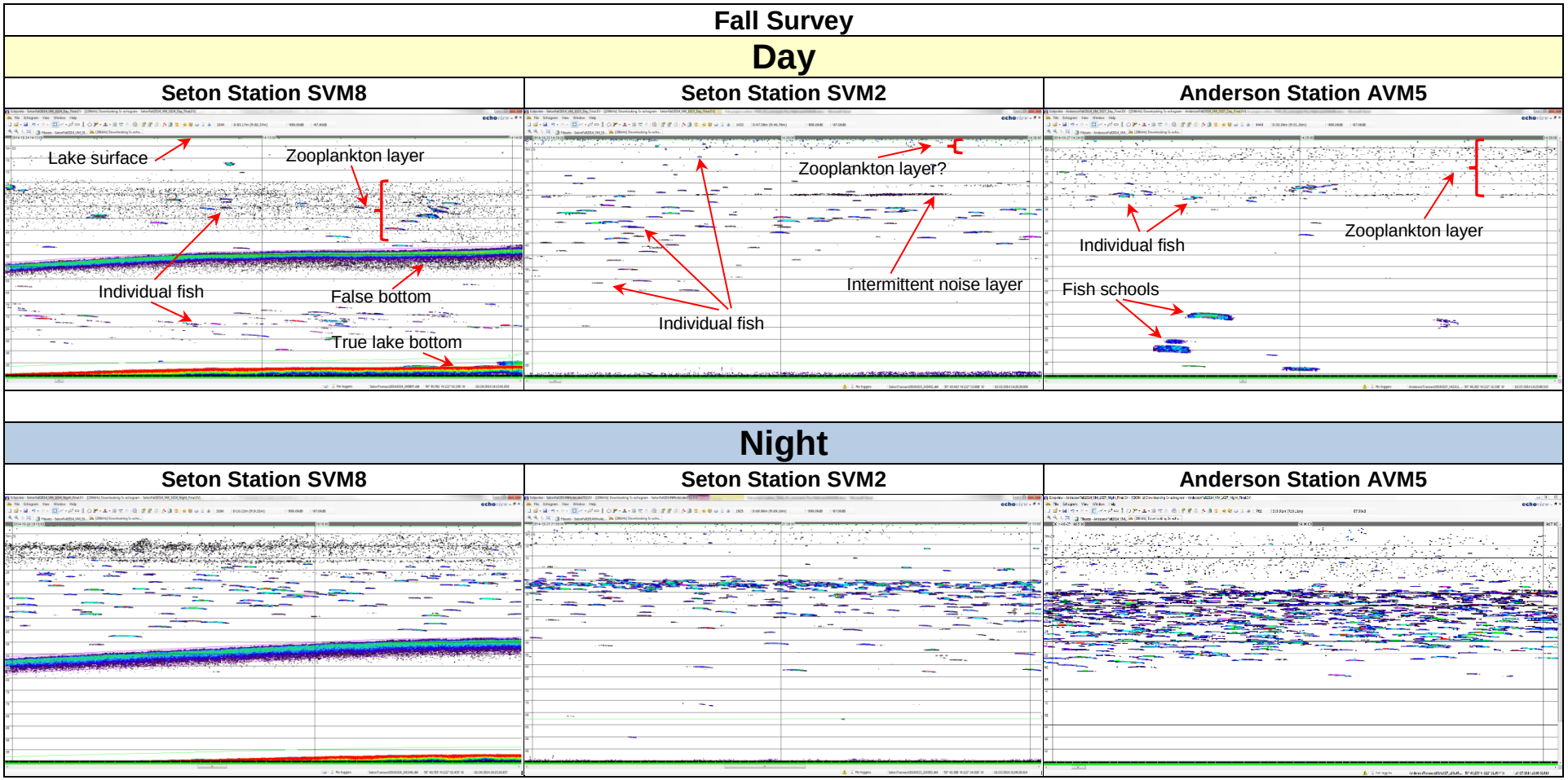


Figure 63 Day and night echograms from fall DVM surveys showing zooplankton and fish in the water column. A low display threshold (-87 dB, 20 log R amplification, and time varied threshold -120 dB @ 1 m) was used to make zooplankton visible. Each cell shows an approximately two minute transect segment with a 2-100 m vertical range gridded by 5 m layers.

4.4.10.2 Acoustic sampling limits for detection of fish

The large range of depths occupied by fish during daytime DVM sampling, the small size of age-0 *O. nerka*, and frequent schooling of fish sometimes exceeded the limits of our acoustic system to quantitatively measure fish distribution patterns. Small fish deep in the water column return a weak echo that can be masked by acoustic system electronic noise (low signal to noise ratio), leading to underestimation of fish density (fish/m^3) and TS (Parker-Stetter et al 2009). This problem was more severe in Anderson Lake where many fish occurred below 80 m during the day in both summer and fall DVM surveys, much deeper than in Seton Lake (Figure 64). Calculations based on in situ acoustic system noise levels and fish sizes from trawling showed that density of the smallest fish present (i.e. minimum length in the trawl catch) was underestimated below about 50 m in the summer and 70 m in the fall (Table 29). These fish were difficult to even detect below 75-85 m in the summer and 95-100 m in the fall. In particular, in Anderson Lake on August 2, when the highest measured fish density was at 109 m, it is likely that many small fish were too deep to detect. Although larger fish of this size class were detectable to greater depths than those in Table 29, it is certain that small fish density was underestimated in deep water during the daytime period of the DVM study, especially in Anderson Lake. This problem is not fully correctable, and we made no adjustment for it because data quality appeared adequate as-is for comparing vertical distributions of fish between the two lakes. This problem had little affect on the medium size class and the large size class was unaffected. Underestimation of small fish density deep in the water column was reduced by our choice of fish tracking rather than echo integration for processing the DVM data. With the lower amplification used for echo integration, many fish in deep water that were below the threshold of detection for echo integration were detectable with tracking (Figure 58). This problem could be eliminated in future DVM studies by making supplementary observations with the transducer lowered closer to the fish in deep water (i.e. within ranges specified in Table 29) in addition to sampling the shallower fish as we did in 2014.

Schooling of fish during daytime DVM sampling reduced the quality of acoustic data to some extent. Generally, patchy distributions of fish due to schooling lead to higher variance of fish density estimates, weakening statistical comparisons (Simmonds and MacLennan 2005). Also, it appears that the TS of fish tracks in schools were occasional over-estimated when multiple fish were erroneously classified as a single fish by the acoustic processing software. We deleted from analysis all fish of the large size class that were part of schools (39 tracks in 16 schools) because close examination suggested that they were probably multiple targets. This was done to avoid error in depth distributions of large fish. Also, it was difficult or impossible to accurately track fish in schools, depending on their density, so counts of fish in schools were only approximate. However, a comparison of daytime fish density estimates from tracked fish and echo integration (the preferred method for schooled fish) showed close to 1:1 correspondence (slope 0.82) and a low but significant regression relationship (R^2=0.47, P<0.01) between the two methods in the 5-60 m depth range where schools were common in both lakes (Figure 64 and Figure 65). In the 60-130 m depth range, where schools were only common in Anderson Lake, the degree of correlation was nearly identical (R^2=0.48, P<0.01), but density

estimates from echo integration were about 1/10 of those from fish tracking. Fish tracking typically underestimates fish density where schools are present, whereas echo integrations does not (Simmonds and MacLennan 2005), so this is additional evidence that fish tracking was the better processing choice for the DVM study.

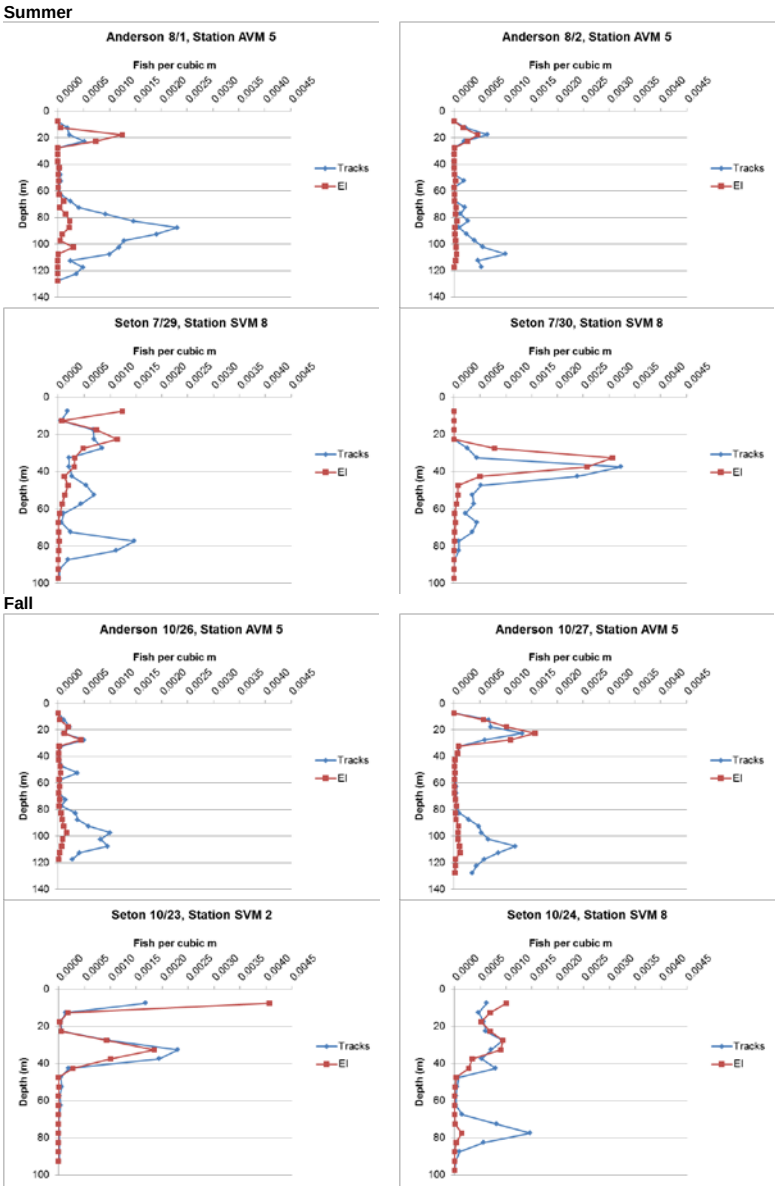


Figure 64 Daytime fish density versus depth during the DVM study, estimated from fish tracks (TR) and echo integration (EI). Data points are means of density estimates from contiguous one minute time intervals on each sampling date.

Table 29 Estimated maximum depths for detection and unbiased estimation of fish density and TS of the smallest fish present during summer and fall 2014 acoustic surveys of Seton and Anderson Lakes. This analysis used the smallest fish of the small size class and acoustic system noise levels measured in situ during the surveys. TS was estimated from Love's (1977) dorsal aspect model corrected for depth according to Boyles Law (Mukai and Iida 1996). Required signal to noise ratios were 9 dB for simple fish detection (on the acoustic beam axis only), and 15 dB for unbiased estimates of beam volume and TS (Parker-Stetter et al 2009). Minimum fish lengths are from trawl sampling during the seasonal surveys.

			Minimum	Dorsal	Depth	Maximum allowable value			
						For detection on echograms		For unbiased estimates	
Lake	Date	Size Class	Length (mm)	aspect TS (dB)	Comp. TS (dB)	Noise (dB)	Depth (m)	Noise (dB)	Depth (m)
Anderson	8/1-2	small	23	-57.2	-62.8	-72	75-85	-78	50-60
Seton	7/29-30	small	27	-55.8	-61.9	-71	85-95	-77	55-65
Anderson	10/26-27	small	31	-54.7	-61.1	-70	95-100	-76	70
Seton	10/23-24	small	36	-53.5	-60.0	-69	100	-75	70

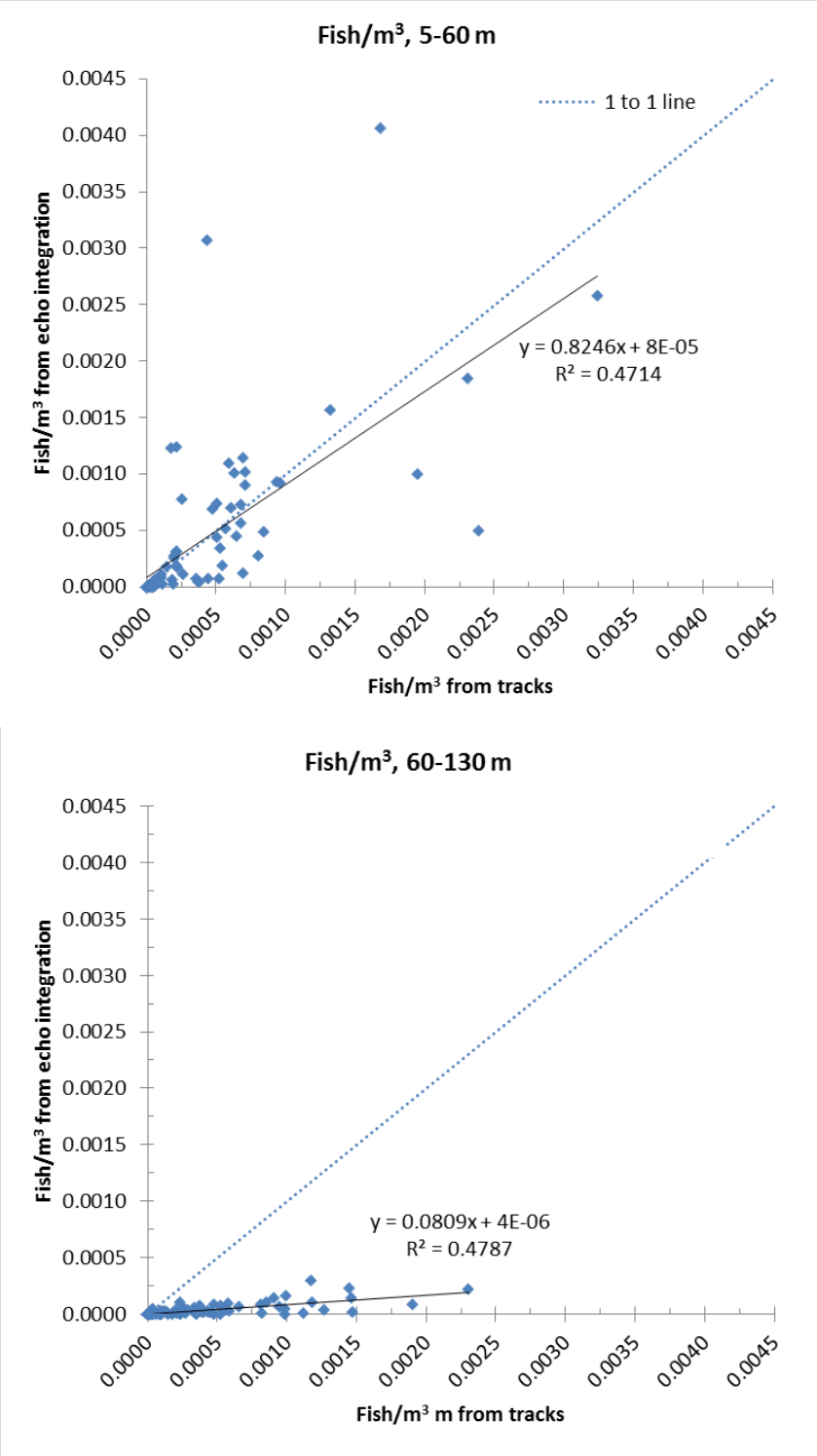


Figure 65 Linear regression of daytime total fish density estimates (all size classes combined) from echo integration and tracked fish in the 5-60 m depth range (upper) and 60-130 m depth range (lower) using data from both lakes combined. Data points are means of density estimates from contiguous one minute time intervals from all sampling dates of the DVM study.

4.4.10.3 Diel patterns in the vertical distribution of fish

General fish behavioral patterns from echograms

Echograms from DVM sampling showed several behavioral patterns of fish in Anderson and Seton Lakes without size class distinction (echograms show all size classes together). Fish appeared on echograms as individual fish tracks (fish seen individually) and as schools (groups of indistinguishably overlapping fish tracks, Figure 66). Schools were seen on all daytime echograms and no night echograms from either lake, whereas individual tracks were common on all echograms from both periods at some depth in the water column. During the day in both seasons at Anderson Lake, schools and individual fish were detected throughout the water column to the limits of detection (120-125 m, Table 30). Most fish were below 65 m in the summer, while most fish were even deeper, below 75 -95 m, in the fall, and schools tended to be slightly shallower than individual fish in both seasons. In Seton Lake during the day in both seasons, schools and individual fish were observed throughout the water column to the limits of sampling (95-98 m), but most fish were above 85 m (Table 30). Schools were much shallower than individual targets in Seton Lake, extending to a maximum of 50 m, except at highly turbid SVM 2 where they were only found to 7 m. At dusk, fish in both lakes migrated to surface waters to feed (not shown), then at night descended to a narrow band (15-30 m thick) centred on about 20 m in the summer and 27 m in the fall (Figure 66). These observations indicate that fish made a DVM in both lakes, but many fish in both lakes, especially those in schools, remained high in the water column during the day, while most fish that did not school descended to great depths at that time.

Size of fish observed with acoustics in the DVM study

The acoustic targets seen in the DVM echograms comprised fish of a wide range of sizes in both lakes. Overall TS values of tracked targets ranged from -64.6 to -28.2 dB in Seton Lake (equivalent to fish lengths of 9-756 mm by Love's 1977 dorsal model), and from -64.9 to -28.5 dB in Anderson Lake (estimated lengths 9-729 mm). These estimated fish lengths roughly corresponded to the minimum and maximum sizes of fish captured in the combined trawl and gill net catches from the lakes of 27 to 750 mm in Seton Lake and 23 to 650 mm in Anderson Lake. As described in Methods, we separated tracked targets by mean TS to examine the vertical distribution patterns of small, medium, and large size classes of fish individually.

Daytime vertical distributions of small and medium size classes of fish

For both lakes and seasons, daytime profiles of small and medium size fish density versus depth (fish/m^3) nearly always showed peaks at more than one depth (usually two, Figure 67 - Figure 70), and these peaks corresponded to layers of schools and individual tracks on echograms. On most sampling dates, the depths of shallow peaks were similar for the small and medium size classes. Deep and shallow water peaks were well defined on most sampling dates, except at Seton Lake in the summer, especially for the medium size group on July 30. The July 30 daytime sample was somewhat anomalous because although it was a daytime sample by our definition of diel periods, it was collected immediately before dusk, and it appears that fish

had already begun to ascend from their maximum daytime depth range, showing peak densities of small and medium fish at a shallower depth than on the previous day and a unimodal depth distribution (Figure 67).

Depths of daytime fish layers were fairly consistent among the DVM surveys of Anderson Lake. The daytime shallow layer of medium and small fish was between 10 and 30 m in summer and fall (Figure 67 and Figure 68). The deep layer was 65 m to >125 m for small fish and 50-100 m for medium fish in the summer, and about 70 m to >120 m for both size groups in the fall, with some variation among days. Vertical distribution patterns were more complex in Seton Lake. For small fish on July 29 at station SVM 8, there two shallow layers (5-30 m and 30-60m) and a deep layer at 70-90 m, while medium fish were 5-55 m deep. On July 30 at the same station, small and medium fish density profiles were both unimodal, with small fish 25-85 m and medium fish 25-40 m. At the two stations sampled in the fall at Seton Lake depths of fish layers differed greatly. At turbid station SVM 2 the upper layer of both size classes was 2-15 m and the deeper layer was 25-45 m. At clearer station SVM 8 there was an upper small fish layer 2-45 m and a deep one 65-90 m, and single layer of medium sized fish 5-45 m. Turbidity and the light attenuation coefficient (k) were 11.1 NTU and -0.69 at SVM 2, compared to 4.1 NTU and -0.41 at SVM 8.

In all cases, the shallow layer of small and medium size fish at least partially overlapped the zooplankton layer, although on some dates many of the shallow fish were outside the zooplankton layer (e.g., July 29 and October 24 at Seton Lake station SVM 8, Figure 67 to Figure 70). In the summer, the shallower fish layers extended into warm surface waters (17-20°C range) in both lakes. In the fall the shallow layer in both lakes experienced cool temperatures (≤13°C). In all cases, the deep water fish layer for small and medium size groups was below the zooplankton layer and in the hypolimnion where temperatures were cold (<5°C).

Daytime vertical distribution of the large size class of fish

During the day, fish of the large size class were found in low densities in a narrow unimodal band on most sampling dates (Figure 67 to Figure 70). In Anderson Lake in the summer, they were in a band above the thermocline from 12 to 22 m, while in the fall they were more dispersed from 17 to 72 m, with highest densities above the thermocline. No large fish were detected in the summer DVM surveys of Seton Lake, but in the fall low densities were detected between 20 and 50 m, at or below the thermocline.

Nocturnal vertical distribution of all size classes of fish

The nocturnal vertical distributions of all size classes of fish were unimodal on all sampling dates, and the depth of peak fish density was quite consistent between lakes and seasons (Figure 70). The night time depth distribution of most small and medium fish was much shallower than their daytime distribution (Figure 67 to Figure 70). At night in the summer in Seton Lake they were found in a narrow band about 15 m wide centred on 18 m, whereas in the fall the band was wider (~30 m) and centred on 24 – 30 m. In Anderson Lake, nocturnal small targets were at about the same depths as in Seton Lake but somewhat more dispersed in the summer (30m wide) and fall (30-5m wide).

Differences in the mean depth of size classes between lakes

As described in Methods, we calculated the weighted mean depth (WMD) of each size class for comparing the depths occupied by fish in Seton and Anderson Lakes and with other studies. Overall, daytime WMDs of small fish were considerably greater in Anderson Lake than in Seton Lake (72-91 m versus 17-57 m, Table 31), while their night time WMDs were similar in both lakes (18-31 m, Table 32). For the medium size class, daytime WMDs were also greater for Anderson Lake (58-75 m versus 13-34 m for Seton Lake), as were night time WMDs (30-38 m versus 20-30 m). For the large size class, daytime WMDs were similar between lakes (Anderson 17-35 m, Seton 24-40 m), whereas night time WMDs were greater in Anderson than in Seton Lake (34-47 m versus 23-34 m). Although WMD did not reflect the bimodality of some of the depth distributions, it still provided an unbiased statistic for describing the central tendency of the entire depth distribution of each size class.

Vertical migration distances and rates for the small size class

As a consequence of the deeper daytime depths in Anderson Lake, vertical migration distances for small fish during the evening DVM were greater in Anderson Lake (45 – 67 m) than in Seton Lake (-2 to 46 m, Table 31 and Table 32). The longer vertical migration distance in Anderson Lake would result in a longer migration time than in Seton Lake. In Anderson Lake we have a complete record through the dusk to night period on October 26 (not shown). On that date, most small targets were at daytime depth of ~90 m until 16:55, after which they began their upward migration, reaching a minimum depth of ~5 m at 18:22 (ascent of 85 m in 87 min, ascent rate = 0.98 m/min). Subsequently, they settled to a nighttime depth of 31 m by 18:50 (descent of 26 m in 28 min, descent rate = 0.93 m/min). Our data record for Seton Lake is not as complete, but due to the shallower daytime depths, vertical migration times would have been shorter, as little as zero minutes at SVM 2 in the fall.

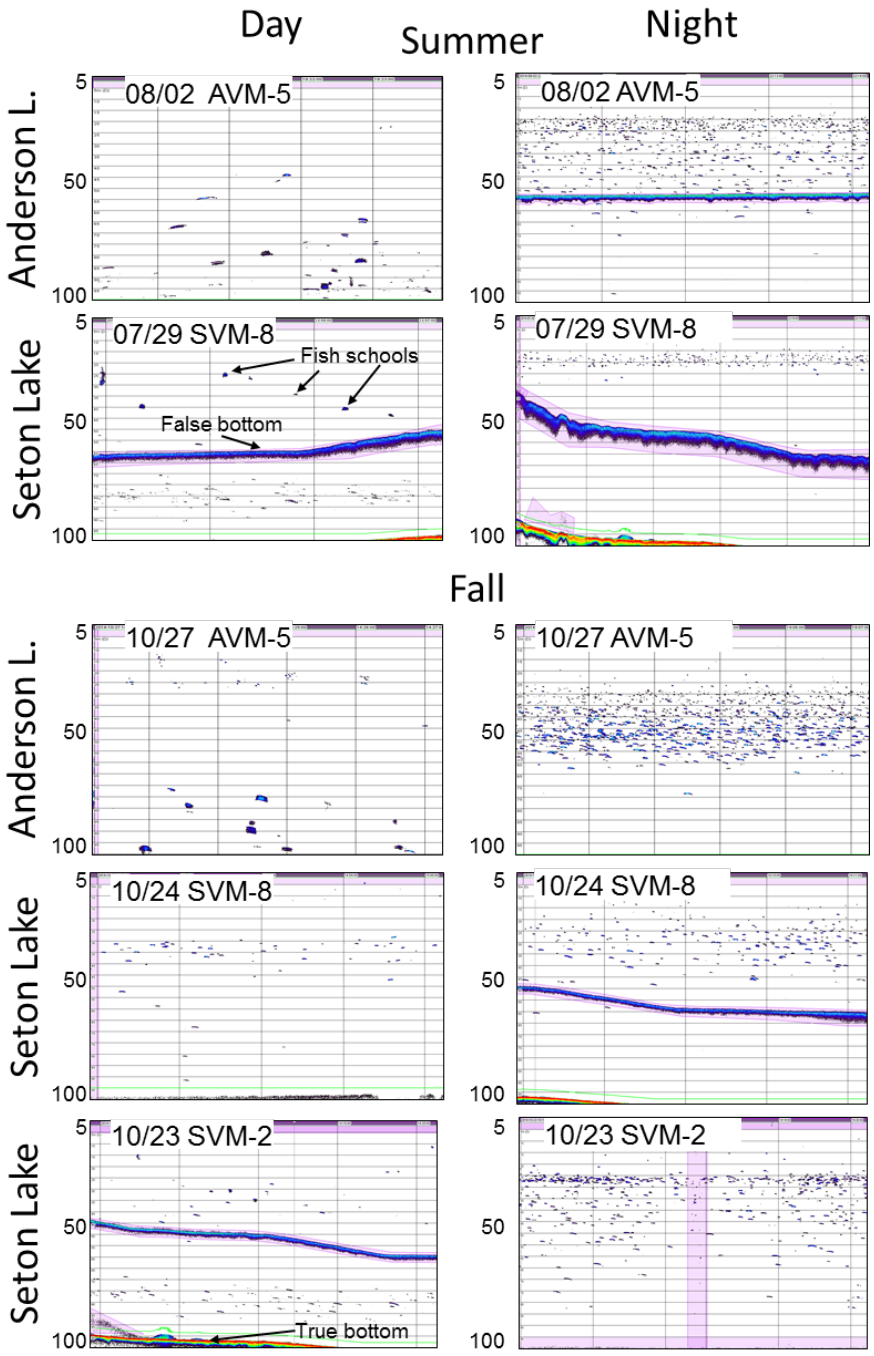


Figure 66 Typical segments of day and night time echograms from summer and fall DVM sampling of Seton and Anderson Lakes. Echogram settings (threshold = -65 dB, range compensation = 40 log R) were sensitive enough to show Sockeye fry to the maximum depth displayed. Gridding is the same on all plots: 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 depth of analysis.

Table 30 Observed daytime depth range of fish schools and individual fish tracks on echograms from 2016 acoustic surveys of Seton and Anderson lakes.

					Schools (depths in m)			Individual fish (depths in m)		
					Total	Layers		Total	Layers	
Lake	Season	Date	Station	Start Time	Range (m)	Major	Minor	Range (m)	Major	Minor
Anderson	Summer	1-Aug	AVM 5	19:19	12-95	65-95	12-25	12-125[a]	80-125[a]	12-25
"	"	2-Aug	AVM 5	16:13	14-110	70-110	13-25	15-120[a]	95-120[a]	13-25
"	Fall	26-Oct	AVM 5	14:56	15-110	80-110	15-30	5-120[a]	80-120[a]	15-30
"	"	27-Oct	AVM 5	13:49	12-120[a]	75-120	12-30	15-130[a]	100-130[a]	15-40
Seton	Summer	29-Jul	SVM 8	12:30	5-50	5-50	none	35-95	70-85	none
"	"	30-Jul	SVM 8	19:25	25-40	25-40	none	20-85	35-50	none
"	Fall	23-Oct	SVM 2	14:30	0-7	0-7	none	6-95[a]	25-45	none
"	"	24-Oct	SVM 8	14:00	8-40	8-40	none	7-98[a]	70-85	20-50

[a]The deepest depth of the range is the maximum depth of data collection

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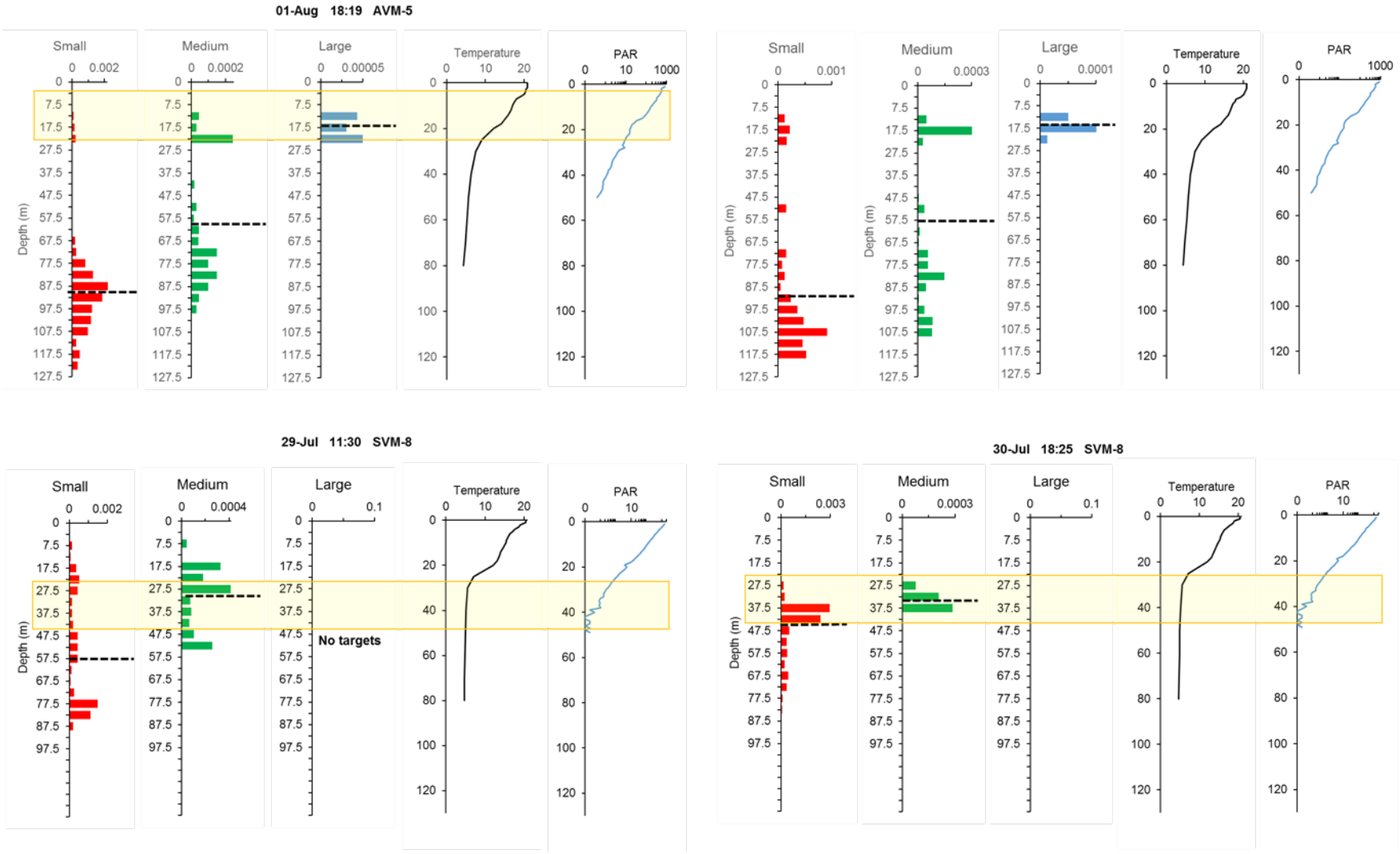

Figure 67 Mean daytime vertical distribution in the summer of small, medium, and large fish targets in both lakes. The mean fish depths (dashed lines) and the zooplankton layer (yellow shading), as well as temperature (°C), and light (PAR, µmoles/m²/s) profiles are also shown. Date, start time of acoustic data collection, and sampling station are indicated atop each set of five plots.

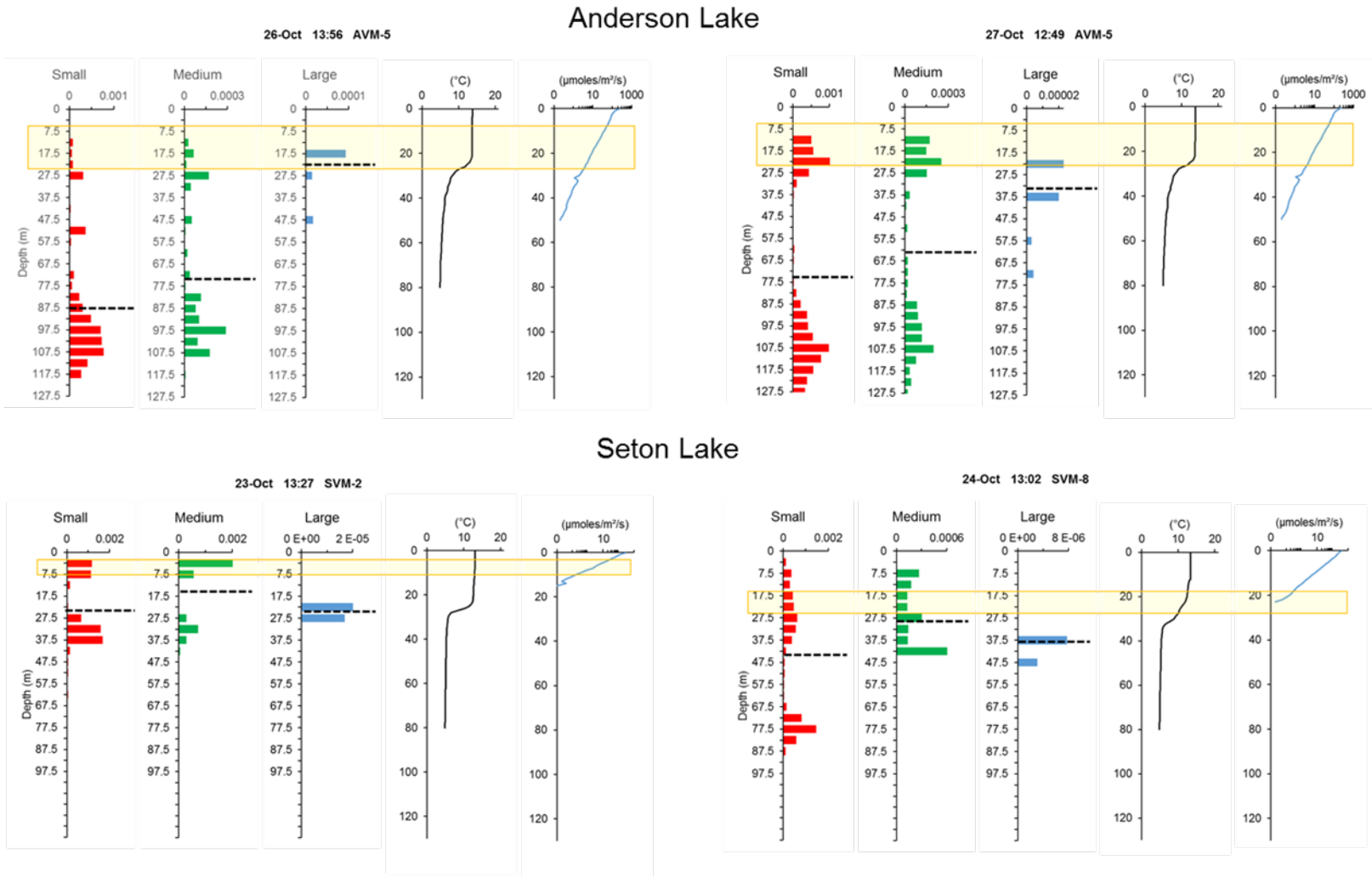

Figure 68 Mean daytime vertical distribution in the fall of small, medium, and large fish targets in both lakes. The mean fish depths (dashed lines) and the zooplankton layer (yellow shading), as well as temperature (°C), and light (PAR, µmoles/m²/s) profiles are also shown. Date, start time of acoustic data collection, and sampling station are indicated atop each set of five plots.

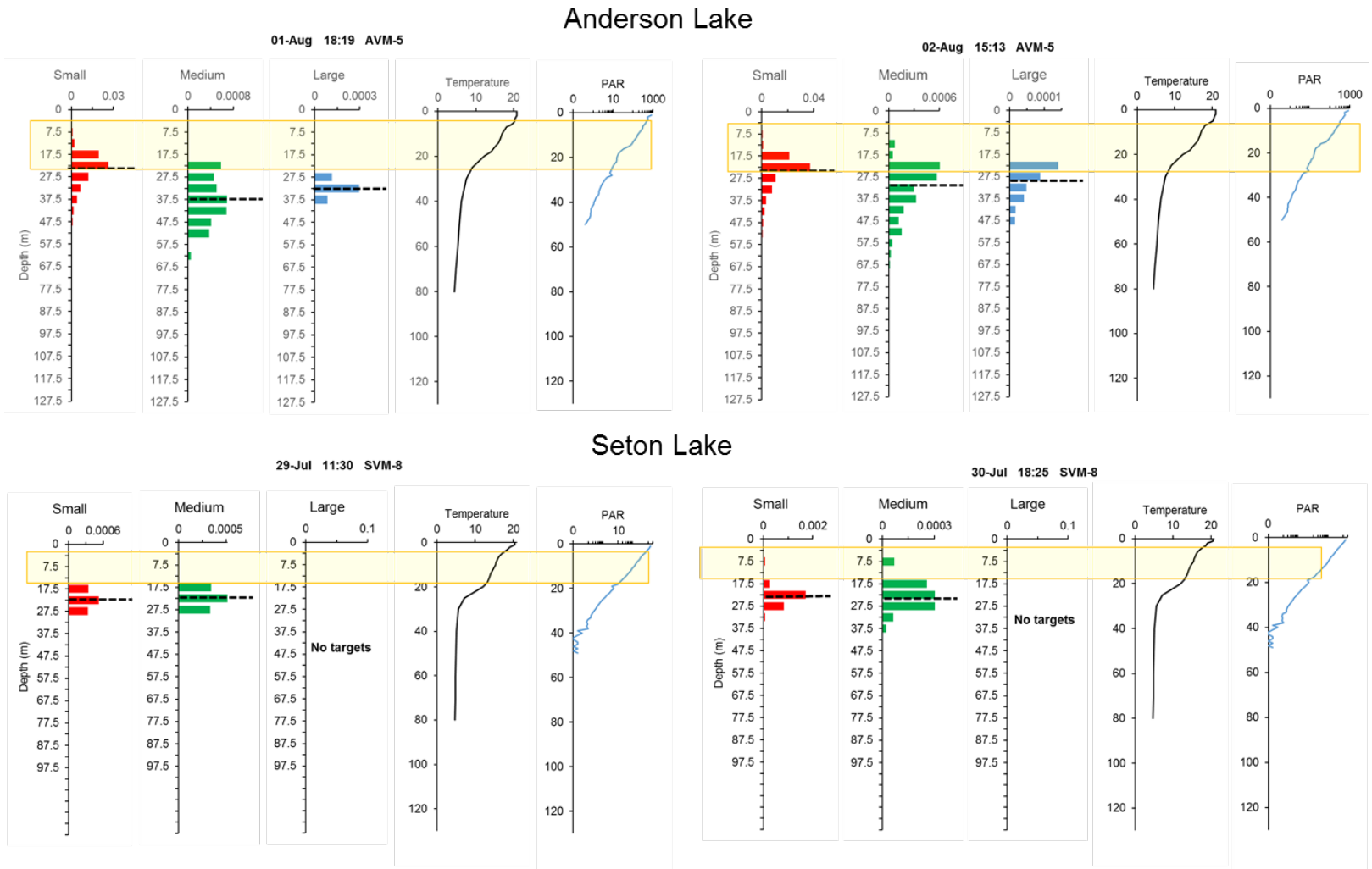

Figure 69 Mean nocturnal vertical distribution in the summer of small, medium, and large fish targets in both lakes. The mean fish depths (dashed lines) and the zooplankton layer (yellow shading), as well as temperature (°C), and light (PAR, µmoles/m²/s) profiles are also shown. Date, start time of acoustic data collection, and sampling station are indicated atop each set of five plots.

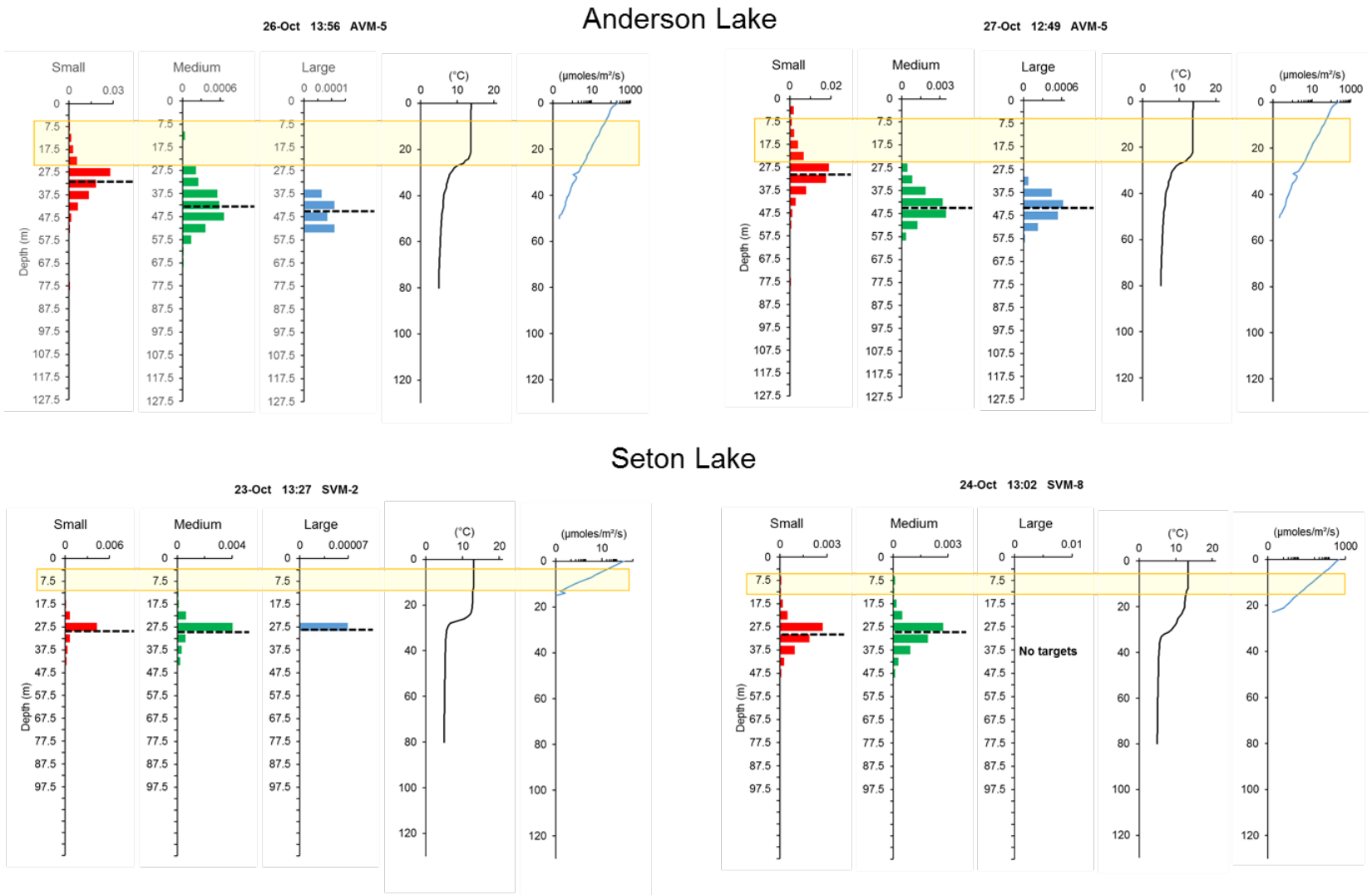

Figure 70 Mean nocturnal vertical distribution in the fall of small, medium, and large fish targets in both lakes. The mean fish depths (dashed lines) and the zooplankton layer (yellow shading), as well as temperature (°C), and light (PAR, µmoles/m²/s) profiles are also shown.

Table 31 Mean daytime depths of small, medium, and large acoustic targets during DVM sampling.

Year	Date	Season	DVM transect	Start time (PST)	Weighted mean daytime depth (m)					
					Small targets		Medium targets		Large targets	
					Mean	±95% CI	Mean	±95% CI	Mean	±95% CI
Seton Lake										
2003	26-Jul	Summer	Tr 6	13:14	17.0					
2014	29-Jul	Summer	SVM 8	11:30	57.4	5.6	31.2	14.1	na	
2014	30-Jul	Summer	SVM 8	18:25	45.6	1.2	34.3	2.2	na	
2003	24-Oct	Fall	Tr 6	15:15	29.0					
2014	23-Oct	Fall	SVM 2	13:27	24.7	4.4	13.5	5.3	24.8	4.4
2014	24-Oct	Fall	SVM 8	13:02	49.5	2.9	28.9	8.7	40.3	4.3
Anderson Lake										
2003	07-Aug	Summer	Tr 3	13:50	72.5					
2014	01-Aug	Summer	AVM 5	18:19	90.4	3.3	59.6	11.0	17.8	127.1
2014	02-Aug	Summer	AVM 5	15:13	90.7	17.1	57.6	25.5	16.4	83.1
2003	15-Oct	Fall	Tr 3	12:50	71.9					
2014	26-Oct	Fall	AVM 5	13:56	87.3	10.0	75.2	10.1	22.7	47.4
2014	27-Oct	Fall	AVM 5	12:49	76.3	14.1	63.1	11.9	34.6	34.3

Table 32 Mean nighttime depths of small, medium, and large acoustic targets during DVM sampling.

Year	Date	Season	DVM transect	Weighted mean nighttime depth (m)					
				Small targets		Medium targets		Large targets	
				Mean	±95% CI	Mean	±95% CI	Mean	±95% CI
Seton Lake									
2003	26-Jul	Summer	Tr 6	21.0					
2014	29-Jul	Summer	SVM 8	18.0	0.0	19.5	0.2	24.5	4.2
2014	30-Jul	Summer	SVM 8	17.9	0.1	19.7	0.2	23.4	2.8
2003	24-Oct	Fall	Tr 6	NA					
2014	23-Oct	Fall	SVM 2	30.7	0.1	29.5	0.3	33.8	3.4
2014	24-Oct	Fall	SVM 8	24.3	0.5	30.1	0.7	31.9	1.5
Anderson Lake									
2003	07-Aug	Summer	Tr 3	NA					
2014	01-Aug	Summer	AVM 5	23.4	0.1	31.9	0.8	37.7	11.5
2014	02-Aug	Summer	AVM 5	24.0	0.2	30.3	0.5	34.4	3.4
2003	15-Oct	Fall	Tr 3	NA					
2014	26-Oct	Fall	AVM 5	30.4	0.3	37.4	1.0	46.5	0.8
2014	27-Oct	Fall	AVM 5	28.9	0.9	37.8	1.4	44.5	0.1

4.4.10.4 Depth of fish versus turbidity and light

In Seton and Anderson lakes, a relatively small increase in turbidity caused significant changes in light transmission (Figure 71A, $R^2 = 0.84$, $P<0.01$), with a corresponding decrease in mean depth of small size class acoustic targets that represented mainly age-0 *O. nerka* (Figure 71B, $R^2 = 0.69$, $P<0.05$). Similarly, Chernoff (1971) and Levy (1990) found that the daytime depths of juvenile Sockeye in the four basins of Owikeno Lake were negatively related to turbidity levels. Juvenile Sockeye in glacially turbid lakes in the Skeena and Nass watersheds were also found much closer to the surface than those in clear lakes (Hume and MacLellan 2008; MacLellan and Hume 2011). The effect of turbidity on Sockeye depth selection is related to its affect on light transmission as measured by the attenuation coefficient (Lloyd 1987, Levy 1990).

In Seton and Anderson lakes, the mean depth of small targets was significantly negatively related to the light attenuation coefficient (Figure 72, $R^2 = 0.73$, $P<0.01$), and there was no significant difference between our relationship and the relationship Levy (1990) found for five other Sockeye and Gwenish rearing lakes in British Columbia (Figure 66, ANCOVA, $P<0.01$). Combining data for small fish from our study with data from Levy (1990) results in a common relationship of: Depth = 89.6 + 107.8•k, ($R^2 = 0.78$). There was less data available for medium and large targets, and a weakly significant relationship was observed for medium targets (Figure 72, $R^2 = 0.61$, $P<0.05$), but none for large targets ($R^2 =0.07$).

Nocturnal depth distribution of juvenile Sockeye appears to be associated with the thermocline (Narver 1970; McDonald 1973; Woodey 1972; Levy 1990). Similarly, in Anderson and Seton lakes, mean nocturnal depths of juvenile *O. nerka* were close to the bottom of the metalimnion (Figure 69 and Figure 70). In Seton Lake, temperatures at depths of the night time age-0 *O. nerka* layer ranged from 5.8 -13.4 °C and averaged 10.9 °C, similar to the mean nocturnal temperatures of 11.0 (range = 8 - 15 °C) found by Levy (1990). The mean nocturnal temperature in Anderson Lake was somewhat lower at 9.4 °C (range 8.2 - 10.3 °C).

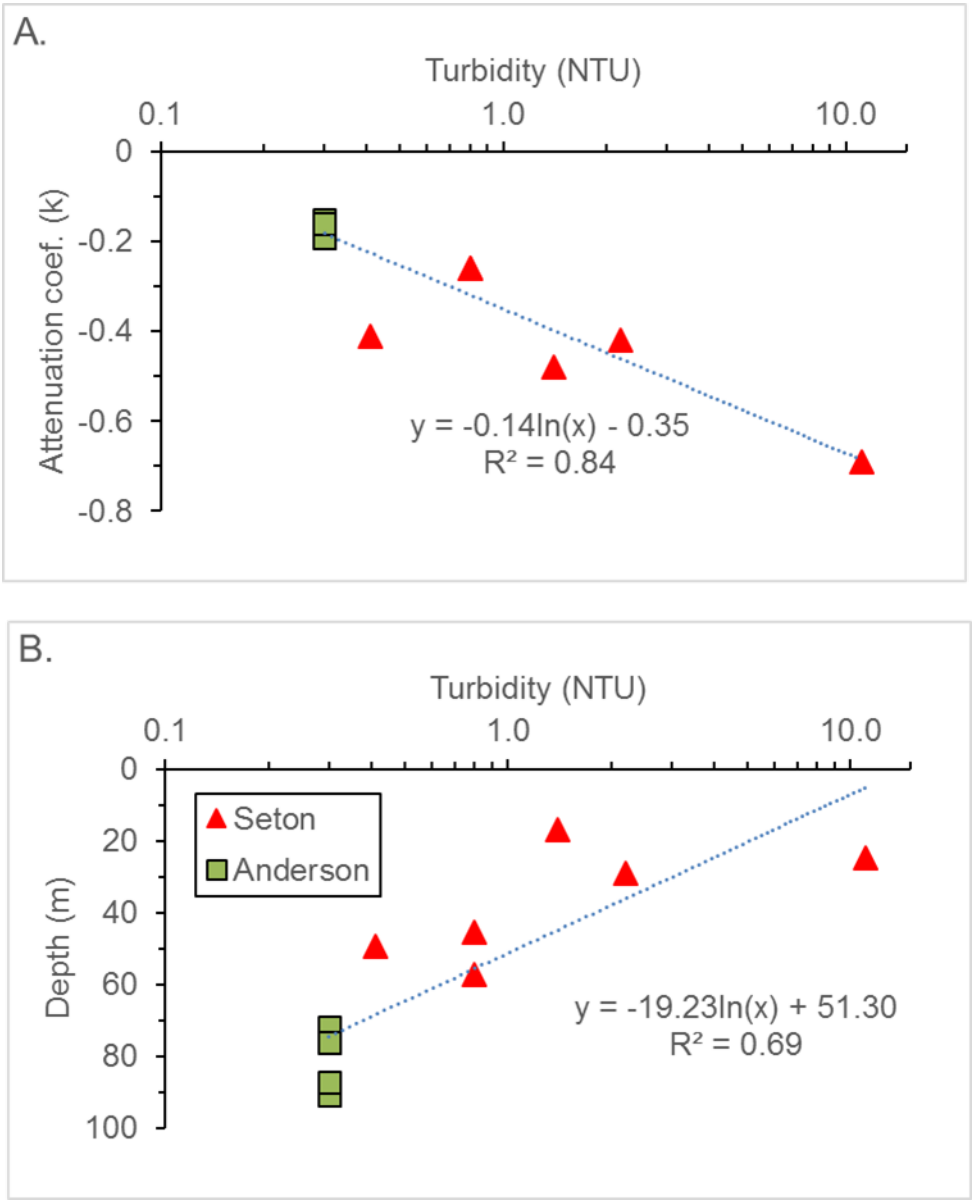

Figure 71 Relationship between A) turbidity and the light attenuation coefficient and B) turbidity and the mean depth of small targets representing age-0 O. nerka in Anderson and Seton lakes.

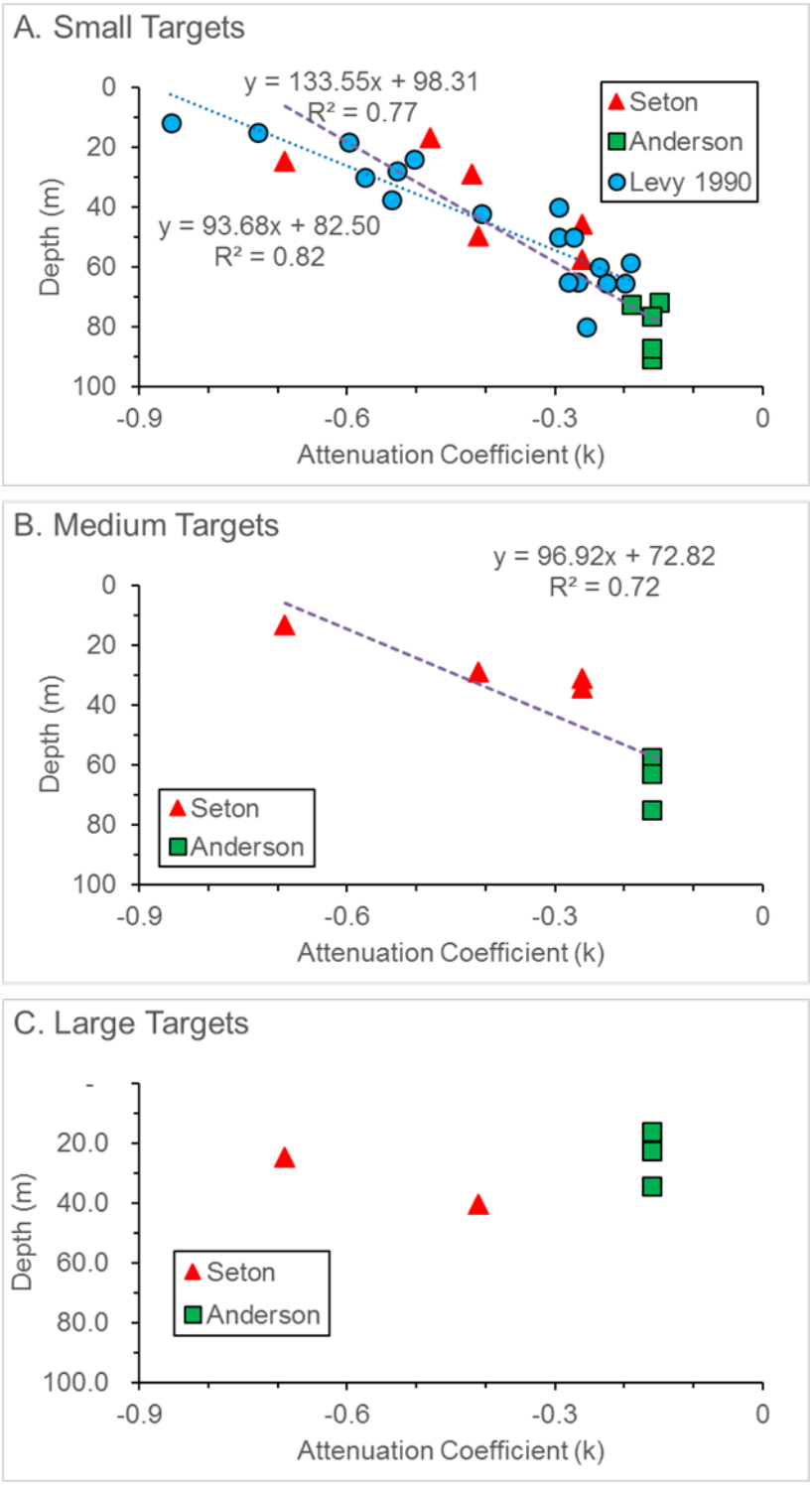

Figure 72. Relationship between the mean depth of fish targets and the light attenuation coefficient in Anderson and Seton lakes. Data from other BC lakes from Levy (1989,1990).

4.4.11 Summary and Discussion

In this study, we examined between-lake differences in the physical limnology (specifically turbidity, light transmission, and temperature) and in juvenile *O. nerka* ecology, biology, and behaviour in Anderson and Seton Lakes to explain some previously observed and puzzling differences in growth and survival rates of *O. nerka* rearing in the lakes. In essence, prior studies indicated that juvenile Sockeye and Gwenish grew faster and survived better in Seton Lake than in Anderson Lake despite higher densities of their prey (zooplankton) in Anderson Lake (Limnotek 2015). We also estimated the extent of the early migration of *O. nerka* fry from Anderson to Seton Lake and investigated its impact on the growth and survival of Gates Creek Sockeye. Based on these lines of enquiry we developed a series of hypotheses and conducted a field study in 2014 to test them. Conclusions about our findings regarding these hypotheses are summarized below.

H1: Sockeye fry remain in the upper water column of turbid Seton Lake throughout the day along with their zooplankton prey, and as a consequence the diel vertical migration (DVM) of Sockeye fry is reduced or absent in Seton Lake while it is extensive in relatively clear Anderson Lake

This hypothesis was not confirmed by 2014 studies because Sockeye fry (and Gwenish) performed an appreciable DVM in both lakes, and it remains unclear whether the reduced vertical migration that we observed in Seton Lake was a factor in the higher growth and survival rates of age-0 *O. nerka* that we measured there.

The premise behind this hypothesis was that turbidity in Seton Lake would provide enough protection against predation by piscivorous fish that age-0 *O. nerka* would not vertically migrate, enabling them to feed continuously in the zooplankton layer during the day rather than just during brief crepuscular periods. Although mean daytime depths of fish were related to light attenuation, it appears that the decrease in illumination levels in surface waters during the study was inadequate for the majority of age-0 *O. nerka* to remain in the epilimnion. Overall, the depth distribution of small fish in Seton Lake was shallower than in Anderson Lake, but on only one sampling day, at the station near the Carpenter Lake outflow when the epilimnetic turbidity (11 NTU) and light attenuation (-0.69) were the highest of the DVM study, were all of the fish found above 45 m. Even then fry formed two layers during the day, with many fish below the plankton layer. It should be noted, however, that our investigation of vertical migrations in Seton Lake was hampered by 1) a strong longitudinal turbidity gradient in the lake that we were unaware of at the time of the summer DVM survey (in the summer we only sampled a station where the water was relatively clear), 2) seasonal variability in the Carpenter outflow turbidity, and 3) the limited number of DVM stations we were able to sample to assess within-lake variation in DVM patterns (one in the summer and two in the fall). In hindsight we should have determined turbidity levels throughout the lake apriori so we could have included a station with higher turbidity in the summer sampling program.

H2: The daytime depth distribution of *O. nerka* fry is related to differences in light penetration in each lake and they fit Levy's (1990) model of mean depth vs the light attenuation coefficient

Although the vertical distribution of age-0 *O. nerka* densities (fish/m^3) was frequently bimodal in both lakes, a between-lakes comparison of weighted mean depth, an unbiased estimator of overall depth distribution for a group of fish, did show that daytime depths of *O. nerka* fry were consistently deeper in Anderson Lake than in Seton Lake. We found that there was not only a significant relationship between mean depth and the light attenuation coefficient within the study lakes but that the relationship was not significantly different from Levy's (1990) model for age-0 Sockeye Salmon and Gwenish in several BC lakes.

H3: The depth distribution of Sockeye fry during daytime and dusk (excluding night) conforms to Scheuerell and Schindler's (2002) antipredation window model

Scheuerell and Schindler (2002) modeled a lake rearing system where Sockeye Salmon fry formed a single modal group during the day that was at a depth well below the Visual Illuminance Threshold (VIT). The VIT is the maximum depth with enough light for a predator to see its prey, and VIT varies among fish species (Henderson and Northcote 1985). As Scheuerell and Schindler's (2002) model poorly represents the multimodal (usually bimodal) daytime depth distribution of *O. nerka* fry in Anderson and Seton lakes, we did not attempt to fit our data to their model.

The unimodal type of DVM described by Scheuerell and Schindler (2002) is well documented for age-0 *O. nerka* in many of their rearing lakes (Narver 1970; McDonald 1973; Woodey 1972; Levy 1990), and the resulting daytime distribution in deep water with inadequate light for visual predation is well established as a mechanism to avoid predation (Levy 1990; Scheuerell and Schindler 2002). Consequently, the depth to which *O. nerka* descend during the day is correlated with water clarity. Levy (1990) found that the mean daytime depth was much shallower in turbid Owikeno Lake (~20 m) than in other more transparent lakes such as Quesnel (60-80 m) and Shuswap Lake (~50 m). This DVM strategy has a cost, though, if by diving deep the *O. nerka* fry forgo better feeding opportunities in shallow water. Although this typical DVM (deep by day, shallow dusk-dawn) is found in many lakes, *O. nerka* are very adaptable, and other predator avoidance strategies have been documented in the wide range of habitats they occupy. For example, Hardiman et al (2004) describes a seasonally changing strategy in a Colorado reservoir where juvenile Gwenish used the warm epilimnion in summer as a thermal refuge from its primary predator, Lake Trout (*S. namaycush*), which require colder water.

We found that although juvenile *O. nerka* performed a DVM in Anderson and Seton Lakes, their day time vertical distribution was more complex than we are aware of elsewhere. Age-0 *O. nerka* were not confined to single part of the water column in either lake but were typically distributed bimodally during the day, sometimes with a large proportion of the fry found in well illuminated waters above the thermocline in both lakes.

These shallow fish were mostly in schools. Both the mean depth and the depth of the deepest modal group was shallower in Seton Lake than in Anderson Lake but the majority of fish in Seton Lake were not shallow enough to be feeding in the zooplankton layer known from sampling during the day. It is possible that a zooplankton layer was present at a depth greater than the 30m zooplankton haul depth. This uncertainty will be resolved with adjusted zooplankton sampling in 2016 to include hauls from depths greater than 30m as well as from 30 m.

These complex vertical distribution patterns suggest that fry in Anderson and Seton Lakes were using two different antipredation strategies during the 2014 DVM study. One strategy appeared to be to form schools in the upper water column where illumination was sufficient for schooling, which reduces but does not eliminate visual predation risk, while allowing feeding if zooplankton were present. The other strategy was to descend to depths where illumination was inadequate for piscivores to prey on them (below the VIT), but where zooplankton was mostly absent. In our study the presence of schools was a useful behavioral indicator of illumination levels at the depths occupied by fish, which included most of the water column: if schools were present then there was enough light for the fry to see each other, i.e., they were above the VIT. As illumination decreases with depth, light becomes insufficient for school formation at some point. The surprising depth to which schools were observed in both lakes (to at least 120 m in Anderson Lake and to 50 m even in turbid Seton Lake) attest to the high visual sensitivity of juvenile *O. nerka*.

H4: Sockeye fry migrated from Anderson Lake to rear in Seton Lake in their first summer of life, but not vice versa

The spring and early summer migration of Gates Creek Sockeye fry from Anderson Lake, through Portage Creek, and into Seton Lake that was documented in the late 1950's and early 1970's (Geen and Andrews 1961; Woodey 1975) must have also occurred in 2014, as many Gates Sockeye fry were identified by DNA analysis in Seton Lake in the summer 2014 survey. Expanding the DNA results with the acoustics resulted in an estimated 1.5 million Gates origin Sockeye fry in each lake, indicating that approximately half of the Gates age-0 Sockeye population left Anderson Lake for Seton Lake. There may have been some upstream movement of Portage Creek fry into Anderson Lake, as DNA analysis of the trawl catch found a small number of age-0 Portage Creek fish rearing in Anderson Lake. But it is also possible that they are offspring of lake spawning adults because spawners from the Portage Creek run have been recorded beach spawning in Anderson Lake in a few recent years. Although none were observed in 2013, lake shore surveys are not always conducted and none were recorded in 2013 (DFO, Lakes Research Program, Cultus lake Salmon Research Laboratory, Data on file).

H5: Age 0 *O. nerka* growth and survival is higher in Seton Lake than in Anderson Lake

Growth rates of age-0 Sockeye (both stocks combined and Gates Creek Sockeye only) were much higher in Seton Lake than in Anderson Lake. Sockeye were at least 40% longer and 300% heavier in Seton Lake and fall sizes of 1.3 g in Anderson Lake and 4.4 g in Seton Lake were near the extremes seen in other Fraser River Sockeye lakes. For example fall Sockeye size ranged from 1.5 - 3.5 g over 21 sample years in Shuswap Lake and from 1.9 – 4.3 g over 19 years in Quesnel Lake. (DFO, Lakes Research Program, Cultus lake Salmon Research Laboratory, Data on file).

The larger mean size of age-0 Sockeye in Seton Lake in 2014 is puzzling for a few reasons. Zooplankton is the only food source for *O. nerka* in these lakes, but the 2014 growing season zooplankton production and biomass were estimated to be about twice as high in Anderson Lake compared to Seton Lake (Limnotek 2015). On the other hand, there appears to be little difference between the diet of the Sockeye in each lake, with the same four common prey types found in the stomachs of fish from each lake, although not necessarily during the same sampling period or year. Also, in 2014 the Seton Lake age-0 *O. nerka* had at least as many prey items in their stomachs as their counterparts in Anderson Lake. Newly available information from this study about the diel vertical distribution of zooplankton in Seton and Anderson Lakes suggests that zooplankton production may have been underestimated in Seton Lake in 2014. Planned changes in 2016 zooplankton sampling methods will seek to clarify this situation. Higher growth rates of age-0 sockeye in Seton Lake may also be related their shorter DVM distances compared to those in Anderson Lake, with accompanying lower energy expenditures. We did not pursue this line of thought because a bioenergetics investigation was beyond the scope of our study.

Apparent summer to fall survival rates of age-0 Sockeye in Seton Lake (68%) in 2014 were double those in Anderson Lake (34%). These survival rates are well within the range observed in other Sockeye rearing lakes. For example, summer to fall survival rates in Shuswap Lake have historically ranged from 20 – 96% (mean = 53%). The higher survival rate in Seton Lake compared to Anderson Lake may be size related as has been shown for smolt or fry to adult life stages (Ricker 1962; Koenings et al 1993; Hume et al 1996). Larger fish presumably have higher survival rates because they can swim faster to avoid predation. Other reasons for increased survival in Seton Lake include: the much lower density of large piscivores in Seton Lake compared to Anderson Lake (1/ha versus 9/ha in 2014): areas of Seton Lake where turbidity is higher than 1-2 NTU may reduce the reaction distance of salmonid predators (Hansen et al 2013); or a combination of these and other factors.

H6: Predator density is lower in the pelagic habitat of Seton Lake than in Anderson Lake

As pointed out in the H5 discussion above, the density of pelagic piscivores (mostly Bull Trout with some Northern Pikeminnow) was estimated to be much lower in Seton Lake than in Anderson Lake (1/ha versus 9/ha in 2014). Although there was appreciable uncertainty in estimates of large predator densities, sensitivity analysis indicated that even if absolute estimates of predator density were inexact, the conclusion about relative abundance of large predators in two lakes was reasonably reliable (i.e., there were more predators in Anderson Lake). In future studies of large predators in these lakes, increased gill netting effort would be especially beneficial.

H7: Juvenile Sockeye losses to predation are lower in Seton Lake than in Anderson Lake

We did not attempt to estimate total consumption of Sockeye fry by predators. However, as pointed out under the H5 and H6 discussion above, with larger, faster swimming age-0 Sockeye fry and fewer predators in Seton Lake, lower predation rates would be expected there than in Anderson Lake.

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?

According to the age model to date, all three cores from Seton Lake appear to exhibit a different sedimentation rate, S3, the core located the closest to the diversion, exhibits the higher sedimentation rate and therefore, covering a shorter time frame than S1 and S2.

Both pigment and diatoms concentrations decreased in the Seton Lake cores concurrent with constant concentrations for both proxies in Anderson Lake. It is likely that changes in both pigment and diatom concentrations are consistent in inferring a decrease in primary productions in Seton Lake following the establishment of the Bridge River Diversion. The development of a strong age model should lead to a better understanding of the sedimentation rate of each core that would allow for a better understanding of the diatoms and pigments data by calculating fluxes. A similar decrease in concentrations in the sub-fossil cladoceran remains occurred in the Seton Lake cores analyzed to date. The reference system, Anderson Lake, by comparison, was relatively stable throughout the period of the diversion. Seton Lake cores exhibit higher pigment and diatom concentrations compared to Anderson Lake prior to the abrupt decrease. Additionally, Seton Lake seems to have experienced a higher meso-eutrophic state compared to the recent time periods which is supported by a shift from meso-eutrophic diatom taxa to more oligotrophic diatom taxa. However, the recent

increase in oligotrophic taxa in Seton Lake is mainly as the result of increases in *Cyclotella comensis* and *Discostella stelligera* and thus may be due more to a regional signal, as similar signal in Anderson Lake. If the timing of changes in the diatom assemblages as well as diatom and pigment concentrations correspond to the beginning of the establishment of the diversion, it appears that Seton Lake cores may provide some evidence that Seton Lake was more productive than Anderson Lake prior to the diversion. To verify this hypothesis the development of a stronger age model is necessary. The completion of the cladoceran analysis could also give additional information on the changes in the trophic state of those lakes.

Several analytical tasks remain to be completed in 2016 for the Question 1 paleolimnology study. Tasks to be completed in 2016 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 May 2016.
- Completion of diatom and cladoceran analyses by the end of April 2016.
- Grain size analyses are currently in progress and are scheduled for completion by the end of May 2016.

After the collection of the cladoceran data for core S1 and the completion of the grain size analysis, we will start data analysis using the BACI design in PRIMER. In order to apply the BACI design, a stronger age model needs be developed for all cores, which will be done using the radioisotopic data outlined in this report. The new age model developed will be then tested by comparing similar distinct peaks in the species composition of the diatom assemblages, as well as changes in the grain-size results. Interpretation of all the proxy data of all cores from both Anderson and Seton lakes will begin to be discussed more thoroughly at the workshop in May/June 2016.

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

One more year 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 to October of 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?

Acoustic processing found distinct zooplankton layers in both lakes. In Anderson lake, zooplankton were found between 2 m and 25 m during day and night in summer and fall. In Seton Lake however, zooplankton were found between 20 m and 45 m during the day (mostly below 30 m) and 2 m and 15 m at night in both summer and fall. Acoustic observations suggest that the densities of zooplankton (the food supply for *O. nerka*) in Seton Lake may have been substantially higher than suggested by results of plankton hauls which sampled only the top 30 m in each lake (section 2). Therefore, in 2016, duplicate zooplankton hauls will occur at three haul depths (0-30m, 0-50m and 0-70m) at each station and time in order to compare density and biomass of zooplankton above and below 30 m and 50 m and revisit the question about differences in food availability for *O. nerka* between Seton and Anderson lakes.

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

One 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.