A Nutrient Mass Balance of Fernan Lake, Idaho, and Directions for Future Research

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Authorization to Submit Thesis

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Abstract

Anthropogenic activities alter the delivery of nutrients, particularly nitrogen-N and phosphorus-P, to water bodies resulting in increased aquatic primary productivity. This process is referred to as cultural eutrophication and often results in blooms of toxic cyanobacteria which produce some of the most potent toxins known to humans, making management strategies imperative. Fernan Lake, a small lake in northern Idaho has experienced harmful algal blooms since the 1990's. It is hypothesized that these blooms result from the presence of excess phosphorus, however, few data were previously available to test this hypothesis. Before any management action can be implemented at Fernan Lake, a detailed analysis of the loading relationships of nutrients and sediment to the lake is required. I performed a detailed mass balance of Fernan Lake using high resolution data to calculate a water and nutrient budget for the lake to quantify the amount of P and sediment retained during one calendar year. Fernan Lake retained 81% of the P and 68% of the sediment that entered it. The majority of the loads came in during the short (January-May) runoff period. I used two methods to quantify internal loading in Fernan Lake and tested if sampling only one site was adequate to describe whole-lake concentrations. Sources of the internal loading that Fernan experiences can be attributed to either wind-induced sediment resuspension or phosphorus excretion by the large fish community. Further research is needed to identify the source of internal loading. This study gives insight into the loading relationships in Fernan Lake, and provides directions for future research.

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Chapter 1: General Introduction

The cultural eutrophication of natural waters, which is the abundance of excess nutrients in waters that lead to excessive algal growth thus decreasing water quality as a result of human activities (Downing 2013), has been recognized for decades (Schindler 1977, Sharpley *et al.* 2000) and has spurred on stringent water quality criteria (e. g., Clean Water Act, in the USA). However, large numbers of people continue to face threats to water-security because of it (Sharpley *et al.* 2000). The continuously expanding human population with concomitant increased modification of the terrestrial landscape contribute nutrients to freshwater ecosystems that result in pollution via eutrophication (USEPA 1996, Environment Agency 1998, European Environment Agency 1998, Sharpley *et al.* 2000).

A consequence of eutrophication is an altered ratio of key elements, especially nitrogen (N) and phosphorus (P) which often alters phytoplankton communities to become dominated by species such as cyanobacteria (Schindler 1977, Schindler *et al.* 2008). The Redfield ratio of carbon, nitrogen, and phosphorus (C: N: P) in a balanced lake system approximates 40:7:1 by mass (Redfield 1958, Schindler *et al.* 2008). Thus, technically, if the N:P ratio is >7, the system is considered phosphorus-limited, a desirable state of freshwaters because P is typically the limiting nutrient in freshwaters (Schindler 1977), while a ratio of <7 indicates nitrogen-limitation and favors an algal community dominated by species capable of fixing atmospheric nitrogen such as cyanobacteria (Schindler *et al.* 2008). Cyanobacteria are the only algal group in which some species have heterocycsts, specialized cells with thick walls and a low oxygen interior capable of 'fixing' atmospheric N to nitrate, allowing them to overcome N-limitation and make use of the excess P. This gives cyanobacteria a competitive advantage over other algal species in low or N-limited conditions (Birand *et al.* 2003, Paerl *et al*. 2011). Although the Redfield ratio indicates dominance of cyanobacteria below a ratio of 7 by mass, many authors have found dominance of cyanobacteria in lakes in which the ratio has ranged from <30 to <75

(Smith 1983, Graham *et al.* 2010, Orihel *et al.* 2012, Harris *et al.* 2014). This is likely the result of the needs of individual species and their fraction of the community as a whole, whereas the Redfield ratio is a generalized ratio.

Dominance of the algal community by cyanobacteria has several negative consequences for aquatic ecosystems. First, because nitrogen-fixing cyanobacteria need access to atmospheric nitrogen, they tend to form large mats or scums at the water surface, which decreases light penetration into the water column and negatively affects the light available to other algae for photosynthesis (Bartram and Chorus 1999, Huisman *et al.* 2004, Paerl *et al.* 2011). Cyanobacteria thrive in high temperatures, having the highest growth rates at 25°C, which is contrary to other algae species, giving cyanobacteria yet another competitive advantage (Paerl and Huisman 2008). In addition, cyanobacteria are not palatable to- or interfere with the filtering mechanism of zooplankton (Gliwicz 1990, Gliwicz and Lampert 1990), decreasing the abundance of large zooplankton species that efficiently transfer energy to higher trophic levels (Stockner and Brandt 2006). When blooms of cyanobacteria decline, sinking and decomposing cells consume dissolved oxygen from the water column and can contribute to the formation of anoxia (lack of oxygen) in the hypolimnion (Pitois *et al.* 2001). It is well-known that iron-bound phosphorus precipitates in sediments at the lake bottom are re-mobilized under anoxic conditions in what is termed 'internal loading' (Bostrom and Pettersson 1982, Nürnberg *et al.* 1986, Bostrom *et al*. 1988, Nürnberg 1998). Thus blooms of cyanobacteria can establish a positive feedback loop via anoxia and internal P loading leading to the predominance of cyanobacteria and a severely impoverished aquatic ecosystem (Paerl *et al.* 2011).

Cyanobacteria also produce some of the most potent toxins known to humans (Codd *et al.* 1989, Keijola *et al.* 1988, Lahti and Hiisvirta 1989, Lawton and Robertson 1999, Bartram and Chorus 1999). Their occurrence in water supplies is highly problematic and poses a threat to the well-being of humans, livestock and pets. Blooms of toxic cyanobacteria are known as harmful algal blooms (HABs). Toxins include hepatotoxins (target the liver), neurotoxins (target the neuromuscular system), and dermatotoxins which irritate the skin via rashes (Birand *et al.* 2003). The most abundantly found toxin in cyanobacteria blooms is microcystin, responsible for many of the poisonings of humans and animals (Codd *et al.* 1989, Yu 1994, Tsuji *et al.* 1995, Dunn 1996, Graham *et al.* 2010). Conventional water treatment methods are ineffective at removing these toxins, making toxic blooms in potable sources a significant problem (Keijola *et al.* 1988, Lahti and Hiisvirta 1989, Lawton and Robertson 1999, Graham *et al.* 2010). For example, in 2014 the City of Toledo (population 500,000) on the USA shores of Lake Erie had to shut down its water supply distribution system for 5 days because of the occurrence of toxic cyanobacteria around its water intake. The annual costs incurred as a result of HABs in the United States is estimated at more than 2 billion dollars in the agricultural, drinking water and recreation sectors (Dodds *et al.* 2009, Paerl *et al.* 2011).

Because the Redfield ratio clearly indicates that 1 unit of P can significantly stimulate algal production, it should be the key nutrient of focus to control eutrophication (Redfield 1958, Schindler *et al.* 2008). This is supported by evidence that eutrophication occurs more quickly at low TN:TP ratios due to nitrogen-fixing cyanobacteria blooms (Schindler 1977, Smith 1983, Schindler 2006). Thus when examining the eutrophication of a water body, it is advisable to quantify phosphorus loads, especially in waterbodies with blooms of cyanobacteria.

Fernan Lake is a small (154 ha, maximum depth 8.2 m, 10.5 km shoreline length) lake in Kootenai County immediately adjacent to the City of Coeur d'Alene in northern Idaho, USA. It has a watershed area of 5, 579 ha which is dominated by steeply sloped terrain covered with conifers on US Forest Service lands in the headwaters, and hay and grazing pastures in the valley bottom and floodplain areas. A large wetland is present at the east end of the lake where Fernan Creek enters the lake, and the City of Fernan Lake Village (71 homes, 169 people, (IDEQ 2013, US Census Bureau) is located on the west shore near the outlet. Because much of the land surrounding the lake is privately owned, stakeholder participation is an important part of any project on the lake (IDEQ 2013). The lake is the most heavily fished lake per unit area in Idaho and much of the north shore is accessible to the public. The lake is also heavily used for recreational activities including swimming, boating, and waterskiing (IDEQ 2013).

Fernan Lake is polymictic because of its east-west orientation, its relatively shallow uniform depth and basin shape similar to a bath tub. It only weakly stratifies for short (1-2 week) periods in summer (IDEQ 2013). During winter the lake is usually ice-covered for approximately three months between December and March.

Since 2007, the Idaho Department of Environmental Quality (IDEQ) has records of reports of nuisance blooms of algae on Fernan Lake. After 2011, when IDEQ developed a protocol to report blooms of cyanobacteria, advisories to avoid human contact with the lake water have been posted in July and September to November (Larson, IDEQ personal communication). Because of these and other issues, Fernan Lake has been listed on the State of Idaho’s 303D list of impaired water bodies that do not meet designated uses, which for Fernan Lake include support of aquatic life, primary and secondary recreational contact, water supply, support of wildlife habitats, and aesthetics. To recover these beneficial uses, IDEQ has established a TMDL for Fernan Lake (IDEQ 2013) which identifies reductions of P input during August and September to 160 kg, such that in-lake concentrations do not exceed 20 $\mu g \cdot L^{-1}$.

Although the TMDL completed by IDEQ identifies targets and loads, not all inflows were quantified, and discharge was based on the 1999 water year, while in-lake P concentrations were based on limited data collected more recently. Thus there is a significant spatial and temporal disjunction among the data, making it difficult to accurately calculate a water or nutrient mass-balance. In addition, internal loading was not measured and assigned as part of the remainder to equalize the mass balance. To obtain a more accurate assessment and develop stage-discharge curves as well as discharge-sediment and discharge-P relationships, a year-long study was completed. The focus of the second chapter of my thesis involves performing an extensive nutrient mass balance on Fernan Lake. I quantified the change in storage of total phosphorus (TP) and total residue (TR, suspended and dissolved solids) in the lake over one calendar year (April 29, 2014 - April 29, 2015).

Scientists often face a trade-off between the cost of a project related to the collection of data, and collecting sufficient data to understand a lake as a whole to make meaningful management decisions. In general, this trade-off is not well-explored in limnology in terms of how many sites should be sampled to adequately represent the state of a lake on any given sampling occasion. I examine this trade-off in my third chapter. Given the near homogeneous basin shape of Fernan Lake, it's relatively small size, and the fact that it is polymictic, I sampled 30 sites on multiple occasions to address to what extent the water column P concentration, temperature and oxygen concentrations were uniform throughout the entire lake. In addition to the examination of this trade off, I explored two different methods to estimate internal loading. I compared the two methods proposed by Nürnberg (2009, 2012) using *in-situ* measurements from two summers (2014 and 2015) and mass balance data from chapter 2.

After quantifying internal loading for the summers of 2014 and 2015, I explored potential sources of internal loading in Fernan Lake, as its polymictic nature makes internal loading through traditional redox-dependent reactions unlikely (Søndergaard *et al.* 1992). For my fourth chapter, I explored the potential for internal loading from sediment resuspension and the influence of the biological community, namely fish. This was completed via a literature review, as data on sediment properties and the fish community (density and mass) that inhabit Fernan Lake were too inadequate to examine this empirically.

Chapter 5 of my thesis compiles all of the conclusions and remediation strategies suggested in chapters 2-4 along with new research questions that have emerged from this study.

Chapters 2-4 are written as individual manuscripts for publication and will thus have detailed introductions and discussions. Additionally, all chapters have been formatted using style guidelines of the peer-reviewed journal *Lake and Reservoir Management*.

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Chapter 2: A nutrient mass balance of Fernan Lake, Idaho, USA using high resolution temporal data

Abstract

Toxic blooms of cyanobacteria, also known as harmful algal blooms (HABs), generally result from anthropogenic changes in land use that alter the delivery of nutrients (nitrogen –N, and phosphorus -P) to aquatic ecosystems. To remediate HABs requires that we understand the nutrient dynamics to select appropriate strategies. Fernan Lake, a small lake in northern Idaho, USA, experiences long-duration HABs annually which are hypothesized to result from the presence of excess phosphorus. I collected high-resolution temporal data (15 minute water levels and daily water samples) for analysis of total phosphorus (TP) and sediment to quantify an annual water, TP and sediment mass balance for the lake. For the calendar year of April 29, 2014 to April 29, 2015, the lake retained 81% of the total phosphorus, and 68% of the sediment that entered it. The majority of the TP and sediment load (93%) entered the lake during the short (January-May) runoff period. This snowmelt-dominated transport during a short window of time presents significant challenges to reduce inflow TP and sediment.

Introduction

Clean water is one of the most important resources for the survival and perpetuation of humans. Access to clean water has resulted in stringent water quality criteria in the USA under the Clean Water Act (CWA) (1972). The National Pollutant Discharge Elimination System (NPDES) of the CWA readily reduced pollution from point sources, but elimination of pollution from non-point sources has not been as successful. Non-point sources include streams leading to water bodies, storm water runoff, atmospheric deposition etc., and are usually more difficult to quantify and monitor than point sources (Ongley *et al.* 2010). Non-point sources were not monitored until revisions to the CWA were made in 1987 (CWA, section 319). Because of the ease in controlling point source pollution, non-point sources now generally account for a greater fraction of the total pollution load and thus must be the dominant focus to recover aquatic ecosystems (Ongley *et al.* 2010).

The expanding human population and concomitant changes in land use mobilize sediment and associated nutrients (particularly N and P) which enter waterbodies via runoff and result in eutrophication. Eutrophication, the presence of excess nutrients in water which stimulate excessive algal growth and decrease water quality (Schindler 1977), is a continuing threat to water quality worldwide (Downing 2013). While eutrophication is a natural process, the term 'cultural' eutrophication (Pitois *et al.* 2001) refers to the acceleration of this process due to anthropogenic activities (Pitois *et al.* 2001). With regard to eutrophication, the water quality of many freshwater lakes and reservoirs hinges on the nitrogen to phosphorus ratio (Redfield 1958). Phosphorus limitation is the ideal case for a healthy water body, but is upset by the introduction of excess P, resulting in accelerated eutrophication often manifested by the proliferation of cyanobacteria (Schindler *et al.* 2008, Redfield 1958). A balanced TN:TP ratio is 7:1 N:P by mass (Redfield 1958). At a ratio greater than 7, in theory, the system is phosphorus limited which limits the growth of harmful algae such as cyanobacteria. However, recent studies have shown that cyanobacteria blooms can proliferate at ratios lower than 75:1 (Harris *et al.* 2014a).

Cyanobacteria have a competitive advantage during N-limitation (excess P) because some of the species' ability to fix atmospheric nitrogen. Because they use atmospheric nitrogen, they tend to form scums at the lake surface, where they outcompete other algae by shading them. This allows them to propagate into large surface blooms, which can induce a positive feedback loop. When the blooms senesce, they deplete water column dissolved oxygen which allows sediment-bound P to re-enter the water column via internal loading (Nürnberg *et al.* 1986) further exacerbating the presence of excess P (Paerl *et al.* 2011). This has led researchers and managers to conclude that management of P in freshwaters is paramount to protecting or recovering them from eutrophication (Schindler *et al.* 2008).

Blooms of cyanobacteria produce some of the most potent toxins known to humans and other vertebrates. The cyanotoxins include hepatotoxins (target the liver), neurotoxins (target the neuromuscular system), and dermatotoxins (skin irritant) (Birand *et al.* 2003). Microcystin is one of the most common toxins produced by cyanobacteria and has been responsible for the poisoning of humans and many animals (Codd *et al.* 1989, Yu 1994, Tsuji *et al.* 1995, Dunn 1996, Graham *et al.* 2010). The unaesthetic appearance of cyanobacteria (thick surface scums) can also have negative economic impacts by decreasing property value and tourism (Dodds *et al.* 2009). Furthermore, water treatment facilities are unable to easily remove the toxins (Keijola *et al.* 1988, Codd *et al.* 1989, Yu 1994, Tsuji *et al.* 1995, Dunn 1996, Birand *et al.* 2003, Graham *et al.* 2010), at times resulting in the shutdown of plants. This was exemplified by the shutdown of the City of Toledo's water system in 2014 due to toxic cyanobacteria surrounding its intake in Lake Erie. Although unfortunate, this shutdown has helped focus attention on the threat of HABs in the Great Lakes basin and the need for action to reduce nutrient inputs.

Best management practices (BMPs) are often implemented to prevent or reduce the entry of P from land to water bodies. These BMPs include the installation of riparian buffer strips, landscape management, and reduction in the use of fertilizers, etc. (Allan 2004), but depend on current land use. Thus, before implementing a specific BMP, a detailed analysis of the loading relationships of nutrients and sediment to a water body is required. This can be accomplished via a mass balance approach (Vollenweider and Kerekes 1980, Wetzel 2001), which quantifies the change in storage of nutrients based on the quantity entering and leaving the system (equation 2.1).

Inflows-Outflow= Δ Storage (equation 2.1)

The main objective of this research was to empirically quantify the annual mass balance of total phosphorus (TP) and sediment (total residue - TR) in Fernan Lake, ID. I also examined the scale of the nutrient load and make suggestions for potential solutions to reduce the TP and sediment loads to Fernan Lake.

Materials and Methods

Study site

Fernan Lake is a small (150 ha) lake located just east of the city Coeur d' Alene in northern Idaho, USA (Figure 2.1). The shoreline length is 10.5 kilometers, while the maximum depth is 8.2 meters (IDEQ 2013). The watershed area is 49.7 km^2 (Table 2.1, IDEQ 2013) comprised primarily of steeply sloped hills that have shallow and easily erodible soils. Historic land use in the watershed was timber production with some agricultural activities in the valley bottom (Fernan Lake Watershed Plan 2003). The outflow of Fernan Lake is intermittent due to a seasonal dam located near the confluence of the outflow of Fernan Lake and French Gulch Creek (Figure 2.2) and enters Lake Coeur d' Alene. The seasonal dam maintains a high lake level for recreation during the summer, and is removed in late fall through spring runoff to avoid flooding. As early as 1990, the lake started to exhibit signs of high algal productivity with a high risk for cyanobacteria blooms (Mossier 1993, Fernan Lake Watershed Plan 2003). Since then, the algal blooms have gradually increased in severity, with "No contact advisories" being issued by the Panhandle Health District at least twice a year. Occurrences of species such as *Anabanea*, *Gleotrichia*, and *Microcystis* are responsible for high concentrations of the cyanotoxin microcystin (Kristin Larson-IDEQ, personal communication). The water quality of Fernan Lake must be maintained to meet the needs of the designated beneficial uses which include: cold water aquatic life, salmonid spawning, primary contact recreation and domestic water supply. Currently these designated uses are not being met and has result in the listing of the lake on the State of Idaho's 303d list, and the development of a total maximum daily load (TMDL) (IDEQ 2013).

The lake is a popular recreation site for residents of Coeur d' Alene for recreation including fishing and swimming. Furthermore, approximately 71 homes located in the community of Fernan Lake Village (US Census Bureau 2010, IDEQ 2013) border the lake on the west end. Not only are the blooms a threat to the health of citizens recreating on the lake, but also to the citizens living immediately adjacent to or near the lake. Historically many of the homes used the water from the lake as a domestic water supply, but with the development of the City of Coeur d'Alene, many switched to its supply and now only a small number of residents still use the lake for their domestic water supply (IDEQ 2013), making it imperative that information about the harmful blooms reaches all residents or citizens with potential to come into contact with the toxins (Personal communication with residents at July 2015 FLCRA meeting).

In 2014, the seasonal dam was installed on May 14 and removed on December 1 (Jurvelin personal communication). In 2015 the dam was installed on March 3 due to the early spring runoff. The dam was not watertight, and leaked throughout the period it was in place. As a result of the early installation, discharge overtopped the dam from March 15 to April 29, 2015.

External sources of phosphorus

Fernan Creek is the main non-point source of phosphorus and sediment to Fernan Lake. The watershed consists mostly of US Forest Service land, with one third of the land, primarily in the lower Fernan Creek area, in private ownership (IDEQ 2013). The watershed area is particularly vulnerable to frequent rain-on-snow events due to warm winter storms from the southwest with heavy rainfall. The creek is especially vulnerable to flooding and transport of large amounts of sediment (Fernan Lake Watershed Plan 2003).

In 2009, the road that runs along the north side of Fernan Lake was rebuilt, resulting in disturbance of the creek channel and the riparian vegetation along it (IDEQ 2013). Historic modifications to the creek for agricultural purposes also have resulted in channelization (FHWA

2004). There are approximately 22 culverts around the lake that directly shunt runoff from the road and surrounding areas into the lake. However, in general, these culverts only transport water during storm events that generate overland flow.

Atmospheric deposition as sources of phosphorus

Atmospheric wet (rain and snow) and dry (dust) deposition can transport nutrients adsorbed to particles into lakes (Anderson and Downing 2006). Transport can be associated with natural activities such as wind and fire, or enhanced through anthropogenic activities such as tillage of fields and travel of vehicles along roadways (Anderson and Downing 2006). Because lakes generally experience the highest atmospheric deposition near the shoreline, and are highly influenced by the immediate environment (Anderson and Downing 2006), it is likely that Fernan Lake receives a significant amount of atmospheric deposition due to its proximity to the I-90 corridor (Figure 2.3). To quantify atmospheric deposition at Fernan Lake, I used daily measured P deposition collected by IDEQ (Harvey 2001 unpublished data), and multiplied it by the number of days with (wet) or without (dry) precipitation during the project.

Potential sources of internal phosphorus

Because the Fernan Lake mixes frequently and is isothermal for most of the year (Appendix A), it is unlikely that internal loading is significant from anoxia near the sediments. However, frequent turbulent mixing may result in internal loading from the disturbance of sediment at the sediment water interface. Because sediment contains a large pool of phosphorus in lakes, it is important to understand the sediment-water dynamics occurring during frequent mixing events, especially in shallow, well-mixed lakes which tend to have a high sediment area to water ratio (Søndergaard *et al.* 2003). Biological communities can also be a large source of internal phosphorus to water bodies via excretion and egestion (Vanni 2002, Eilers *et al.* 2011). Due to the abundant fish community in Fernan Lake, it is possible that it could contribute a significant load of P into the water column, this is addressed in chapter 4.

Inputs of P and total residue to Fernan Lake

To quantify the amount of total phosphorus (TP) and total residue (TR) entering and leaving Fernan Lake, I continuously monitored two inflows (North Side- NS, and Mike Webb – MW), and the lake outlet (OF) for one calendar year (April 29, 2014 to April 29 2015) with automated water samplers (Teledyne ISCOs, Lincoln, NE) (Figure 2.1, Appendix B). The daily samples of 500 ml were partitioned into 125 ml for the analysis of total phosphorus (TP) by the City of Coeur d' Alene wastewater treatment plant using a Lachat 8500 series 2 autoanalyzer (Lachat Instruments, Hach Company, Loveland CO) via method 10-115-01-1-F (EPA 365.1 Rev. 2.0 1993). Another 20 ml aliquot was used for the analysis of turbidity using an Orbeco TB-200 turbidity meter. The remainder was analyzed for total residue using Standard Method 2450 (Eaton *et al.* 2005) by evaporation to dryness at 105 °C in a pre-weighed ceramic dish. Samples were cooled in a desiccator and re-weighed. Event-based sampling was not feasible because of the distance (1.5 hr driving time) between Moscow and Fernan Lake. To account for storm events, I checked and ensured that the daily samples were adequately capturing the rising and falling limbs of the hydrograph.

Equipment malfunction, below freezing air temperature, and dry periods decreased the number of samples collected to fewer than 365. The MW sampler did not take samples on 6 Nov, 29 Dec in 2014, 4, 6 and 10 Jan in 2015 (total of 5 days).The NS sampler failed on 12, 20-23 Jun, 19-21 Jul, and 30-31 Dec in 2014, and on 1 Jan, 3-5 Jan, and 5-12 Mar in 2015 (total of 22 days). No water was present at the site from 2 Aug -28 Nov 2014 (total of 118 days). The OF sampler failed to take samples on 15-19 Aug, 11-20, and 30 Nov, 5, 26, 27, and 29-31 Dec in 2014, 1-5, 9-12, 14-15 Jan, 11-18 Feb, 26 Mar-1 Apr, and 23-29 April in 2015 (total of 50 days). Missed samples were estimated using linear interpolation between the adjacent dates.

To estimate the fraction of dissolved phosphorus (DP) entering the lake on a daily basis, I established a relationship between DP as a function of TP by collecting grab samples every two weeks for simultaneous analysis of TP and DP (filtered through a 0.45 μm filter). These samples were analyzed as above.

To calculate the TN:TP ratio during the growing season from June to November, samples for the analysis of total nitrogen (TN), nitrate+nitrite (NO_3) and ammonia (NH_4) were collected every two weeks. Samples were submitted to SVL Analytical Laboratories in Coeur d'Alene, where they were analyzed using method 353.2 (1993) for NO_3, method 350.1 (1974) for NH_4, and ASTMD-5176 (2003) for TN (USEPA 1993, 1974, 2003).

To quantify discharge, cross-sectional wetted area and velocity measurements were taken at all in- and outflow sites every two weeks during regular site visits. To determine cross-sectional area, I measured the wetted width of each in- or outflow, and the depth at predetermined stations permanently marked on metal or concrete box culverts. Velocity was recorded at each of these stations with a Hach 950 portable velocity meter and top-setting wading rod at 60% depth from the stream bottom (USGS 1983)(Appendix C). Water level was recorded continuously at each site at 15 minute intervals using HOBO (Onset Computer Corporation, Bourne, MA) water level loggers (model number U20-001-01) and HOBOware BoxCar software. One additional level logger was mounted out of the water to record barometric pressure to allow for correction of atmospheric pressure. When data from the barometric pressure logger was not available (due to download malfunctions, 1 two-week period), airport data were used (Appendix D). Stage was recorded at each of the sample sites at a designated permanent reference site, e.g., immovable boulder or rebar stake. I used stage and level measurements to create a stage-level relationship to convert 15 minute water level data to stage (Appendix C). Stage data were converted to discharge via a stage-discharge relationship (rating curve).

Discharge and constituent concentration were used to calculate daily and annual constituent load using the nonparametric smearing approach (Duan 1983, Collin 1995, Helsel and Hirsch 2002, Appendix E). To determine if a relationship existed between the amount of TP and TR, daily values of TR were regressed as a function of TP using least-squares regression. To determine if turbidity could be used as a predictor for TP or TR in this system, I also regressed daily values of turbidity as a function of TP and TR. Outliers were removed from the analysis if there was a significant rain event on that day or substantial algae growth was present in the small bay at the dam.

Discharge at the dam was separated into three periods, i) dam out and water flowing normally, ii) dam in and leaking, and iii) dam overflowing. When the dam was out and water was flowing, discharge was calculated with cross-section and velocity measurements as described for the inflow sites. When the dam was in and leaking, discharge was measured bi-weekly using a 2000 ml plastic graduated cylinder, stopwatch and a piece of metal pressed against the downstream side of the dam to capture water into the cylinder. Triplicates of time and volume of water collected were used to estimate discharge from leakage. Linear interpolation was used to calculate daily discharge for biweekly intervals while the dam was in place and leaking. When water flowed over the dam, discharge was measured using traditional cross section and velocity measurements at a concrete culvert approximately 30 meters downstream of the dam. The downstream site was dry when the dam was in place and not overflowing. Linear interpolation was used to calculate daily discharge between bi-weekly visits when water was flowing over the dam.

Water budget of Fernan Lake

I calculated a daily water balance for the lake. Daily discharge values were added together for the in- and outflow to give the amount of water that entered and left the lake via streams only. LiDAR data collected for EPSCoR's Managing Idaho's Landscapes for Ecosystem Services (MILES) program, and bathymetry data (Wilhelm and LaCroix 2015) were used to calculate a daily volume and surface area for the lake (Appendix F) which allowed me to calculate a daily water balance for the lake. Daily precipitation was recorded from a weather station mounted at the MW site on 19 June 2015 which also measured wind speed and direction as well as air temperature. On 2 December 2014 a precipitation adapter collar with antifreeze and ethanol was added to the tipping bucket rain gage to melt and quantify precipitation values from snow. Daily precipitation values were multiplied by the surface area of the lake on that day to calculate the volume of water gained from precipitation. The volume of water lost to evaporation was obtained my multiplying regional monthly average evaporation rates by the daily surface area (Appendix G).

The Rathdrum Prairie Aquifer Working Group estimated that Fernan Lake loses approximately 30,805.5 $m^3 \cdot d^{-1}$ to the Rathdrum Prairie Aquifer (Spokane County 2009). I estimated loss/gain from the aquifer using daily volumes from the GIS-bathymetry derived volume as the "true" volume of the lake, and measured changes in volume from the water balance. The daily measured change was subtracted from the "true" change in volume over the course of the day. Gain and loss measurements were compared to aquifer water levels in a City of Coeur d'Alene groundwater well located close to Fernan Lake (Appendix H) to check that these calculations followed recorded seasonal groundwater patterns.

Nutrient budget of Fernan Lake

To predict TP and TR values based on discharge, I regressed daily constituent concentrations as a function of daily discharge using the smearing method (Duan 1983) (Appendix E). Dry and wet deposition were calculated from values supplied by the Idaho Department of Environmental Quality in December 1999 to December 2000 (Geoff Harvey-IDEQ, unpublished data). The average wet deposition concentration was multiplied by the daily precipitation. I multiplied the dry deposition ($g \cdot m^{-2}$) by the daily surface area of the lake.

To determine the amount of P released in the water column from wetland sediments via internal loading, I used data collected by MILES undergraduate research intern (MURI), Jessica

Balbiani (2014). Briefly, sediment cores were collected from the wetland using a Kajak-Brinkhurst (K-B) corer with 0.05 diameter × 61 cm long polycarbonate core tubes. All cores were incubated for approximately one month. Balbiani (2014) estimated loading rates from the wetland sediments of 21.1 mg $P \cdot m^{-2} \cdot day$. Wetland observations showed very little interaction with the center of the wetlands and the lake itself; thus we deemed a 10m buffer area at the wetland-lake interface and a 2 meter buffer at the 3 Fernan Creek flow paths through the wetland area as the only areas that contributed in P loads into the lake. The wetland-lake interface and the flow path areas were summed, multiplied by the loading rate and divided by the total volume of the wetland-lake interface and the flow paths to obtain a daily load in $mg \cdot day^{-2}$. The wetland was assumed to be anoxic during the period that the dam was in place (206 days). The daily loads were summed for the period in which the wetland was assumed to be anoxic to give the annual wetland load in kg.

To estimate the load of P from the culverts, I measured discharge and TP concentrations on five separate days. Discharge was quantified using a 2 L graduated cylinder and a stopwatch. Samples for the analysis of total phosphorus were taken in triplicate at each culvert with discharge and analyzed using the methods above (Appendix I).

Because I did not directly measure internal loading, estimates of internal phosphorus loading were calculated by taking the sum of the inputs and subtracting the outputs, and by subtracting the residual from the change of the mass (kg) of phosphorus in the lake (Welch and Jacoby 2001).

Results

Annual loads of total phosphorus (TP) and total residue (TR) to Fernan Lake

Hydrographs for the three sites showed that the majority of the discharge occurred during spring when snow melted (Figure 2.4). In 2015, runoff began in January, causing the hydrograph to increase (Figure 2.4). Because of this, I considered January-May to be the wet period of the study year, as rain events usually resulted in distinct spikes in the hydrograph (Figure 2.5). Discharge at the NS site ceased on August 1, 2014 and began again on November 18, 2014. The maximum discharge at this site was 0.53 $m^3 \cdot s^{-1}$ with a mean of 0.08 $m^3 \cdot s^{-1}$. Discharge at the MW site was continuous throughout the study and ranged from a minimum of 0.01 $m^3 \cdot s^{-1}$ to a maximum of 2.5 $m^3 \cdot s^{-1}$(Figure 2.4). During the dry period, the mean discharge was 0.05 $m^3 \cdot s^{-1}$, while during the wet period it was 0.6 $m^3 \cdot s^{-1}$. The dam resulted in highly variable discharge at the OF site. When the dam was not in place, discharge varied from 0.08 $m^3 \cdot s^{-1}$ to 2.5 $m^3 \cdot s^{-1}$, with a mean of 0.75$m^3 \cdot s^{-1}$. When water was flowing over the dam, discharge varied from 0.03 $m^3 \cdot s^{-1}$ to 0.87 $m^3 \cdot s^{-1}$, with a mean of 0.49 $m^3 \cdot s^{-1}$. Mean discharge while the dam leaked was 0.0001 $m^3 \cdot s^{-1}$.

Daily total phosphorus concentrations for the three sites varied from a minimum of 0.015 mg $TP \cdot L^{-1}$ to a maximum of 0.777 mg $TP \cdot L^{-1}$ (Table 2.2, Appendix J). At the inflow sites, mean TP concentrations ranged from 0.065 mg $TP \cdot L^{-1}$ at the NS site to 0.092 mg $TP \cdot L^{-1}$ at the MW site (Table 2.2).

Daily total residue concentrations for the three sites ranged from a minimum of 2 mg $TR \cdot L^{-1}$ to a high of 1757 mg $TR \cdot L^{-1}$ (Table 2.2, Appendix K). Mean concentrations of total residue ranged from a low of 103 mg $TR \cdot L^{-1}$ at the OF site to a high of 148 mg $TR \cdot L^{-1}$ at the MW site (Table 2.2).

Bi-weekly particulate and dissolved phosphorus concentrations varied by site. At the NS site, the DP fraction was 57.31% of TP (Table 2.3, Appendix L), which remained relatively consistent, even when accounting for the wet (January-May) and dry period (June-December) (Figure 2.6). Conversely, at the MW site, the DP fraction was 48.12 % of TP for the year (Table 2.3). However, during spring runoff it decreased, and only represented 36% of TP, while the remaining fraction (64%) was in particulate form (Figure 2.6). At the OF site, the DP fraction was only 45% of TP during the dry period (Figure 2.6).

Total phosphorus and residue flux over the course of the year closely followed discharge values for the NS and MW sites, however, this was not the case at the OF site (Figures 2.7and 2.8). The estimate of TP load using the smearing method at the MW site was 1125 kg (areal annual loading rate of 686.3 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$), of which 133.15 kg came from the NS site (Table 2.4). The majority of the TP load (93%) entered the lake during the wet period (January-May). During the month of February the lake received 312 kg of TP which was 27% of the annual load (Table 2.4). At the OF site, 265 kg of TP left the lake.

Estimates of TR loads using the smearing method for the MW site was 2297 tonnes, of which the NS site contributed 254.3 tonnes (Table 2.5). During the month of February, 675 tonnes of residue (29% of the annual load) entered the lake (Table 2.5).

Least squares linear relationships of total phosphorus as a function of total residue at the MW and OF sites were significant ($p < 0.001$, $R^2 = 0.64$ and 0.56, respectively, Table 2.6). At the NS site the relationship was also significant (p-value <0.001, Table 2.6), but the R^2 was only 0.19. Least squares linear relationships of total phosphorus as a function of turbidity at the MW and NS sites were also significant ($p < 0.001$, $R^2 = 0.74$ and 0.76, respectively, Table 2.6). The OF site was also significant (p-value <0.001, Table 2.6) but the R^2 value was only 0.45 (Figure 2.10, Table 2.6). Least squares linear relationships of total residue and turbidity at the MW site was significant (p-value <0.001) with an R^2 value of 0.81. The NS and OF sites (p-values <0.001, R^2 values of 0.60 and 0.58, respectively, Table 2.6). The NS site had a number of consecutive days (Dec 16-28, 2014) on which total phosphorus concentrations were high and total residue and turbidity remained low (Figure 2.10, 2.11).

Annual water budget of Fernan Lake

Discharge at the MW site accounted for 64% ($8.3x10^6 m^3{\cdot}y^{-1}$) of the water that entered Fernan Lake. Gains from groundwater sources or the aquifer contributed another 26% ($3.4x10^6$ $m^3{\cdot}y^{-1}$). Precipitation was 10% ($1.3x10^6$ $m^3{\cdot}y^{-1}$) of the annual water budget. Discharge from the road culverts along the north side of the lake was estimated to be 0.14% ($1.8x10^4$ $m^3 \cdot y^{-1}$) of the annual water budget (Table 2.7).

Outflow via the outlet accounted for 70 %($8.8x10^6 m^3 \cdot y^{-1}$) of the water loss from Fernan Lake. Evaporation accounted for 14% ($1.7x10^6$ $m^3 \cdot y^{-1}$), while loss to the aquifer accounted for 16% ($2.0x10^6$ $m^3 \cdot y^{-1}$, Table 2.7).

Annual phosphorus budget of Fernan Lake

A total of 1403kg (855.93 $mg \cdot m^{-2} \cdot yr^{-1}$ total areal loading rate) of P entered the lake from April 29, 2014 to April 29, 2015. Of this, 1125 kg of P entered via Fernan Creek (MW site), while wet deposition added 145 kg, and dry deposition contributed 99 kg. The wetland on the east side of the lake contributed 33 kg and the road culverts contributed 1 kg. A total of 264 kg of P left the lake via the outflow. As a result, the lake retained 1139 kg, resulting in a retention efficiency of 81% (Table 2.8).

TN:TP ratios

The TN:TP ratios from Fernan Lake recorded from June 2014 to October 2015 remained consistently low with a minimum TN:TP of 8.7:1 and a maximum TN:TP of 20.1:1 (Table 2.9, Appendix M).

Internal phosphorus load

On June 24, 2014 the lake experienced a spike of 143 kg in internal phosphorus loading to the water column (Table 2.10). A second spike occurred on July 22, 2014 when 135 kg entered the water column. A smaller spike of 54 kg occurred on September (Table 2.10).

Discussion

The areal loading rate of Fernan Lake, 855.93 $mg \cdot m^{-2} \cdot yr^{-1}$, is consistent with other lakes in the Pacific Northwest. For example, in a study of western Washington lakes by Welch and Jacoby (2001), the polymictic Long-Kitsap Lake had an external load of 885 $mg \cdot m^{-2} \cdot yr^{-1}$, similarly, they found that Lake Ballinger and Lake Stevens had annual loading rates of 840 and

860 mg·m^{-2}·yr^{-1}, respectively. Thus the load I measured for Fernan is not uncommon in the Pacific Northwest.

Previous phosphorus mass balance studies on small lakes have shown that the inflow typically is the largest contribution to the total annual load (Havens and James 2005, Wagner 2010). The MW site which contributed 64% of the water budget for the 2014-2015 sampling period is consistent with the two studies mentioned above. Additionally, the large amount of P (1125 kg) that entered with this inflow is the largest contributor of P to the lake for the year. It is not surprising that Fernan Creek was the primary focus of the TMDL completed by the IDEQ in 2013. It is interesting that the majority of the water and P entered the lake during the short January to May wet season. However, the algal blooms are mainly a problem during the summer and thus the TMDL focused on summer and fall as the "critical" loading period (Aug 15 to Sep 15) during which input loads should be reduced. A load of 160 kg for the critical period was suggested in the TMDL from the inflow with a lake standing mass of P at 250 kg. Based on the detailed daily estimates, I found that only 5 kg of phosphorus entered the lake during the critical period (Table 2.4), suggesting that the TMDL for inflow is being met. However, the water column mass of TP during the critical period varied from a low of 134 kg to a high of 367 kg (Table 2.9), suggesting a large internal source of P which may be responsible for the severe booms of cyanobacteria. Interestingly, the estimates of internal phosphorus loading show large increases of internal P release around June 24, July 22 and September 2, 2014 (Table 2.9), which coincide roughly with the dates of July 8 and September 9, 2014, when health advisories were placed on the lake by the Panhandle Health District. It is apparent that Fernan Lake likely experiences some internal phosphorus cycling during the summer months which is likely the source of P that contributes to the cyanobacteria blooms.

The increase in the water phosphorus concentration during summer is consistent with other studies on shallow eutrophic lakes (Søndergaard *et al*. 2003) and suggests that P sequestered in the sediments from runoff, or other in-lake processes is released into the water column via internal loading (Søndergaard *et al.* 2001,Welch and Jacoby 2001). Shallow lakes that are weakly stratified and frequently mix experience increased interactions between water and sediment which can result in P release (Søndergaard *et al.* 2003). This type of P release differs the typical redox-dependent reactions characteristic of lakes with distinct anoxia-related internal loading as discussed by Nürnberg *et al.* (1986). In shallows lakes, any soluble P released from the sediments is generally immediately available for uptake by phytoplankton because it does not need to travel from a deep hypolimnion to the epilimnion (Søndergaard *et al.* 2003, Welch and Jacoby 2001). The lack of stratification during 2014/2015 combined with the increase in P concentrations during the summer of 2014 suggest that Fernan Lake experienced internal phosphorus loading that may be related to the large mass of sediment and P retained in the lake from spring runoff. There is an indication that there may be a relationship between internal loading and wind events (Figure 2.12), but there is not sufficient data to form a predictive relationship. The internal load estimates in Table 2.10 show a large load initially, that decreases over the duration of the summer. These potential relationships give some indication of processes that may be occurring, and should be the focus of future studies. For example, measures of in-lake turbidity should be related to anemometer data to specifically investigate the validity of any potential relationships.

Sediment release of phosphorus is a well-studied phenomenon in limnology (Nürnberg *et al.* 1986, Nürnberg 2009, 2012), but the suspected large fish community in Fernan Lake bodes the question of how fish excretion influences the internal phosphorus availability in the lake. Additionally, the basin shape and polymictic nature make it susceptible to wind-induced sediment resuspension. The potential for phosphorus release from the biological community and sediment resuspension are compelling questions and are addressed in detail in chapter 4.

The NS site relationship between total phosphorus as a function of total residue and turbidity showed several outliers with high phosphorus concentrations and low total residue and turbidity, which may be indicative of processes related to snowmelt. For 12 consecutive days (Dec 16-28, 2014) during which air temperature steadily increased, high TP values were measured with low TR and low turbidity. This could be related to phosphorus loading from the release of DP from the snowpack. A study by Stoddard (1995) showed that snowmelt can be a source of major ion loads in Sierra Nevada lakes. With the extremely light snow pack across the Pacific Northwest in 2014/2015(NRCS 2015), it is likely not a regular occurrence and only occurs at the very beginning of snowmelt. If the typical snowmelt for Fernan Lake is more prolonged such a spike in phosphorus may not occur. However, it is interesting that at the NS site, DP was a higher fraction of TP, which was amplified during the wet season with 60% of the total phosphorus coming in the dissolved form (Figure 2.6). The large proportion of dissolved phosphorus could be due to the NS site's proximity to agricultural fields (Cooke and Prepas 1998). However, given that the load entering from this site (133.5 kg) is only a small portion of the total inflow measured at the MW site (1125 kg), it may not be important to explore in future studies.

The large influx and retention of P and sediment into Fernan Lake concentrated during spring runoff (often times resulting in floods, Figure 2.13) provide a challenge for remediation to lower water column P during the entire year. Analysis of historic air photos has shown that portions of the creek in the valley bottom have experienced significant stream channel modifications (an example is shown in Figure 2.14) that may increase the velocity of water coming into the lake, possibly increasing P and sediment loads. Management decisions for the lake must consider the strong seasonality of the P and sediment influx. While only focusing on the inflow would decrease the accumulation of new P and sediment, the large quantity of P and sediment already in the lake and wetlands will likely result in a long time lag between treatment application and noticeable changes in the system. While remediation focused on only in-lake concentrations may decrease the occurrence of blooms, it will not stop the influx of P and sediment entering the lake each year. For a rapid response, remediation should focus on a two-pronged approach: reduce influx and treat in-lake sources of P.

Survey of remediation options

Sediment delivery in streams is dependent on activities within the watershed. Before any remediation strategy can be implemented to reduce sediment and P loads from a watershed, there has to be an understanding of the avenues of sediment delivery to the creek (Roni and Beechie 2012). This begins at the headwaters of a watershed; in many watersheds there can be a large amount of sediment delivered to streams via runoff from road networks, whether they be logging roads or roads used for recreational activities involving all-terrain vehicles (Black *et al.* 2012). Thus before any strategy is implemented, a road network analysis such as a GRAIP (Geomorphic Road Analysis and Inventory Package) can be used to determine the extent of the delivery of sediment from roads to creeks. Additionally, a basin wide assessment can be used to determine which sub-catchments in the watershed are contributing the largest sediment and P loads (Roni and Beechi 2012). There are several potential strategies that can be implemented lower in watersheds or even in the lakes themselves, but these are a small portion of a whole-watershed approach. Without the understanding of the watershed as a whole, there is a disjunction in solving the broader problem.

Sediment concentrations in streams typically depend on unit stream power (USP), defined as the rate of energy expenditure per unit weight of water (Yang and Stall 1974). Streamflow is the movement of water to disperse excess energy. This energy can be lost via movement of sediment in the channel itself due to friction, or erosion of banks (Brooks *et al.* 2003). Often after channel morphology has been straightened, a stream will naturally tend to return to a sinuous state by eroding sides of the channel during the remeander process (Karr and Schlosser 1978). Re-meandering of streams can result in the retention of sediment and consequently adsorbed nutrients such as phosphorus in the stream channel due to slowing down the water. A highly sinuous stream channel also promotes high habitat diversity for invertebrates and submerged macrophytes (Jeppesen *et al.* 1999). As a result, re-meandering may cause a stream to become a sink for sediments and nutrients, which would lower the transport of constituents to the lake. For example, after re-meandering the River Gelsa in Denmark, retention of phosphorus as high as 67 kg of $P{\cdot}km^{-1}$ in the river channel was achieved (Jeppesen *et al.* 1999). Similarly, restoration of Middle Tepee Creek, Idaho, which involved reduction of forestry roads in the headwaters and complete re-meander and revegetation of 2,194.6 meters of stream, resulted in an estimated 22% reduction of sediment load (Eagle Stromberg 2012). Reduction of forestry roads and in-stream restoration via bank stabilization in Yellowdog Creek, Idaho, resulted in increased stream macroinvertebrate index (SMI) and stream habitat index (SHI) scores followed by proposed de-listed for sediment impairment (Eagle Stromberg 2012).

Across the US, stream restoration by re-meandering the stream channel is increasing in frequency (Formica and Van Eps 2012). For example, in the Beaver Lake and Illinois River watersheds, significant restoration activities have focused to increase bank stability and riparian vegetation with the purpose of reducing channel erosion and reducing the intrusion of nutrients and sediments through the riparian zone. These projects were highly successful with reductions in sediment and TP loads by as much as 3600 tons (3265.87 tonnes) and 3500 lbs (1587.57 kg) (Formica and Van Eps 2012), respectively. Although these restorations have been highly successful, it is important to note that success depends on targeting areas with significant bank erosion or segments where riparian vegetation enhancement would be beneficial. Thus before any restoration project is undertaken, a basin-wide assessment is required to optimize and identify areas for restoration activities (Roni and Beechie 2012).

Dredging

Dredging, the physical removal of sediment from the lake bottom, would remove material and reduce the loading of P internally. Such an operation would benefit from a lake-wide sediment and nutrient inventory to identify areas with the highest P and sediment accumulation to optimize cost and effort (Murphy 1999). While removal of the highest P-laden sediment can be an effective means of preventing phosphorus loading (Anadotter *et al.* 1999, Murphy 1999), exposure of deep, phosphorus rich sediments may negate such an operation (Anadotter *et al.* 1999). Also, because the bottom will be disturbed, sediment may become resuspended in the water column resulting in highly turbid water (Murphy 1999). Dredging is also expensive (Cooke *et al.* 2005). For example, Cooke *et al.* (2005) give costs of \$0.46·m^{-3} \$26.88·m^{-3} (in 2002 dollars) with a range of \$2.88·m^{-3} to \$7.23·m^{-3} for hydraulic dredging. If contaminated sediments are involved, costs can rise in excess of \$52.00·m^{-3}. In addition to costs, dredging also requires disposal of the removed material. Additionally, if external loading is not reduced first, the P and sediment transported to the lake in subsequent years would quickly negate the dredging benefits (Søndergaard *et al.* 2007).

The addition of Alum

The addition of aluminum sulfate or sodium aluminate to the water column inactivates P by binding to inorganic forms and transporting it to the sediment (Welch and Cooke 1999). This method is entirely focused on controlling P released from lake sediment via internal loading (Welch and Cooke 1999). A mesocosm study at Willow Creek Reservoir, OR showed a 65% reduction in water column TP concentration after the addition of alum. However, the TN:TP ratio remained unchanged because nitrogen was also reduced, allowing cyanobacteria to continue to dominate the algal community (Harris *et al.* 2014b). Because the application of alum is focused on reducing P, few studies concurrently measure nitrogen. The concurrent reduction of nitrogen which left the TN:TP ratio unchanged is important to consider when implementing alum as a management strategy because it may not reduce the cyanobacteria abundance and can negatively affect other biota in the lake (Harris *et al.* 2014b). In Oregon, the treatments decreased the relative volume of cyanobacteria in the mesocosms, but only an inedible non-toxic dinoflagellate and an inedible potentially toxic cyanobacteria species were left as the options for zooplankton to consume (Harris *et al.* 2014b). However, similar to dredging, if inflows are not modified, the alum treatment will be short-lived because continued high influx of sediment will bury the alum layer, rendering it ineffective (Welch and Cooke 1999). In lakes where alum has been used, e.g., Steven's Lake, WA (5.7 km^2), the cost was estimated at $100,000 in 2013 (with 3% inflation this would be $106,000 today) (City of Lake Stevens 2013). Cooke *et al.* (2002) give a range of 105\cdot ha^{-1}$ to 2610\cdot ha^{-1}$ for the application of alum. While this is a relatively cost-efficient method compared to other options, the lack of the reduction of the TN:TP ratio would need further scrutiny (Harris *et al.* 2014b).

Addition of nitrogen to rebalance water column TN:TP

The addition of nitrogen to lakes to manipulate the TN: TP ratio and remove the competitive advantage of nitrogen-fixing cyanobacteria is an option to control cyanobacteria blooms, but is controversial because it is considered a pollutant under the Clean Water Act. The dominance of cyanobacteria and their toxins are greatly reduced at TN:TP ratios > 75:1 as demonstrated in large *in-situ* mesocosm experiments in Willow Creek Reservoir, OR (Harris *et al.* 2014a). When the TN:TP ratio was increased from an average of 18:1 to > 75:1, the phytoplankton community shifted to non-toxic Chlorophyta, increasing the overall diversity of the algal community which was readily accessible to zooplankton grazers (Harris *et al.* 2014a). Increasing the TN:TP ratio of lakes to suppress cyanobacteria growth is only a viable option if there is no risk for ammonia toxicity (Harris *et al.* 2014b). Acute ammonia toxicity occurs at concentrations above 2.79 $mg\cdot L^{-1}$ for freshwater fish (Randall and Tsui 2002). Although the addition of N to rebalance the N:P ratio is logical, adding N to the aquatic environment requires a NDPES permit because it is considered a pollutant under the Clean Water Act. Numerous examples exist throughout the world demonstrating the utility of adding nitrogen (Downing and McCauley 1992, Schindler 2006, Schindler *et al.* 2008) including in Dworshak Reservoir, Idaho (IDFG 2012) where it reduces cyanobacteria to just 5% of the algal community (IDFG 2012). Similar to the other in-lake treatment strategies, it is a temporary fix of the symptoms, but requires continued effort and should be used as one facet of a whole-watershed restoration approach. A decision to pursue the addition of nitrogen will require rigorous monitoring of the N species that contribute to the TN:TP ratio to achieve target ratios and avoid toxicity from ammonia.

Addition of superoxide radicals

A new emerging technology involves the addition of negatively charged oxygen ions in the form of a superoxide anion. The superoxide anions are added to the lake using a Kria reactor (Premier Materials 2013) with the objective to increase the dissolved oxygen concentration and enhance/promote oxidation reactions. Though limited results are available from deployment in natural systems, units were used during a severe cyanobacteria bloom in Grand Lake, St. Mary's, OH (Premier Materials 2013). During this deployment, phosphorus concentrations in the vicinity of the injector dropped by 20%, while nitrogen and ammonia concentrations dropped to near non-detectable concentrations (Premier Materials 2013). Laboratory trials have also shown a rapid reduction of the microcystin toxin, suggesting this approach may be a viable strategy. However, the technology is new, and it is unclear how removal of both N and P will affect the lake or the TN:TP ratio. As well, to-date there has been no consideration of the effects of supersaturation of oxygen in natural waters with respect to its effects on fish or other biological entities. These remain to be discovered, and only then can this strategy be evaluated and compared to the others.

Conclusions

Similar to other studies, it appears that the cyanobacteria blooms in Fernan Lake are fueled by an unbalanced N:P ratio. The lake efficiently retains P and sediment (81 and 67%, respectively) from the inflow of Fernan Creek, which has a typical western US snowpack dominated hydrograph. This means that the majority (93%) of the annual flow arrives in a very short period of time in spring. During this study, the unusually low winter snow pack and early spring rain-on-snow events resulted in an early and short runoff period. This type of hydrograph limits the remediation strategies. Lake managers should start from a whole-watershed perspective to pin point the sources of sediment entering the creek and then proceed to decide on the most plausible management practice. Until a watershed assessment can be performed, nitrogen additions to improve water quality can potentially be a viable option for the short term. Future studies should focus on careful monitoring of the TN:TP ratio and the viability of nitrogen additions as a potential management strategy at Fernan Lake. Overall, without controlling the non-point sources to the creek, Fernan Lake will continue to experience a large influx of P and sediment which will contribute to ongoing cyanobacteria blooms.

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Table 2.1: Watershed and lake characteristics for Fernan Lake, ID. Watershed area obtained from IDEQ (2013). Lake characteristics calculated from ARC GIS using bathymetry data collected during this study and LiDAR data collected by MILES.

Characteristic	
Watershed Area	49.7 km^2
Max pool elevation	650.95 m
Flood elevation	651.23 m
Min pool elevation	650.49 m
Volume at flood pool	$8.8x10^6$ m^3
Volume at max pool	$8.3x10^6$ m^3
Volume at min pool	$7.5x10^6$ m^3
Surface area at flood	1.75km^2
Surface area at max pool	1.69 km^2
Surface area at min pool	1.60 km^2

Table 2.2: Mean, median and range of concentrations of total phosphorus (TP) and total residue (TR) for the North Side (NS), Mike Webb (MW), and Outflow (OF) sampling sites at Fernan Lake, ID for the 2014-2015 study period.

Constituent	Site	Mean ($mg \cdot L^{-1}$)	Median ($mg \cdot L^{-1}$)	Range ($mg \cdot L^{-1}$)
TP	NS	0.065	0.059	0.027-0.260
	MW	0.092	0.055	0.019-0.690
	OF	0.066	0.039	0.015-0.777
TR	NS	129	116	12-1120
	MW	148	115	3-1757
	OF	103	88	2-1149

Table 2.3: Mean and range concentrations of dissolved and particulate phosphorus for the North Side (NS), Mike Webb MW, and Outflow (OF) sampling sites at Fernan Lake, ID during the 2014-2015 study period.

Constituent	Site	Mean (%)	Range (%)	N
Dissolved P	NS	57.31	17.57-85.07	19
	MW	48.12	8.11-83.03	24
	OF	47.03	36.73-86.66	24
Particulate P	NS	42.69	14.93-82.43	19
	MW	51.88	16.97-91.89	24
	OF	52.97	13.34-87.21	24

Table 2.4: Annual and monthly total phosphorus (TP) loads for the North Side (NS), Mike Webb (MW), and Outflow (OF) sites in Fernan Lake, ID calculated using the smearing method. Dashes (--) indicate days where the site had no flow and could not be sampled.

	Monthly TP Loads (kg)		
Month	NS	MW	OF
January	27	201	45
February	31	312	91
March	28	280	39
April	15	151	48
May	14	101	25
June	8	25	0.07
July	3	12	0.03
August	0.01	3	0.04
September	--	2	0.02
October	--	2	0.003
November	0.14	4	0.007
December	7	32	17
Annual	133.15	1125	264. 7

Table 2.5: Annual and monthly total residue (TR) loads for the Mike Webb, North Side and Outflow sampling sites at Fernan Lake, ID calculated using the smearing method. Dashes (--) indicate days when the site had no flow and could not be sampled.

	Monthly TR Loads (tonnes)		
Month	NS	MW	OF
January	51	412	139
February	58	675	263
March	54	596	80
April	29	294	171
May	27	191	75
June	16	40	0.07
July	6	18	0.04
August	0.02	5	0.05
September	--	3	0.02
October	--	3	0.003
November	0.28	5	0.008
December	13	55	34
Annual	254.30	2297	728.14

Table 2.6: Descriptive relationships between total phosphorus, total residue, and turbidity for the 2014-2015 study period and each individual location. Regression equations are in the form y=mx+b, where m and b are the slope and intercept respectively. Total phosphorus (TP) is measured in $mg \cdot L^{-1}$; total residue (TR) is measured in $mg \cdot L^{-1}$; turbidity is in nephlometric turbidity units (NTU); CI is the confidence interval; N is the number of observations; *P* is the probability; and R^2 is the coefficient of determination.

Site	Y	X	Slope	95% CI	Intercept	95% CI	N	*P*	R^2
NS	TP	TR	$1.5 \cdot 10^{-4}$	$(1.1 \cdot 10^{-4}, 1.9 \cdot 10^{-4})$	0.04	(0.035, 0.049)	209	< 0.001	0.19
	TP	turbidity	$2.6 \cdot 10^{-3}$	$(2.4 \cdot 10^{-3}, 2.8 \cdot 10^{-3})$	0.03	(0.029, 0.035)	222	< 0.001	0.76
	TR	turbidity	3.0	(2.6, 3.3)	88.93	(79.72, 98.14)	221	< 0.001	0.60
MW	TP	TR	$6.8 \cdot 10^{-4}$	$(6.2 \cdot 10^{-4}, 7.3 \cdot 10^{-4})$	-0.01	(-0.019, 0.001)	349	< 0.001	0.64
	TP	turbidity	$3.2 \cdot 10^{-3}$	$(3.0 \cdot 10^{-3}, 3.4 \cdot 10^{-3})$	0.03	(0.021, 0.035)	337	< 0.001	0.74
	TR	turbidity	3.8	(3.6, 4.0)	77.75	(70.96, 84.54)	334	< 0.001	0.81
OF	TP	TR	$5.2 \cdot 10^{-4}$	$(4.7 \cdot 10^{-4}, 5.8 \cdot 10^{-4})$	0.01	(0.001, 0.017)	293	< 0.001	0.56
	TP	turbidity	$2.6 \cdot 10^{-3}$	$(2.3 \cdot 10^{-3}, 2.9 \cdot 10^{-3})$	0.04	(0.033, 0.056)	294	< 0.001	0.45
	TR	turbidity	2.4	(2.2, 2.7)	75.72	(67.44, 84.00)	293	< 0.001	0.58

Table 2.7: Annual water budget for Fernan Lake, ID during the 2014-2015 study period.

Annual water budget			
Inputs	$m^3 \cdot y^{-1}$	Depth ($m \cdot y^{-1}$)	Source
Fernan Creek	$8.3 \cdot 10^6$	5.05	This study
Precipitation	$1.3 \cdot 10^6$	0.77	This study
Culverts	$1.8 \cdot 10^4$	0.01	This study
Gain from groundwater	$3.4 \cdot 10^6$	2.05	This study
Outputs			
Fernan Dam	$8.8 \cdot 10^6$	5.38	This study
Evaporation	$1.7 \cdot 10^6$	1.03	Regional averages (see Appendix G)
Loss to Aquifer	$2.0 \cdot 10^6$	1.23	This study

Table 2.8: Annual phosphorus budget for Fernan Lake, ID for the 2014-2015 study period.

Annual phosphorus budget		
Inputs	$kg \cdot y^{-1}$	Source
Fernan Creek	1125	This study
Wet deposition	145	Harvey IDEQ 2001
Dry deposition	99	Harvey IDEQ 2001
Wetland	33	Balbiani 2014, this study
Road culverts	1	This study
Annual	1403	
Outputs		
Fernan Dam	264	This study
Δ P Storage	1139	This study

Table 2.9: Total nitrogen: total phosphorus (TN:TP) ratios for Fernan Lake, ID for bi-weekly collections between June 2014 and October 2014. Total nitrogen was measured using method ASTM D-5176 (USEPA 2003).

Date	TP ($ug \cdot L^{-1}$)	TN ($ug \cdot L^{-1}$)	TN:TP
10-Jun-14	15.33	223.00	14.55
24-Jun-14	21.23	267.00	12.57
8-Jul-14	24.81	498.00	20.07
22-Jul-14	17.36	252.00	14.52
5-Aug-14	16.21	302.00	18.64
20-Aug-14	21.46	380.00	17.71
2-Sep-14	26.67	360.00	13.50
16-Sep-14	34.73	302.00	8.69
30-Sep-14	29.46	277.00	9.40
14-Oct-14	26.36	491.00	18.62
28-Oct-14	30.10	324.00	10.76

Table 2.10: Estimates of internal sources of phosphorus in Fernan Lake, ID for the 2014 study period and 5 day moving average for wind speed ($m \cdot s^{-1}$). Negative numbers in the Δ in lake column indicate a loss of phosphorus from the previous two weeks. Negative numbers in the Internal Estimate column indicate P removal to the sediment, while positive numbers indicate P entered the water column.

		Total phosphorus (kg)			
Date	5 day moving average wind speed ($m \cdot s^{-1}$)	Sum of daily inputs	Standing lake mass	Δ in lake	Internal estimate
24-Jun-14	0.69	23.26	305.85	166.85	143.59
8-Jul-14	0.56	24.17	216.76	-89.09	-113.25
22-Jul-14	0.94	15.07	367.72	150.95	135.89
5-Aug-14	0.90	11.74	266.70	-101.02	-112.76
20-Aug-14	0.79	13.57	227.21	-39.49	-53.06
2-Sep-14	1.48	8.74	290.38	63.17	54.43
16-Sep-14	0.86	10.34	233.65	-56.73	-67.07
30-Sep-14	0.96	9.65	217.05	-16.60	-26.24
14-Oct-14	0.93	7.74	263.30	46.25	38.50
28-Oct-14	1.02	10.86	222.06	-41.24	-52.10

Figure 2.1: Site of automated streamflow samplers (white triangles) on the inflows and outflow of Fernan Lake, Idaho, USA. Image from USGS 2013 0.5m orthoimagery. The North side site (NS) and Mike Webb site (MW) are the two inflow sites and the Outflow site (OF) is the site at the outlet of the lake.

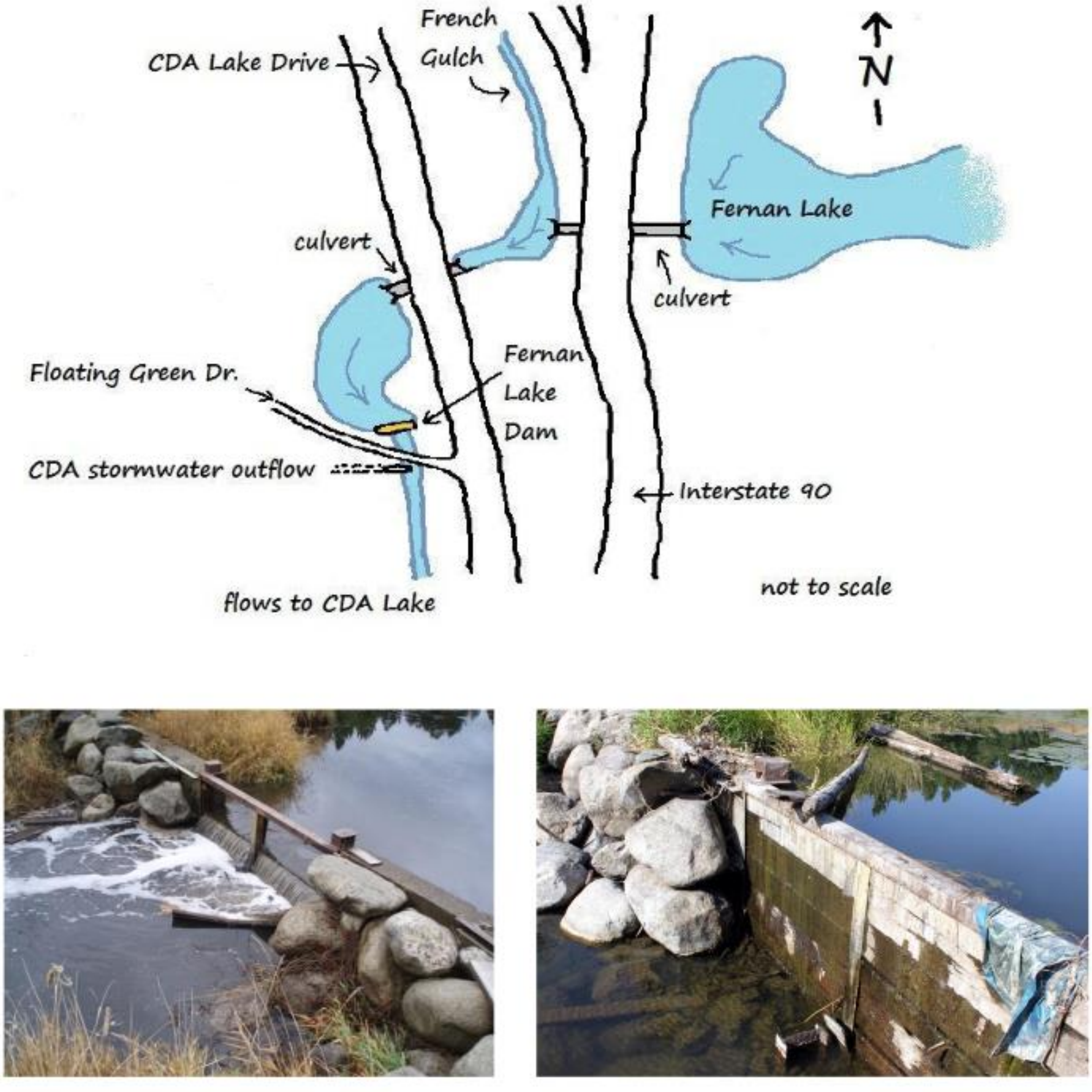

Figure 2.2: Map of the outflow of Fernan Lake, Idaho, USA, showing the confluence of French Gulch Creek and the culvert leaving Fernan Lake. Images below the map show the dam when open and closed. Image modified from IDEQ (2013).

Figure 2.3: Aerial image of the proximity of the I-90 corridor to Fernan Lake, Idaho, USA. Image from USGS (2013) 0.5 meter orthoimagery.

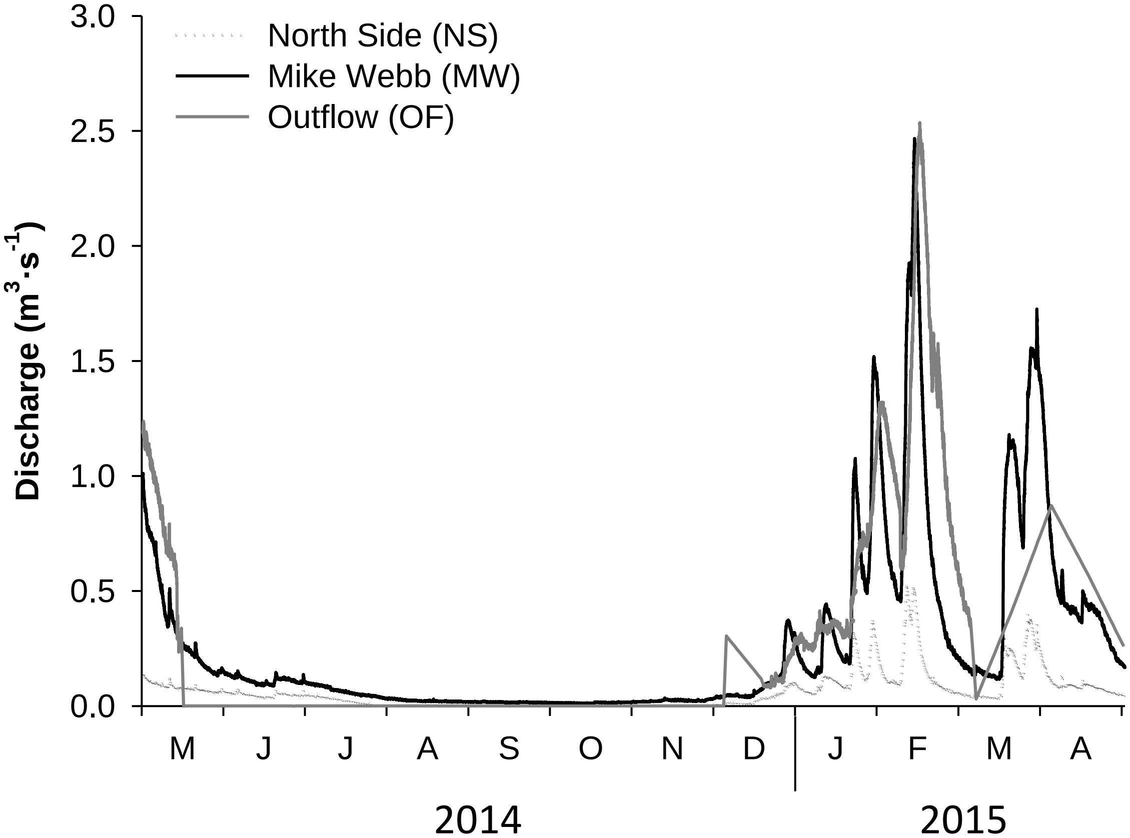

Figure 2.4: Hydrograph of discharge for the North Side (NS), Mike Webb (MW) and Outflow (OF) sampling sites during the 2014-2015 sampling period.

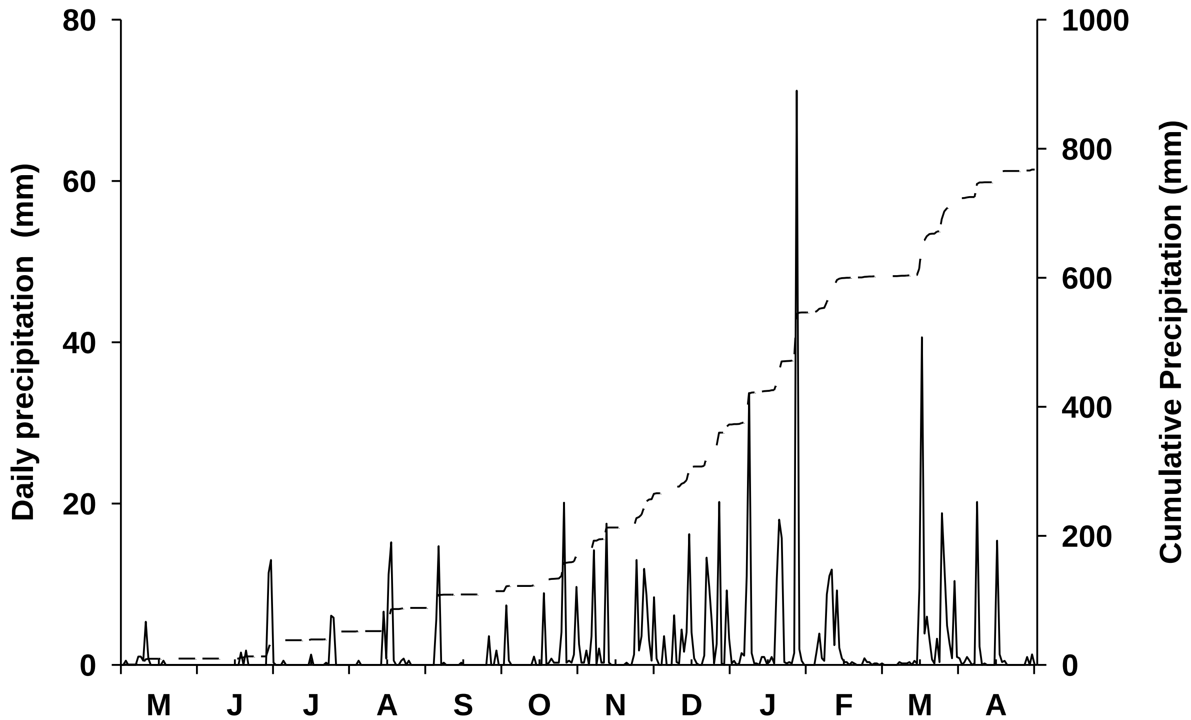

Figure 2.5: Daily precipitation (left axis, solid line) and cumulative precipitation (right axis, dashed line) as a function of time during 2014-2015 collected at the Mike Webb (MW) weather station.

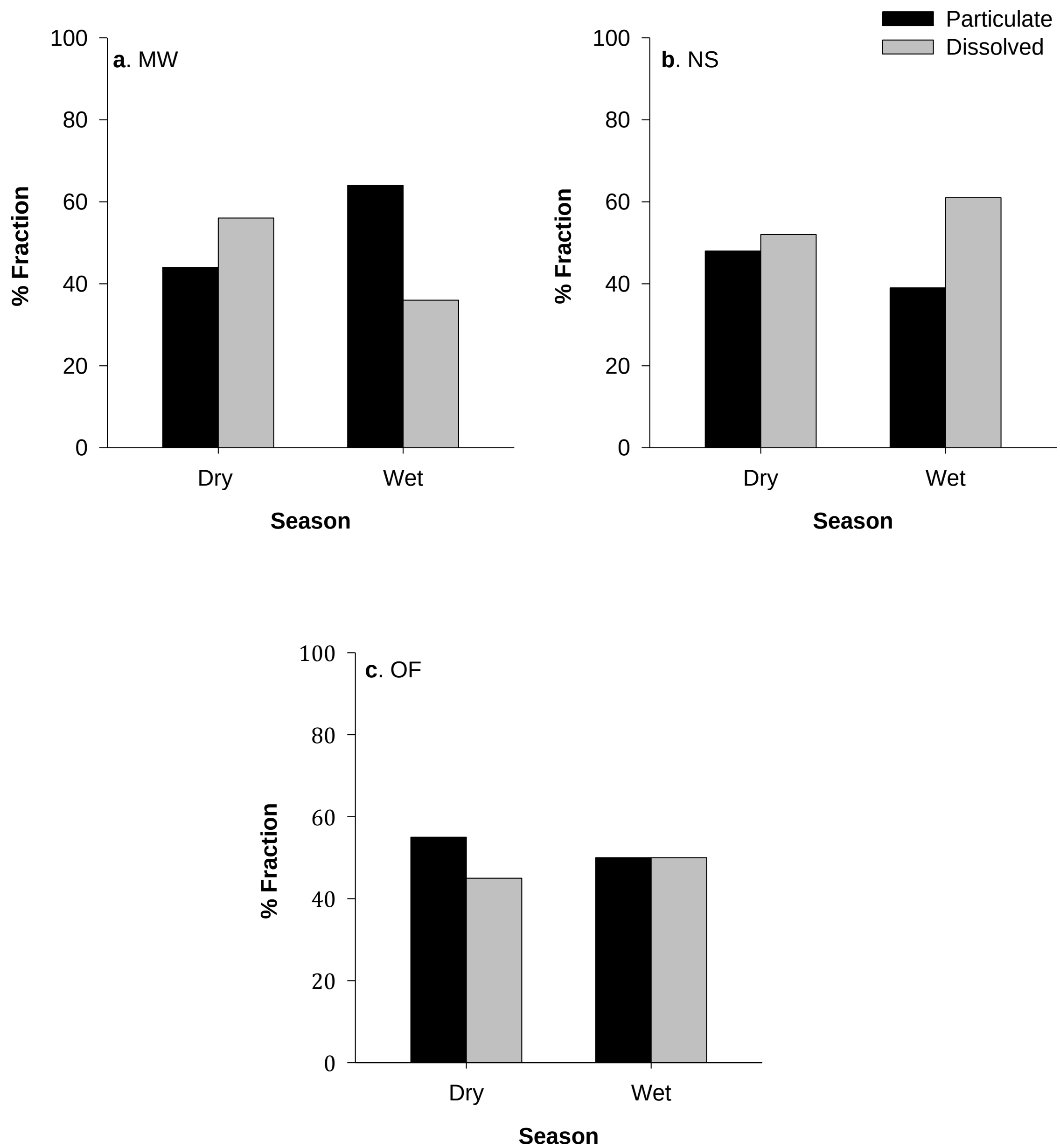

Figure 2.6: % Fractions of dissolved and particulate phosphorus separated by dry and wet season for the (a) North Side (NS), (b) Mike Webb (MW), and (c) Outflow (OF) sites.

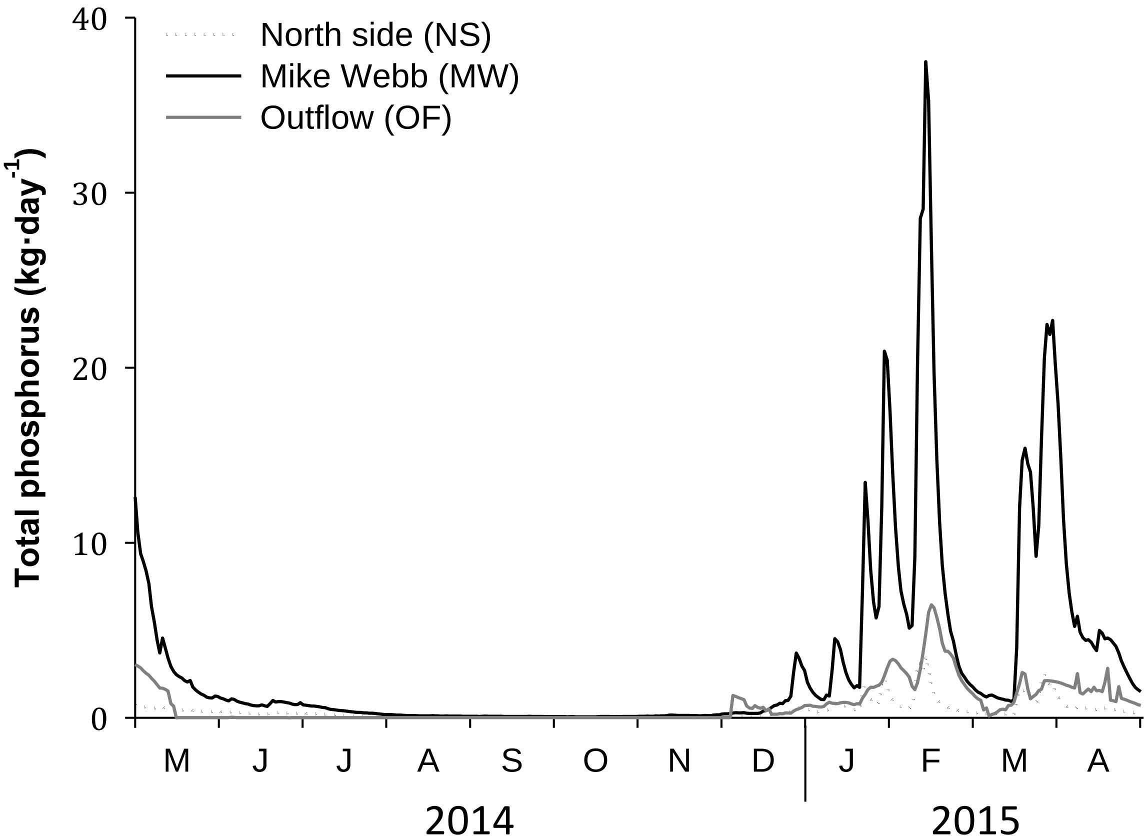

Figure 2.7: Total phosphorus flux (kg·day-1) as a function of time during 2014-2015 for the North Side (NS), Mike Webb (MW) and Outflow (OF) sites.

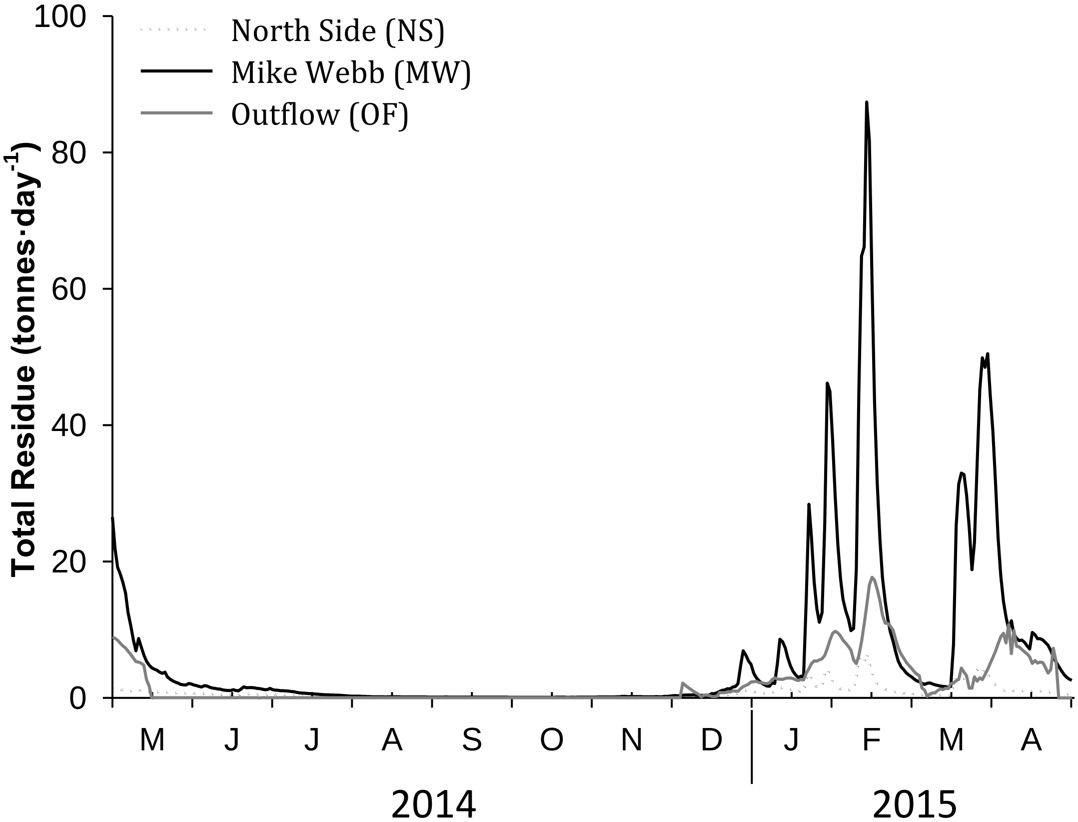

Figure 2.8: Total residue flux (tonnes·day^{-1}) as a function of time during 2014-2015 for the North Side (NS), Mike Webb (MW), and Outflow (OF) sites.

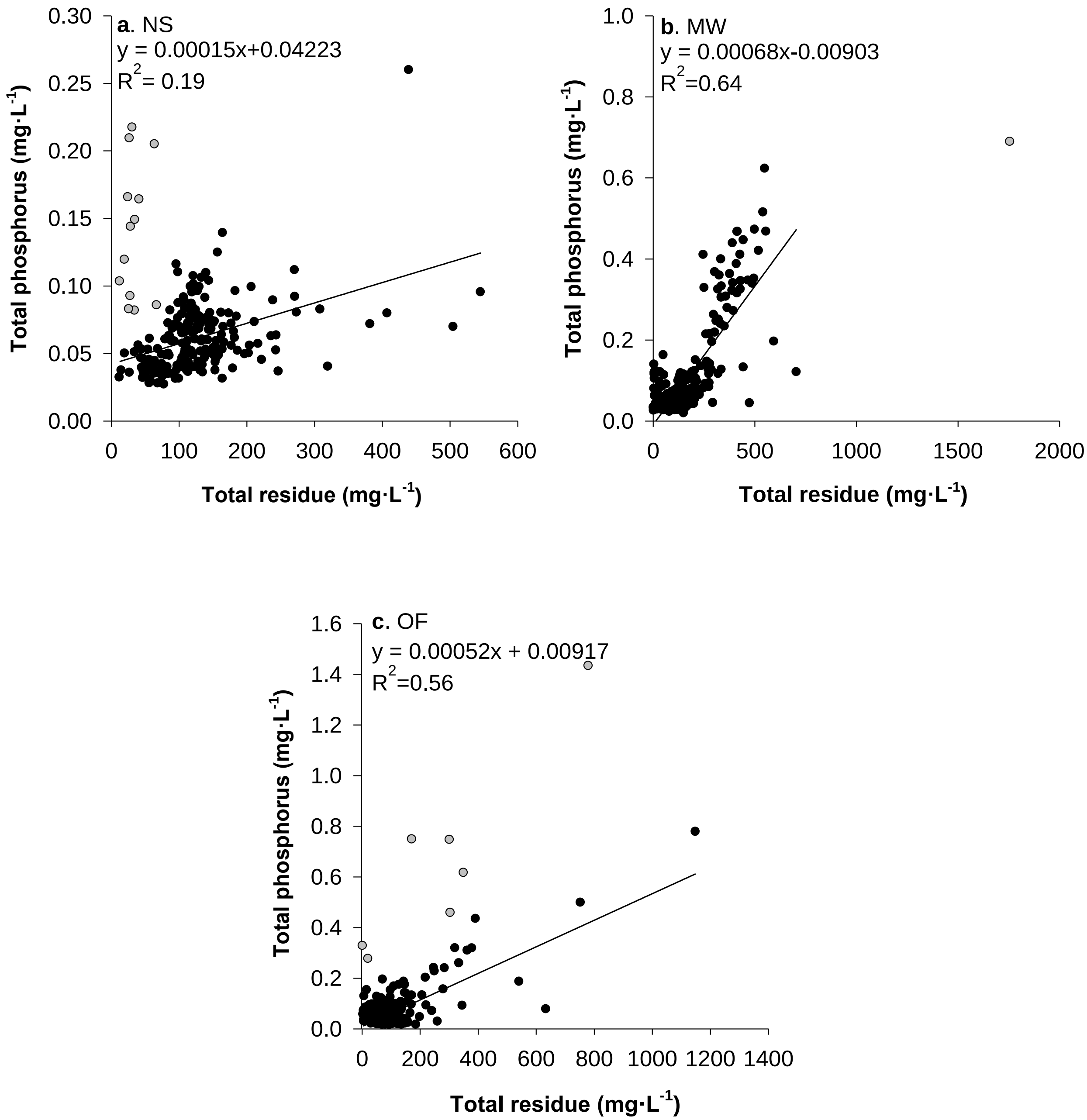

Figure 2.9: Linear regressions of total phosphorus (mg·L^{-1}) as a function of total residue (mg·L^{-1}) for the (a) North Side (NS), (b) Mike Webb (MW), and (c) Outflow (OF) sites. Light gray points indicate outliers which were not included in the regression analyses.

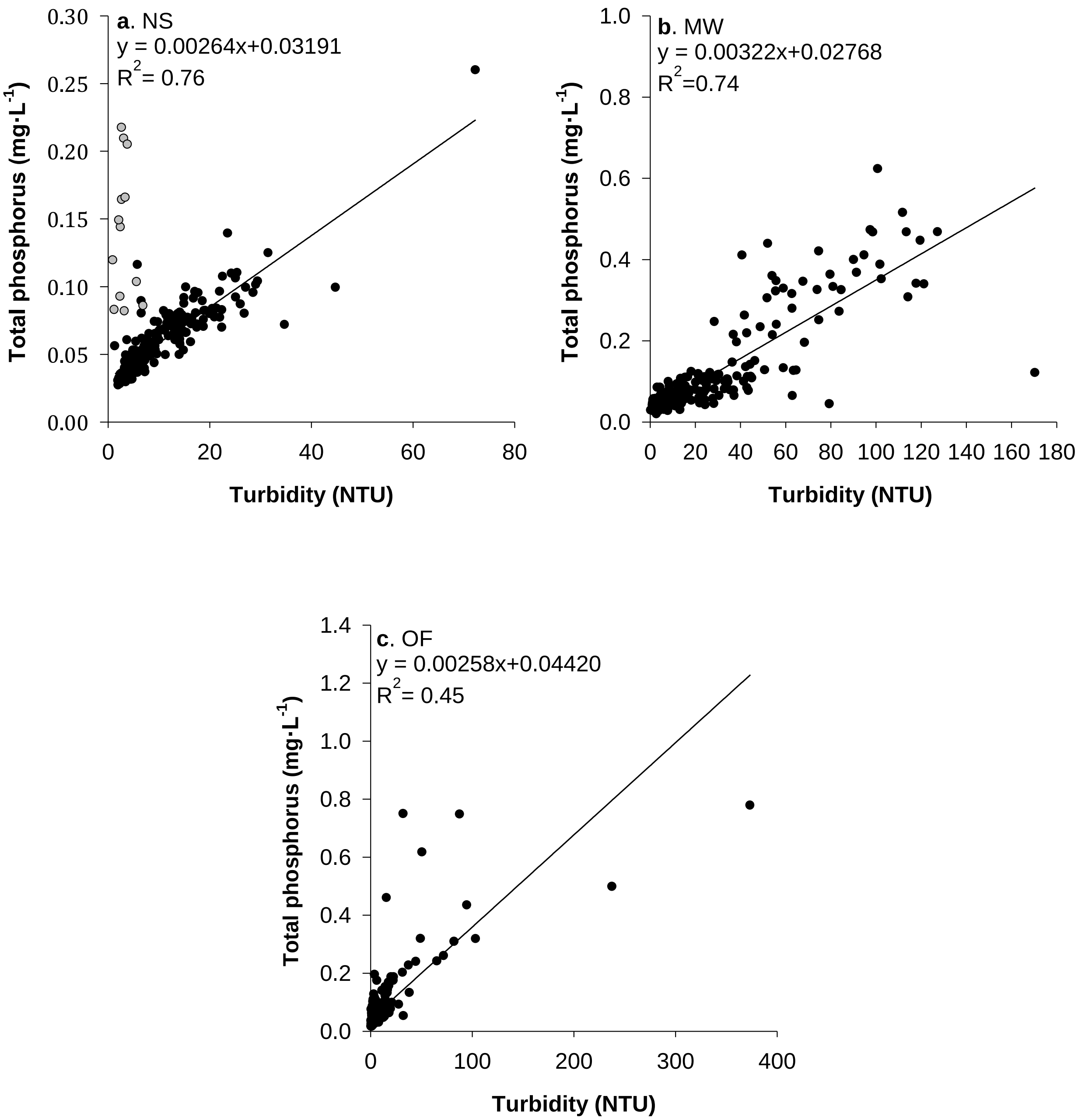

Figure 2.10: Linear regressions of total phosphorus (mg·L^{-1}) as a function of turbidity (NTU) for the (a) North Side (NS), (b) Mike Webb (MW), and (c) Outflow (OF) sites. Light gray points indicate outliers which were not included in the regression analyses.

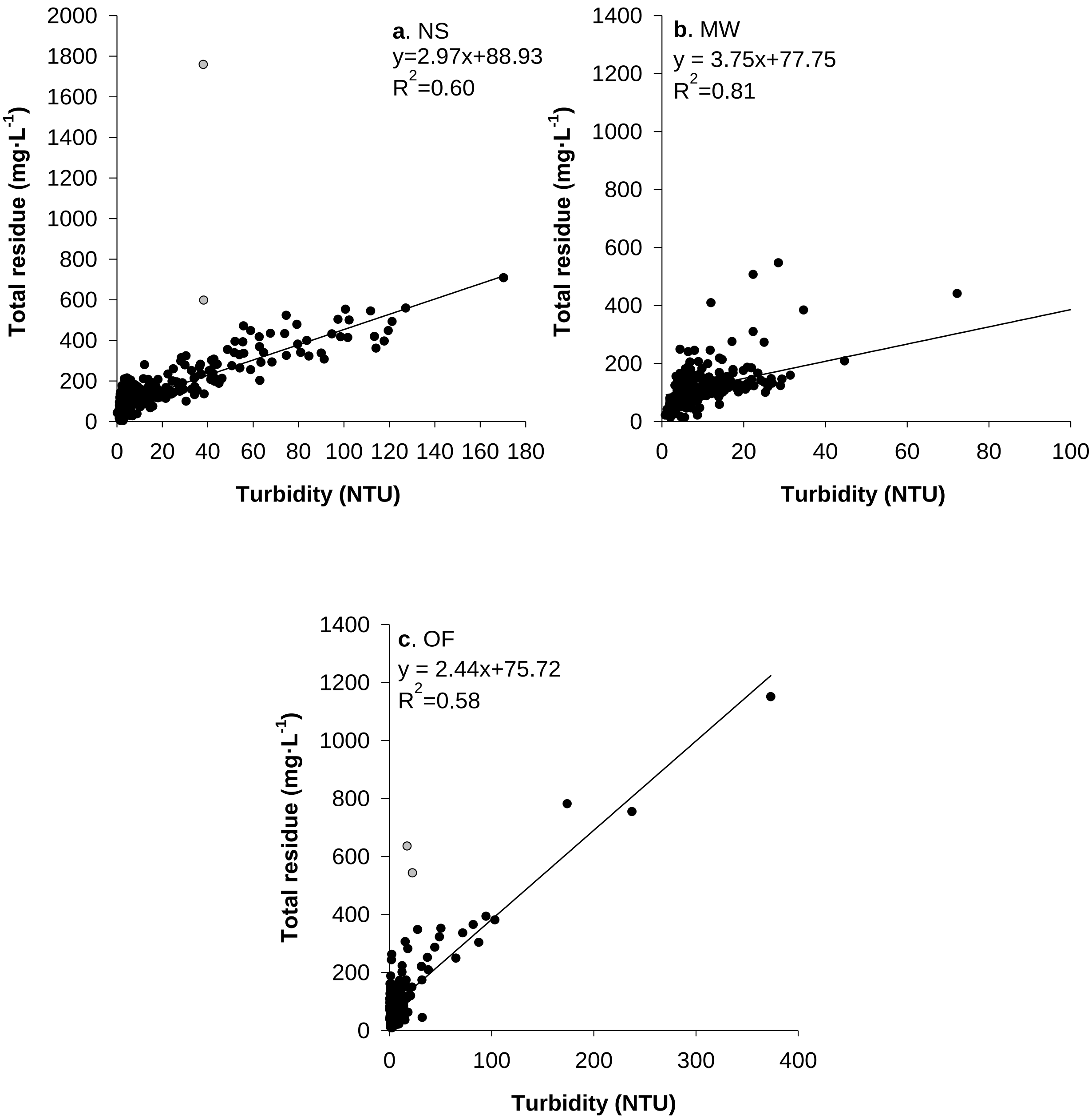

Figure 2.11: Linear regressions of total residue (mg·L^{-1}) as a function of turbidity (NTU) for the (a) North Side (NS), (b) Mike Webb (MW), and (c) Outflow (OF) sites. Light gray points indicate outliers which were not included in the regression analyses.

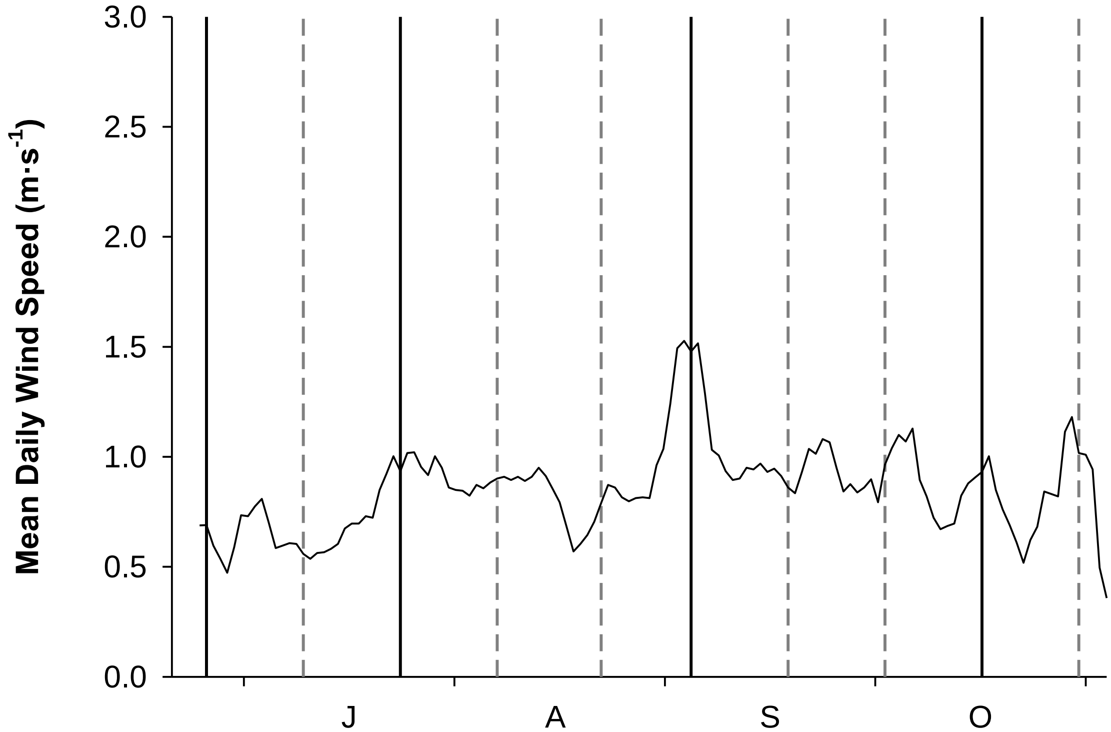

Figure 2.12: Five day moving average of wind speed (m·s^{-1}) from June to October 2014 shown with 2014 lake sampling dates (vertical lines). Solid black vertical lines are indicative of internal loading events that were observed, dashed gray lines were sampling dates where internal loading was not observed.

Figure 2.13: Image of Fernan Creek from March 2014, showing that the creek is perched above the floodplain and channel capacity is minimal. Photo: Frank Wilhelm.

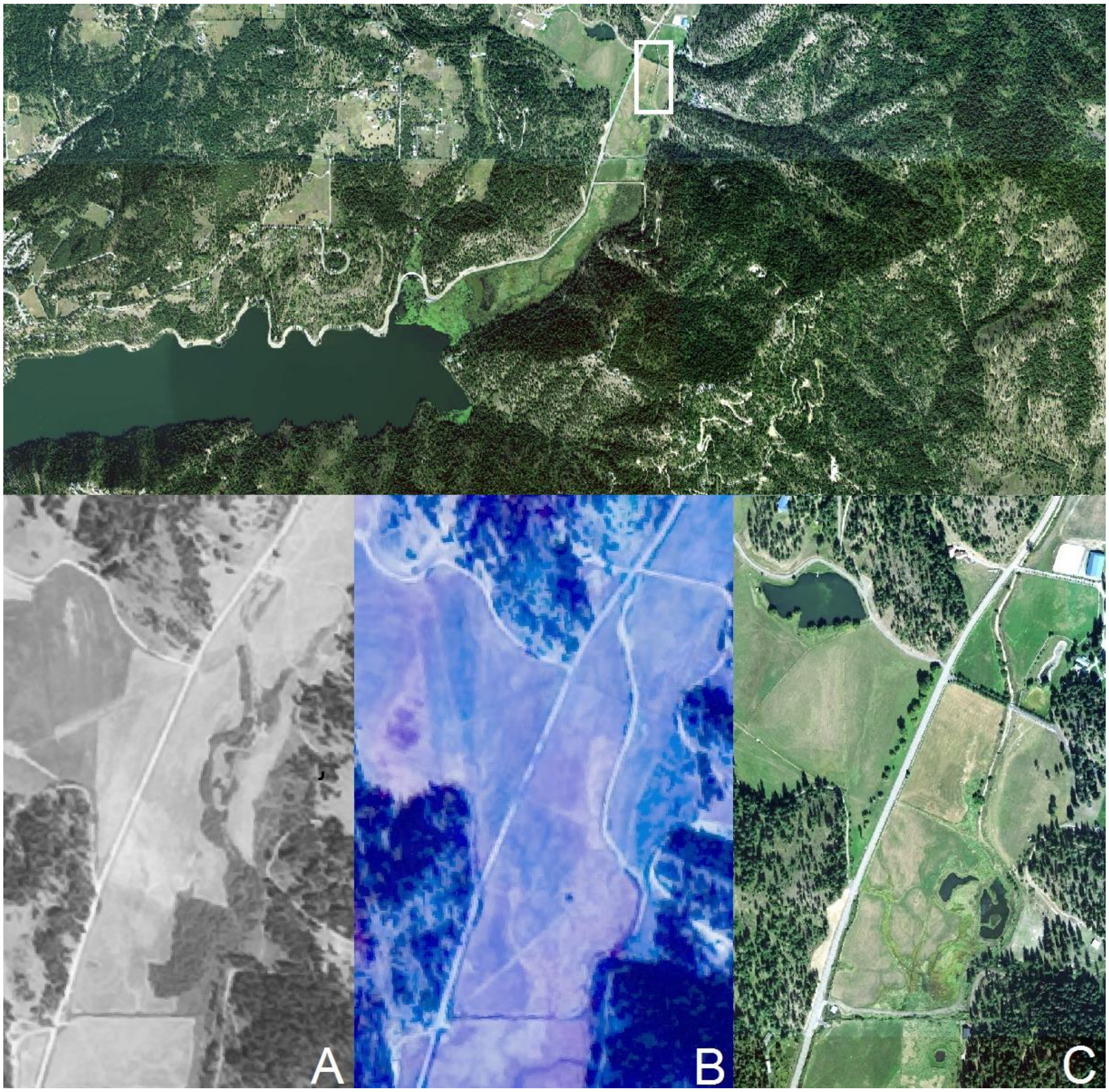

Figure 2.14: Historic Air Photos of Fernan Creek from (A) 1954, (B) 1971, and (C) 2013. Images from USGS earth explorer (A and B) and USGS 2013 0.5m orthoimagery (C).

Chapter 3: Estimates of internal loading in Fernan Lake, Idaho, using *in-situ* lake measurements and high resolution mass balance data

Abstract

Harmful algal blooms (HABs) often result from an excess supply of phosphorus (P) derived from either external or internal sources which favor the proliferation of toxic cyanobacteria. High external loads of P can accumulate in a receiving water body and lead to legacy effects lasting well into the future, even if the external source is reduced. This legacy effect manifests itself via internal loading, particularly when external sources are reduced either permanently via remediation, or seasonally when inflows decrease to a minimum. Internal loading of P can occur as a result of changes in redox potential, wind-induced mixing, and metabolic processes of the biological community. Quantifying the internal load which contributes to cyanobacteria blooms is often difficult, but is vital to make optimal management decisions. Fernan Lake receives a large (856 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$) external load of P that enters the lake primarily in the spring. Interestingly, in summer when HABs are prevalent, external inputs are at a minimum, suggesting that HABs in Fernan Lake may be driven by internal loading. The primary objective of this chapter was to quantify the internal load of P in Fernan Lake using two methods. In addition, I examine the adequacy of using one sampling site to estimate the mean whole-lake P concentration.

Internal loading rates were calculated from *in-situ* measurements obtained in 2014 and 2015 from the deepest location in Fernan Lake. One calendar year of daily samples for P analysis also were collected to calculate a P mass balance for Fernan Lake. Based on the *in-situ* measurements, internal loading rates differed significantly between 2014 (183 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$) and 2015 (50 and 71 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$), while the mass balance estimate for 2015 was 17 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$. The difference in internal loading estimates between the *in-situ* rates suggests that the rates depend on external loads from the previous spring runoff period. Additionally, the use of one site to

quantify internal loading resulted in an overestimate of 21 mg·m^{-2}·yr^{-1}, which indicates that using one sampling site may be inadequate to estimate rates in Fernan Lake.

Introduction

Phosphorus (P) loading to lakes from external sources often fuels harmful algal blooms (HABs) because it results in N-limitation and favors an algal community dominated by cyanobacteria which produce some of the most potent toxins known to humans (Birand *et al.* 2003). However, in many cases even if external loads of P are reduced, cyanobacteria continue to dominate during summer (Søndergaard *et al.* 2003) because of phosphorus loading from internal sources such as the release from sediment under anoxic/ hypoxic conditions in the hypolimnion in stratified lakes (Nürnberg 1984, Søndergaard *et al.* 2003, Nürnberg 2009, 2012), sediment resuspension from wind-induced mixing/sheer stress (Søndergaard *et al.* 2003), and metabolic releases from the biological community (e. g., fish Eilers *et al.* 2011). Phosphorus released into the hypolimnion from changes in redox due to anoxia can reach the epilimnion via metalimnetic entrainment (Wetzel 2001) and become available to algae. In shallow lakes, redox-dependent reactions typically do not dominate internal loading because weak stratification is easily broken down by wind-induced mixing (Søndergaard *et al.* 1992). Instead, in shallow lakes, sediment–released P becomes directly available to algae (Søndergaard *et al.* 2003). This means that internal P loading in well-mixed lakes with high dissolved oxygen (DO) frequently can sustain blooms throughout the summer even when external P sources are low. To select an appropriate management strategy to reduce water column P requires a good understanding of the seasonally important sources of P.

Different methods to estimate internal phosphorus loads are available (Nürnberg 1984, 2009, 2012), each requiring different data, which often determines the method chosen. In one method (referred to as method 2, from hereon, after Nürnberg 2012), internal loading is calculated as a difference from a mass balance approach, where inflow and outflow P concentrations are measured for at least one calendar year to estimate change in storage of P in the lake. The change in stored P is considered the pool that can contribute to internal loading in subsequent summers (Nürnberg 2009, 2012). This method works best with multiple years to account for legacy sediments accumulated from loading in previous years (Søndergaard *et al.* 2003, Nürnberg 2009, 2012). Furthermore, this method yields 'net' estimates, which can be converted to 'gross' estimates to include the portion of the load that has settled back to the sediment (Nürnberg 2012). Another method (referred to as method 1, from hereon, after Nürnberg 2012) that is somewhat simpler, due to the nature of the data required, involves measuring the P concentration in the water column directly when external P loading is minimal (Nürnberg 2012). Changes in P concentration in the lake are monitored over time, and increases in P concentration not accounted for by external sources are considered to be from internal sources (Nürnberg 2009, 2012). These estimates of internal loading are termed 'partial net' estimates because they do not account for the proportion of the internal load that has settled back down to the sediment. Due to data availability and ease in sampling, method 1 is used most commonly (Welch and Jacoby 2001, North *et al.* 2015). Here I use both methods to quantify the internal load of P in Fernan Lake, Idaho, USA and compare the results.

Given both methods rely on a number of measured parameters including whole-lake P concentration, lake volume and area, small errors associated with each parameter can result in widely different loading rates. For example, the instantaneous mass of P in a lake is typically derived by multiplying a measured water P concentration by the lake volume. Thus, small differences in P concentration can result in large errors of the whole-lake P mass given the large volumes typically associated with lakes. Thus it is imperative that the water column P concentration is measured accurately. In addition it is also important to have accurate lake volume information. For Fernan Lake, which is a polymictic lake (lakes that only stratify for short periods and are frequently mixed by wind) it is often assumed that the whole-lake P concentration can be accurately measured from one sample (Fernan Lake Watershed Plan 2003, IDEQ 2013). However, spatial variation within lakes and natural variability in total phosphorus concentrations are known to occur and can vary by up to 20% (Nürnberg 2009, 2012). To reduce this error raises the interesting question of how many samples are needed to accurately estimate the in-lake P concentration. I address this question with in-lake samples from 2014 and 2015 collected monthly for 6 months at 30 sites. My hypothesis is that there is no difference in the whole-lake mean estimated from 30 sites compared to the mean measured at the deep site.

Materials and Methods

Study Site

Fernan Lake is a small (150 ha) lake located just east of the city Coeur d' Alene in northern Idaho, USA (Figure 3.1). The shoreline length is 10.5 kilometers, while the maximum depth is 8.2 meters (IDEQ 2013). The watershed area is 49.7 km^2 (IDEQ 2013) comprised primarily of steeply sloped hills that have shallow and easily erodible soils. Historic land use in the watershed was timber production with some agricultural activities in the valley bottom (Fernan Lake Watershed Plan 2003). The outflow of Fernan Lake is intermittent due to a seasonal dam located near the confluence of the outflow of Fernan Lake and French Gulch Creek (Figure 2.2) and enters Lake Coeur d' Alene. The seasonal dam maintains a high lake level for recreation during the summer, and is removed in late fall through spring runoff to avoid flooding. As early as 1990, the lake started to exhibit signs of high algal productivity with a high risk for cyanobacteria blooms (Mossier 1993, Fernan Lake Watershed Plan 2003). Since then, the algal blooms have gradually increased in severity, with "No contact advisories" being issued by the Panhandle Health District at least twice a year. Occurrences of species such as *Anabanea*, *Gleotrichia*, and *Microcystis* are responsible for high concentrations of the cyanotoxin microcystin (Kristin Larson-IDEQ, personal communication). The water quality of Fernan Lake must be maintained to meet the designated beneficial uses which include: cold water aquatic life, salmonid spawning, primary contact recreation and domestic water supply. Currently these designated uses are not being met, which has resulted in the listing of the lake on the State of Idaho's 303d list, and the development of a total maximum daily load (TMDL) to identify sources and acceptable loads from each (IDEQ 2013).

The lake is a popular destination for residents of the City of Coeur d' Alene for recreation including fishing and swimming. Furthermore, approximately 71 homes located in the community of Fernan Lake Village (US Census Bureau 2010, IDEQ 2013) border the lake on the west end. Not only are the blooms a threat to the health of humans recreating on the lake, but also to the citizens living adjacent to the lake. Historically many of the lakeshore homes used the water from the lake as a domestic water supply, but with the development of the City of Coeur d'Alene, many switched to its supply and now only a small number of residents still use the lake for their domestic water supply (IDEQ 2013), making it imperative that information about the harmful blooms reaches all residents or citizens with potential to come into contact with the toxins (Personal communication with residents at July 2015 FLCRA meeting).

Estimates of internal load using a mass balance (method 2)

I used the mass balance data collected and reported in chapter 2 for this approach. Given data were collected from April 29, 2014 to April 29, 2015, the loading rate reflects the 2015 runoff period and the mass of material available for internal loading during 2015. By the end of April 2014, all snowmelt runoff had occurred and was not captured during my sampling. Net internal loading was calculated using the annual aerial external P load, and the measured and predicted retention. Measured P retention for Fernan Lake was calculated using the equation from Nürnberg (2009, 2012):

$R_{meas} = (L_{ext} - L_{out})/L_{ext}$ *(equation 3.1)*

Where: R_{meas} is the measured lake retention as a proportion of the P retained in the lake on a whole lake basis; L_{ext} is the measured external load that entered the lake (in $mg \cdot m^{-2} \cdot yr^{-1}$); and L_{out} is the measured load that left the lake via the outflow (in $mg \cdot m^{-2} \cdot yr^{-1}$ - summarized from Nürnberg 2012). The predicted retention is representative of the proportion of P lost to settling and sedimentation. It is modeled by the equation presented by Nürnberg (1984, 2009, 2012):

$R_{pred}=1\ 5/(18+q_s)$ *(equation 3.2)*

Where R_{pred} is the predicted retention as a proportion of P retained in the lake on a whole-lake basis; q_s is the annual water load (in $m \cdot yr^{-1}$) derived by dividing the annual average outflow volume by the average lake area; and 15 and 18 are constants (summarized from Nürnberg 2012). Nürnberg (1984) used 54 oxic lakes and a non-linear least squares algorithm "DUD (doesn't use derviatives)" approach by Ralson and Jennrich (1978) to develop these constants, which had no associated error indicated. This equation was developed by Nürnberg (1984) to estimate phosphorus retention in oxic stratified lakes, but has also been applied in polymictic lakes Nürnberg (2005).

Releases of P from the sediments were quantified as the difference between the predicted retention and measured retention of phosphorus multiplied by the annual aerial external load. Predicted retention is expected to overestimate retention by the amount of P released by the sediments. Thus according to Nürnberg (2009, 2012), the internal load of phosphorus can be estimated by:

Net $L_{int\_2}= L_{ext}$ x $(R_{pred} - R_{meas})$ *(equation 3.3)*

Where Net L_{int_2} is the net internal load estimate (in $mg \cdot m^{-2} \cdot yr^{-1}$); L_{ext} is the measured external load (in $mg \cdot m^{-2} \cdot yr^{-1}$ - summarized from Nürnberg 2012); and R_{pred} and R_{meas} are as defined above. This model makes the assumption that the residual error in the model (the difference between R_{pred} and R_{meas}) is representative of internal phosphorus loading (Nürnberg 1984).

Gross internal loading can be calculated using the model given by Nürnberg (2009, 2012):

Gross L_{int_2}=Net L_{int_2}/ (1-R_{pred}) *(equation 3.4)*

Where Gross L_{int2} is the gross internal load estimate (in $mg \cdot m^{-2} \cdot yr^{-1}$- summarized from Nürnberg 2012), and Net L_{int_2} and R_{pred} are as defined above.

Estimates of internal load using changes in whole-lake P mass (method 1)

Sample collection in 2014

Water samples for the analysis of total phosphorus were taken from a depth of 7 m at biweekly intervals between April 29, 2014 and October 11, 2014 at the deepest spot in Fernan Lake (Figure 3.2). Samples were collected using a 1 L Van Dorn water sampler, transferred to 125 ml bottles, and placed on ice for transport to the University of Idaho Limnology Laboratory. Samples were analyzed using method 4500-P (Eaton *et al.* 2005).

Sample collection in 2015

Water samples for the analysis of total phosphorus were taken from a depth of 7 m at monthly intervals from April 16, 2015 to August 11, 2015 at the deepest spot in Fernan Lake (Figure 3.2). Samples were handled and analyzed as above.

Estimates of internal loading

To estimate whole-lake P mass, the phosphorus concentrations from each sample date were multiplied by the volume of the lake on that day. The first dates on which internal loading were noted in 2014 and 2015 were June 26, 2014 and May 20, 2015, respectively. External loading rates were a small percentage of the total external load (4%, 45 kg) between the sampling dates of June 24, 2014 and October 14, 2014 (see mass balance in chapter 2). Additionally, in the summer of 2015, external loading between the sampling dates of May 20, 2015 and August 11, 2015 was also low (35 kg). Because external loading rates were low for both years' sampling periods, they were not included in this analysis as recommended by

Nürnburg (2012) to avoid complicating the calculation with external loads. Changes in whole-lake P were used to calculate internal loading according to Nürnberg (2009, 2012):

L_{int1}= (P_t_2 x V_t_2 - P_t_1 x V_t_1)/ (A_o) *(equation 3.1)*

Where L_{int_1} is the estimate of internal loading; t_1 and t_2 are successive sampling dates in Julian days; P is the lake water phosphorus concentration at time t_1 and t_2; V is the lake volume at time t_1 and t_2; and A_o is the lake surface area, which was averaged between the two sampling dates to account for small differences in surface area. This yielded the aerial loading rate ($mg{\cdot}m^{-2}{\cdot}$ number of $days^{-1}$) for the time period under consideration, which was divided by the number of days between the sampling dates to give a daily loading rate ($mg{\cdot}m^{-2}$ day^{-1}).

The dataset in 2015 was limited due to time constraints and equipment availability; thus the 2015 dataset ends on August 11, 2015; before the end of internal loading. To account for any internal loading not measured in September and October similar to 2014, I took the proportion of the internal load that was observed in September and October in 2014 and applied it to the estimates in 2015.

Estimating error associated with measurements of whole-lake P concentration

To test the hypothesis that bottom samples (7 m) from a single site were reflective of the whole-water column P concentration, I collected additional mid-water samples from 30 sites on five occasions in 2015 (April 16, May 20, June 17, July 16 and August 11, 2015). The 30 samples were chosen haphazardly and marked with an eTrex 20 handheld GPS receiver at the time of sample collection. At each site, the water depth was measured using a handheld depth sounder (Hawkeye model H22PX, Orlando, FL), and one sample for the analysis of total phosphorus was taken from the middle of the water column. Concentrations from all 30 sites were used to calculate a mean lake concentration for each sampling occasion. Analyses of internal loading were completed as described above under method 1

To calculate changes in water temperature for the lake over time, I used temperature and oxygen profiles collected during the 2014-2015 sampling period (Appendix A), and the volume of each depth (layer) in the water column. I multiplied the temperature of each layer by the corresponding volume, summed the values and divided by the total volume; this gave a volume weighted average temperature for each sampling day.

Results

Estimates of internal load using method 2

The net and gross estimates from the one year of mass balance data, equations 3.2-3.5, were 17.27 $mg \cdot m^{-2} \cdot yr^{-1}$ (net estimate) and 101.59 $mg \cdot m^{-2} \cdot yr^{-1}$(gross estimate), respectively. The measured areal annual external loading rate was 855.93 $mg \cdot m^{-2} \cdot yr^{-1}$.

Estimates of internal load using method 1

The mean in-lake TP concentrations for the summer of 2014 was 28 $\mu g \cdot L^{-1}$, with a minimum of 16 $\mu g \cdot L^{-1}$ and a maximum of 46 $\mu g \cdot L^{-1}$ (Table 3.1). Using equation 3.1, I estimated an aerial phosphorus load from internal loading of 258 $mg \cdot m^{-2} \cdot yr^{-1}$. The proportion of internal loading that occurred in September and October 2014 was 26%.

The mean TP concentration of the deep site in 2015 was 19 $\mu g \cdot L^{-1}$, with a minimum of 10 $\mu g \cdot L^{-1}$, and a maximum of 25 $\mu g \cdot L^{-1}$ (Table 3.1). The internal loading estimated using method 1 was 56.15 $mg \cdot m^{-2} \cdot yr^{-1}$; and adjusting this value to account for September and October increased it to 70.7 $mg \cdot m^{-2} \cdot yr^{-1}$.

The mean TP concentration for the 30 sites in 2015 was 15 $\mu g \cdot L^{-1}$, with a minimum of 6 $\mu g \cdot L^{-1}$ and a maximum of 47$\mu g \cdot L^{-1}$ (Table 3.1). Using the average lake concentration of the 30 sites resulted in a loading estimate of 39.6 $mg \cdot m^{-2} \cdot yr^{-1}$ and after adjusting this value to account for potential loading in September and October, it was 49.9 $mg \cdot m^{-2} \cdot yr^{-1}$.

Error estimates for method 1

If only the deep site was used to estimate whole-lake P, the P loading rate in the lake would have been over-estimated by 20.8 $mg \cdot m^{-2} \cdot yr^{-1}$ (the 30 site mean was 29% lower than the

7 m site). This means the loading rate calculated for 2014 (258 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$) was also an overestimate and was reduced to 183.18 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$.

Discussion

The greater than two-fold higher internal loading rate in 2014 compared to 2015 could be related to the low snowpack and subsequent low discharge experienced throughout the Pacific Northwest in the winter of 2014-2015. Additionally, the lake experienced increases in temperature earlier in 2015 than 2014 (Figure 3.4). The hypothesis that Fernan Lake experiences large inter-annual variability of external and internal P loads is supported by the mass balance presented in the TMDL (IDEQ 2013). It was estimated that 4100 kg (2501.3 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$) entered the lake and 570 kg (347.74 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$) entered from internal loading (IDEQ 2013). Maximum discharge values in this study were two-fold higher than those I reported in chapter 2 (Figure 3.3). My estimated partial net loading rates for 2015 from *in-situ* measurements were lower than the 2013 TMDL rate of 347.74 $mg{\cdot}m^{-2}{\cdot}yr^{-1}$, supporting the hypothesis that 2015 was a below normal year in terms of hydrology, and lake processes responded accordingly. The large differences in the calculated loading rates show high inter-annual variability in Fernan Lake, suggesting that managers should be wary of mixing and matching parameters measured in different years.

Given these calculations rely on the sequential use of values, error propagation is of great concern. For example, outcomes calculated with method 1, which depends on lake volume, lake surface area, in-lake P concentrations, and repeated sampling (Nürnberg 2012), means that small errors in lake volume and surface area can greatly alter estimates for Fernan Lake because surface areas and volumes varied greatly throughout the year from the operation of the seasonal dam (see chapter 2). Additionally, if the correct volumes and surface areas are calculated and combined with a small error in the TP analysis, deviations similar to the one shown can be greatly magnified given the typically large lake volumes and surface areas. The differences between the mean TP estimates from the 30 sites and those obtained from just the deep site exemplify the potential for error when trying to quantify in-lake P dynamics. Thus, it is imperative for any scientist attempting to calculate internal loading to ensure that (1) they have correct lake volumes and surface areas for the days on which internal loading occurs; (2) they ensure the analysis has minimal error, or at minimum a way to account for it; and (3) multiple sites need to be sampled to obtain meaningful estimates of the whole-lake mean.

Conclusions

The 2014 and 2015 data from Fernan Lake offer several insights. First, although the lake remained well-mixed throughout the year, seasonal variability in spatial and vertical P concentrations occurred. This means that sampling one site in Fernan Lake (the deep site) can yield very different loading rates when scaled to the entire lake. To reduce the margin of error, multiple samples should be collected from the lake to estimate the average lake concentration.

Estimates of internal loading in lakes for comparative purposes are likely most accurate with multiple years of data to dampen the undue influence of abnormal years like 2015 during which the snowpack was especially low (NRCS 2015). Interestingly, method 1 consistently had higher estimates than the net estimate for the year given by method 2 (Table 3.2). It is possible that lake managers can use these estimates to gain insight to the scale of internal loading. Although using these methods to estimate internal loading in Fernan Lake was insightful, they do not identify the source and thereby cannot be used to establish concrete actions, such as application of alum, or manipulation of the fish community to reduce the internal load of P. I use a literature review to explore the contribution from different sources in chapter 4. Additionally, if internal loading in Fernan is indeed driven by the external load received during the previous spring, management actions to control the external load become more vital to the overall recovery of lake water quality, and supports a whole-watershed approach to restoration.

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Table 3.1: Mean and range of concentrations of total phosphorus (TP) measured in Fernan Lake, ID during 2014 and 2015.

Constituent	Year	Location	Mean ($\mu g \cdot L^{-1}$)	Range ($\mu g \cdot L^{-1}$)
TP	2014	Deep Site	28	16-46
TP	2015	All Locations	15	6-47
		Deep Site	19	10-25

Table 3.2: Comparisons of partial net *in-situ* internal loading estimates (method 1) and gross estimates from a detailed mass balance as shown by Nürnberg (2008, 2012) and this study.

Lake	Year	Internal load in $mg{\cdot}m^{-2}{\cdot}yr^{-1}$		
		Method 1 partial net	Method 2 gross	Reference
Fernan Lake	2015	40	102	This Study
Cherry Creek Reservoir	1992	204	255	Nürnberg 2008
	1993	183	330	“
	1994	220	234	“
	1995	98	64	“
	1996	165	332	“
	1997	407	213	“
	1998	338	-31	“
	1999	78	170	“
	2000	158	164	“
	2001	154	650	“
	2002	141	315	“
	2003	175	275	“
	2004	584	667	“
	2005	660	241	“
	2006	195	943	“
Lake Säkylän Pyhäjärvi	1990	109	29	Nürnberg 2012
	1992	44	66	“
	1999	46	66	“
	2003	51	74	“

Figure 3.1: Location of Fernan Lake in Northern Idaho, USA, close to the City of Coeur'd Alene, aerial photo from USGS 0.5m orthoimagery (2013).

Figure 3.2: Location of the deep sampling site used for *in-situ* measurements during the 2014 sampling period. Aerial photo from USGS 0.5m orthoimagery (2013).

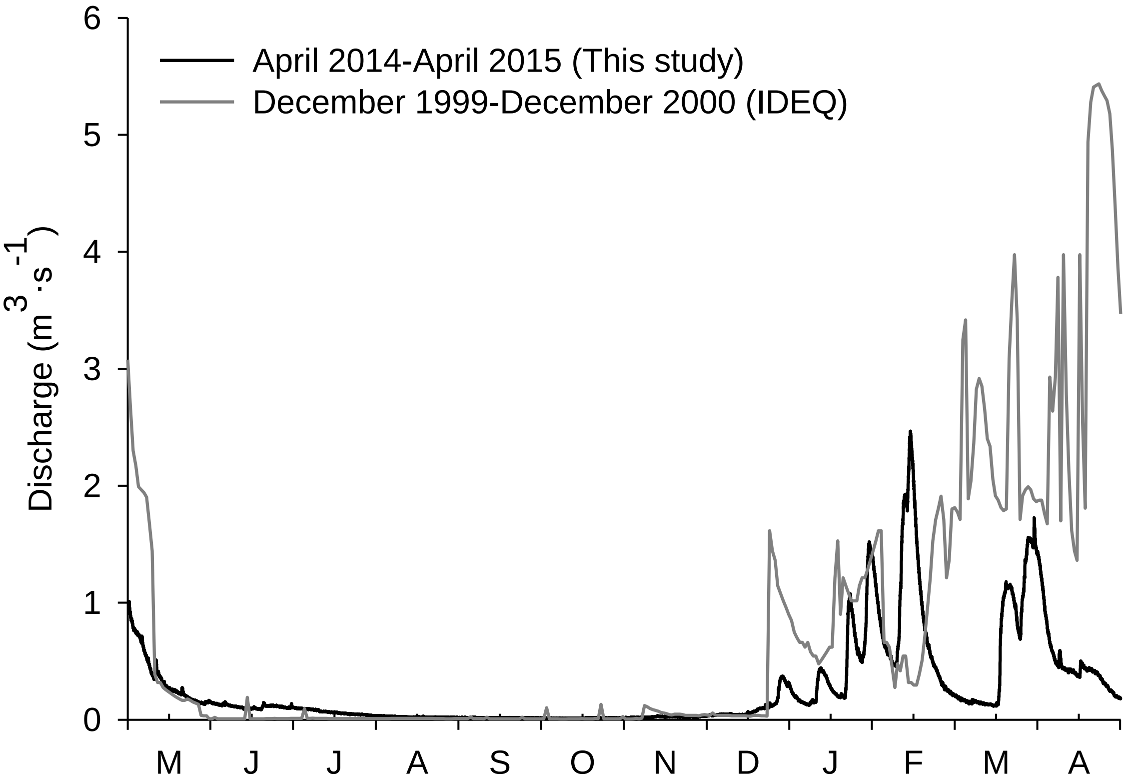

Figure 3.3: Annual discharge from the Total Maximum Daily Load (TMDL) assessment performed by IDEQ in 2013 using data from December 1999-December 2000 (Harvey-IDEQ unpublished data) in gray in comparison to discharge measured during this study from April 2014 to April 2015 in black.

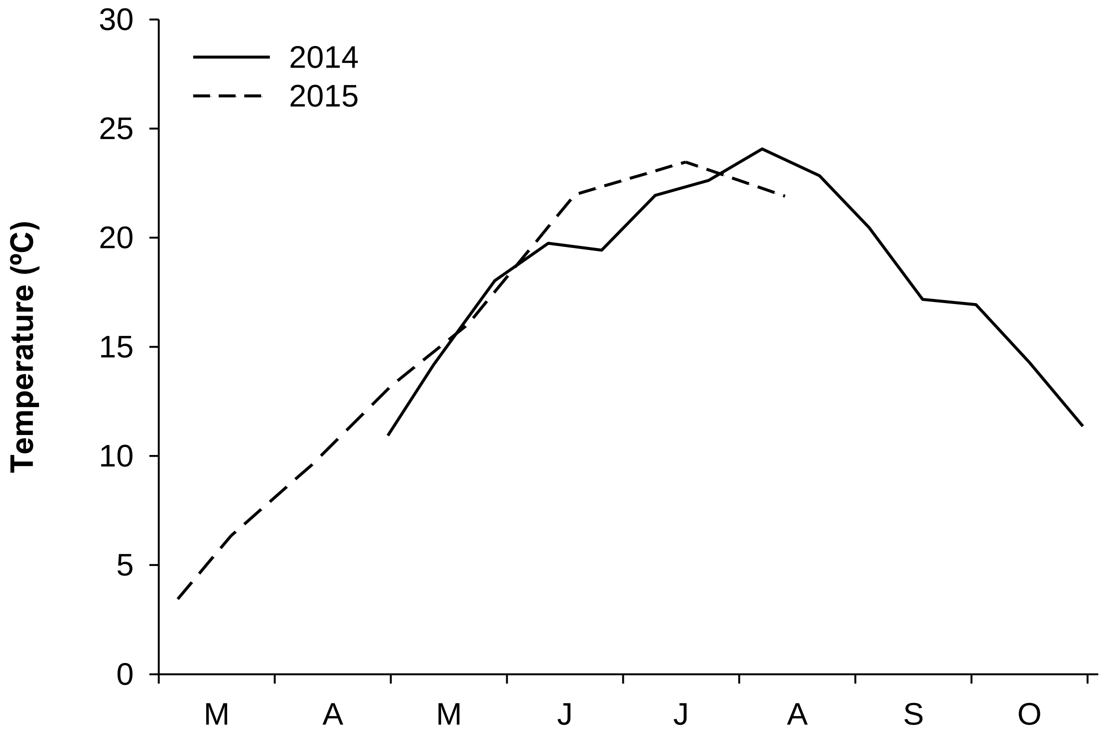

Figure 3.4: Volume weighted water temperatures for Fernan Lake as a function of time for the 2014 and 2015 study period.

Chapter 4: Sources of internal loading in a small polymictic lake in northern Idaho

Abstract

Internal loading estimates in Fernan Lake described in chapter 3 raise the question of the source(s) of the internal P load. Because Fernan Lake experiences low external inputs of P during the summer when cyanobacteria blooms are most prevalent, it is important to determine the source of this internal load. Because the bathymetry of Fernan Lake is a relatively shallow bathtub-shaped basin with east-west orientation and it has a large fish population suggests that water movement/sheer stress leading to sediment resuspension from wind mixing or the metabolic outputs of the biological community may be the source of the internal P load. Here I use a literature review to investigate these two potential sources of internal loading in Fernan Lake. I used a literature review because insufficient empirical data were available for Fernan Lake to estimate the contributions directly. This review will help focus future studies to quantify the components of the internal sources of P.

Introduction

Every lake has a finite lifespan, eventually evolving to solid land (Wetzel 2001). The addition of excessive nutrients accelerates a lake's evolutionary trajectory, decreasing its time as a lake. Generally, the input of excessive nutrients and sediment to aquatic ecosystems can be linked to anthropogenic activities across the landscape, and often increase the occurrence and severity of harmful algal blooms (HABs) (Downing 2013). Laws such as the Clean Water Act (CWA) in the USA have been enacted to control point source pollution, decreasing direct sources of nutrients to lakes and reservoirs. While the Clean Water Act has been successful at decreasing point source pollution, the control of non-point sources remains problematic and now represents the primary source of phosphorus (P) to many lakes and reservoirs (Ongley *et al.* 2010). Although some aquatic ecosystems have responded positively to non-point source controls (e.g., Eagle Stromberg 2012), many continue to have persistent algal blooms

(Søndergaard *et al.* 2003, 2007). This lack of a response is likely related to lag effects such as the continued availability of P from internal sources (Søndergaard *et al.* 2003) given that P does not have a gas phase and is very difficult to remove from aquatic ecosystems.

Internal loading of P is common and well known in lakes that experience periods of anoxia in bottom waters when redox-mediated reactions release soluble P into the water column (Nürnberg 1984, Søndergaard *et al*. 2003). Typically, the P increase in hypolimnetic waters does not become fully available to phytoplankton in the epilimnion until fall turnover (Søndergaard *et al.* 2003), however, periods of metalimnetic entrainment (Wetzel 2001) can introduce P to the epilimnion. In polymictic lakes, stratification only occurs for short periods of time which can result in internal loading of P that subsequently becomes available for uptake by algae when the lake is re-mixed. This can contribute to increase the frequency and intensity of cyanobacteria blooms (Søndergaard *et al.* 2003).

In shallow lakes with a long fetch, wind-induced sediment resuspension is common (Hamilton and Mitchell 1996) and can alter biological and chemical processes, resulting in high nutrient concentrations and subsequently algal blooms including HABs throughout the water column (Søndergaard *et al.* 1992, 2003). Sediment can be re-suspended and redistributed multiple times before being buried permanently (Søndergaard *et al.* 2003). However, Søndergaard *et al.* (2007) reviewed the restoration of 70 lakes in the Netherlands and concluded that lakes returned to algal dominated turbid states because of sediment-released P most likely related to wind-induced mixing. Furthermore, it is well known that even during periods with no sediment resuspension and oxic conditions, P is released from lake sediments (Lee *et al.* 1977). These releases of P from sediments highlight the persistence of this element once introduced into aquatic ecosystems.

High water temperatures in summer result in high metabolic activity (excretion and egestion) of biota including the release of P (Vanni 2002). Because much of this P is readily available for direct uptake and use by phytoplankton, blooms of cyanobacteria can occur in lakes with large populations of fish (Søndergaard *et al*. 2007, Schindler 1992, Eilers *et al.* 2011). When excretion by fish is the dominant source of internal P, such as in Diamond Lake, OR (Eilers *et al.* 2011), intentional removal of the fish can often reduce HABs (Søndergaard *et al*. 2007). Because such biomanipulations are labor intensive and costly, it is important to correctly identify that the fish community is the dominant source of internal P.

Given the commonality of contributions to internal loading in shallow lakes by wind-induced sediment resuspension, redox-dependent reactions, and the biotic community, creates a disparity in identifying the dominant source. For example, in Fernan Lake, ID, which suffers annual HABs, water column P concentrations increased during the summer when the contribution from external sources was low, suggesting that P is derived from internal sources. Because the lake is polymictic, it is unlikely that internal loading is due to redox-dependent reactions from the sediment. However, the lake is oriented east-west and experiences high winds throughout the summer that keep it mixed. Thus, wind-induced sediment resuspension may contribute to internal loading. In addition, Fernan Lake has an abundant fish community which may also contribute to the increasing water column P during summer. Here I use a literature review to the compare the internal loading rates calculated in chapter 3 to internal loading rates for sediment resuspension and release from biota reported in the literature to determine if sediment-released P from wind mixing or biota could account for those observed in Fernan Lake. I used this approach because insufficient empirical data were available to estimate the contributions directly in Fernan Lake. This review will help focus future studies to quantify the components of the internal sources of P. Knowing the dominant source should then help guide managers to identify appropriate remediation strategies.

Materials and Methods

Description of Fernan Lake study site

Fernan Lake is a small (150 ha) lake located just east of the city Coeur d' Alene in northern Idaho, USA (Figure 2.1). The shoreline length is 10.5 kilometers, while the maximum depth is 8.2 meters (IDEQ 2013). The watershed area is 49.7 km^2 (Table 2.1, IDEQ 2013) comprised primarily of steeply sloped hills that have shallow and easily erodible soils. Historic land use in the watershed was timber production with some agricultural activities in the valley bottom (Fernan Lake Watershed Plan 2003). The outflow of Fernan Lake is intermittent due to a seasonal dam located near the confluence of the outflow of Fernan Lake and French Gulch Creek (Figure 2.2) and enters Lake Coeur d' Alene. The seasonal dam maintains a high lake level for recreation during the summer, and is removed in late fall through spring runoff to avoid flooding. As early as 1990, the lake started to exhibit signs of high algal productivity with a high risk for cyanobacteria blooms (Mossier 1993, Fernan Lake Watershed Plan 2003). Since then, the algal blooms have gradually increased in severity, with "No contact advisories" being issued by the Panhandle Health District at least twice a year. Occurrences of species such as *Anabanea*, *Gleotrichia*, and *Microcystis* are responsible for high concentrations of the cyanotoxin microcystin (Kristin Larson-IDEQ, personal communication). The water quality of Fernan Lake must be maintained to meet the needs of the designated beneficial uses which include: cold water aquatic life, salmonid spawning, primary contact recreation and domestic water supply. Currently these designated uses are not being met and has result in the listing of the lake on the State of Idaho's 303d list, and the development of a total maximum daily load (TMDL) (IDEQ 2013).

The lake is a popular destination for residents of the City of Coeur d' Alene for recreation including fishing and swimming. Furthermore, some of the 71 homes located in the community of Fernan Lake Village (US Census Bureau 2010, IDEQ 2013) border the lake on the west end. Not only are the blooms a threat to the health of humans recreating on the lake, but also to the citizens living adjacent to the lake. Historically many of the lakeshore homes used the water from the lake as a domestic water supply, but with the development of the City of Coeur d'Alene, many switched to its supply and now only a small number of residents still use the lake for their domestic water supply (IDEQ 2013), making it imperative that information about the harmful blooms reaches all residents or citizens with potential to come into contact with the toxins (Personal communication with residents at July 2015 FLCRA meeting).

The lake is stocked annually with triploid rainbow trout (*Oncorhynchus mykiss*) between April and September. The number of rainbow trout stocked in the lake each year varies from 20,000 to 30, 000 fingerlings (approx. 12 cm in length). In some years, approximately 3,000 channel catfish (*Ictalurus punctatus*) were stocked into the lake, but this did not occur in 2012 or 2014. In addition to regular stocking, the lake also has populations of yellow perch (*Perca flavescens*), black crappie (*Pomoxis nigromaculatus*), largemouth bass (*Micropterus salmoides*), pumpkinseed (*Lepomis gibbosus*), smallmouth bass (*Micropterus dolomieu*), bluegill (*Lepomis macrochicrus*), brown bullhead (*Ameiurus nebulosus*), tench (*Tinca tinca*), northern pike (*Esox lucius*) and occasional brook trout (*Salvelinus fontinalis*) and cutthroat trout (*Oncorhynchus clarkii)* (IDFG 2011).

Estimates of internal loading from wind-induced sediment resuspension and the fish community

To obtain estimates of internal loading for phosphorus loading from wind-induced mixing, I searched Google Scholar with combinations of the keywords and key phrases: *wind-induced internal loading*, *oxic release rates, oxic internal loading, phosphorus release in shallow lakes, polymictic lake phosphorus release.* From this search, I eliminated sources that focused solely on internal loading resulting from anoxia-induced P release in stratified lakes.

Estimates and methods for internal loading resulting from fish excretion was primarily derived from information in Eilers *et al.* (2011). Citations of this publication related to nutrient cycling in relation to fish were also investigated. Additionally, I searched Google Scholar using combinations of the keywords and phrases: *fish excretion and phosphorus release, nutrient cycling by fish, phosphorus increases due to fish, fish biometrics, biomanipulation, biological nutrient loading, fish excretion.* From this search, I eliminated publications related to hatchery fish, or cage-reared fish if they were given an artificial diet as they were typically not representative of ecosystem processes in the wild. I used the Idaho Department of Fish and Game Panhandle Region Fishery Management Annual Report (2011) to identify angler effort for lakes in the region and to identify fish species in Fernan Lake. No empirical estimates of fish densities were available for Fernan Lake.

The discussion section encompasses the results and knowledge acquired with the literature search. It is divided into two sections; wind-induced mixing and the fish population. I converted annual loading rates calculated in chapter 3 to daily loading rates to more easily compare among internal loading estimates reported in the literature. Because there are insufficient data on sediment properties in Fernan Lake, and the lack of empirical data on the density of fish, I could only compare the daily loading rates from chapter 3 to estimates from other lakes. Additionally, I suggest methods that could be used in Fernan Lake in future studies to determine the dominant source of internal loading.

Results

Estimates of the contribution of wind-induced mixing to internal loading

The initial literature search yielded 24 publications with rates of P loading from sediment resuspension and oxic release. These led to an additional five relevant publications, for a total of 29 published release rates of internal loading from sediment resuspension under oxidized conditions (Table 4.1). Eleven of these reported P loading in individual lakes with mean depths ranging from 1.3 to 12 m and surface areas from 0.03 to 2338 km^2. Loading rates of P ranged from 0.01 to 37 $mg \cdot m^{-2} \cdot day^{-1}$ (Table 4.1). In addition, 17 publications outlined details on methods used to measure sediment resuspension (Table 4.2).

Estimates of the contribution of fish excretion to internal loading

The literature search resulted in 20 publications of fish excretion rates that were relevant to internal loading in lakes. An additional six publications were added from the reference sections of the initial 20. Of the 26 publications, four used a bioenergetic approach to estimate P excretion by fish. Eleven presented species-specific and lake-specific rates of P excretion ranging from 0.02 to 5.46 $mg \cdot m^{-2} \cdot day^{-1}$ (Table 4.3). Six publications related increased water clarify as a result of biomanipulation experiments. Seventeen publications outlined details of methods used to measure fish excretion directly or through lake responses to biomanipulation (Table 4.4).

Estimates of internal loading rates in Fernan Lake

Internal loading rates in Fernan Lake calculated in chapter 3 showed internal loading rates of 183.18 $mg \cdot m^{-2} \cdot yr^{-1}$ in 2014 and 49.9 $mg \cdot m^{-2} \cdot yr^{-1}$ in 2015. The daily loading rates for the lake in 2014 and 2015 were 3.3 and 0.8 $mg \cdot m^{-2} \cdot day^{-1}$, respectively. These loading rates fall within the range of wind-induced resuspension rates found in the literature (Table 4.1) and the range of loading rates from fish excretion (Table 4.3).

Discussion

The daily loading rates in Fernan Lake were consistent with lakes of comparable mean depths (Table 4.1). For example, Loch Leven in Scotland (mean depth 3.9m), Moses Lake in Washington, USA (mean depth 5.6m) and Lake Pyhäjärvi in Finland (mean depth 5.4m) had the most similar mean depths to Fernan Lake (mean depth 5.1m). Loading rates in these three lakes ranged from 0.006 to 12 $mg \cdot m^{-2} \cdot day^{-1}$ which encompassed the 0.8 and 3.1 $mg \cdot m^{-2} \cdot day^{-1}$ assessed for 2014 and 2015, respectively, in Fernan Lake. These data suggest that wind-induced mixing could be the source of loading in Fernan Lake. However, the Fernan Lake loading rates are also within the range of fish-related loading rates reported in the literature (Table 4.3), meaning that fish may be the dominant source of internal loading in Fernan Lake. Thus, without empirical data I cannot identify the source of internal loading with certainty; this will require additional research. Below I discuss methods that could be used in Fernan Lake to assess and identify the dominant source.

Methods to measure wind-induced contributions to internal loading in Fernan Lake

Studies reported in the literature show that sediments naturally release phosphorus under oxic conditions (Lee *et al.* 1977, Andersen and Ring 1999). For these, release rates highly depend on the composition (organic content and bulk density) of the sediment (Carrick *et al.* 1993, Schelske *et al*. 1995, Søndergaard *et al.* 2003). Assessing the sediment composition in Fernan Lake would be straightforward and methods outlined by Schallenberg and Burns (2004) could be used. They removed the top 3 mm from sediment cores, homogenized the sediment with a stirring rod and analyzed bulk density, water content and organic content. These properties are important in determining the sediment's resistance to suspension (Schallenberg and Burns 2004).

Because wind-induced sediment mixing is related to the shear stress applied to the sediment as a result of wave action (Hamilton and Mitchell 1996, Bailey and Hamilton 1997, Cózar *et al.* 2005), determining the currents that occur as a result of wave action would provide insight to susceptibility of sediment resuspension (Kaitaranta 2013). This can be done using fluorescent dye (James and Barko 1991) and can be used to identify the difference between wind-influenced and convective currents resulting from cooling overnight. Additionally, estimates of shear stress from wind-induced waves can be calculated theoretically (Hamilton and Mitchell 1996).A number of other studies reported the use of empirical and site-specific models (Hamilton and Mitchell 1996, Chung *et al.* 2008, Chung *et al.* 2009, Cózar *et al.* 2005, Bailey and Hamilton 1996) which may be modified for use on Fernan Lake. Unfortunately, although models are useful, most rely heavily on abundant data, and have limited use if data are sparse, which may be the case for Fernan Lake.

The most direct measures of P release from suspended sediment have been from *in-situ* experiments and experiments using intact sediment cores. For example, in an experiment to quantify the amount of P released during a resuspension event in Lake Arreso, Denmark, Søndergaard *et al.* (1992) used field–collected sediment cores in a continuous flow system and simulated sediment resuspension using a piston moving up and down in the water column above the cores without touching the sediment. This method effectively measured the increase in soluble reactive phosphorus (SRP) in the water column. Interestingly, this study noted that the amount of SRP released into the water column depended on the equilibrium condition of SRP in the water column and the sediment. Releases of SRP continued until equilibrium concentrations were reached in the water column (Søndergaard *et al.* 1992). Although this method effectively measured SRP, the authors concluded it overestimated release rates when scaled to the entire water body (Søndergaard *et al.* 1992). However, this is the most direct measurement P released during sediment suspension and could be effectively applied in a laboratory setting using cores from Fernan Lake if supplemented with water column P measurements to account for overestimation.

In addition to laboratory experiments, *in-situ* lake concentrations of phosphorus, chlorophyll and turbidity have been measured after significant mixing events (Schallenberg and Burns 2004). For example, in Lake Waihola, sediment resuspension was a common problem, and the use of *in-situ* measurements allowed for the calculation of the relationship between turbidity, total phosphorus and chlorophyll in the water column (Schallenberg and Burns 2004), thus monitoring of lake water quality after large wind events at Fernan Lake could provide insight to the water column response to sediment resuspension. Measuring water column concentrations could be a method to account for overestimates from laboratory derived release rates from experiments using sediment core disturbances (Søndergaard *et al.* 1992).

Methods to determine the contribution of fish excretion to internal loading in Fernan Lake

Because Fernan Lake has a large fish population consisting of mostly omnivorous species, the excretion of P may drive the availability of P in the water (Ketola and Richmond 1994, Schindler and Eby 1997, Vanni 2002). Ideally, to assess the potential release of phosphorus into the water a bioenergetics approach should be used which would account for body size, density of fish, and temperature-dependent metabolic rates (Carpenter *et al.* 1992).

Four of the papers identified in the literature search used a bioenergetic approach to estimate P release from fish excretion in a whole-lake context (Table 4.4). In each study, a mass balance equation was presented to account for factors contributing to the flow of phosphorus in individual fish. The most basic version of the mass balance equation takes into account the amount of P consumed and allocated to growth (Kraft 1992, Schindler and Eby 1997, Schindler *et al.* 2001):

$U_p=C_p-G_p-E_p$ *(equation 4.1)*

Where U_p is urinary excreted P; C_p is the amount of P consumed by the fish; G_p is the amount of P allocated for growth; and E_p is the amount lost to egestion (feces, not readily available for algal uptake) (Kraft 1992, Schindler and Eby 1997, Schindler *et al.* 2001). The crux here is to identify the amount of P in the food consumed by fish. This can be derived empirically, or from the literature. Both have pros and cons which should be evaluated in light of the question under investigation (Schindler *et al.* 2001).

A bioenergetics model also requires estimates of species-specific density, their lengths and weights. Such data can be collected with standard fisheries techniques including gill nets (Nakashima and Legget 1980, Gido 2002), trap nets (Eilers *et al.* 2011), electrofishing combined with mark recapture techniques (Schindler *et al.* 2011, Vanni *et al.* 2002), seine fishing (Gido

2002, Vanni *et al.* 2002), or shoreline surveys (Eilers *et al.* 2011). With these data in hand, they can be used in the model by Hewett and Johnson (1992) which can be used to estimate fish consumption rates for multiple species (Schindler *et al.* 1997). While accurate, bioenergetic models require a significant amount of data which does not currently exist for Fernan Lake. In addition, many studies that use a bioenergetic approach have missing data and require assumptions to compensate for these (Kraft 1992, Schindler *et al.* 1997, Schindler *et al.* 2001) which introduces errors that should be assessed. To collect these data for Fernan Lake would require a well-funded multi-researcher effort.

Another approach to assessing the effect of fish on water quality is through biomanipulation to achieve improved water quality. This occurs through either natural, accidental or deliberate processes (Shapiro 1990, Benndorf 1990), and requires knowledge of the existing fish community. Strategies that have been implemented typically focus on piscivore additions to decrease zooplanktivore populations, increase zooplankton size and grazing on phytoplankton. Other manipulation experiments have completely eliminated the fish populations using treatments such as rotenone to improve water quality (Eilers *et al.* 2011). Because Fernan Lake has 11 different species inhabiting the lake, most of which are opportunistic omnivores, deliberate biomanipulation would require some knowledge of the dominant populations inhabiting the lake. Without this knowledge, biomanipulations can cause instability in the food web (Benndorf 1990).

Conclusions

Internal loading occurs each summer in Fernan Lake, the source of which should be determined to optimize and effectively target remediation strategies. My analysis of the literature has shown that wind-induced sediment resuspension and P released from the metabolism of fish could account for the increase in water column P in Fernan Lake. I would recommend investigating the sediment dynamics of the lake before the fish because of the potential relationship to wind events observed in chapter 2 and the inter-annual variability observed in chapter 3. This also would be the more cost-effective approach given the data required to assess the release from the fish community. Until the dominant source of internal loading is established, attempting to mitigate internal loading will be imprudent.

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Table 4.1: Loading rates of phosphorus (P) from wind-induced sediment resuspension rates and physical characteristics for 11 lakes.

Lake	Mean depth (m)	Area (km^2)	Loading rate ($mg \cdot m^{-2} \cdot day^{-1}$)	Reference
Lake Hiidenvesi, Finland	1.1	1.6	37	Niemisto and Horppila 2007
Salton Sea, CA, USA	9	963	0.0045	Chung *et al.* 2009
Taihu Lake, China	2	2338	10.77-23.1	Fan and Zhang 2001
Moses Lake, WA, USA	5.6	27	0.27-2.01	Jones and Welch 1990
Lake Pyhäjärvi, Finland	5.4	154	0.006-0.150	Elkholm *et al.* 1995
Lake Okeechobee, FL, USA	2.7	1732	0.29-1.09	Reddy *et al.*1995
S-9, AB, Canada	2.5	0.03	0.01-0.52	Shaw and Prepas 1990
Tucker, AB, Canada	7	7.2	0.15-0.22	Shaw and Prepas 1990
Loch Leven, Scotland	3.9	13.3	12	Spears *et al.* 2007
Lake Onondaga, NY, USA	12	11.7	10.3	Penn *et al.* 2000
Lake Arreskov, Denmark (Littoral)	1.3	3.17	1.27-2.48	Anderson and Ring 1999
Lake Arreskov, Denmark (Profundal)	3.3		0.82-0.95	
Lake Apopoka, FL, USA	2	125	1	Reddy *et al.*1996
Lake Arreso, Denmark	3	41	60-70	Søndergaard *et al.* 1992

Table 4.2: Methods to measure wind-induced sediment resuspension reported in the literature.

Method type	Lake	Reference	Brief description
model	South Island New Zealand Lakes	Hamilton and Mitchell 1996	Wind-wave theory
model	Salton Sea, CA, USA	Chung 2008, 2009	One-dimensional numeric model
laboratory	Lake Arreso, Denmark	Søndergaard *et al.* 1992	Sediment cores with piston
in-situ	Lake Waihola, New Zealand	Schallenberg and Burns, 2004	Water column measurements
in-situ	Nine Alberta Lakes, Canada	Shaw and Prepas 1990	Peepers to measure pore water
in-situ	Eau Galle Reservoir, WI, USA	James and Barko 1991	Fluorescent dye to assess currents
laboratory	Lake Arreskov, Denmark	Andersen and Ring 1999	Sediment core incubations
model	Thompsons Lake, Australia	Bailey and Hamilton 1997	Horizontal circulation model
in-situ	Gulf of Finland	Kaitaranta *et al.* 2013	Sedimentation Traps
model	Laguna Galarza, Argentina	Cózar *et al.* 2005	Empirical model and wind-wave theory
in-situ	Loch Leven, Scotland	Spears *et al.* 2007	Pore water and water column measurements
in-situ	Lake Hiidenvesi, Finland	Niemisto and Horppila 2007	Cylindrical sediment traps
in-situ	Lake Taihu, China	Kelderman *et al.* 2005	Several year mass budget
in-situ	Norfolk Broads, England	Phillips *et al.* 1994	Pore water and bioturbation by chironomids
in-situ	Lake Pyhäjärvi, Finland	Elkholm *et al.* 1997	Mass balance/sedimentation measurements
in-situ	Lake Apopka, FL, USA	Reddy *et al.* 1996	Core incubations
in-situ	Lake Okeechobee, FL, USA	Reddy *et al.* 1995	Sediment property analyses

Table 4.3: Phosphorus excretion rates for different fish species reported in the literature. Dashes indicate data that were not given.

Lake or Stream	Area (km^2)	Mean depth (m)	Species	Temperature (°C)	P excretion rate $mg \cdot m^{-2} \cdot day^{-1}$	Reference
Lake Gjersjøen	2.7	23	roach (*Rutilus rutilis*)	8-20	1.21-2.95	Brabrand *et al*. 1992
Lake Memphremagog	102	15.5	yellow perch (*Perca flavescens*)	15-23.8	0.35-2.07	Kraft 1992
Lake Michigan	58	85	alewife (*Alosa pseudoharengus*)	17	2.19	Kraft 1993
Lake Finjasjön	11	2.5	roach (*Rutilus rutilis*), bream (*Abramis brama*)	16	0.53	Persson 1997
Acton Lake	2.3	3.9	gizzard shad (*Doromosa cepedianum*)	--	5.46	Schaus *et al.* 1997
Lake Pend Orielle, ID	383	164	kokanee salmon (*Oncorhynchus nerka*)	4-8	0.02	Chipps and Bennett 2000
Diamond Lake, OR	12.26	6.9	tui chub (*Gila bicolor*)	20	2.3	Eilers *et al.* 2011

Table 4.4: Methods to measure phosphorus excretion rates in fish reported in the literature.

Method	Species measured	Lake	Reference
Bioenergetics model	rainbow trout	Lake Rototi, New Zealand	Blair *et al.* 2013
Bioenergetics model	stocked trout spp.	Sierra Nevada Lakes	Schindler *et al.* 2001
Bioenergetics/Fish mass balance	multiple spp.	multiple	Schindler and Eby 1997
Top down control effects (review)	multiple spp.	233 Danish Lakes	Jeppesen *et al.* 1997
Biomanipulation (review)	multiple spp.	multiple	Benndorf 1990
Laboratory tanks with natural diet	bream, perch, roach	Lake Wingra, WI, USA	Brabrand 1990
Bioenergetics model	perch	Lake Memphremagog, Canada	Kraft 1992
Biomanipulation	tui chub	Diamond Lake, OR, USA	Eilers 2011
Lake comparison study	multiple spp.	Peter Lake and West Long Lake	Schindler 1993
Direct capture and measurement	gizzard shad, smallmouth buffalo, river carpsucker	Lake Texoma, OK, USA	Gido 2011
Biomanipulation	multiple spp.	Lake Lilla Stockelidsvatten, Sweden	Stenson *et al.* 1978
Biomanipulation	replaced planktivores with piscivores	Tuesday Lake, WI, USA	Carpenter *et al.* 1992
Biomanipulation	bream, roach, pike perch	Lake Worderwijd/Nuldernauw, The Netherlands	Boers *et al.* 1991
Ecological Stoichiometry	several vertebrate spp.	Rio Las Marías,Venezuela	Vanni *et al.* 2002
Diet analysis and individual P-budgets	perch	Lake Memphremagog, Canada	Nakashima and Legget 1980
Mesocosm experiments	gizzard shad	Acton Lake, OH, USA	Schaus and Vanni 2000
Laboratory tanks with natural diet	roach and bream	Finjasjön, Sweden	Persson 1997

Chapter 5: Summary and Conclusions

Fernan Lake, ID has experienced blooms of cyanobacteria since the early 1990's (Mossier 1993), decreasing its aesthetic appeal. The intensity of these blooms has increased in recent years, resulting in health advisories and further decreases in aesthetic value. Low concentrations of nutrients (Nitrogen (N) and Phosphorus (P)) are necessary to maintain high water quality (low algal biomass) and must be balanced to maintain a diverse algal community (including small green algae) that is easily grazed by zooplankton to shunt carbon energy to higher trophic levels (IDFG 2012). Phosphorus limitation is ideal, because when N is limited an algal community dominated by cyanobacteria is favored (Paerl *et al.* 2011). This is because cyanobacteria can fix atmospheric nitrogen to overcome the N-limitation. Redfield (1958) suggested that a TN:TP ratio of 7:1 (by mass) was balanced, however, recent studies show that cyanobacteria dominated at ratios lower than 75:1 (Harris *et al.* 2014a). Fernan Creek is the main inflow to Fernan Lake and contributes most of the water and sediment to the lake. Historic activities in the upstream watershed such as road construction, stream restoration and agricultural activities on private lands have caused large changes in stream morphology (FHWA 2004, IDEQ 2013), ultimately resulting in the transport of large amounts of P to the lake. In 2013 a Total Maximum Daily Load (TMDL) assessment was completed for Fernan Lake by the Idaho Department of Environmental Quality to identify a target load of P to the lake to restore its designated beneficial uses. Unfortunately, this document was based on limited data. My thesis was aimed to contribute additional detailed data to better understand nutrient and sediment loading to the lake.

In chapter 2, I addressed the need to quantify the loading relationships via a mass balance approach (Vollenweider and Kerekes 1980, Wetzel 2001). This required quantifying in- and outflow loads to calculate the change in storage of nutrients and sediment. I found that 2297 tonnes of total residue entered the water body and 783 tonnes exited, thus 1514 tonnes of sediment remained in the lake. A total of 1125 kg of P entered the lake from the Fernan Creek inflow. Estimates of wet and dry deposition were 145 kg and 99 kg respectively, while the wetland and road culverts only contributed 33 kg and 1 kg, respectively. Thus the annual total load was 1403 kg, of which 264 kg of P left the lake via the outflow, resulting in a change in P storage of 1139 kg (81% retention efficiency). Additionally, I found that 93% of the load from Fernan Creek entered during the short snowmelt spring runoff between January and May. The large load that enters the lake during a short period of time makes management decisions for remediation difficult.

To remediate Fernan Lake, reducing the influx of P and sediment and treating the in-lake sources of P would be the best approach. Controlling the external load will require a whole-watershed approach. Because of the lack of sinuosity in lower Fernan Creek, it is likely that the water has a higher unit stream power (USP) during the spring which increases the velocity of the water and can cause erosion of the stream channel (Stall and Yang 1972). If this is the case in the lower Fernan Creek, re-meandering of the stream could increase the retention of the sediment and phosphorus by slowing down the water. For example, this was successful in Middle Tepee Creek, Idaho which resulted in a 22% reduction in sediment load (Eagle Stromberg 2012). This approach for Fernan Creek would require a respectful and tactful approach, as much of the lower Fernan Creek floodplain is privately owned.

Use of in-lake strategies alone are a temporary measure to increase water quality. As previously stated, they are most effective when coupled with reductions of the external load. I examined dredging, alum addition, nitrogen addition and introduction of superoxide radicals as possible remediation strategies. Dredging to remove the P contaminated sediment can be effective to reduce internal loads from legacy sediments (Murphy 1999), but is costly and requires space to de-water the removed sediment. Alum additions inactivate P in the water column and transports it back to the sediment (Welch and Cooke 1999), but the alum layer can become buried if inflows are not controlled and can promote the occurrence of inedible phytoplankton, which would negatively affect the zooplankton community (Harris *et al.* 2014b). The addition of nitrogen would re-balance the TN:TP ratio and reduce dominance of cyanobacteria, but is a controversial practice because nitrogen is considered a pollutant under the Clean Water Act (USEPA 1972). In addition to this, it requires continuous monitoring of the TN:TP ratio, making it a labor intensive strategy. A new emerging technology, called the Kria reactor, involves the addition of negatively charged ions in the form of a superoxide anion to reduce nitrogen and phosphorus in the system (Premier Materials 2013). Though it has been shown to be successful in laboratory studies, it is unclear how the removal of both N and P will affect the lake, the TN:TP ratio, or the concentration of dissolved gas in the lake. Further research is needed to understand the potential of this strategy. The approach for Fernan Lake that would likely be the most rapid and cost-effective would be the addition of nitrogen; the high flushing rate that Fernan Lake experiences makes it a good candidate for the consistent monitoring that nitrogen additions require.

In chapter 3, I estimated internal loading in Fernan Lake using two methods developed by Nürnberg (2009, 2012). I used method 2 (Nürnberg 2009, 2012) and the mass balance data from chapter 2 to calculate a net internal loading rate of 17.3 $mg \cdot m^{-2} \cdot yr^{-1}$and a gross internal loading rate of 101.6 $mg \cdot m^{-2} \cdot yr^{-1}$. Using method 1, I determined partial net internal loading rates of 183.18 $mg \cdot m^{-2} \cdot yr^{-1}$ and 49.9 $mg \cdot m^{-2} \cdot yr^{-1}$ in 2014 and 2015, respectively. The much lower 2015 rates were likely due to the low snowpack during the 2014-2015 winter. This positive relationship between flow (P and sediment delivery) and internal loading is supported by the large internal loading rates reported for the approximately two-fold higher maximum discharge in the 2013 TMDL (IDEQ 2015). Because I collected monthly samples from 30 sites in 2015, I was able to error check my whole-lake water column P mass I estimated from samples from one location (the deep site) in 2014. I found that data from one site resulted in an overestimate of the rate by 20.8 $mg \cdot m^{-2} \cdot yr^{-1}$. Though the deep site was historically the main site for sample collection (IDEQ 2013), results from my analysis reinforce the importance of multiple lake sites to estimate average lake P concentrations.

In chapter 4, I explored two potential sources of internal loading in Fernan Lake; wind-induced sediment mixing and contributions from the fish community to attempt to identify the most likely source in Fernan Lake. I used the rates calculated in chapter 3 and compared them to rates published in the literature, because I did not have empirical data of P loading from either wind-induced sediment resuspension or fish excretion in Fernan Lake. I found that the rates calculated in chapter 3 fell within the range reported for both wind-induced sediment resuspension and fish excretion (Tables 4.1 and 4.3). To evaluate the potential for wind-induced sediment resuspension in Fernan Lake, I recommend core incubations and sediment property analyses (Søndergaard *et al.* 1992, Reddy *et al.* 1995). To evaluate contributions by the fish community, I recommend first obtaining fish density data for the 11 different species in the lake, followed by a bioenergetic approach (Kraft 1992, Schindler and Eby 1997, Schindler *et al.* 2001).

Research presented in this thesis provides detailed information on nutrient and sediment relationships and their delivery to Fernan Lake, ID. Additionally, it provides insight to internal loading processes that occur in the lake. While a large amount of data was collected during this project, additional research should be undertaken to identify the dominant source of internal loading to optimize remediation strategies. The overall remediation of the lake should benefit from a two-pronged approach; one to reduce the external nutrient load, and one to reduce internal loading. Unless these are carried out simultaneously, it is likely that Fernan Lake will continue to experience cyanobacteria blooms.

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Appendix A: Temperature and oxygen profiles

Table A.1: Profiles of depth, temperature (°C), dissolved oxygen ($mg \cdot L^{-1}$), and conductivity ($\mu s \cdot cm^{-1}$) for the 2014 and 2015 study period in Fernan Lake, ID, measured at the deep site (Longitude: 520551, Latitude: 5279988, UTM WGS 1984).

Date	Depth (m)	Temp (°C)	DO ($mg \cdot L^{-1}$)	Conductivity ($\mu s \cdot cm^{-1}$)
1-Apr-14	0.14	7.06	12.12	38
1-Apr-14	0.97	6.36	12.24	38
1-Apr-14	1.98	6.15	11.97	39
1-Apr-14	3.02	6.11	11.51	39
1-Apr-14	3.96	6.11	11.02	39
1-Apr-14	4.97	6.1	10.63	39
1-Apr-14	6.06	6.09	10.28	39
1-Apr-14	6.05	6.09	10.17	39
1-Apr-14	6.96	6.08	9.92	39
15-Apr-14	0.21	10.51	10.69	38
15-Apr-14	1.09	10.52	10.61	37
15-Apr-14	1.98	10.48	10.49	38
15-Apr-14	3.02	10.27	10.25	38
15-Apr-14	4.10	10.15	9.84	38
15-Apr-14	5.00	9.85	9.57	38
15-Apr-14	6.05	9.43	8.91	38
15-Apr-14	7.04	9.37	8.44	38
29-Apr-14	0.41	12.52	11.02	37
29-Apr-14	1.00	12.36	10.4	37
29-Apr-14	0.98	12.31	10.35	37
29-Apr-14	2.01	11.65	10.19	37
29-Apr-14	3.01	10.61	10.03	37
29-Apr-14	4.00	10.32	9.66	37
29-Apr-14	5.01	10.02	9.16	37
29-Apr-14	5.99	9.98	8.98	37
29-Apr-14	7.01	9.97	8.73	37
29-Apr-14	7.52	9.97	8.63	37
27-May-14	0.22	19.35	9.03	38
27-May-14	0.94	19.21	8.69	38
27-May-14	2.00	18.90	8.49	38
27-May-14	3.01	17.98	8.19	38
27-May-14	4.03	17.71	7.75	38
27-May-14	5.01	17.60	7.57	38
27-May-14	6.15	16.15	6.98	39
27-May-14	6.85	16.06	5.17	40

Table A.1 continued

Date	Depth (m)	Temp (°C)	DO ($mg \cdot L^{-1}$)	Conductivity ($\mu s \cdot cm^{-1}$)
10-Jun-14	0.16	19.76	8.57	39
10-Jun-14	1.00	19.75	8.09	39
10-Jun-14	2.00	19.75	7.81	39
10-Jun-14	2.99	19.74	7.50	39
10-Jun-14	4.03	19.75	7.24	39
10-Jun-14	4.95	19.75	7.00	39
10-Jun-14	6.01	19.74	6.80	39
10-Jun-14	7.02	19.74	6.63	39
10-Jun-14	7.39	19.74	6.45	39
10-Jun-14	7.51	19.73	6.30	39
24-Jun-14	0.24	20.28	8.65	39
24-Jun-14	0.98	20.32	8.58	39
24-Jun-14	2.01	20.22	8.44	39
24-Jun-14	3.03	20.12	8.12	39
24-Jun-14	4.00	19.06	7.91	39
24-Jun-14	4.99	18.70	7.50	39
24-Jun-14	6.02	17.63	6.96	39
24-Jun-14	7.01	17.59	6.06	39
24-Jun-14	7.25	17.58	5.92	39
8-Jul-14	0.17	25.17	8.41	40
8-Jul-14	1.05	24.27	8.66	40
8-Jul-14	2.04	23.08	8.72	39
8-Jul-14	3.03	22.34	8.51	40
8-Jul-14	4.00	21.48	8.19	39
8-Jul-14	5.05	19.92	7.72	40
8-Jul-14	6.00	19.24	5.69	41
8-Jul-14	7.02	19.19	4.10	41
8-Jul-14	7.38	19.17	3.42	42
22-Jul-14	0.28	23.27	7.64	41
22-Jul-14	1.02	22.95	7.65	40
22-Jul-14	2.04	22.87	7.49	41
22-Jul-14	3.01	22.83	7.31	40
22-Jul-14	3.98	22.81	7.02	41
22-Jul-14	4.97	22.79	6.69	41
22-Jul-14	6.00	21.70	5.53	45
22-Jul-14	6.99	20.69	2.57	51

Table A.1 continued

Date	Depth (m)	Temp (°C)	DO (mg·L^{-1})	Conductivity (μs·cm^{-1})
5-Aug-14	0.19	25.78	9.66	42
5-Aug-14	1.00	25.32	9.77	42
5-Aug-14	2.07	25.05	9.45	42
5-Aug-14	3.02	24.15	9.14	42
5-Aug-14	3.97	23.99	8.92	42
5-Aug-14	5.07	23.13	8.40	42
5-Aug-14	5.94	21.93	5.49	46
5-Aug-14	7.02	21.84	2.68	46
20-Aug-14	0.09	23.15	9.50	43
20-Aug-14	1.00	23.02	9.75	43
20-Aug-14	2.00	22.96	9.47	43
20-Aug-14	3.01	22.93	9.14	43
20-Aug-14	3.99	22.92	8.78	43
20-Aug-14	5.02	22.70	8.10	42
20-Aug-14	6.03	22.39	5.64	43
20-Aug-14	7.00	22.34	4.79	43
20-Aug-14	7.42	22.34	4.58	44
2-Sep-14	0.10	20.55	10.37	42
2-Sep-14	0.97	20.53	9.72	42
2-Sep-14	2.01	20.51	9.42	42
2-Sep-14	3.01	20.51	9.12	42
2-Sep-14	4.03	20.50	8.78	42
2-Sep-14	5.02	20.49	8.41	42
2-Sep-14	6.00	20.32	7.96	42
2-Sep-14	7.04	19.87	7.32	42
2-Sep-14	7.21	19.88	6.91	42
16-Sep-14	0.14	17.68	9.78	40
16-Sep-14	1.01	17.55	9.88	40
16-Sep-14	2.01	17.22	9.62	40
16-Sep-14	3.00	17.06	8.81	40
16-Sep-14	4.03	17.04	8.41	40
16-Sep-14	5.00	17.04	8.05	40
16-Sep-14	6.00	17.01	7.71	40
16-Sep-14	7.03	16.99	7.45	40

Table A.1 continued

Date	Depth (m)	Temp (°C)	DO (mg·L^{-1})	Conductivity (μs·cm^{-1})
30-Sep-14	0.12	17.02	9.28	39
30-Sep-14	0.98	17.03	8.86	39
30-Sep-14	1.98	16.95	8.56	39
30-Sep-14	3.01	16.90	8.25	39
30-Sep-14	4.01	16.91	8.01	39
30-Sep-14	5.03	16.91	7.75	39
30-Sep-14	6.02	16.89	7.47	39
30-Sep-14	7.01	16.73	6.99	39
30-Sep-14	7.20	16.73	6.69	39
14-Oct-14	0.12	14.31	10.55	38
14-Oct-14	1.00	14.29	10.31	38
14-Oct-14	1.99	14.28	10.03	38
14-Oct-14	3.01	14.28	9.64	38
14-Oct-14	4.02	14.28	9.30	38
14-Oct-14	5.03	14.27	8.95	38
14-Oct-14	6.02	14.27	8.62	38
14-Oct-14	7.01	14.27	8.35	38
14-Oct-14	7.24	14.27	8.25	38
28-Oct-14	0.32	11.40	12.46	38
28-Oct-14	1.00	11.39	12.18	38
28-Oct-14	2.02	11.36	11.75	38
28-Oct-14	3.01	11.35	11.39	38
28-Oct-14	4.00	11.35	11.03	38
28-Oct-14	5.00	11.35	10.58	38
28-Oct-14	6.02	11.35	10.19	38
28-Oct-14	7.01	11.35	9.83	38
28-Oct-14	7.15	11.36	9.57	38
5-Mar-15	0.31	3.43	17.72	39
5-Mar-15	1.00	3.39	17.54	39
5-Mar-15	2.00	3.43	16.96	39
5-Mar-15	3.00	3.47	16.34	39
5-Mar-15	4.02	3.42	15.61	39
5-Mar-15	5.01	3.46	15.10	39
5-Mar-15	5.98	3.50	14.57	39
5-Mar-15	7.01	3.52	14.17	39
5-Mar-15	7.29	3.52	13.92	39

Table A.1 continued

Date	Depth (m)	Temp (°C)	DO ($mg \cdot L^{-1}$)	Conductivity ($\mu s \cdot cm^{-1}$)
19-Mar-15	0.20	6.96	16.04	40
19-Mar-15	1.01	6.47	16.07	39
19-Mar-15	2.01	6.40	15.69	39
19-Mar-15	3.02	6.37	15.15	40
19-Mar-15	3.99	6.30	14.62	40
19-Mar-15	5.00	6.22	13.98	40
19-Mar-15	6.01	6.17	13.44	40
19-Mar-15	6.98	6.15	12.89	40
19-Mar-15	7.44	6.15	12.63	40
19-Mar-15	7.75	6.16	12.44	41
9-Apr-15	0.29	10.78	14.13	39
9-Apr-15	1.00	10.58	13.95	39
9-Apr-15	2.01	10.11	13.69	39
9-Apr-15	2.99	9.83	13.31	39
9-Apr-15	4.01	9.02	13.19	39
9-Apr-15	5.15	8.84	12.63	39
9-Apr-15	6.01	8.80	12.30	39
9-Apr-15	7.06	8.77	11.97	39
9-Apr-15	7.50	8.69	11.61	39
30-Apr-15	0.29	13.32	13.79	39
30-Apr-15	1.00	13.33	12.82	39
30-Apr-15	2.00	13.33	12.51	39
30-Apr-15	3.00	13.33	12.15	39
30-Apr-15	4.00	13.33	11.77	40
30-Apr-15	5.01	13.33	11.39	39
30-Apr-15	6.00	12.96	11.14	40
30-Apr-15	7.02	12.35	10.85	39
30-Apr-15	7.72	12.33	10.48	39
20-May-15	0.16	18.17	9.48	39
20-May-15	1.04	18.11	9.43	39
20-May-15	1.99	17.16	9.62	39
20-May-15	3.03	15.82	9.84	39
20-May-15	3.99	15.35	9.53	39
20-May-15	5.07	15.00	7.71	39
20-May-15	6.02	14.53	6.41	40
20-May-15	7.05	14.29	4.34	40

Table A.1 continued

Date	Depth (m)	Temp (°C)	DO ($mg{\cdot}L^{-1}$)	Conductivity ($\mu s{\cdot}cm^{-1}$)
17-Jun-15	0.12	22.23	8.49	41
17-Jun-15	1.03	22.25	8.13	41
17-Jun-15	1.99	22.23	7.96	41
17-Jun-15	3.01	22.20	7.66	41
17-Jun-15	4.04	22.17	7.39	41
17-Jun-15	5.03	22.16	7.08	41
17-Jun-15	5.98	21.45	6.42	42
17-Jun-15	7.00	19.62	3.83	49
17-Jun-15	7.54	19.43	1.25	50

Table A.2: Locations and profiles of depth, temperature (°C), dissolved oxygen (mg·L^{-1}), and conductivity (µs·cm^{-1}) for April 16, 2015, coordinates are in UTM WGS 1984.

Date	Longitude	Latitude	Depth (m)	Temp (°C)	DO (mg·L^{-1})	Conductivity (µs·cm^{-1})
16-Apr-15	521635	5280350	0.31	10.15	14.01	39
16-Apr-15	521635	5280350	1.02	10.10	13.79	39
16-Apr-15	521635	5280350	2.00	10.04	13.37	39
16-Apr-15	521635	5280350	3.01	9.78	13.03	39
16-Apr-15	521635	5280350	4.02	9.47	12.64	39
16-Apr-15	521635	5280350	4.86	9.41	12.16	39
16-Apr-15	521422	5280126	0.29	10.27	14.43	39
16-Apr-15	521422	5280126	1.00	10.19	13.56	39
16-Apr-15	521422	5280126	1.99	9.79	13.19	39
16-Apr-15	521422	5280126	4.00	9.13	12.58	39
16-Apr-15	521422	5280126	4.99	9.05	12.24	39
16-Apr-15	521422	5280126	6.01	9.01	11.87	39
16-Apr-15	521422	5280126	7.06	8.95	11.53	39
16-Apr-15	521422	5280126	7.57	8.95	11.35	39
16-Apr-15	520318	5280269	0.32	10.18	14.43	39
16-Apr-15	520318	5280269	1.04	10.06	13.77	39
16-Apr-15	520318	5280269	2.01	9.75	13.47	39
16-Apr-15	520318	5280269	3.02	9.53	13.14	39
16-Apr-15	520318	5280269	4.01	9.45	12.68	39
16-Apr-15	520318	5280269	5.02	9.37	12.24	39
16-Apr-15	520318	5280269	6.00	9.12	11.92	39
16-Apr-15	520318	5280269	6.50	8.95	11.78	39
16-Apr-15	519829	5279934	0.29	10.25	14.12	39
16-Apr-15	519829	5279934	1.00	10.12	13.47	39
16-Apr-15	519829	5279934	2.00	9.64	13.23	39
16-Apr-15	519829	5279934	3.02	9.54	12.83	39
16-Apr-15	519829	5279934	4.02	9.29	12.47	39
16-Apr-15	519829	5279934	5.01	9.04	12.16	39
16-Apr-15	519829	5279934	5.99	8.95	11.81	39
16-Apr-15	519829	5279934	6.99	8.82	11.45	39
16-Apr-15	519829	5279934	7.33	8.81	11.34	39

Table A.2 continued

Date	Longitude	Latitude	Depth (m)	Temp (°C)	DO (mg·L^{-1})	Conductivity (µs·cm^{-1})
16-Apr-15	519438	5279976	0.35	9.81	13.36	39
16-Apr-15	519438	5279976	1.00	9.64	13.32	39
16-Apr-15	519438	5279976	1.99	9.28	13.12	39
16-Apr-15	519438	5279976	3.01	9.21	12.73	39
16-Apr-15	519438	5279976	4.01	9.15	12.37	39
16-Apr-15	519438	5279976	5.02	9.04	11.98	39
16-Apr-15	519438	5279976	6.01	8.87	11.67	39
16-Apr-15	519438	5279976	6.37	8.86	11.47	39
16-Apr-15	519553	5280226	0.28	10.20	17.59	39
16-Apr-15	519553	5280226	1.01	10.05	14.40	39
16-Apr-15	519553	5280226	1.97	9.75	14.08	39
16-Apr-15	519553	5280226	3.00	9.40	13.54	39
16-Apr-15	519553	5280226	4.01	9.33	13.06	39
16-Apr-15	519553	5280226	5.00	9.30	12.61	39
16-Apr-15	519553	5280226	6.03	9.11	12.20	39
16-Apr-15	519553	5280226	6.13	9.08	12.12	39
16-Apr-15	520096	5280283	0.30	10.48	14.88	39
16-Apr-15	520096	5280283	1.00	10.38	14.06	39
16-Apr-15	520096	5280283	1.99	9.72	13.69	39
16-Apr-15	520096	5280283	3.00	9.53	13.30	39
16-Apr-15	520096	5280283	4.02	9.40	12.77	39
16-Apr-15	520096	5280283	5.02	9.34	12.20	39
16-Apr-15	520096	5280283	6.00	9.14	11.77	39
16-Apr-15	520096	5280283	6.50	8.99	11.68	39
16-Apr-15	521078	5279970	0.27	11.23	15.45	39
16-Apr-15	521078	5279970	0.99	10.77	13.99	39
16-Apr-15	521078	5279970	2.00	10.46	13.50	39
16-Apr-15	521078	5279970	3.02	10.37	13.02	39
16-Apr-15	521078	5279970	3.99	9.57	12.85	39
16-Apr-15	521078	5279970	5.00	9.12	12.45	39
16-Apr-15	521078	5279970	6.01	9.01	11.99	39
16-Apr-15	521078	5279970	6.83	8.98	11.53	39

Table A.3: Locations and profiles of depth, temperature (°C), dissolved oxygen ($mg \cdot L^{-1}$), and conductivity ($\mu s \cdot cm^{-1}$) for July 16, 2015, coordinates are in UTM WGS 1984.

Date	Longitude	Latitude	Depth (m)	Temp (°C)	DO ($mg \cdot L^{-1}$)	Conductivity ($\mu s \cdot cm^{-1}$)
16-Jul-15	521504	5280192	0.26	24.10	7.51	44
16-Jul-15	521504	5280192	0.34	24.11	7.52	44
16-Jul-15	521504	5280192	0.34	24.11	7.53	44
16-Jul-15	521504	5280192	0.26	24.11	7.60	44
16-Jul-15	521504	5280192	1.05	24.13	7.51	44
16-Jul-15	521504	5280192	1.98	24.13	7.32	44
16-Jul-15	521504	5280192	3.03	24.12	6.95	44
16-Jul-15	521504	5280192	4.04	24.10	6.67	44
16-Jul-15	521504	5280192	4.99	23.12	1.78	49
16-Jul-15	521504	5280192	5.14	22.79	1.69	51
16-Jul-15	521112	5280177	0.26	23.86	7.91	44
16-Jul-15	521112	5280177	1.04	23.87	7.68	44
16-Jul-15	521112	5280177	2.01	23.90	7.37	44
16-Jul-15	521112	5280177	3.02	23.89	7.05	44
16-Jul-15	521112	5280177	4.00	23.88	6.75	44
16-Jul-15	521112	5280177	4.99	23.87	6.50	44
16-Jul-15	521112	5280177	6.00	22.35	0.15	59
16-Jul-15	520364	5279958	0.22	23.54	7.72	44
16-Jul-15	520364	5279958	1.02	23.56	7.46	44
16-Jul-15	520364	5279958	2.05	23.56	7.19	44
16-Jul-15	520364	5279958	3.02	23.55	6.88	44
16-Jul-15	520364	5279958	4.00	23.49	6.41	44
16-Jul-15	520364	5279958	5.01	23.47	6.16	44
16-Jul-15	520364	5279958	5.99	23.43	5.74	45
16-Jul-15	520364	5279958	6.98	22.42	0.10	56
16-Jul-15	520364	5279958	7.46	22.32	0.01	57
16-Jul-15	520120	5280002	0.25	23.31	7.72	44
16-Jul-15	520120	5280002	1.05	23.34	7.24	44
16-Jul-15	520120	5280002	2.01	23.33	6.95	44
16-Jul-15	520120	5280002	3.01	23.34	6.61	44
16-Jul-15	520120	5280002	4.01	23.34	6.29	44
16-Jul-15	520120	5280002	5.02	23.35	6.00	44
16-Jul-15	520120	5280002	5.99	23.33	5.81	44
16-Jul-15	520120	5280002	6.88	23.15	3.84	46

Table A.3 continued

Date	Longitude	Latitude	Depth (m)	Temp (°C)	DO ($mg \cdot L^{-1}$)	Conductivity ($\mu s \cdot cm^{-1}$)
16-Jul-15	519865	5280069	0.18	23.57	7.63	44
16-Jul-15	519865	5280069	1.02	23.57	7.36	44
16-Jul-15	519865	5280069	2.00	23.54	7.07	44
16-Jul-15	519865	5280069	3.01	23.48	6.72	44
16-Jul-15	519865	5280069	3.99	23.40	6.36	44
16-Jul-15	519865	5280069	5.02	23.35	6.03	44
16-Jul-15	519865	5280069	6.08	23.27	5.58	44
16-Jul-15	519546	5279894	0.28	23.45	7.50	44
16-Jul-15	519546	5279894	1.00	23.44	7.20	44
16-Jul-15	519546	5279894	2.00	23.44	6.90	44
16-Jul-15	519546	5279894	3.02	23.42	6.58	44
16-Jul-15	519546	5279894	4.02	23.41	6.26	44
16-Jul-15	519546	5279894	5.02	23.40	5.97	44
16-Jul-15	519546	5279894	6.01	23.36	5.77	44
16-Jul-15	519546	5279894	6.68	23.32	5.59	44
16-Jul-15	519885	5279920	0.38	23.62	7.57	44
16-Jul-15	519885	5279920	1.15	23.62	7.33	44
16-Jul-15	519885	5279920	2.03	23.6	7.06	44
16-Jul-15	519885	5279920	3.01	23.49	6.74	44
16-Jul-15	519885	5279920	4.01	23.38	6.41	44
16-Jul-15	519885	5279920	5.05	23.35	6.11	44
16-Jul-15	519885	5279920	6.04	22.75	2.03	52
16-Jul-15	519380	5280072	0.28	23.66	7.82	44
16-Jul-15	519380	5280072	1.02	23.66	7.5	44
16-Jul-15	519380	5280072	2.04	23.56	7.01	45
16-Jul-15	519380	5280072	2.99	23.42	6.55	44
16-Jul-15	519380	5280072	3.97	23.34	6.06	44
16-Jul-15	519380	5280072	5.10	23.31	5.53	45
16-Jul-15	519380	5280072	5.61	23.18	4.42	48
16-Jul-15	519380	5280072	5.59	23.18	2.86	48

Table A.3 continued

Date	Longitude	Latitude	Depth (m)	Temp (°C)	DO ($mg \cdot L^{-1}$)	Conductivity ($\mu s \cdot cm^{-1}$)
16-Jul-15	519965	5280103	0.27	23.96	8.62	44
16-Jul-15	519965	5280103	0.99	23.96	8.15	44
16-Jul-15	519965	5280103	2.05	23.94	7.87	44
16-Jul-15	519965	5280103	2.87	23.75	7.37	44
16-Jul-15	519965	5280103	4.00	23.67	6.74	44
16-Jul-15	519965	5280103	4.97	23.53	6.44	44
16-Jul-15	519965	5280103	6.01	22.83	4.79	50
16-Jul-15	519965	5280103	6.04	22.96	2.54	50
16-Jul-15	519965	5280103	6.37	22.35	0.74	60
16-Jul-15	520606	5280049	0.22	24.14	8.02	44
16-Jul-15	520606	5280049	1.03	24.17	7.88	44
16-Jul-15	520606	5280049	2.05	24.09	7.62	44
16-Jul-15	520606	5280049	3.04	24.05	7.28	44
16-Jul-15	520606	5280049	4.06	23.88	6.56	44
16-Jul-15	520606	5280049	5.08	23.31	3.17	55
16-Jul-15	520606	5280049	5.48	23.15	2.45	77
16-Jul-15	521002	5280253	0.27	24.14	7.86	44
16-Jul-15	521002	5280253	1.01	24.16	7.74	44
16-Jul-15	521002	5280253	2.00	24.10	7.56	44
16-Jul-15	521002	5280253	3.06	24.05	7.24	44
16-Jul-15	521002	5280253	4.02	23.98	6.89	44
16-Jul-15	521002	5280253	5.04	23.70	6.17	46
16-Jul-15	521002	5280253	5.44	23.11	2.79	63
16-Jul-15	521093	5280089	0.25	24.27	7.97	44
16-Jul-15	521093	5280089	1.03	24.23	7.97	44
16-Jul-15	521093	5280089	1.98	24.18	7.76	44
16-Jul-15	521093	5280089	3.01	24.10	7.04	44
16-Jul-15	521093	5280089	4.02	23.87	5.66	45
16-Jul-15	521329	5280214	0.20	24.32	8.03	44
16-Jul-15	521329	5280214	0.98	24.26	7.98	44
16-Jul-15	521329	5280214	2.02	24.21	7.77	44
16-Jul-15	521329	5280214	3.02	23.93	7.34	44
16-Jul-15	521329	5280214	3.91	23.75	6.28	44

Table A.4: Locations and profiles of depth, temperature (°C), dissolved oxygen ($mg \cdot L^{-1}$), and conductivity ($\mu s \cdot cm^{-1}$) for August 11, 2015, coordinates are in UTM WGS 1984.

Date	Longitude	Latitude	Depth (m)	Temp (°C)	DO ($mg \cdot L^{-1}$)	Conductivity ($\mu s \cdot cm^{-1}$)
11-Aug-15	519392	5279959	0.07	22.60	8.60	41
11-Aug-15	519392	5279959	1.01	22.22	9.06	43
11-Aug-15	519392	5279959	1.99	21.97	9.06	43
11-Aug-15	519392	5279959	3.01	21.76	8.97	43
11-Aug-15	519392	5279959	4.01	21.66	8.73	43
11-Aug-15	519392	5279959	5.02	21.45	8.47	43
11-Aug-15	519392	5279959	5.40	21.43	8.09	43
11-Aug-15	519925	5280047	0.22	22.96	9.79	43
11-Aug-15	519925	5280047	1.02	22.41	9.64	44
11-Aug-15	519925	5280047	2.03	22.08	9.64	44
11-Aug-15	519925	5280047	3.02	21.95	9.38	43
11-Aug-15	519925	5280047	4.01	21.83	9.09	43
11-Aug-15	519925	5280047	5.07	21.66	8.89	43
11-Aug-15	519925	5280047	6.02	21.48	8.51	43
11-Aug-15	519925	5280047	6.01	21.47	8.27	44
11-Aug-15	520551	5279988	0.18	22.88	9.56	44
11-Aug-15	520551	5279988	1.01	22.82	9.47	44
11-Aug-15	520551	5279988	2.01	22.08	9.48	43
11-Aug-15	520551	5279988	3.00	21.66	8.81	43
11-Aug-15	520551	5279988	4.00	21.61	8.58	43
11-Aug-15	520551	5279988	4.97	21.46	8.18	43
11-Aug-15	520551	5279988	6.01	21.40	7.18	43
11-Aug-15	520551	5279988	6.81	21.36	6.52	44
11-Aug-15	520551	5279988	6.99	21.37	6.33	48
11-Aug-15	521077	5280127	0.26	23.06	9.53	44
11-Aug-15	521077	5280127	1.03	22.96	9.37	44
11-Aug-15	521077	5280127	2.02	22.27	9.52	44
11-Aug-15	521077	5280127	3.00	21.99	9.26	44
11-Aug-15	521077	5280127	3.99	21.59	8.51	44
11-Aug-15	521077	5280127	5.02	21.32	6.85	44
11-Aug-15	521077	5280127	5.93	21.27	5.91	43

Table A.4 continued

Date	Longitude	Latitude	Depth (m)	Temp (°C)	DO ($mg \cdot L^{-1}$)	Conductivity ($\mu s \cdot cm^{-1}$)
11-Aug-15	521347	5280151	0.26	23.09	8.78	44
11-Aug-15	521347	5280151	1.02	22.82	9.15	44
11-Aug-15	521347	5280151	2.01	22.25	9.32	44
11-Aug-15	521347	5280151	3.01	22.01	9.04	43
11-Aug-15	521347	5280151	4.01	21.47	7.90	43
11-Aug-15	521347	5280151	5.01	21.34	6.79	43
11-Aug-15	521347	5280151	5.53	21.29	6.25	44
11-Aug-15	521610	5280256	0.21	23.30	8.89	44
11-Aug-15	521610	5280256	1.04	23.00	9.33	44
11-Aug-15	521610	5280256	2.01	22.34	9.51	44
11-Aug-15	521610	5280256	3.03	21.93	9.23	44
11-Aug-15	521610	5280256	4.02	21.60	8.19	43
11-Aug-15	521610	5280256	4.60	21.45	7.35	48

Appendix B: Reference pictures of sampler locations

Figure B.1: North Side location of sampler box from Fernan Creek Road looking west, the sampler tube runs into the middle of the creek.

Figure B.2: Culvert on the opposite side of the road from the North side box location. This is where cross section and velocity measurements were taken starting from the left side of the culvert. The culvert was marked on the underside beginning at 0.1 meters to 1.85 meters from left to right.

Figure B.3: Picture standing on top of the culvert at the North Side location where cross sections are taken. The pole at the top of the picture is the location of the PVC tube that houses the level logger.

Figure B.4: Picture of the Mike Webb Location. The level logger PVC tube is attached to the intake PVC tube for the sampler. Cross section and velocity measurements were taken at the concrete culvert starting from the right side. Distances were marked on the top of the concrete culvert from 0.2 m to 4.1 m from right to left of the picture.

Figure B.5: Picture of the outflow location. The dam is installed and overflowing. The box has a sampler tube that runs across the concrete of the dam. The level logger PVC is barely visible behind the bush on the right side.

Figure B.6: Image of the Outflow location with the dam removed. Discharge was measured by spreading a tape across the opening of the dam and measuring velocity from the left side (looking downstream).

Figure B.7: Image looking down on the dam when it is in place. Leakage was measured with a metal piece, 2L graduated cylinder and a stopwatch. Two measurements were taken, one from the left side and one from the right. The metal piece is placed against the side of the dam and funnels water into the cylinder.

Figure B.8: Image of concrete culvert downstream of the Outflow location (Dam). This is where discharge was measured when the dam was overflowing. Distances were not marked here, a measuring tape was placed on the top of the culvert and the person on the top directed the person below with the velocity meter.

Figure B.9: Image of the Outflow location PVC housing the level logger tube. It is painted black to reduce the effects of ice cover.

Appendix C: Stage discharge rating curves.

This project followed the procedures put into place by a previous MS student for the development of stage discharge rating curves (Rajkovich 2014). Stage (water height) was recorded every two weeks at the same location (immovable boulder or concrete pad) with a meter stick. Discharge was measured by traditional cross-sections with an OTT MF Pro velocity meter (USGS 1983, Rajkovich 2014). Discharge was regressed as a function of stage, to form a rating curve and analyzed with the relationship of:

$$y = a \times x^b$$

Least squares regression was performed using Microsoft Excel's solver function (Rajkovich 2014). It should be noted that Sigma-plot yields the same regression equation as the solver function in Microsoft Excel and can be used to check methodology. Measured discharge values and stages for each location are shown in Table C.1., fitted parameters are shown in Table C.2 and site specific rating curves are shown in Figures C.1-C.3.

Stage was measured in 15 minute intervals using HOBO water level loggers (model number U20-001-01) and HOBOware Pro software (Onset computer corporation, Bourne MA).

References

Rajkovich H. 2014. Research in the Willow Creek watershed: estimates of sediment and phosphorus loads from sub-catchments; gauging public response to a constructed wetland; and a quantitative assessment of a conceptual constructed wetland. [MS Thesis]. [Moscow (ID)]: University of Idaho.

United States Geological Survey (USGS). 1983. Measurement and computation of streamflow: volume 2. Computation of discharge.

Table C.1: Discharge (Q) and stage values measured at the North Side, Mike Webb and Outflow sampling locations of Fernan Lake during the 2014-2015 sampling period.

	North Side		Mike Webb		Outflow	
Date	Q ($m^3 \cdot s^{-1}$)	Stage (m)	Q ($m^3 \cdot s^{-1}$)	Stage (m)	Q ($m^3 \cdot s^{-1}$)	Stage (m)
15-Apr-14	0.38	0.15	0.51	0.64	0.95	0.40
29-Apr-14	0.37	0.14	0.99	0.85	1.09	0.40
12-May-14	0.29	0.08	0.29	0.53	0.37	0.25
27-May-14	0.25	0.06	0.11	0.38	--	--
10-Jun-14	0.22	0.04	0.05	0.34	--	--
24-Jun-14	0.22	0.05	0.07	0.37	--	--
8-Jul-14	0.19	0.03	0.04	0.30	--	--
22-Jul-14	0.11	0.01	0.03	0.25	--	--
5-Aug-14	--	--	0.00	0.20	--	--
20-Aug-14	--	--	0.01	0.18	--	--
2-Sep-14	--	--	0.01	0.16	--	--
16-Sep-14	--	--	0.02	0.17	--	--
30-Sep-14	--	--	0.00	0.15	--	--
14-Oct-14	--	--	0.00	0.14	--	--
28-Oct-14	--	--	0.00	0.14	--	--
18-Nov-14	--	--	0.01	0.19	--	--
2-Dec-14	0.12	0.01	0.04	0.24	0.30	0.28
16-Dec-14	0.19	0.03	0.07	0.30	0.15	0.20
6-Jan-15	0.28	0.07	0.14	0.39	0.32	0.27
22-Jan-15	0.36	0.13	0.60	0.67	0.59	0.36
5-Feb-15	0.33	0.11	0.49	0.63	0.85	0.39
19-Feb-15	0.31	0.09	0.50	0.62	1.26	0.47
5-Mar-15	0.21	0.04	0.18	0.38	--	--
19-Mar-15	0.47	0.23	1.14	0.90	--	--
2-Apr-15	0.33	0.11	0.71	0.74	--	--
16-Apr-15	0.31	0.10	0.47	0.61	--	--
30-Apr-15	0.22	0.04	0.16	0.41	--	--

Table C.2: Fitted parameters of a and b and corresponding R2 values obtained for each location used to calculate continuous discharge.

Location	a	b	R^2
North Side (NS)	1.16	2.13	0.84
Mike Webb (MW)	1.50	2.47	0.99
Outflow (OF)	8.54	2.43	0.95

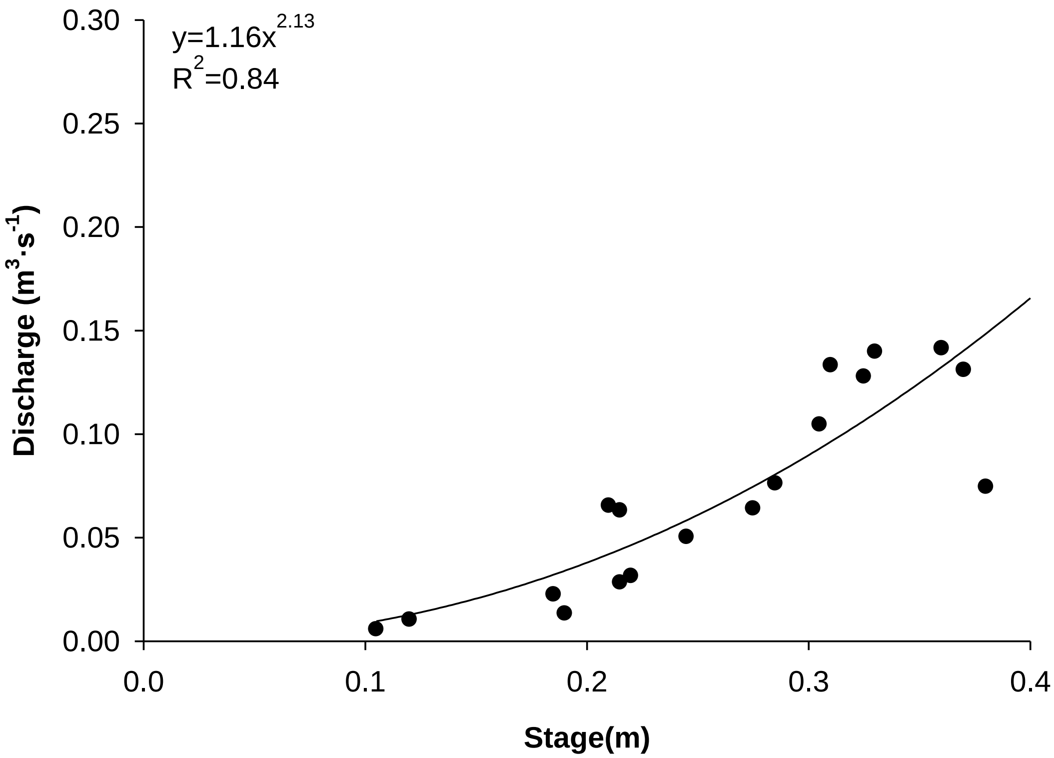

Figure C.1: Discharge as a function of stage for the North Side location at Fernan Lake, ID. The least square regression in the form $y = a \cdot x^b$ is presented as the solid line.

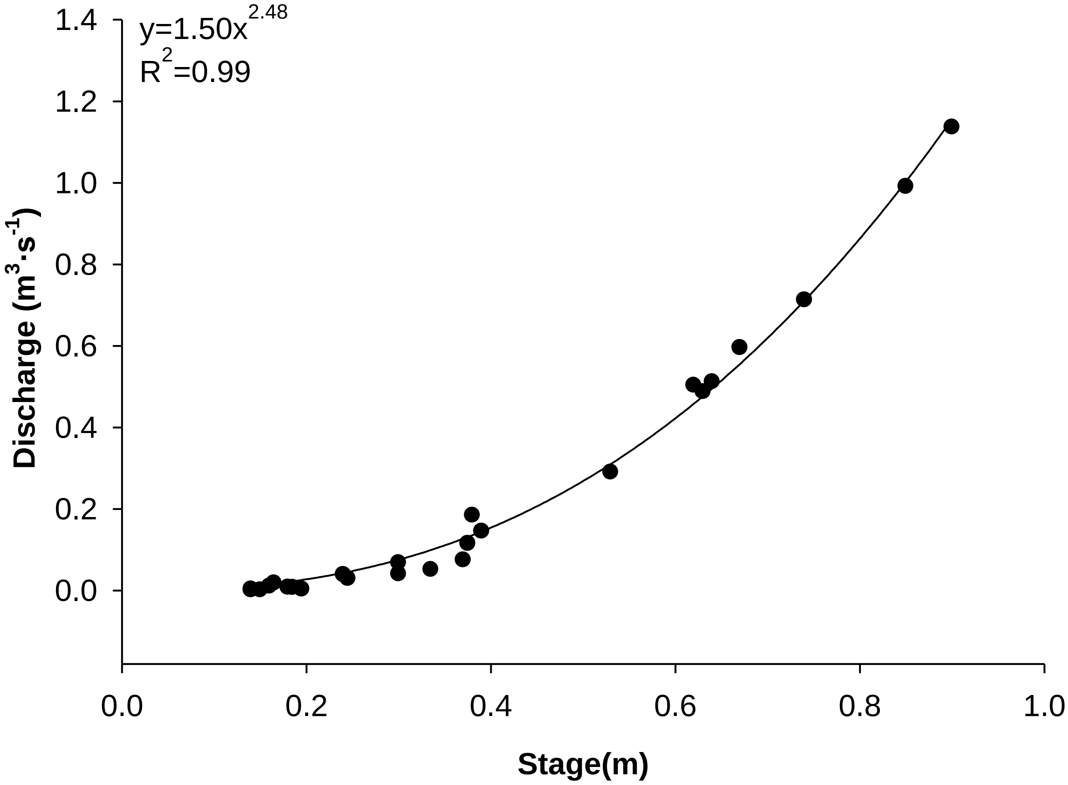

Figure C.2: Discharge as a function of stage for the Mike Webb location at Fernan Lake, ID. The least square regression in the form $y = a \cdot x^b$ is presented as the solid line.

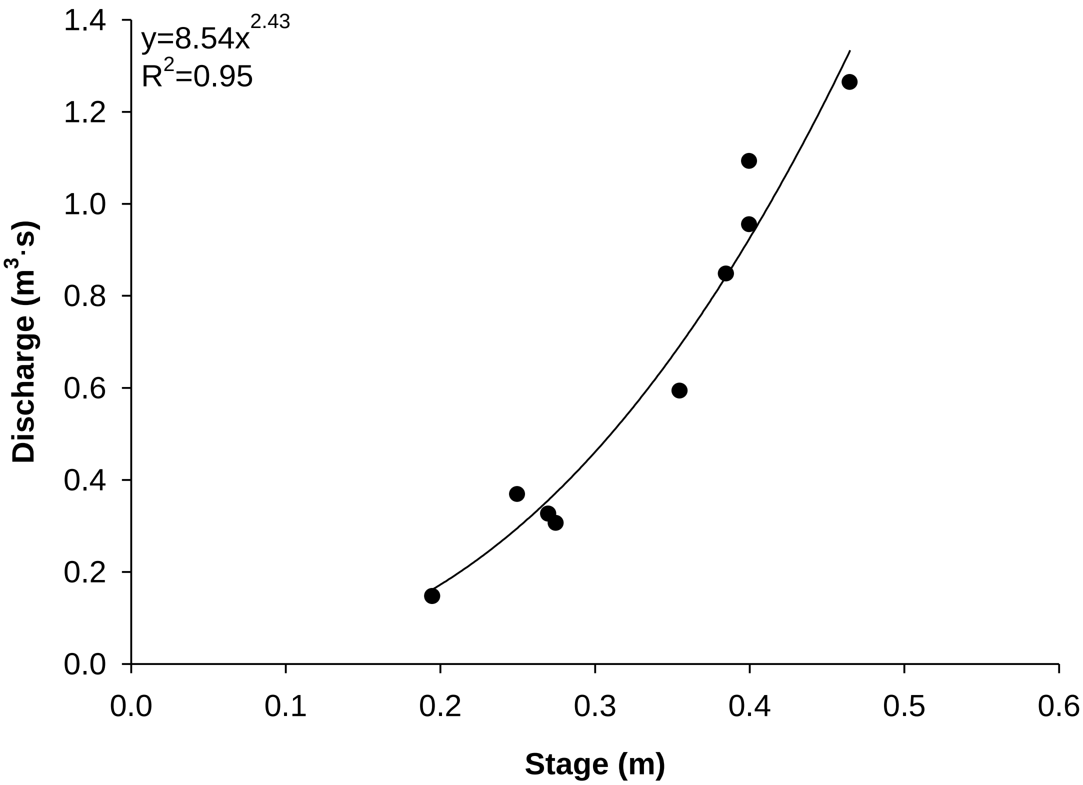

Figure C.3: Discharge as a function of stage for the Outflow location at Fernan Lake, ID. The least square regression in the form $y = a{\cdot}x^b$ is presented as the solid line.

Appendix D: Use of airport barometric pressure data to process HOBO-onset level loggers

Barometric pressure data measured by a HOBO-onset level logger was lost for the time period of June 10, 2014 to June 24, 2014. The level logger was mounted in a box to process the data collected by four other level loggers measuring water level. The water level loggers mounted in the water measure absolute pressure, which in this case is the atmospheric pressure and that due to water pressure. For the software to calculate water level, a reference atmospheric pressure reading is needed. Barometric pressure is fairly consistent over wide areas. When the barometric data file was lost, data from the Coeur d' Alene Airport (NCDC 2014) was used as the barometric pressure reference. To ensure that the Coeur d' Alene airport barometric data was comparable to the data being recorded by the logger, the data recorded from the airport was compared to the data the logger recorded for the two months prior to the loss of the file. This ensures that the data from the airport could be substituted for the barometric pressure recorded by the level logger. The airport reported data in 20 minute intervals, while the deployed water level loggers were recording data in 15 minute intervals. To make the datasets line up correctly in time, data were interpolated to compensate for the time differences. Data from the airport was reported in UTC (Coordinated Universal Time) and thus had to be corrected by 7 hours to PDT (Pacific Daylight Time) to match the water level logger time zone. Sigma plot was used for the regression analysis to account for the small elevation difference between Fernan Lake(650.95 m) and Coeur d' Alene airport (707.2 m). The regression showed a strong relationship (R^2= 0.9962, Figure D.1). The regression equation was then applied to the data for the two week period from the airport and water-level for the two week period was then calculated by HOBO onset level logger software.

$$y = 0.6833 + 0.9989x$$

References

(NCDC) National Climate Data Center. 2014. National Oceanic and Atmospheric Administration; (cited on 25 July 2014) Available from http://www.ncdc.noaa.gov/

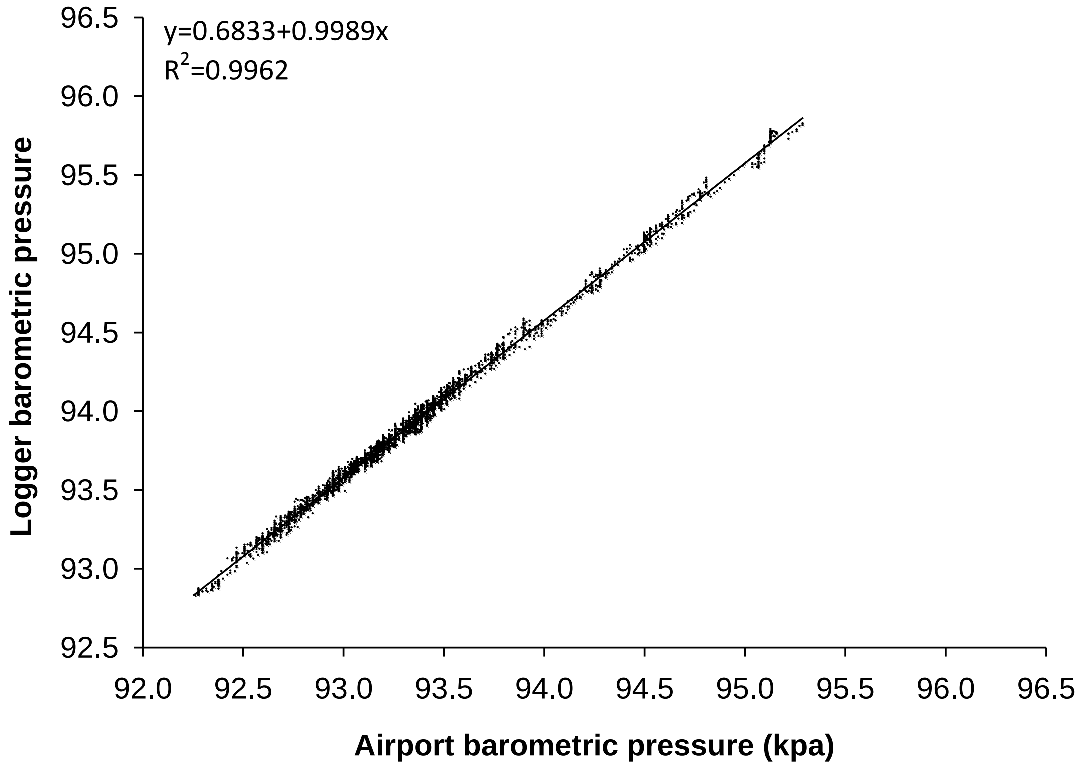

Figure D.1: Linear regression of barometric pressure of Hobo Onset water level logger barometric pressure data as a function of barometric pressure data collected from the Coeur d' Alene airport.

Appendix E: Calculation of annual total phosphorus (TP) and total residue (TR) loads using the smearing method (Duan 1983)

The following section follows the detailed description of the smearing method (Duan 1983, Collin 1995, Helsel and Hirsch 2002) provided in Appendix G of Rajkovich (2014) with a small change in the following equation:

$$Load = \exp\left[b_0 + b_1 \ln(Q)\right] * \frac{\sum_{i=1}^{n} \exp(ei)}{n}$$

Where Q is discharge (m^3/s), e_i are the residuals, n is the number of residuals, and b_o and b_1 are sub-catchment-specific fitted parameters. The daily loading estimates were summed to estimate annual loads. The Outflow location loads were calculated using this method only when the dam was not in place, otherwise it was estimated using daily averages and linear interpolation.

Annual TP (Figure E.1-E.4) and TR (Figure E.5-E.8) loads were graphed versus discharge and a nutrient specific linear relationship was determined for each location. The fitted parameters, bias correction factor and corresponding R^2 values for TP and TR loading are presented Table E.1 and E.2, respectively. Linear relationships determined were then applied to continuous discharge data to estimate 15 minute interval TP and TR loading for each location.

References

Collin TA. 1995. Recent advances in statistical methods for the estimation of sediment and nutrient transport in rivers. Review of Geophysics 33: 1117.

Duan N. 1983. Smearing estimate: a nonparametric retransformation method. Journal of American Statistical Association 78: 605–610.

Helsel DR., Hirsch R. 2002. Statistical methods in water resources techniques of water resources investigations, Book 4, chapter A3. Geological Survey.

Rajkovich H. 2014. Research in the Willow Creek watershed: estimates of sediment and phosphorus loads from sub-catchments; gauging public response to a constructed wetland; and a quantitative assessment of a conceptual constructed wetland. [MS Thesis]. [Moscow (ID)]: University of Idaho.

Table E.1: Fitted parameters, bias-correction factor and corresponding R2 values used to calculate location specific continuous TP loading calculated using the smearing method.

Location	b_o	b_1	bias-correction factor	R^2
North Side (NS)	11.28	1.1	1.11	0.88
Mike Webb (MW)	11.67	1.25	1.29	0.89
Outflow (OF)	10.26	1.04	1.03	0.93

Table E.2: Fitted parameters, bias-correction factor and corresponding R2 values used to calculate sub-catchment specific continuous TR loading calculated using the smearing method.

Location	b_o	b_1	bias-correction factor	R^2
North Side (NS)	11.84	1.09	1.17	0.78
Mike Webb (MW)	12.39	1.37	1.32	0.86
Outflow (OF)	11.32	0.94	1.07	0.77

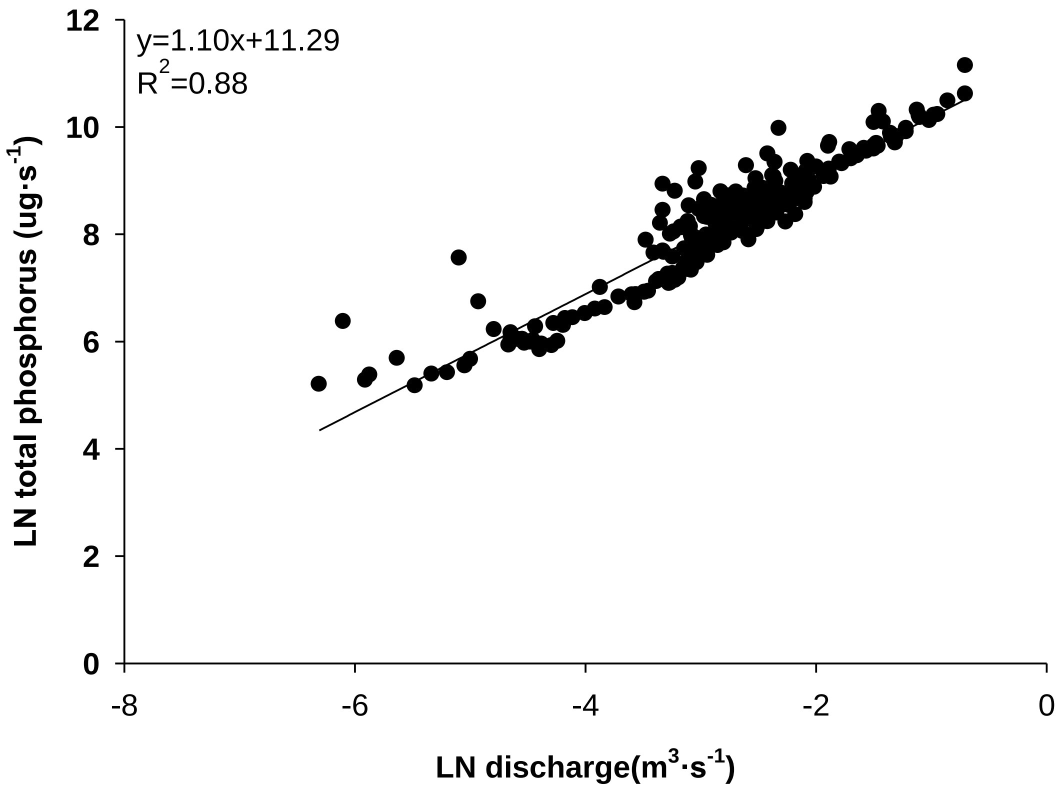

Figure E.1: Ln total phosphorus as a function of Ln discharge for the North Side location at Fernan Lake, ID during the sampling period of April 29,2014-April 29 2015.

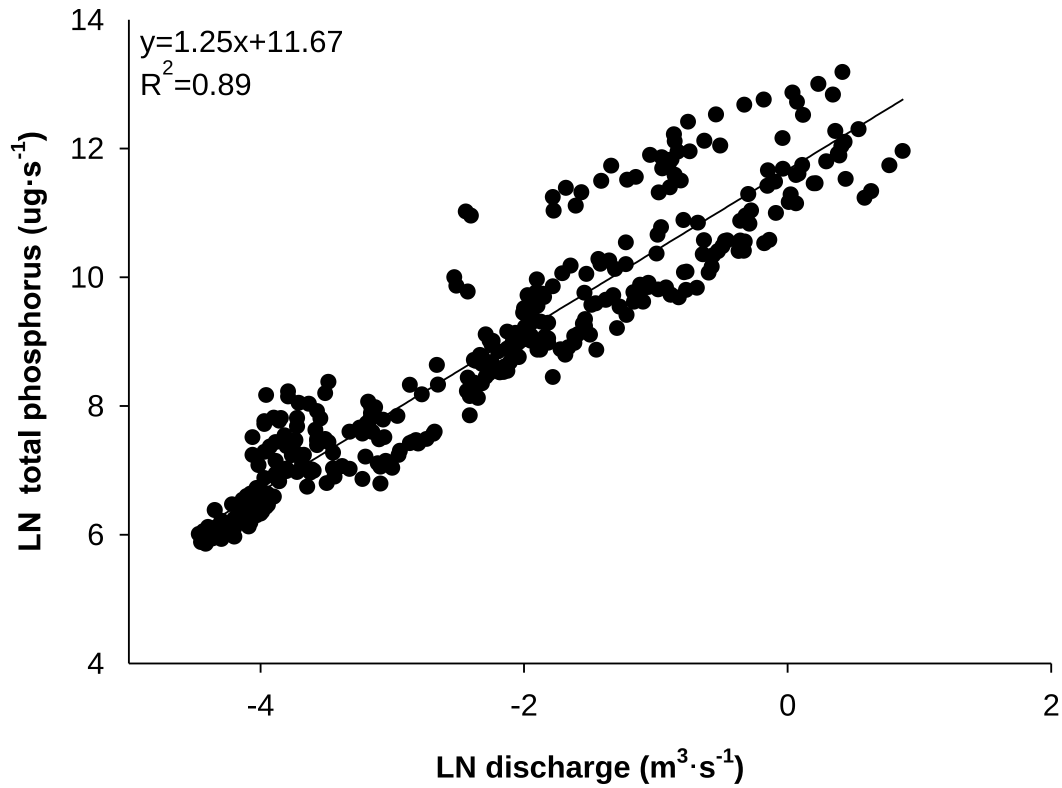

Figure E.2: Ln total phosphorus as a function of Ln discharge for the Mike Webb location at Fernan Lake, ID during the sampling period of April 29, 2014-April 29 2015.

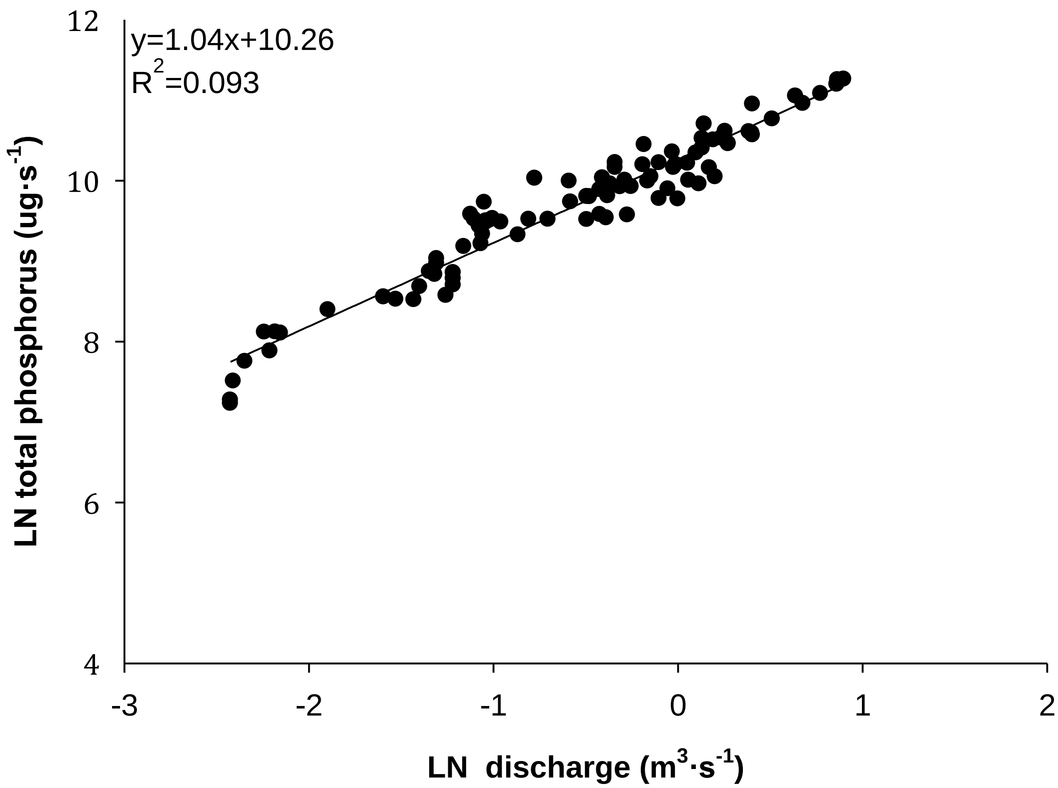

Figure E.3: Ln total phosphorus as a function of Ln discharge for the Outflow location at Fernan Lake, ID during the sampling period of April 29, 2014-April 29 2015.

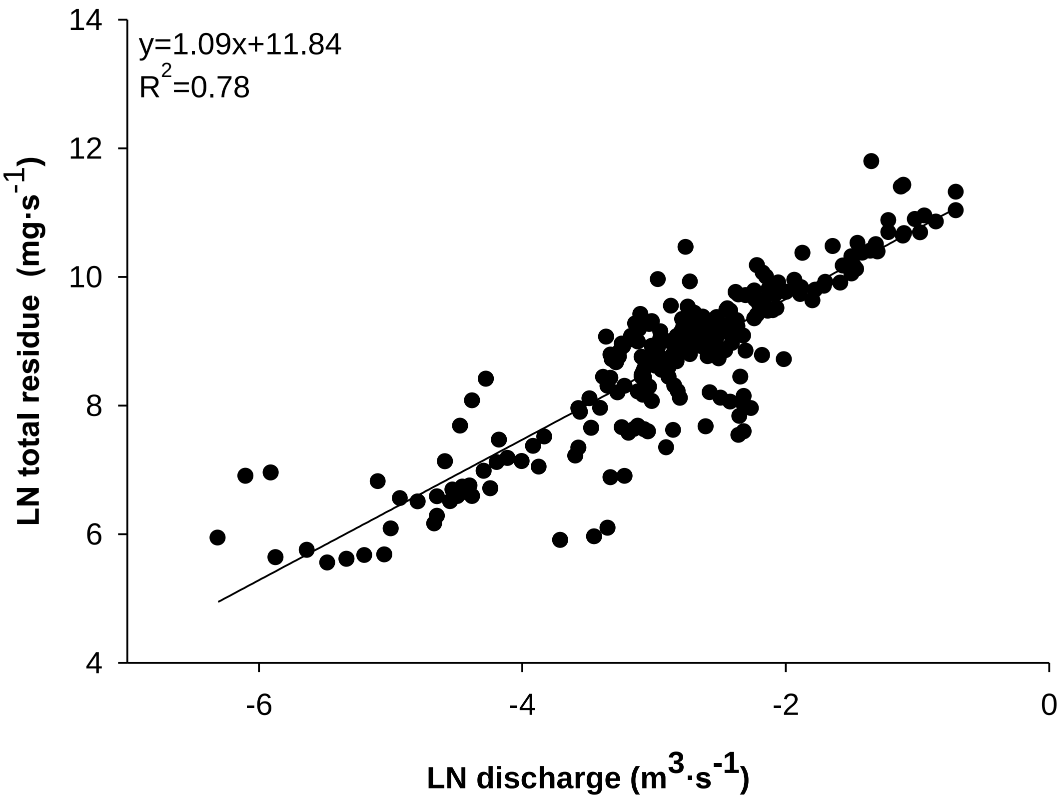

Figure E.4: Ln total residue as a function of Ln discharge for the North Side location at Fernan Lake, ID during the sampling period of April 29,2014-April 29 2015.

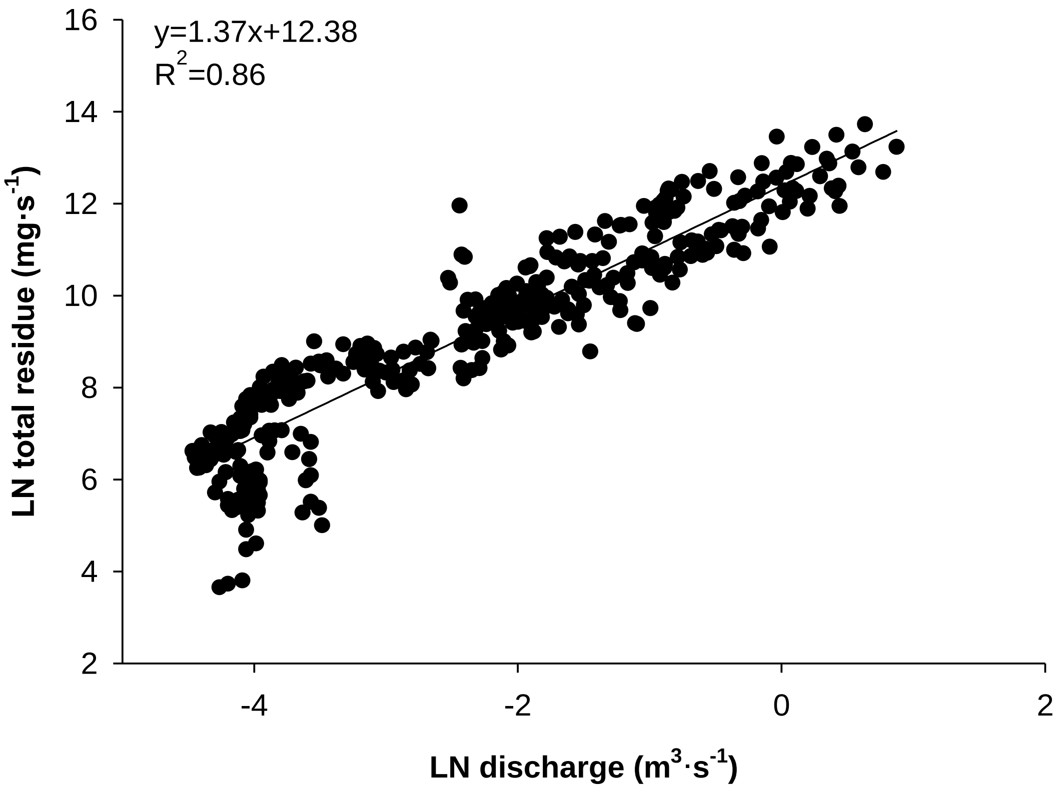

Figure E.5: Ln total residue as a function of Ln discharge for the Mike Webb location at Fernan Lake, ID during the sampling period of April 29, 2014-April 29 2015.

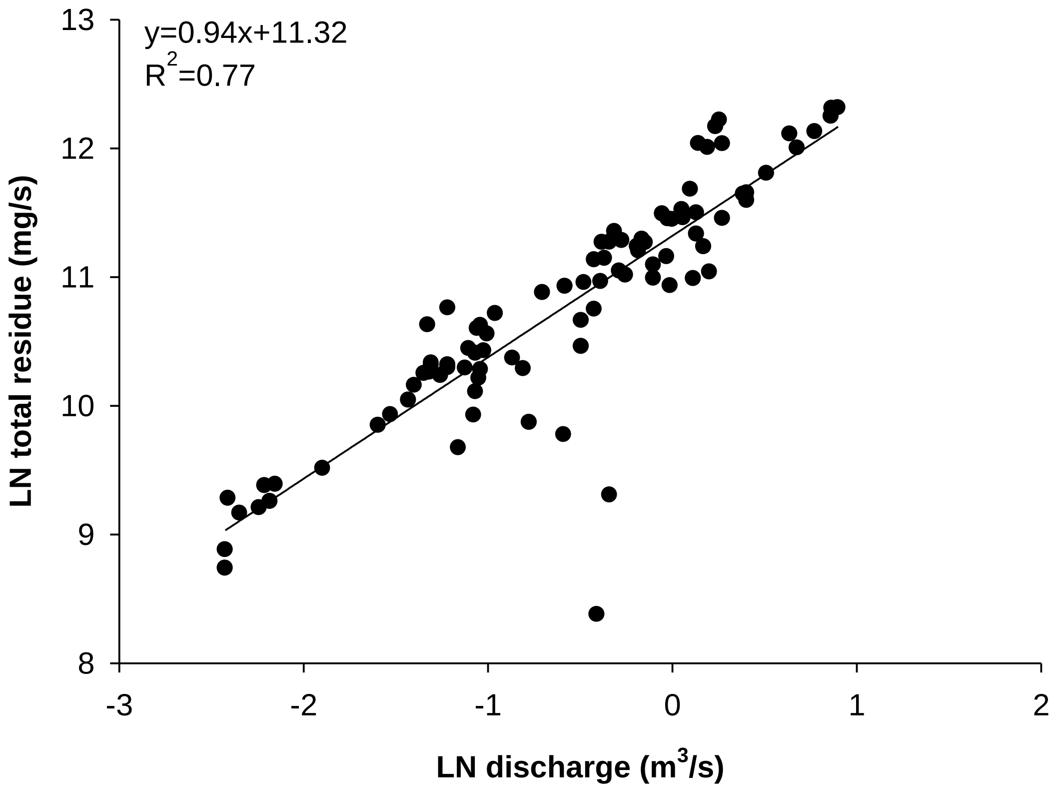

Figure E.6: Ln total residue as a function of Ln discharge for the Outflow location at Fernan Lake, ID during the sampling period of April 29, 2014-April 29 2015.

Appendix F: Lake elevations and corresponding volumes and surface areas

Lake volumes and surface areas were calculated using 15 minute interval water level data from a HOBO onset water level logger mounted at the Outflow location. Stage was measured using a meter stick and was measured from the base of the concrete pad on the upstream side of the dam. 15 minute water level data was averaged for each day and used to calculate the elevation of the lake each day.

Accurate volumes and surface areas were calculated using LiDAR data collected for the MILES project, bathymetry data collected by Frank Wilhelm and myself on 11 October 2014, 1 July 2015 and 17 July 2015. Data was then input into ARC GIS and the surface volume tool under 3D analyst tools was used to calculate the volume and surface area of the lake for each day of the study (Table F.1).

Table F.1: Dam Stages and the associated lake elevation, surface areas and volumes.

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
29-Apr-14	0.445	650.785	1654083.620	7994156.405
30-Apr-14	0.439	650.779	1652833.820	7983078.240
1-May-14	0.436	650.776	1652260.705	7977955.345
2-May-14	0.428	650.768	1650814.915	7964908.210
3-May-14	0.418	650.758	1649086.483	7949068.708
4-May-14	0.410	650.750	1647590.143	7935222.644
5-May-14	0.400	650.740	1645519.879	7919909.696
6-May-14	0.388	650.728	1642830.325	7899850.790
7-May-14	0.375	650.715	1639845.632	7877364.502
8-May-14	0.360	650.700	1636663.546	7853118.389
9-May-14	0.352	650.692	1634940.580	7839868.401
10-May-14	0.348	650.688	1634242.399	7834474.249
11-May-14	0.342	650.682	1632789.617	7823202.994
12-May-14	0.264	650.604	1616986.221	7697764.959
13-May-14	0.254	650.594	1614884.945	7680959.233
14-May-14	0.322	650.662	1628714.905	7791240.376
15-May-14	0.316	650.656	1627503.405	7781634.535
16-May-14	0.307	650.647	1625624.721	7766670.148
17-May-14	0.293	650.633	1622912.473	7745067.385
18-May-14	0.282	650.622	1620614.457	7726741.463
19-May-14	0.276	650.616	1619356.086	7716697.555
20-May-14	0.267	650.607	1617451.597	7701484.562
21-May-14	0.257	650.597	1615510.768	7685966.346
22-May-14	0.249	650.589	1613856.325	7672725.943
23-May-14	0.240	650.580	1612164.726	7659176.657
24-May-14	0.259	650.599	1615975.373	7689682.555
25-May-14	0.281	650.621	1620289.547	7724148.740
26-May-14	0.285	650.625	1621183.331	7731279.980
27-May-14	0.291	650.631	1622505.318	7741821.967
28-May-14	0.298	650.638	1623849.626	7752534.937
29-May-14	0.305	650.645	1625236.760	7763581.830
30-May-14	0.311	650.651	1626441.423	7773174.281
31-May-14	0.315	650.655	1627360.130	7780495.333
1-Jun-14	0.316	650.656	1627503.405	7781634.535
2-Jun-14	0.318	650.658	1627872.276	7784564.373
3-Jun-14	0.335	650.675	1631450.636	7812757.430
4-Jun-14	0.340	650.680	1632454.103	7820590.799

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
5-Jun-14	0.340	650.680	1632558.896	7821407.053
6-Jun-14	0.337	650.677	1631889.085	7816183.936
7-Jun-14	0.338	650.678	1632077.262	7817652.721
8-Jun-14	0.340	650.680	1632537.933	7821243.798
9-Jun-14	0.340	650.680	1632433.150	7820427.555
10-Jun-14	0.338	650.678	1632077.262	7817652.721
11-Jun-14	0.340	650.680	1632370.304	7819937.835
12-Jun-14	0.341	650.681	1632663.739	7822223.358
13-Jun-14	0.340	650.680	1632454.103	7820590.799
14-Jun-14	0.355	650.695	1628139.086	7786680.781
15-Jun-14	0.360	650.700	1636642.195	7852954.723
16-Jun-14	0.360	650.700	1636620.845	7852791.060
17-Jun-14	0.375	650.715	1639997.223	7878512.447
18-Jun-14	0.409	650.749	1647523.090	7934728.377
19-Jun-14	0.416	650.756	1648677.019	7945276.281
20-Jun-14	0.420	650.760	1649479.822	7952697.131
21-Jun-14	0.422	650.762	1649713.021	7954841.606
22-Jun-14	0.424	650.764	1650090.938	7958306.400
23-Jun-14	0.423	650.763	1649892.795	7956491.409
24-Jun-14	0.429	650.769	1651033.166	7966889.319
25-Jun-14	0.434	650.774	1652279.142	7978120.572
26-Jun-14	0.436	650.776	1652279.142	7978120.572
27-Jun-14	0.444	650.784	1653802.492	7991675.491
28-Jun-14	0.464	650.804	1657652.379	8025120.952
29-Jun-14	0.462	650.802	1657284.367	8021971.762
30-Jun-14	0.465	650.805	1657768.848	8026115.578
1-Jul-14	0.467	650.807	1658099.505	8028934.066
2-Jul-14	0.468	650.808	1658333.500	8030923.926
3-Jul-14	0.469	650.809	1658587.545	8033079.925
4-Jul-14	0.468	650.808	1658431.142	8031753.117
5-Jul-14	0.468	650.808	1658431.142	8031753.117
6-Jul-14	0.465	650.805	1658313.982	8030758.094
7-Jul-14	0.464	650.804	1657807.698	8026447.136
8-Jul-14	0.466	650.806	1657671.782	8025286.718
9-Jul-14	0.467	650.807	1658099.505	8028934.066
10-Jul-14	0.463	650.803	1658138.470	8029265.690
11-Jul-14	0.462	650.802	1657477.905	8023629.144
12-Jul-14	0.458	650.798	1657129.781	8020645.997
13-Jul-14	0.456	650.796	1656436.828	8014681.578
14-Jul-14	0.456	650.796	1656015.519	8011037.881

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
15-Jul-14	0.453	650.793	1655500.702	8006567.335
16-Jul-14	0.451	650.791	1655083.048	8002925.693
17-Jul-14	0.447	650.787	1654403.154	7996968.619
18-Jul-14	0.440	650.780	1653000.819	7984565.866
19-Jul-14	0.433	650.773	1651727.514	7973164.562
20-Jul-14	0.431	650.771	1651416.279	7970356.890
21-Jul-14	0.426	650.766	1650434.150	7961441.899
22-Jul-14	0.423	650.763	1650000.822	7957481.377
23-Jul-14	0.421	650.761	1649623.262	7954016.772
24-Jul-14	0.416	650.756	1648677.019	7945276.281
25-Jul-14	0.412	650.752	1647916.310	7938188.607
26-Jul-14	0.408	650.748	1647076.530	7931433.777
27-Jul-14	0.404	650.744	1646208.042	7925011.873
28-Jul-14	0.401	650.741	1645630.743	7920732.483
29-Jul-14	0.397	650.737	1644767.334	7914316.208
30-Jul-14	0.394	650.734	1643995.075	7908560.874
31-Jul-14	0.392	650.732	1643576.876	7905437.681
1-Aug-14	0.387	650.727	1642457.918	7897058.295
2-Aug-14	0.384	650.724	1641998.685	7893609.615
3-Aug-14	0.383	650.723	1641649.387	7890982.697
4-Aug-14	0.376	650.716	1640257.322	7880480.600
5-Aug-14	0.376	650.716	1640040.553	7878840.451
6-Aug-14	0.371	650.711	1639110.730	7871790.277
7-Aug-14	0.366	650.706	1637926.820	7862778.427
8-Aug-14	0.358	650.698	1636194.280	7849518.245
9-Aug-14	0.352	650.692	1635067.756	7840849.403
10-Aug-14	0.347	650.687	1633946.861	7832186.517
11-Aug-14	0.342	650.682	1632894.571	7824019.415
12-Aug-14	0.344	650.684	1633209.732	7826468.994
13-Aug-14	0.344	650.684	1633314.886	7827285.625
14-Aug-14	0.345	650.685	1633609.583	7829572.472
15-Aug-14	0.349	650.689	1634263.524	7834637.675
16-Aug-14	0.351	650.691	1634728.780	7838233.566
17-Aug-14	0.349	650.689	1634263.524	7834637.675
18-Aug-14	0.345	650.685	1633567.459	7829245.754
19-Aug-14	0.342	650.682	1632936.566	7824345.999
20-Aug-14	0.339	650.679	1632202.803	7818632.005
21-Aug-14	0.334	650.674	1631138.000	7810310.489
22-Aug-14	0.331	650.671	1630534.847	7805581.063
23-Aug-14	0.326	650.666	1629602.257	7798245.756

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
24-Aug-14	0.320	650.660	1628406.235	7788797.535
25-Aug-14	0.316	650.656	1627441.989	7781146.293
26-Aug-14	0.311	650.651	1626584.067	7774312.840
27-Aug-14	0.309	650.649	1626094.579	7770409.625
28-Aug-14	0.303	650.643	1624930.592	7761144.204
29-Aug-14	0.298	650.638	1623910.779	7753022.101
30-Aug-14	0.294	650.634	1623034.656	7746041.169
31-Aug-14	0.287	650.627	1621610.223	7734684.913
1-Sep-14	0.280	650.620	1620269.244	7723986.712
2-Sep-14	0.277	650.617	1619619.792	7718802.890
3-Sep-14	0.284	650.624	1620939.484	7729334.706
4-Sep-14	0.284	650.624	1620898.849	7729010.522
5-Sep-14	0.280	650.620	1620269.244	7723986.712
6-Sep-14	0.277	650.617	1619558.930	7718317.013
7-Sep-14	0.272	650.612	1618626.228	7710869.187
8-Sep-14	0.268	650.608	1617775.479	7704072.744
9-Sep-14	0.264	650.604	1616965.993	7697603.261
10-Sep-14	0.259	650.599	1615874.352	7688874.593
11-Sep-14	0.249	650.589	1613977.277	7673694.293
12-Sep-14	0.245	650.585	1613070.537	7666433.436
13-Sep-14	0.242	650.582	1612466.560	7661595.131
14-Sep-14	0.240	650.580	1612064.138	7658370.600
15-Sep-14	0.238	650.578	1611581.473	7654502.225
16-Sep-14	0.235	650.575	1611099.073	7650635.009
17-Sep-14	0.233	650.573	1610596.853	7646607.889
18-Sep-14	0.231	650.571	1610235.433	7643709.140
19-Sep-14	0.228	650.568	1609613.334	7638718.374
20-Sep-14	0.225	650.565	1609071.866	7634373.149
21-Sep-14	0.224	650.564	1608931.540	7633246.848
22-Sep-14	0.222	650.562	1608490.661	7629707.684
23-Sep-14	0.226	650.566	1609192.164	7635338.629
24-Sep-14	0.224	650.564	1608911.495	7633085.956
25-Sep-14	0.218	650.558	1607689.633	7623275.323
26-Sep-14	0.218	650.558	1607589.556	7622471.504
27-Sep-14	0.213	650.553	1606669.385	7615078.708
28-Sep-14	0.211	650.551	1606169.699	7611062.660
29-Sep-14	0.207	650.547	1605480.525	7605121.120
30-Sep-14	0.207	650.547	1605498.913	7605281.669
1-Oct-14	0.199	650.539	1604123.605	7593245.589
2-Oct-14	0.196	650.536	1603557.378	7588273.684

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
3-Oct-14	0.191	650.531	1602519.660	7579136.367
4-Oct-14	0.189	650.529	1602319.907	7577373.705
5-Oct-14	0.190	650.530	1602392.525	7578014.647
6-Oct-14	0.188	650.528	1602011.521	7574650.023
7-Oct-14	0.188	650.528	1602047.782	7574970.429
8-Oct-14	0.186	650.526	1601685.424	7571766.696
9-Oct-14	0.185	650.525	1601504.449	7570165.101
10-Oct-14	0.181	650.521	1600781.909	7563760.529
11-Oct-14	0.180	650.520	1600655.688	7562640.026
12-Oct-14	0.175	650.515	1599756.047	7554638.998
13-Oct-14	0.174	650.514	1599576.527	7553039.332
14-Oct-14	0.176	650.516	1599827.893	7555278.915
15-Oct-14	0.179	650.519	1600349.429	7559919.172
16-Oct-14	0.181	650.521	1600745.839	7563440.376
17-Oct-14	0.179	650.519	1600421.454	7560559.326
18-Oct-14	0.180	650.520	1600691.744	7562960.161
19-Oct-14	0.179	650.519	1600367.433	7560079.208
20-Oct-14	0.177	650.517	1600115.496	7557838.870
21-Oct-14	0.177	650.517	1600061.544	7557358.843
22-Oct-14	0.174	650.514	1599594.473	7553199.290
23-Oct-14	0.196	650.536	1603520.892	7587952.976
24-Oct-14	0.200	650.540	1604324.838	7595010.236
25-Oct-14	0.202	650.542	1604581.191	7597256.470
26-Oct-14	0.195	650.535	1603338.544	7586349.547
27-Oct-14	0.195	650.535	1603338.544	7586349.547
28-Oct-14	0.218	650.558	1607669.616	7623114.556
29-Oct-14	0.253	650.593	1614662.985	7679182.982
30-Oct-14	0.249	650.589	1613957.117	7673532.897
31-Oct-14	0.250	650.590	1614118.409	7674824.127
1-Nov-14	0.251	650.591	1614259.563	7675954.059
2-Nov-14	0.250	650.590	1614037.759	7674178.496
3-Nov-14	0.249	650.589	1613816.011	7672403.176
4-Nov-14	0.265	650.605	1617107.600	7698735.187
5-Nov-14	0.272	650.612	1618504.643	7709898.048
6-Nov-14	0.274	650.614	1618950.536	7713459.249
7-Nov-14	0.275	650.615	1619234.402	7715725.978
8-Nov-14	0.272	650.612	1618464.119	7709574.351
9-Nov-14	0.283	650.623	1620776.956	7728038.020
10-Nov-14	0.290	650.630	1622220.419	7739550.659
11-Nov-14	0.286	650.626	1621345.933	7732576.992

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
12-Nov-14	0.278	650.618	1619802.403	7720260.630
13-Nov-14	0.273	650.613	1618666.761	7711192.917
14-Nov-14	0.271	650.611	1618241.265	7707794.163
15-Nov-14	0.268	650.608	1617815.972	7704396.303
16-Nov-14	0.270	650.610	1618079.225	7706499.635
17-Nov-14	0.267	650.607	1617532.556	7702131.559
18-Nov-14	0.265	650.605	1617148.064	7699058.613
19-Nov-14	0.265	650.605	1617148.064	7699058.613
20-Nov-14	0.265	650.605	1617148.064	7699058.613
21-Nov-14	0.265	650.605	1617148.064	7699058.613
22-Nov-14	0.265	650.605	1617148.064	7699058.613
23-Nov-14	0.265	650.605	1617148.064	7699058.613
24-Nov-14	0.265	650.605	1617148.064	7699058.613
25-Nov-14	0.265	650.605	1617148.064	7699058.613
26-Nov-14	0.265	650.605	1617148.064	7699058.613
27-Nov-14	0.265	650.605	1617148.064	7699058.613
28-Nov-14	0.265	650.605	1617148.064	7699058.613
29-Nov-14	0.265	650.605	1617148.064	7699058.613
30-Nov-14	0.265	650.605	1617148.064	7699058.613
1-Dec-14	0.151	650.491	1595485.083	7516296.261
2-Dec-14	0.151	650.491	1595485.083	7516296.261
3-Dec-14	0.151	650.491	1595485.083	7516296.261
4-Dec-14	0.151	650.491	1595485.083	7516296.261
5-Dec-14	0.151	650.491	1595485.083	7516296.261
6-Dec-14	0.151	650.491	1595485.083	7516296.261
7-Dec-14	0.151	650.491	1595485.083	7516296.261
8-Dec-14	0.151	650.491	1595485.083	7516296.261
9-Dec-14	0.151	650.491	1595485.083	7516296.261
10-Dec-14	0.151	650.491	1595485.083	7516296.261
11-Dec-14	0.151	650.491	1595485.083	7516296.261
12-Dec-14	0.151	650.491	1595485.083	7516296.261
13-Dec-14	0.151	650.491	1595485.083	7516296.261
14-Dec-14	0.151	650.491	1595485.083	7516296.261
15-Dec-14	0.151	650.491	1595485.083	7516296.261
16-Dec-14	0.151	650.491	1595485.083	7516296.261
17-Dec-14	0.159	650.499	1596917.779	7529225.487
18-Dec-14	0.158	650.498	1596598.631	7526351.322
19-Dec-14	0.171	650.511	1599056.687	7548401.314
20-Dec-14	0.169	650.509	1598591.596	7544244.371
21-Dec-14	0.167	650.507	1598270.148	7541367.196

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
22-Dec-14	0.194	650.534	1603210.981	7585227.254
23-Dec-14	0.210	650.550	1606149.718	7610902.044
24-Dec-14	0.220	650.560	1608070.030	7626330.295
25-Dec-14	0.231	650.571	1610295.659	7644192.219
26-Dec-14	0.246	650.586	1613332.389	7668530.598
27-Dec-14	0.248	650.588	1613614.469	7670789.461
28-Dec-14	0.250	650.590	1614017.598	7674017.093
29-Dec-14	0.242	650.582	1612526.939	7662078.880
30-Dec-14	0.240	650.580	1612104.372	7658693.017
31-Dec-14	0.237	650.577	1611380.441	7652890.744
1-Jan-15	0.234	650.574	1610898.150	7649024.010
2-Jan-15	0.240	650.580	1612023.905	7658048.191
3-Jan-15	0.261	650.601	1616278.507	7692106.746
4-Jan-15	0.272	650.612	1618484.381	7709736.199
5-Jan-15	0.265	650.605	1617107.600	7698735.187
6-Jan-15	0.263	650.603	1616824.409	7696471.435
7-Jan-15	0.263	650.603	1616662.625	7695178.040
8-Jan-15	0.268	650.608	1617694.497	7703425.650
9-Jan-15	0.271	650.611	1618302.038	7708279.645
10-Jan-15	0.271	650.611	1617633.766	7702940.351
11-Jan-15	0.267	650.607	1617633.766	7702940.351
12-Jan-15	0.260	650.600	1616116.823	7690813.788
13-Jan-15	0.255	650.595	1615147.334	7683058.754
14-Jan-15	0.262	650.602	1616601.964	7694693.050
15-Jan-15	0.259	650.599	1615813.744	7688389.840
16-Jan-15	0.299	650.639	1623931.164	7753184.493
17-Jan-15	0.321	650.661	1628529.649	7789774.616
18-Jan-15	0.344	650.684	1633356.961	7827612.292
19-Jan-15	0.356	650.696	1635896.160	7847227.782
20-Jan-15	0.360	650.700	1636663.546	7853118.389
21-Jan-15	0.360	650.700	1636748.971	7853773.071
22-Jan-15	0.365	650.705	1637819.493	7861959.490
23-Jan-15	0.378	650.718	1640648.011	7883433.415
24-Jan-15	0.404	650.744	1646230.273	7925176.495
25-Jan-15	0.432	650.772	1651672.519	7972669.052
26-Jan-15	0.454	650.794	1655786.406	8009050.800
27-Jan-15	0.461	650.801	1656994.696	8019486.053
28-Jan-15	0.456	650.796	1656111.127	8011865.913
29-Jan-15	0.446	650.786	1654215.074	7995314.310
30-Jan-15	0.433	650.773	1651672.519	7972669.052

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
31-Jan-15	0.423	650.763	1649892.795	7956491.409
1-Feb-15	0.412	650.752	1647916.310	7938188.607
2-Feb-15	0.398	650.738	1645032.674	7916290.088
3-Feb-15	0.361	650.701	1636813.061	7854264.105
4-Feb-15	0.344	650.684	1633335.923	7827448.957
5-Feb-15	0.376	650.716	1640148.912	7879660.498
6-Feb-15	0.427	650.767	1650651.547	7963422.550
7-Feb-15	0.482	650.822	1661219.540	8055156.575
8-Feb-15	0.531	650.871	1673688.985	8135683.414
9-Feb-15	0.581	650.921	1684974.771	8220990.041
10-Feb-15	0.598	650.938	1688883.777	8249330.333
11-Feb-15	0.592	650.932	1687361.196	8238357.543
12-Feb-15	0.570	650.910	1682485.045	8202637.418
13-Feb-15	0.543	650.883	1676477.963	8157291.934
14-Feb-15	0.507	650.847	1666245.782	8096083.978
15-Feb-15	0.485	650.825	1661680.779	8058977.910
16-Feb-15	0.485	650.825	1661700.874	8059144.080
17-Feb-15	0.478	650.818	1660421.657	8048513.295
18-Feb-15	0.464	650.804	1657691.189	8025452.486
19-Feb-15	0.435	650.775	1652076.524	7976303.176
20-Feb-15	0.407	650.747	1646898.130	7930116.187
21-Feb-15	0.384	650.724	1641976.839	7893445.417
22-Feb-15	0.368	650.708	1638442.690	7866710.070
23-Feb-15	0.351	650.691	1634834.655	7839050.957
24-Feb-15	0.336	650.676	1631680.190	7814552.152
25-Feb-15	0.323	650.663	1628879.713	7792543.414
26-Feb-15	0.308	650.648	1625767.695	7767808.135
27-Feb-15	0.293	650.633	1622851.388	7744580.520
28-Feb-15	0.286	650.626	1621366.260	7732739.127
1-Mar-15	0.303	650.643	1624808.154	7760169.282
2-Mar-15	0.335	650.675	1631471.495	7812920.576
3-Mar-15	0.341	650.681	1632747.650	7822876.441
4-Mar-15	0.348	650.688	1634073.472	7833166.923
5-Mar-15	0.358	650.698	1636215.589	7849681.866
6-Mar-15	0.364	650.704	1637540.675	7859830.506
7-Mar-15	0.372	650.712	1639261.844	7872937.708
8-Mar-15	0.379	650.719	1640713.189	7883925.619
9-Mar-15	0.388	650.728	1642698.821	7898865.131
10-Mar-15	0.395	650.735	1644391.930	7911520.423
11-Mar-15	0.401	650.741	1645608.566	7920567.921

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
12-Mar-15	0.413	650.753	1648145.631	7940331.047
13-Mar-15	0.467	650.807	1658274.955	8030426.435
14-Mar-15	0.525	650.865	1672579.573	8126983.119
15-Mar-15	0.588	650.928	1686569.801	8232621.861
16-Mar-15	0.660	651.000	1682438.794	8359271.236
17-Mar-15	0.712	651.052	1698005.351	8439748.163
18-Mar-15	0.746	651.086	1707350.656	8498151.232
19-Mar-15	0.769	651.109	1714446.419	8538184.802
20-Mar-15	0.775	651.115	1716219.496	8547276.173
21-Mar-15	0.787	651.127	1720254.816	8568410.585
22-Mar-15	0.802	651.142	1725206.174	8594423.838
23-Mar-15	0.825	651.165	1732623.168	8634188.437
24-Mar-15	0.853	651.193	1741885.732	8682657.360
25-Mar-15	0.872	651.212	1748531.152	8715988.915
26-Mar-15	0.891	651.231	1755563.347	8749452.559
27-Mar-15	0.893	651.233	1756257.563	8752964.382
28-Mar-15	0.892	651.232	1755946.656	8751383.890
29-Mar-15	0.884	651.224	1752950.701	8736997.541
30-Mar-15	0.876	651.216	1749822.588	8722285.953
31-Mar-15	0.865	651.205	1745968.679	8703758.479
1-Apr-15	0.850	651.190	1740833.381	8677433.163
2-Apr-15	0.833	651.173	1735189.749	8647539.386
3-Apr-15	0.814	651.154	1729098.719	8615322.188
4-Apr-15	0.812	651.152	1728304.449	8611000.433
5-Apr-15	0.807	651.147	1726904.356	8603744.548
6-Apr-15	0.796	651.136	1723366.388	8584767.889
7-Apr-15	0.788	651.128	1720574.339	8570130.999
8-Apr-15	0.780	651.120	1718064.108	8556720.283
9-Apr-15	0.772	651.112	1715441.908	8543158.179
10-Apr-15	0.763	651.103	1712525.937	8527561.191
11-Apr-15	0.758	651.098	1711004.798	8519344.699
12-Apr-15	0.763	651.103	1712495.005	8527389.940
13-Apr-15	0.761	651.101	1711907.558	8524136.757
14-Apr-15	0.755	651.095	1710071.871	8514042.032
15-Apr-15	0.752	651.092	1708999.220	8507887.708
16-Apr-15	0.752	651.092	1709058.509	8508229.514
17-Apr-15	0.749	651.089	1708150.408	8503103.706
18-Apr-15	0.747	651.087	1707625.165	8499858.720
19-Apr-15	0.744	651.084	1706832.765	8494907.758

Table F.1 continued

Date	Dam Stage (m)	Lake Elevation (m)	Surface Area (m^2)	Volume (m^3)
20-Apr-15	0.737	651.077	1704838.612	8482284.624
21-Apr-15	0.730	651.070	1703123.119	8471208.731
22-Apr-15	0.728	651.068	1702508.090	8467462.564
23-Apr-15	0.721	651.061	1700630.116	8455721.705
24-Apr-15	0.719	651.059	1700027.155	8452151.018
25-Apr-15	0.714	651.054	1698732.345	8444163.924
26-Apr-15	0.710	651.050	1697494.586	8436861.964
27-Apr-15	0.705	651.045	1696171.777	8428886.838
28-Apr-15	0.711	651.051	1697836.499	8438729.411
29-Apr-15	0.703	651.043	1695543.439	8425155.950

Appendix G: Calculation of monthly evaporation values

Evaporation data used to calculate the water balance for Fernan Lake, ID was estimated from standard daily pan evaporation measured using a four-foot diameter Class A evaporation pan (WRCC2015).The pan water level reading is adjusted when precipitation is measured to obtain the actual evaporation. Most Class A pans are installed above ground and effects such as radiation on the side walls and heat exchanges with the pan material increase the evaporation totals. To adjust the evaporation rates to levels closer to natural water bodies the numbers were multiplied by 0.7 (WRCC 2015).

Monthly Evaporation Values (Table G.1) from the University of Idaho (Moscow, ID), Sandpoint Experimental Station (Sandpoint, ID) and the Spokane Airport (Spokane, WA) were averaged and multiplied by 0.7 to estimate values for Fernan Lake, ID. These values were applied as daily evaporation rates dependent on month. Differences in Evaporation rates along with the average of the three stations are shown in Figure G.1. Values were not given for January, February and December. To account for this, I applied the average of March, November and October to these three months.

References

WRCC (Western Regional Climate Center) Evaporation Stations; (cited 9 July 2015). Available from http://www.wrcc.dri.edu/htmlfiles/westevap.final.html

Table G.1: Averages of monthly evaporation rates of the three sites and the daily evaporation rates used in the water balance.

Month	Evaporation (mm)		
	Monthly Average	Daily Average	Adjusted *0.7
January	75.00	2.42	1.69
February	75.00	2.68	1.88
March	76.96	2.48	1.74
April	108.08	3.60	2.52
May	151.47	4.89	3.42
June	174.50	5.82	4.07
July	231.73	7.48	5.23
August	213.61	6.89	4.82
September	136.91	4.56	3.19
October	76.96	2.48	1.74
November	72.39	2.41	1.69
December	75.00	2.42	1.69

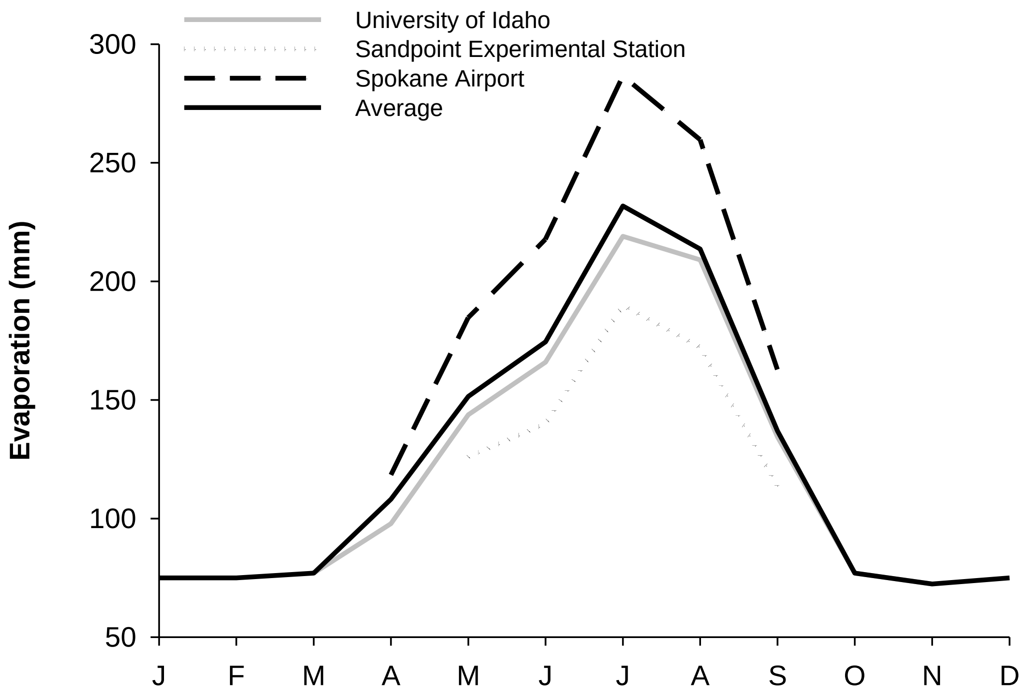

Figure G.1: Monthly evaporation rates for each site and the average of the three sites.

Appendix H: Aquifer recharge and seepage calculations

Differences between the daily water balance data and the actual volume of the lake calculated using ARC GIS were assumed to be aquifer recharge if there was a loss of water, or groundwater/aquifer seepage if there was a gain of water. To see if there was a correlation between aquifer levels and the differences calculated by the daily water balance, I used the water levels of a nearby aquifer monitoring well (Figure H.1). The monitoring well used was well number 50N 04W 13DAA1, data was available from 2007 to 2012 (IDWR 2015). Values for each year were averaged to get a general seasonal trend. Data were plotted with the differences between the estimated volume from the daily water balance (Table H.1) and the true volumes (Appendix F) to see if there was a correlation between the two (Figure H.2). The differences between the volumes is considered an unaccounted for or unmeasured gain or loss to the lake. The differences are largest during the Spring Runoff period which is the time of greatest water influx. Due to lack of available data, it is not possible to attribute the loss or gain of water to only the aquifer as there is likely a groundwater component affecting the lake that was not measured.

References

(IDWR) Idaho Department of Water Resources. Ground water level data; (cited 8 July 2015); Available from http://www.idwr.idaho.gov/hydro.online/gwl/

Table H.1: Daily changes in volume predicted by the water balance, the changes observed in ARC GIS and the resulting aquifer estimate. Negative and positive numbers indicate loss or gain of water.

	Volume estimates (m^3)		
Date	Predicted Δ Volume	GIS Δ Volume	Aquifer Estimate
29-Apr-14	-22836.75		
30-Apr-14	-30989.62	-11078.16	19911.46
1-May-14	-35173.83	-5122.90	30050.93
2-May-14	-33396.04	-13047.13	20348.91
3-May-14	-31493.42	-15839.50	15653.92
4-May-14	-32207.89	-13846.06	18361.82
5-May-14	-34484.86	-15312.95	19171.91
6-May-14	-32370.30	-20058.91	12311.39
7-May-14	-32595.41	-22486.29	10109.13
8-May-14	-32139.67	-24246.11	7893.56
9-May-14	-17973.27	-13249.99	4723.28
10-May-14	-27234.39	-5394.15	21840.24
11-May-14	-29010.63	-11271.25	17739.38
12-May-14	-7833.87	-125438.04	-117604.17
13-May-14	-4176.99	-16805.73	-12628.73
14-May-14	17148.74	110281.14	93132.41
15-May-14	16301.20	-9605.84	-25907.04
16-May-14	16514.28	-14964.39	-31478.67
17-May-14	14604.17	-21602.76	-36206.93
18-May-14	13979.47	-18325.92	-32305.39
19-May-14	14708.40	-10043.91	-24752.31
20-May-14	11774.05	-15212.99	-26987.05
21-May-14	10457.03	-15518.22	-25975.24
22-May-14	9371.76	-13240.40	-22612.16
23-May-14	8590.35	-13549.29	-22139.64
24-May-14	7969.77	30505.90	22536.13
25-May-14	6968.97	34466.18	27497.21
26-May-14	6679.09	7131.24	452.15
27-May-14	7014.36	10541.99	3527.62
28-May-14	7650.16	10712.97	3062.81
29-May-14	7381.81	11046.89	3665.08
30-May-14	6612.41	9592.45	2980.04
31-May-14	6159.60	7321.05	1161.45
1-Jun-14	4532.40	1139.20	-3393.19
2-Jun-14	4112.26	2929.84	-1182.42
3-Jun-14	5159.92	28193.06	23033.14
4-Jun-14	4843.79	7833.37	2989.58

Table H.1 continued

	Volume estimates (m^3)		
Date	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
5-Jun-14	3926.46	816.25	-3110.20
6-Jun-14	3385.98	-5223.12	-8609.09
7-Jun-14	3039.44	1468.78	-1570.66
8-Jun-14	2660.15	3591.08	930.93
9-Jun-14	2451.00	-816.24	-3267.25
10-Jun-14	1868.34	-2774.83	-4643.17
11-Jun-14	1530.39	2285.11	754.73
12-Jun-14	1480.91	2285.52	804.61
13-Jun-14	1446.37	-1632.56	-3078.93
14-Jun-14	2069.74	-33910.02	-35979.76
15-Jun-14	1405.16	66273.94	64868.78
16-Jun-14	3631.45	-163.66	-3795.12
17-Jun-14	2633.26	25721.39	23088.12
18-Jun-14	7221.11	56215.93	48994.82
19-Jun-14	3489.44	10547.90	7058.46
20-Jun-14	3657.75	7420.85	3763.10
21-Jun-14	3633.14	2144.48	-1488.67
22-Jun-14	3458.09	3464.79	6.70
23-Jun-14	3116.39	-1814.99	-4931.38
24-Jun-14	2918.17	10397.91	7479.74
25-Jun-14	2426.75	11231.25	8804.50
26-Jun-14	2042.88	0.00	-2042.88
27-Jun-14	21106.69	13554.92	-7551.77
28-Jun-14	24545.30	33445.46	8900.16
29-Jun-14	2398.24	-3149.19	-5547.43
30-Jun-14	1714.78	4143.82	2429.03
1-Jul-14	-367.79	2818.49	3186.28
2-Jul-14	-630.56	1989.86	2620.42
3-Jul-14	176.39	2156.00	1979.61
4-Jul-14	-846.07	-1326.81	-480.74
5-Jul-14	-1032.48	0.00	1032.48
6-Jul-14	-1402.93	-995.02	407.90
7-Jul-14	-1564.58	-4310.96	-2746.38
8-Jul-14	-2154.22	-1160.42	993.80
9-Jul-14	-2653.31	3647.35	6300.66
10-Jul-14	-2685.93	331.62	3017.55
11-Jul-14	-2910.11	-5636.55	-2726.44
12-Jul-14	-3142.86	-2983.15	159.71
13-Jul-14	-3282.75	-5964.42	-2681.66
14-Jul-14	-1284.94	-3643.70	-2358.75

Table H.1 continued

	Volume estimates (m^3)		
Date	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
15-Jul-14	-3675.45	-4470.55	-795.09
16-Jul-14	-3906.08	-3641.64	264.44
17-Jul-14	-4089.25	-5957.07	-1867.83
18-Jul-14	-4228.49	-12402.75	-8174.26
19-Jul-14	-4458.29	-11401.30	-6943.01
20-Jul-14	-4002.81	-2807.67	1195.14
21-Jul-14	-4631.73	-8914.99	-4283.27
22-Jul-14	5404.42	-3960.52	-9364.95
23-Jul-14	4768.15	-3464.61	-8232.76
24-Jul-14	-4873.58	-8740.49	-3866.91
25-Jul-14	-5034.65	-7087.67	-2053.03
26-Jul-14	-5262.96	-6754.83	-1491.87
27-Jul-14	-5474.04	-6421.90	-947.87
28-Jul-14	-5691.69	-4279.39	1412.30
29-Jul-14	-5789.03	-6416.28	-627.25
30-Jul-14	-5804.20	-5755.33	48.87
31-Jul-14	-5861.47	-3123.19	2738.28
1-Aug-14	-5296.70	-8379.39	-3082.69
2-Aug-14	-4601.08	-3448.68	1152.40
3-Aug-14	-5532.88	-2626.92	2905.96
4-Aug-14	-5609.29	-10502.10	-4892.81
5-Aug-14	-5749.44	-1640.15	4109.29
6-Aug-14	-5844.50	-7050.17	-1205.67
7-Aug-14	-5861.24	-9011.85	-3150.61
8-Aug-14	-5869.37	-13260.18	-7390.81
9-Aug-14	-5910.52	-8668.84	-2758.32
10-Aug-14	-6015.11	-8662.89	-2647.77
11-Aug-14	-6082.06	-8167.10	-2085.05
12-Aug-14	4748.31	2449.58	-2298.73
13-Aug-14	-4785.84	816.63	5602.47
14-Aug-14	12230.32	2286.85	-9943.47
15-Aug-14	18948.96	5065.20	-13883.76
16-Aug-14	-5218.97	3595.89	8814.86
17-Aug-14	-6103.53	-3595.89	2507.63
18-Aug-14	-6144.57	-5391.92	752.65
19-Aug-14	-5280.52	-4899.76	380.77
20-Aug-14	-4850.06	-5713.99	-863.93
21-Aug-14	-6230.91	-8321.52	-2090.60
22-Aug-14	-5385.27	-4729.43	655.84
23-Aug-14	-6225.27	-7335.31	-1110.04

Table H.1 continued

	Volume estimates (m^3)		
Date	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
24-Aug-14	-6223.47	-9448.22	-3224.76
25-Aug-14	-6224.27	-7651.24	-1426.97
26-Aug-14	-6265.02	-6833.45	-568.43
27-Aug-14	-6294.54	-3903.21	2391.33
28-Aug-14	-6302.10	-9265.42	-2963.32
29-Aug-14	-6311.23	-8122.10	-1810.87
30-Aug-14	-6292.38	-6980.93	-688.55
31-Aug-14	-6279.97	-11356.26	-5076.29
1-Sep-14	-3734.08	-10698.20	-6964.12
2-Sep-14	5382.45	-5183.82	-10566.27
3-Sep-14	20378.57	10531.82	-9846.75
4-Sep-14	-3653.21	-324.18	3329.03
5-Sep-14	-3299.98	-5023.81	-1723.83
6-Sep-14	-3712.08	-5669.70	-1957.61
7-Sep-14	-3701.26	-7447.83	-3746.57
8-Sep-14	-3689.32	-6796.44	-3107.13
9-Sep-14	-3687.30	-6469.48	-2782.19
10-Sep-14	-3713.16	-8728.67	-5015.51
11-Sep-14	-3788.67	-15180.30	-11391.63
12-Sep-14	-3420.50	-7260.86	-3840.35
13-Sep-14	-3822.16	-4838.31	-1016.14
14-Sep-14	-3810.90	-3224.53	586.37
15-Sep-14	-3828.84	-3868.37	-39.53
16-Sep-14	-3731.12	-3867.22	-136.09
17-Sep-14	-3700.85	-4027.12	-326.27
18-Sep-14	-3698.08	-2898.75	799.33
19-Sep-14	-3678.98	-4990.77	-1311.78
20-Sep-14	-3762.54	-4345.22	-582.68
21-Sep-14	-3806.45	-1126.30	2680.14
22-Sep-14	-3829.41	-3539.16	290.25
23-Sep-14	1938.51	5630.94	3692.43
24-Sep-14	-3816.93	-2252.67	1564.26
25-Sep-14	-3852.69	-9810.63	-5957.94
26-Sep-14	-994.00	-803.82	190.18
27-Sep-14	-3895.49	-7392.80	-3497.31
28-Sep-14	-3907.20	-4016.05	-108.85
29-Sep-14	-3938.24	-5941.54	-2003.30
30-Sep-14	7915.97	160.55	-7755.43
1-Oct-14	-827.15	-12036.08	-11208.93
2-Oct-14	-1645.09	-4971.91	-3326.82

Table H.1 continued

Date	Volume estimates (m^3)		
	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
3-Oct-14	-1669.37	-9137.32	-7467.95
4-Oct-14	-1664.52	-1762.66	-98.14
5-Oct-14	-1659.33	640.94	2300.27
6-Oct-14	-1677.22	-3364.62	-1687.41
7-Oct-14	-1689.32	320.41	2009.72
8-Oct-14	-1715.46	-3203.73	-1488.27
9-Oct-14	-1738.15	-1601.59	136.55
10-Oct-14	-1757.00	-6404.57	-4647.57
11-Oct-14	-85.97	-1120.50	-1034.54
12-Oct-14	-1741.90	-8001.03	-6259.13
13-Oct-14	-1756.35	-1599.67	156.69
14-Oct-14	-1504.00	2239.58	3743.58
15-Oct-14	12921.35	4640.26	-8281.10
16-Oct-14	-1405.87	3521.20	4927.08
17-Oct-14	-1024.05	-2881.05	-1857.00
18-Oct-14	-233.96	2400.83	2634.80
19-Oct-14	-1125.53	-2880.95	-1755.43
20-Oct-14	-1116.83	-2240.34	-1123.51
21-Oct-14	-1068.36	-480.03	588.33
22-Oct-14	4988.46	-4159.55	-9148.01
23-Oct-14	30835.48	34753.69	3918.20
24-Oct-14	-1047.79	7057.26	8105.05
25-Oct-14	-587.20	2246.23	2833.44
26-Oct-14	-976.71	-10906.92	-9930.22
27-Oct-14	656.03	0.00	-656.03
28-Oct-14	14206.95	36765.01	22558.05
29-Oct-14	2864.13	56068.43	53204.29
30-Oct-14	-839.83	-5650.09	-4810.25
31-Oct-14	-739.55	1291.23	2030.78
1-Nov-14	1843.20	1129.93	-713.26
2-Nov-14	-1044.65	-1775.56	-730.91
3-Nov-14	4687.82	-1775.32	-6463.14
4-Nov-14	22163.02	26332.01	4169.00
5-Nov-14	-914.19	11162.86	12077.05
6-Nov-14	2411.49	3561.20	1149.71
7-Nov-14	-374.48	2266.73	2641.21
8-Nov-14	-242.95	-6151.63	-5908.68
9-Nov-14	28213.02	18463.67	-9749.35
10-Nov-14	186.65	11512.64	11325.99
11-Nov-14	-430.25	-6973.67	-6543.42

Table H.1 continued

	Volume estimates (m^3)		
Date	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
12-Nov-14	-495.51	-12316.36	-11820.86
13-Nov-14	-482.30	-9067.71	-8585.42
14-Nov-14	-467.18	-3398.75	-2931.57
15-Nov-14	-464.41	-3397.86	-2933.45
16-Nov-14	-510.42	2103.33	2613.75
17-Nov-14	-256.67	-4368.08	-4111.41
18-Nov-14	-655.33	-3072.95	-2417.61
19-Nov-14	-755.85	0.00	755.85
20-Nov-14	1240.61	0.00	-1240.61
21-Nov-14	20167.28	0.00	-20167.28
22-Nov-14	2330.97	0.00	-2330.97
23-Nov-14	4999.85	0.00	-4999.85
24-Nov-14	18532.45	0.00	-18532.45
25-Nov-14	13690.18	0.00	-13690.18
26-Nov-14	4841.94	0.00	-4841.94
27-Nov-14	828.44	0.00	-828.44
28-Nov-14	14006.44	0.00	-14006.44
29-Nov-14	1930.66	0.00	-1930.66
30-Nov-14	603.62	0.00	-603.62
1-Dec-14	861.20	-182762.35	-183623.55
2-Dec-14	-19412.26	0.00	19412.26
3-Dec-14	-25495.87	0.00	25495.87
4-Dec-14	-26143.32	0.00	26143.32
5-Dec-14	-26828.64	0.00	26828.64
6-Dec-14	-17432.02	0.00	17432.02
7-Dec-14	-27737.42	0.00	27737.42
8-Dec-14	-28720.27	0.00	28720.27
9-Dec-14	-22665.75	0.00	22665.75
10-Dec-14	-27694.50	0.00	27694.50
11-Dec-14	-24363.80	0.00	24363.80
12-Dec-14	-5215.68	0.00	5215.68
13-Dec-14	-24101.57	0.00	24101.57
14-Dec-14	-29544.59	0.00	29544.59
15-Dec-14	-9103.53	0.00	9103.53
16-Dec-14	-3013.86	0.00	3013.86
17-Dec-14	-1943.95	12929.23	14873.18
18-Dec-14	140.89	-2874.16	-3015.05
19-Dec-14	19431.47	22049.99	2618.52
20-Dec-14	13734.97	-4156.94	-17891.92
21-Dec-14	6460.42	-2877.18	-9337.60

Table H.1 continued

	Volume estimates (m^3)		
Date	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
22-Dec-14	-1260.91	43860.06	45120.97
23-Dec-14	4928.64	25674.79	20746.15
24-Dec-14	39693.69	15428.25	-24265.44
25-Dec-14	12252.02	17861.92	5609.91
26-Dec-14	7924.33	24338.38	16414.04
27-Dec-14	17307.19	2258.86	-15048.32
28-Dec-14	2367.15	3227.63	860.48
29-Dec-14	-7941.32	-11938.21	-3996.90
30-Dec-14	-10401.43	-3385.86	7015.57
31-Dec-14	-11420.48	-5802.27	5618.21
1-Jan-15	-12193.12	-3866.73	8326.39
2-Jan-15	-10251.64	9024.18	19275.82
3-Jan-15	-11251.19	34058.55	45309.75
4-Jan-15	3268.77	17629.45	14360.68
5-Jan-15	36922.34	-11001.01	-47923.35
6-Jan-15	-18514.06	-2263.75	16250.31
7-Jan-15	-7036.61	-1293.39	5743.21
8-Jan-15	5528.58	8247.61	2719.03
9-Jan-15	4277.46	4853.99	576.53
10-Jan-15	1455.18	-5339.29	-6794.48
11-Jan-15	-4412.61	0.00	4412.61
12-Jan-15	-9769.89	-12126.56	-2356.67
13-Jan-15	-11737.74	-7755.03	3982.70
14-Jan-15	-10653.53	11634.30	22287.82
15-Jan-15	-12305.75	-6303.21	6002.54
16-Jan-15	3816.64	64794.65	60978.01
17-Jan-15	15848.49	36590.12	20741.64
18-Jan-15	36471.55	37837.68	1366.13
19-Jan-15	39432.13	19615.49	-19816.64
20-Jan-15	18946.11	5890.61	-13055.51
21-Jan-15	-1711.97	654.68	2366.65
22-Jan-15	-12239.71	8186.42	20426.13
23-Jan-15	-17877.10	21473.92	39351.02
24-Jan-15	98865.06	41743.08	-57121.98
25-Jan-15	11496.99	47492.56	35995.57
26-Jan-15	42374.04	36381.75	-5992.30
27-Jan-15	24286.46	10435.25	-13851.21
28-Jan-15	-2638.79	-7620.14	-4981.35
29-Jan-15	-24189.14	-16551.60	7637.54
30-Jan-15	-37953.44	-22645.26	15308.18

Table H.1 continued

Date	Volume estimates (m³)		
	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
31-Jan-15	-45015.23	-16177.64	28837.59
1-Feb-15	-42828.45	-18302.80	24525.65
2-Feb-15	-38920.80	-21898.52	17022.29
3-Feb-15	-41648.97	-62025.98	-20377.01
4-Feb-15	-40635.77	-26815.15	13820.62
5-Feb-15	-9055.48	52211.54	61267.02
6-Feb-15	24530.73	83762.05	59231.33
7-Feb-15	69793.09	91734.03	21940.93
8-Feb-15	68805.00	80526.84	11721.84
9-Feb-15	50329.18	85306.63	34977.44
10-Feb-15	42540.93	28340.29	-14200.64
11-Feb-15	-8636.08	-10972.79	-2336.71
12-Feb-15	-60662.39	-35720.12	24942.26
13-Feb-15	-89430.35	-45345.48	44084.86
14-Feb-15	-97305.02	-61207.96	36097.07
15-Feb-15	-94963.09	-37106.07	57857.02
16-Feb-15	-82562.49	166.17	82728.66
17-Feb-15	-77499.80	-10630.78	66869.02
18-Feb-15	-84931.43	-23060.81	61870.62
19-Feb-15	-85196.35	-49149.31	36047.04
20-Feb-15	-80708.77	-46186.99	34521.79
21-Feb-15	-69666.66	-36670.77	32995.89
22-Feb-15	-59579.74	-26735.35	32844.40
23-Feb-15	-52336.51	-27659.11	24677.40
24-Feb-15	-46574.58	-24498.80	22075.78
25-Feb-15	-41258.40	-22008.74	19249.66
26-Feb-15	-37091.76	-24735.28	12356.48
27-Feb-15	-32885.05	-23227.62	9657.43
28-Feb-15	-29232.89	-11841.39	17391.49
1-Mar-15	-25507.56	27430.16	52937.72
2-Mar-15	-23908.39	52751.29	76659.68
3-Mar-15	-21826.13	9955.86	31781.99
4-Mar-15	-9744.13	10290.48	20034.62
5-Mar-15	7933.04	16514.94	8581.90
6-Mar-15	6162.37	10148.64	3986.27
7-Mar-15	2740.51	13107.20	10366.69
8-Mar-15	-427.18	10987.91	11415.09
9-Mar-15	-3303.93	14939.51	18243.44
10-Mar-15	-5908.21	12655.29	18563.50
11-Mar-15	-9258.87	9047.50	18306.37

Table H.1 continued

Date	Volume estimates (m^3)		
	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
12-Mar-15	-11064.20	19763.13	30827.32
13-Mar-15	-14713.11	90095.39	104808.50
14-Mar-15	-707.22	96556.68	97263.90
15-Mar-15	70362.42	105638.74	35276.32
16-Mar-15	54546.95	126649.37	72102.42
17-Mar-15	69875.97	80476.93	10600.96
18-Mar-15	66346.12	58403.07	-7943.05
19-Mar-15	58911.80	40033.57	-18878.23
20-Mar-15	48728.32	9091.37	-39636.95
21-Mar-15	40199.61	21134.41	-19065.20
22-Mar-15	17413.44	26013.25	8599.81
23-Mar-15	56217.77	39764.60	-16453.17
24-Mar-15	68348.64	48468.92	-19879.72
25-Mar-15	76076.67	33331.56	-42745.12
26-Mar-15	78972.79	33463.64	-45509.15
27-Mar-15	70079.52	3511.82	-66567.70
28-Mar-15	87952.18	-1580.49	-89532.67
29-Mar-15	56878.96	-14386.35	-71265.31
30-Mar-15	42648.68	-14711.59	-57360.27
31-Mar-15	22071.89	-18527.47	-40599.37
1-Apr-15	625.57	-26325.32	-26950.89
2-Apr-15	-15022.61	-29893.78	-14871.17
3-Apr-15	-23757.63	-32217.20	-8459.57
4-Apr-15	-29087.42	-4321.76	24765.66
5-Apr-15	-32251.60	-7255.88	24995.72
6-Apr-15	7932.55	-18976.66	-26909.21
7-Apr-15	-26961.78	-14636.89	12324.89
8-Apr-15	-30882.02	-13410.72	17471.31
9-Apr-15	-29785.78	-13562.10	16223.68
10-Apr-15	-27811.11	-15596.99	12214.12
11-Apr-15	-26893.15	-8216.49	18676.66
12-Apr-15	-26822.27	8045.24	34867.52
13-Apr-15	-26369.43	-3253.18	23116.25
14-Apr-15	9451.74	-10094.72	-19546.47
15-Apr-15	-13718.32	-6154.32	7564.00
16-Apr-15	-15217.40	341.81	15559.20
17-Apr-15	-12923.44	-5125.81	7797.63
18-Apr-15	-12315.70	-3244.99	9070.72
19-Apr-15	-11455.22	-4950.96	6504.26
20-Apr-15	-10779.87	-12623.13	-1843.26

Table H.1 continued

	Volume estimates (m^3)		
Date	Predicted ΔVolume	GIS Δ Volume	Aquifer Estimate
21-Apr-15	-11355.89	-11075.89	280.00
22-Apr-15	-12526.39	-3746.17	8780.22
23-Apr-15	-12953.58	-11740.86	1212.72
24-Apr-15	-13310.11	-3570.69	9739.43
25-Apr-15	-13511.22	-7987.09	5524.12
26-Apr-15	-12080.79	-7301.96	4778.83
27-Apr-15	-13442.62	-7975.13	5467.50
28-Apr-15	-10218.59	9842.57	20061.16
29-Apr-15	-11480.73	-13573.46	-2092.73

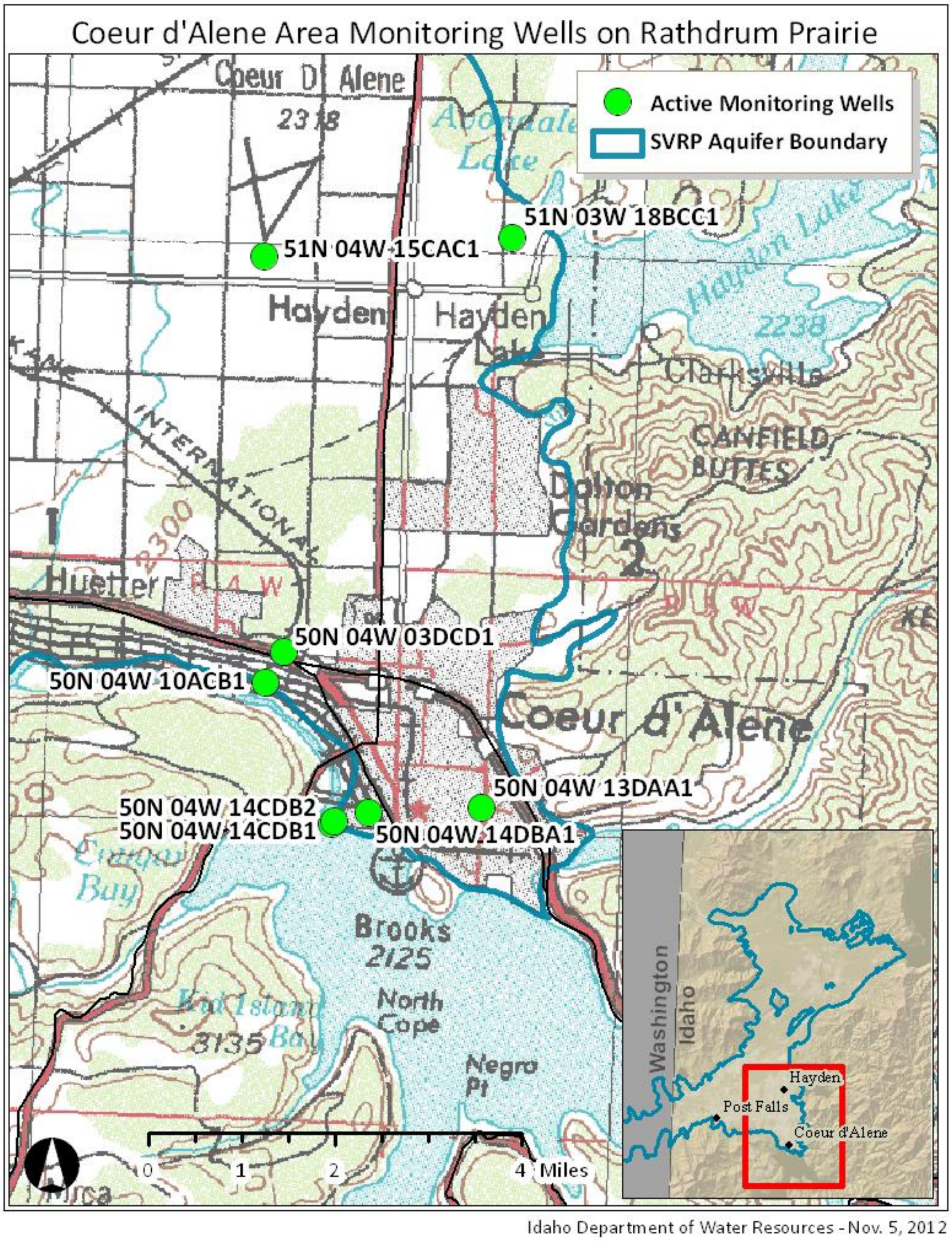

Figure H.1: Locations of monitoring wells in Coeur d' Alene for the Rathdrum Prarie Aquifer. The well closest to Fernan Lake is well# 50N 04W 13DAA1 located northwest of the lake. Image from the Idaho Department of Water Resources.

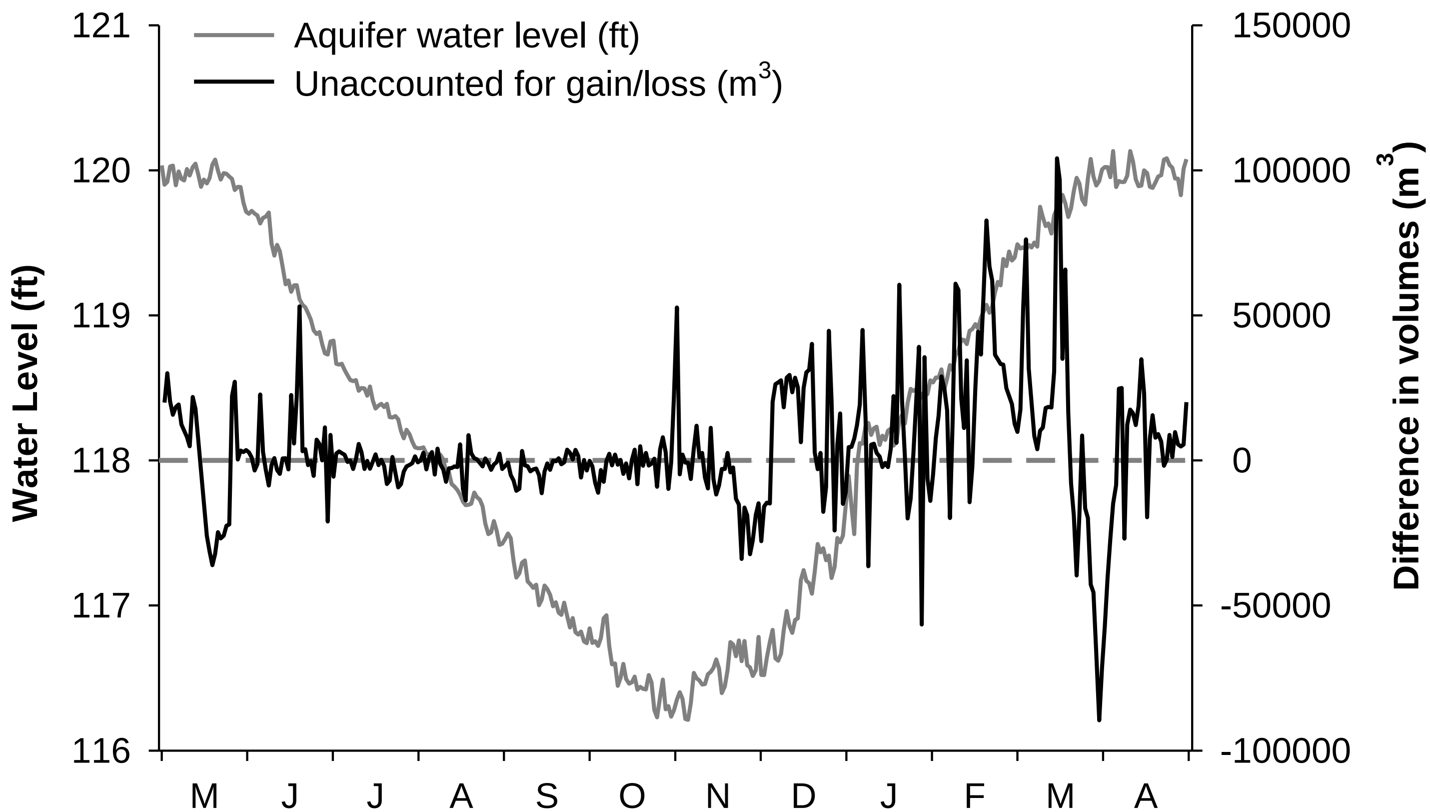

Figure H.2: Average water level at the Rathdrum Prairie aquifer monitoring well 50N 04W 13DAA1 plotted with the unaccounted for water volume resulting from the daily water balance calculations.

Appendix I: Calculation of road culvert loads

Calculation of loads from the culverts that run alongside the road were calculated based upon limited data. Due to the distance between Fernan Lake and the University of Idaho (1.5 hours) it was unrealistic to undertake event-based sampling for storm events that resulted in discharge from the culverts. I was able to measure discharge from culverts on five days. On these days I measured discharge with a 2L graduated cylinder and a stopwatch. Total phosphorus samples were collected in triplicate at each running culvert. To estimate the amount of water entering the lake from the culverts, I used the hydrograph from Fernan Creek and separated baseflow from the direct runoff using the Web Based Hydrograph Analysis tool (WHAT) (Purdue 2015). I then regressed the discharge values I measured on days when the culverts were running as a function of the direct runoff values from Fernan Creek. One culvert had a good relationship (R^2=0.65) (Figure I.1). This relationship was applied to the direct runoff values which were then multiplied by the average TP concentrations from the culverts on all the sampling days. Only three culverts consistently discharged water due to storm events, so these were the only ones to which I applied the direct runoff relationship to. This gave me the resulting load estimate of 1kg for the year.

Due to the lack of data, the above method yields very approximate estimate of the total load from the culverts. However, proportionately the road would still likely contribute a small fraction compared to the Fernan Creek load and would thus likely be negligible if accurately quantified. To fully quantify the road culvert contribution would require equipping all the culverts (Figure I.2) with event-based samplers and level loggers to trigger the sampler when runoff occurs. This would be a costly endeavor as there are approximately 22 culverts along the side of Fernan, most of which only discharge infrequently or during severe storms. Additionally, it would require personnel willing to go out and measure discharge from each culvert at any hour during severe weather.

References

Purdue University. 2015. Web Based Hydrograph Analysis Tool (WHAT); (cited 10 July 2015) Available from https://engineering.purdue.edu/~what/

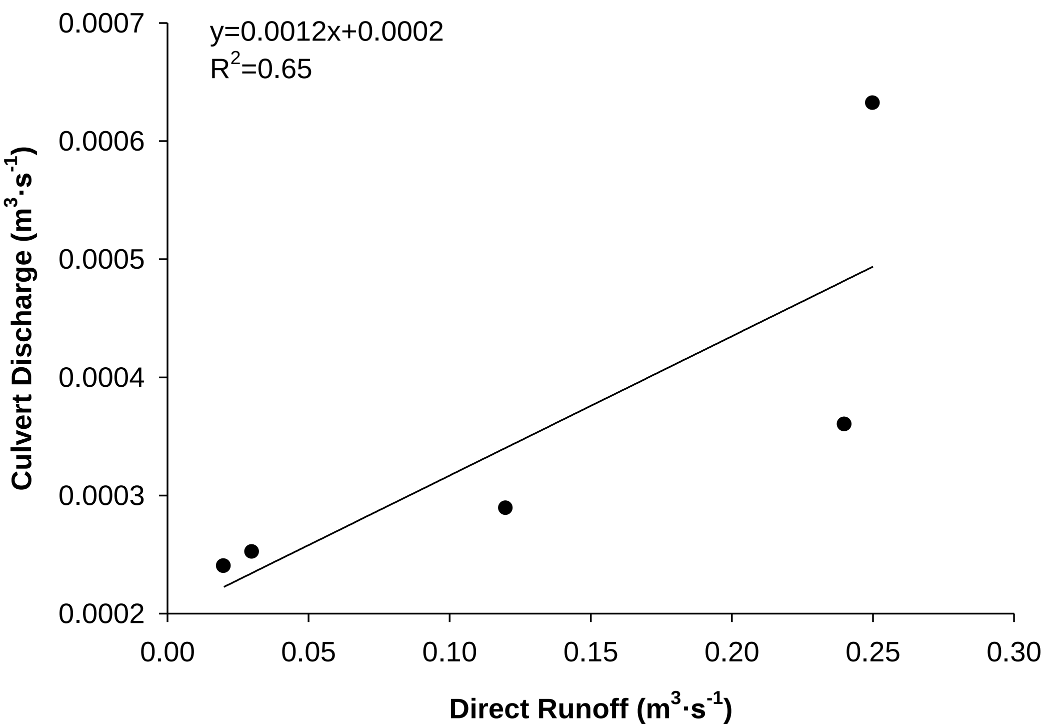

Figure I.1: Culvert discharge as a function of direct runoff from Fernan Creek.

Figure I.2: Aerial image of Fernan Lake showing the locations of the culverts that run underneath Fernan Lake Road.

Appendix J: Daily total phosphorus concentrations for the inflows and outflow of Fernan Lake

Table J.1: Total phosphorus concentrations (TP) (μg·L^{-1}) as measured for daily samples for all three sample locations at Fernan Lake during the 2014-2015 sample period. Blank stand errors (SE) signify interpolated (TP).

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
29-Apr-14	52.98	0.15	120.57	0.41	18.79	0.05
30-Apr-14	49.96	0.13	409.85	0.37	21.71	0.27
1-May-14	55.86	0.33	80.55	0.02	18.78	0.41
2-May-14	53.28	0.07	77.56	0.02	20.81	0.32
3-May-14	49.91	0.71	74.22	0.09	17.47	0.03
4-May-14	50.84	0.09	60.55	0.13	20.88	0.04
5-May-14	48.25	0.13	42.33	0.37	19.43	0.12
6-May-14	59.12	0.09	99.69	0.35	25.71	0.06
7-May-14	52.76	0.08	46.30	0.18	18.79	0.01
8-May-14	49.76	0.03	53.44	0.64	20.38	0.22
9-May-14	60.89	0.34	38.36	0.21	26.56	0.03
10-May-14	48.52	0.12	112.15	0.83	21.99	0.05
11-May-14	49.41	0.07	59.02	0.09	22.14	0.04
12-May-14	62.37	0.16	126.60	0.00	23.66	0.10
13-May-14	57.24	0.00	91.12	0.25	26.58	0.05
14-May-14	59.25	0.79	108.68	0.35	22.38	0.15
15-May-14	58.91	0.09	110.28	0.30	19.30	1.00
16-May-14	72.31	0.03	120.65	0.65	16.41	1.63
17-May-14	80.13	1.79	62.43	0.06	16.31	0.19
18-May-14	43.46	0.20	79.23	0.36	20.80	0.12
19-May-14	73.96	0.11	104.83	0.23	27.86	0.04
20-May-14	60.40	0.19	134.69	0.37	19.55	0.08
21-May-14	70.23	0.10	127.26	0.14	18.17	0.14
22-May-14	82.60	0.23	111.24	0.37	21.14	0.44
23-May-14	95.40	0.06	101.00	0.45	15.23	0.21
24-May-14	73.28	0.03	107.40	0.03	22.19	0.11
25-May-14	61.44	0.18	80.18	0.14	24.25	0.17
26-May-14	53.98	0.24	117.79	1.01	26.42	0.15
27-May-14	51.81	0.02	115.42	0.31	27.90	0.08
28-May-14	48.39	0.25	70.41	0.29	24.25	0.48
29-May-14	46.43	0.56	140.39	0.03	35.32	0.15
30-May-14	41.41	0.16	93.39	2.74	29.14	0.02
31-May-14	41.43	0.27	92.57	0.41	29.55	0.31
1-Jun-14	37.73	0.13	63.42	0.36	23.78	0.04
2-Jun-14	50.38	0.02	77.99	0.08	28.44	0.18
3-Jun-14	47.82	0.29	71.95	0.08	31.41	115.43
4-Jun-14	52.55	0.24	60.69	0.09	34.38	0.47

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
5-Jun-14	45.82	0.04	58.65	0.20	37.35	0.08
6-Jun-14	49.83	0.07	43.93	0.05	167.40	0.84
7-Jun-14	43.86	0.11	43.72	1.20	35.33	0.16
8-Jun-14	48.72	1.28	46.69	0.01	37.78	0.07
9-Jun-14	46.58	0.14	54.73	0.05	35.15	1.17
10-Jun-14	52.21	0.02	42.25	0.28	26.75	0.31
11-Jun-14	49.40	0.03	64.56	0.13	44.43	2.14
12-Jun-14	128.96	0.55	38.02	0.16	92.70	0.28
13-Jun-14	31.24	0.24	45.54	0.18	259.08	1.12
14-Jun-14	38.81	0.17	54.99	0.63	240.44	1.46
15-Jun-14	37.49	0.22	44.35	0.18	28.65	0.04
16-Jun-14	36.67	0.05	42.07	0.47	87.47	0.74
17-Jun-14	57.01	0.15	61.45	0.02	23.46	0.08
18-Jun-14	63.34	0.02	60.27	0.34	32.77	0.48
19-Jun-14	55.77	1.02	42.90	0.02	22.44	0.45
20-Jun-14	51.87		42.29	0.41	31.27	0.33
21-Jun-14	47.96		46.25	0.16	22.01	2.33
22-Jun-14	44.06		45.23	0.21	70.70	1.05
23-Jun-14	40.15		44.37	0.05	185.85	2.10
24-Jun-14	36.25	0.08	75.68	1.39	226.73	0.98
25-Jun-14	32.92	0.05	77.80	0.32	201.21	2.71
26-Jun-14	39.99	0.06	64.19	0.01	95.28	0.38
27-Jun-14	39.63	0.17	87.93	0.58	61.66	0.88
28-Jun-14	40.89	0.04	71.65	0.04	27.19	0.11
29-Jun-14	37.33	0.08	57.34	0.36	20.57	0.16
30-Jun-14	34.79	0.27	66.73	0.08	37.38	0.07
1-Jul-14	36.37	0.20	62.52	0.06	22.21	0.06
2-Jul-14	36.49	0.04	51.48	0.30	433.69	0.61
3-Jul-14	31.88	0.24	622.37	0.28	318.06	0.27
4-Jul-14	31.39	0.18	195.79	0.50	131.97	0.65
5-Jul-14	31.13	0.35	689.68	1.08	97.69	0.11
6-Jul-14	35.95	0.01	271.20	0.75	91.05	0.15
7-Jul-14	36.29	0.01	233.27	1.22	76.87	2.08
8-Jul-14	32.86	0.03	79.32	0.13	151.69	0.11
9-Jul-14	29.22	0.22	27.98	0.01	173.82	0.32
10-Jul-14	33.68	0.00	28.61	0.37	141.59	0.05
11-Jul-14	34.02	0.37	27.14	0.22	308.27	0.15
12-Jul-14	37.46	0.10	26.91	0.30	317.50	0.20
13-Jul-14	34.66	0.00	28.94	0.23	497.64	0.90
14-Jul-14	36.78	0.38	28.89	0.24	777.70	0.52

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
15-Jul-14	36.97	0.15	28.90	0.02	238.97	0.52
16-Jul-14	38.06	0.06	27.67	0.11	155.50	0.73
17-Jul-14	39.99	0.08	26.17	0.15	748.81	1.69
18-Jul-14	40.23	0.07	22.52	0.45	746.84	1.20
19-Jul-14	30.20	0.13	25.02	0.16	458.93	1.95
20-Jul-14	34.50		26.31	0.29	194.25	1.40
21-Jul-14	38.81		26.93	0.16	98.00	2.61
22-Jul-14	43.11	0.04	19.29	0.07	82.14	0.06
23-Jul-14	42.47	0.17	23.70	0.19	80.10	0.09
24-Jul-14	39.39	0.11	55.84	0.16	47.89	0.04
25-Jul-14	40.47	0.13	32.82	0.23	37.78	0.05
26-Jul-14	45.27	0.02	30.65	0.05	45.94	0.09
27-Jul-14	41.98	0.05	33.82	0.09	32.18	0.33
28-Jul-14	81.81	0.01	30.39	0.03	35.93	0.14
29-Jul-14	71.66	0.01	34.95	0.29	41.29	0.08
30-Jul-14	76.03	0.19	44.71	0.43	46.52	0.02
31-Jul-14	259.72	0.74	29.18	0.09	62.11	0.16
1-Aug-14	99.12	0.08	83.55	0.32	53.49	0.05
2-Aug-14	--		39.04	0.05	53.56	0.16
3-Aug-14	--		38.86	0.21	115.12	0.02
4-Aug-14	--		47.45	0.12	126.76	0.59
5-Aug-14	--		100.51	0.30	51.67	0.43
6-Aug-14	--		88.32	0.16	47.06	0.05
7-Aug-14	--		72.26	0.09	39.35	1.04
8-Aug-14	--		43.52	0.19	38.89	0.17
9-Aug-14	--		149.92	5.53	31.70	0.06
10-Aug-14	--		48.08	0.23	25.33	0.43
11-Aug-14	--		79.06	0.20	46.11	0.63
12-Aug-14	--		43.04	0.13	38.72	0.11
13-Aug-14	--		110.50	0.05	44.28	0.05
14-Aug-14	--		49.53	0.55	41.52	0.24
15-Aug-14	--		49.53	0.05	42.45	
16-Aug-14	--		60.64	0.10	43.37	
17-Aug-14	--		45.08	0.14	44.30	
18-Aug-14	--		36.39	0.08	45.22	
19-Aug-14	--		38.36	0.22	46.15	
20-Aug-14	--		32.01	0.23	47.07	1.03
21-Aug-14	--		30.62	0.40	41.33	0.02
22-Aug-14	--		39.41	0.63	54.66	0.13
23-Aug-14	--		32.56	0.01	87.01	0.02

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
24-Aug-14	--		33.45	0.06	71.60	0.36
25-Aug-14	--		50.76	0.09	65.12	0.10
26-Aug-14	--		37.23	0.11	60.67	1.70
27-Aug-14	--		33.85	0.56	92.17	0.59
28-Aug-14	--		35.97	0.06	75.57	1.82
29-Aug-14	--		28.14	0.55	70.76	0.03
30-Aug-14	--		29.97	0.97	93.42	0.30
31-Aug-14	--		33.37	0.01	96.09	0.08
1-Sep-14	--		33.39	0.16	120.72	0.57
2-Sep-14	--		37.92	0.08	69.11	0.10
3-Sep-14	--		35.40	0.57	73.21	0.03
4-Sep-14	--		31.23	0.17	76.48	0.36
5-Sep-14	--		30.11	0.02	87.95	0.25
6-Sep-14	--		38.84	0.05	77.31	0.14
7-Sep-14	--		31.31	0.01	70.15	0.09
8-Sep-14	--		32.02	0.03	53.23	0.13
9-Sep-14	--		32.30	0.16	74.26	0.01
10-Sep-14	--		44.17	0.14	107.61	0.08
11-Sep-14	--		33.28	0.22	139.21	0.10
12-Sep-14	--		31.26	0.02	60.39	0.18
13-Sep-14	--		36.34	0.34	75.85	0.19
14-Sep-14	--		31.07	0.28	75.26	6.27
15-Sep-14	--		34.40	0.11	75.85	0.13
16-Sep-14	--		34.97	0.10	77.42	0.07
17-Sep-14	--		37.00	0.04	125.19	0.49
18-Sep-14	--		28.48	0.10	66.27	0.19
19-Sep-14	--		26.98	0.28	70.14	0.06
20-Sep-14	--		33.77	0.05	90.70	0.44
21-Sep-14	--		43.00	0.17	106.09	0.14
22-Sep-14	--		29.47	0.34	74.20	0.23
23-Sep-14	--		30.78	0.50	82.06	0.15
24-Sep-14	--		30.62	0.02	69.61	0.55
25-Sep-14	--		31.43	0.12	66.94	0.30
26-Sep-14	--		31.67	0.04	75.20	0.13
27-Sep-14	--		32.88	0.01	131.41	0.23
28-Sep-14	--		36.14	0.05	185.93	0.17
29-Sep-14	--		30.96	0.06	72.29	0.05
30-Sep-14	--		30.02	0.13	59.95	0.05
1-Oct-14	--		27.29	0.41	63.55	0.09
2-Oct-14	--		31.94	0.08	61.15	0.05

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
3-Oct-14	--		28.88	0.11	50.55	0.05
4-Oct-14	--		28.83	0.02	56.61	0.18
5-Oct-14	--		29.40	0.06	55.73	0.21
6-Oct-14	--		29.33	0.36	48.96	0.25
7-Oct-14	--		30.59	0.23	58.26	0.41
8-Oct-14	--		30.31	0.13	50.51	0.29
9-Oct-14	--		28.48	0.09	56.26	0.09
10-Oct-14	--		35.23	0.23	51.51	0.05
11-Oct-14	--		44.89	0.00	64.47	0.16
12-Oct-14	--		36.36	0.32	68.05	0.03
13-Oct-14	--		34.92	0.04	52.05	0.11
14-Oct-14	--		43.89	0.06	57.54	0.01
15-Oct-14	--		31.19	0.02	57.95	0.73
16-Oct-14	--		29.67	1.39	49.31	0.00
17-Oct-14	--		37.54	0.27	58.91	0.13
18-Oct-14	--		25.58	0.16	58.82	0.09
19-Oct-14	--		30.78	0.14	59.85	0.13
20-Oct-14	--		33.38	0.40	48.34	0.02
21-Oct-14	--		30.71	0.17	48.08	0.45
22-Oct-14	--		29.69	0.03	45.28	0.06
23-Oct-14	--		46.08	0.27	42.98	0.30
24-Oct-14	--		29.05	0.32	44.24	0.45
25-Oct-14	--		31.49	0.03	47.78	0.15
26-Oct-14	--		42.68	0.31	55.09	0.04
27-Oct-14	--		30.60	0.20	50.46	0.17
28-Oct-14	--		64.68	0.28	276.49	0.39
29-Oct-14	--		104.37	0.21	327.47	0.81
30-Oct-14	--		79.45	0.42	152.81	6.43
31-Oct-14	--		117.59	0.59	83.39	0.14
1-Nov-14	--		181.18	0.31	71.70	0.20
2-Nov-14	--		76.08	0.41	71.52	0.27
3-Nov-14	--		122.55	0.24	71.31	0.09
4-Nov-14	--		162.43	0.43	66.27	0.55
5-Nov-14	--		120.65	0.00	56.80	0.16
6-Nov-14	--		81.23	0.52	81.72	0.04
7-Nov-14	--		113.19	0.57	88.83	0.07
8-Nov-14	--		125.47	0.27	110.56	0.19
9-Nov-14	--		139.15	0.29	128.31	0.22
10-Nov-14	--		119.25	0.26	123.47	
11-Nov-14	--		114.64	0.05	119.04	

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
12-Nov-14	--		61.84	0.24	114.98	
13-Nov-14	--		60.48	0.52	111.25	
14-Nov-14	--		72.70	0.28	107.84	
15-Nov-14	--		96.13	1.74	104.71	
16-Nov-14	--		40.21	0.15	101.84	
17-Nov-14	--		32.02	2.49	99.21	
18-Nov-14	--		53.88	0.65	96.80	
19-Nov-14	--		58.62	0.15	94.59	
20-Nov-14	--		70.44	0.10	92.56	
21-Nov-14	--		84.38	0.19	70.28	0.47
22-Nov-14	--		57.73	0.26	107.90	0.09
23-Nov-14	--		72.62	0.30	173.85	0.16
24-Nov-14	--		47.64	1.17	91.88	0.23
25-Nov-14	--		56.71	0.24	65.82	0.02
26-Nov-14	--		58.48	0.26	101.18	0.30
27-Nov-14	--		54.41	0.02	85.39	0.27
28-Nov-14	--		54.49	0.09	62.63	0.16
29-Nov-14	115.98	0.02	53.80	0.19	48.34	0.07
30-Nov-14	60.34	0.00	54.62	0.15	48.35	
1-Dec-14	49.14	0.28	68.66	0.03	48.36	
2-Dec-14	30.40	0.17	65.68	0.49	48.37	
3-Dec-14	27.10	0.02	50.97	1.92	48.39	
4-Dec-14	27.95	0.18	38.44	0.18	48.40	
5-Dec-14	27.84	0.02	38.64	0.38	48.41	
6-Dec-14	35.69	0.47	48.45	0.55	48.42	0.36
7-Dec-14	35.14	0.04	58.06	0.45	33.16	0.02
8-Dec-14	35.84	0.07	75.38	0.23	29.13	0.00
9-Dec-14	35.03	0.00	45.73	0.84	30.28	0.13
10-Dec-14	39.72	0.22	48.07	0.08	42.25	0.25
11-Dec-14	39.38	0.53	61.89	0.25	38.78	0.11
12-Dec-14	44.45	0.06	64.30	0.02	38.59	0.13
13-Dec-14	52.78	0.18	71.55	0.13	47.12	0.27
14-Dec-14	34.87	0.20	56.23	0.46	34.86	0.27
15-Dec-14	32.25	0.03	57.99	0.05	50.05	0.25
16-Dec-14	85.69	0.31	28.22	0.03	15.56	0.47
17-Dec-14	103.31	0.05	34.78	0.06	16.24	0.35
18-Dec-14	209.25	0.25	45.38	0.14	20.21	0.32
19-Dec-14	164.08	0.92	59.87	0.10	29.64	0.20
20-Dec-14	165.62	0.47	46.73	0.15	24.24	0.54
21-Dec-14	204.85	0.19	47.57	0.15	28.41	0.10

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
22-Dec-14	92.50	0.05	56.08	0.07	24.11	0.15
23-Dec-14	81.76	0.38	45.76	0.17	31.39	0.25
24-Dec-14	143.71	1.13	90.38	0.03	29.38	0.05
25-Dec-14	148.85	0.26	84.45	0.18	25.50	0.08
26-Dec-14	217.25	0.30	53.70	0.38	23.17	
27-Dec-14	119.32	0.06	40.94	0.16	20.84	
28-Dec-14	82.74	0.10	35.84	0.00	18.51	0.29
29-Dec-14	55.96	0.04	39.57		20.26	
30-Dec-14	53.93		43.29		22.01	
31-Dec-14	51.90		47.02		23.76	
1-Jan-15	49.87		50.74		25.51	
2-Jan-15	47.84		54.47		27.25	
3-Jan-15	45.80		58.19		29.00	
4-Jan-15	43.77		61.92		30.75	
5-Jan-15	41.74		65.64	0.21	32.50	
6-Jan-15	39.71	0.15	92.51	0.35	34.25	0.18
7-Jan-15	51.46	0.13	44.27	0.81	29.10	0.04
8-Jan-15	59.29	0.08	36.15	0.11	36.70	1.45
9-Jan-15	46.67	0.11	40.09	0.23	36.88	
10-Jan-15	43.45	0.08	47.64		37.07	
11-Jan-15	37.47	0.19	55.18	0.16	37.25	
12-Jan-15	35.72	0.27	61.49	21.36	37.44	
13-Jan-15	45.56	0.05	29.86	0.12	37.62	0.27
14-Jan-15	41.87	0.28	52.50	1.38	41.01	
15-Jan-15	42.18	0.04	39.27	0.01	44.41	
16-Jan-15	43.13	0.27	49.62	0.09	47.80	0.38
17-Jan-15	35.24	0.12	34.98	0.00	30.90	0.12
18-Jan-15	72.08	0.00	52.19	0.21	49.13	0.57
19-Jan-15	67.35	0.00	63.86	0.13	39.23	0.19
20-Jan-15	64.95	0.18	44.38	0.56	34.16	0.13
21-Jan-15	64.92	0.82	47.24	0.11	36.26	0.14
22-Jan-15	49.86	0.21	55.14	0.22	27.81	0.06
23-Jan-15	51.96	2.05	36.48	0.48	38.56	0.42
24-Jan-15	59.78	0.48	53.50	0.13	29.39	0.41
25-Jan-15	91.99	0.45	68.95	0.23	32.27	0.36
26-Jan-15	80.21	0.25	96.74	0.22	32.30	0.23
27-Jan-15	69.69	0.05	146.06	0.15	38.50	0.36
28-Jan-15	66.06	0.00	75.17	1.32	31.32	0.08
29-Jan-15	55.76	0.82	76.58	0.04	26.43	0.01
30-Jan-15	53.50	0.21	43.88	0.10	29.47	0.08

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
31-Jan-15	52.91	0.02	54.63	0.31	30.01	0.11
1-Feb-15	50.11	0.64	61.10	0.68	28.05	0.19
2-Feb-15	50.07	0.78	45.31	0.08	25.83	0.17
3-Feb-15	89.28	35.62	72.52	0.64	27.17	0.08
4-Feb-15	51.77	0.05	50.68	0.21	30.33	0.66
5-Feb-15	47.26	0.09	51.35	0.42	29.55	0.46
6-Feb-15	65.84	0.02	66.45	0.19	29.97	0.13
7-Feb-15	139.12	0.20	64.08	0.20	41.22	0.00
8-Feb-15	83.80	0.41	43.72	0.28	32.46	0.27
9-Feb-15	68.08	0.20	41.49	0.10	37.85	0.29
10-Feb-15	81.99	0.19	64.14	0.13	33.22	0.13
11-Feb-15	70.93	0.47	56.75	0.21	32.39	
12-Feb-15	60.40	0.26	125.94	0.12	31.57	
13-Feb-15	71.39	0.12	97.78	0.02	30.74	
14-Feb-15	68.21	0.31	99.38	0.10	29.92	
15-Feb-15	76.14	0.08	104.40	0.41	29.09	
16-Feb-15	72.52	0.08	45.51	0.02	28.27	
17-Feb-15	71.08	0.08	57.45	0.31	27.44	
18-Feb-15	63.71	0.35	58.89	0.15	26.62	
19-Feb-15	48.88	0.13	116.06	0.48	25.79	0.14
20-Feb-15	77.83	0.21	39.91	0.17	26.62	0.18
21-Feb-15	52.71	0.07	56.99	0.22	28.79	0.24
22-Feb-15	73.64	0.12	47.45	0.20	26.49	0.08
23-Feb-15	58.90	0.11	48.98	0.21	26.64	0.48
24-Feb-15	70.25	0.38	60.44	0.41	26.28	0.02
25-Feb-15	69.61	0.12	61.82	0.16	30.39	1.08
26-Feb-15	99.35	0.06	47.10	0.22	29.05	0.06
27-Feb-15	76.42	0.05	43.83	0.77	30.08	0.19
28-Feb-15	76.81	0.04	38.49	0.17	27.43	0.28
1-Mar-15	79.64	0.04	39.56	0.15	30.49	0.00
2-Mar-15	45.25	0.10	27.27	2.50	26.58	0.33
3-Mar-15	111.70	0.00	54.24	1.16	31.11	0.19
4-Mar-15	62.73	0.34	49.18	0.11	32.15	0.32
5-Mar-15	57.75	0.11	446.04	0.96	35.06	0.31
6-Mar-15	84.07	0.27	51.41	0.31	36.45	0.07
7-Mar-15	80.41	0.46	46.66	0.16	31.40	0.02
8-Mar-15	79.41		74.74	0.05	33.71	0.13
9-Mar-15	78.40		73.14	0.17	36.82	0.07
10-Mar-15	77.40	0.25	98.57	0.38	32.49	0.09
11-Mar-15	58.05	0.37	48.10	0.63	25.91	0.08

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
12-Mar-15	60.43		63.61	0.12	35.07	0.29
13-Mar-15	62.81	0.13	56.45	0.09	31.50	0.29
14-Mar-15	74.83	0.12	61.31	0.77	35.07	0.16
15-Mar-15	107.30	0.11	123.11	0.05	51.26	0.09
16-Mar-15	124.70	0.01	194.93	0.80	64.93	0.31
17-Mar-15	106.02	0.16	99.12	0.09	79.64	0.38
18-Mar-15	99.12	0.26	110.69	0.11	71.28	0.09
19-Mar-15	68.65	0.25	239.21	1.71	43.71	0.13
20-Mar-15	78.86	0.10	367.11	0.05	26.86	0.25
21-Mar-15	101.21	0.00	104.68	0.05	28.33	0.37
22-Mar-15	76.94	0.41	106.25	0.23	29.18	0.83
23-Mar-15	66.42	0.10	64.02	0.13	32.08	0.08
24-Mar-15	82.06	0.19	76.29	0.23	32.04	0.04
25-Mar-15	72.20	0.09	100.69	0.09	39.36	0.14
26-Mar-15	77.93	0.18	114.80	0.26	37.52	
27-Mar-15	75.36	0.14	109.85	0.31	35.69	
28-Mar-15	66.89	0.12	346.64	0.62	33.85	
29-Mar-15	70.28	0.20	261.74	0.83	32.01	
30-Mar-15	66.04	0.07	345.08	1.09	30.17	
31-Mar-15	65.68	0.06	306.92	0.24	28.34	
1-Apr-15	64.90	0.41	132.19	0.38	26.50	
2-Apr-15	68.13	0.07	438.67	4.51	24.66	0.06
3-Apr-15	79.62	0.17	278.86	1.35	24.65	0.06
4-Apr-15	73.69	0.04	338.71	0.43	24.37	0.04
5-Apr-15	76.94	0.05	514.83	0.68	24.40	0.02
6-Apr-15	91.17	0.09	467.20	0.49	37.28	0.07
7-Apr-15	78.73	0.06	351.04	0.29	21.83	0.13
8-Apr-15	67.50	0.20	472.02	1.41	21.26	2.17
9-Apr-15	74.49	0.31	314.94	0.35	24.41	0.36
10-Apr-15	72.31	0.05	419.91	2.63	27.34	0.09
11-Apr-15	79.78	0.47	324.69	1.31	25.68	0.72
12-Apr-15	79.25	0.16	304.37	0.78	31.01	0.28
13-Apr-15	76.09	0.08	214.51	1.16	28.31	0.18
14-Apr-15	91.51	0.39	321.47	0.25	29.62	0.06
15-Apr-15	95.17	0.12	218.25	0.47	29.54	0.31
16-Apr-15	77.29	0.19	246.30	0.27	41.42	0.20
17-Apr-15	80.76	0.08	250.02	2.05	60.51	0.32
18-Apr-15	103.72	0.13	213.60	0.23	22.34	0.03
19-Apr-15	87.28	0.06	340.04	0.84	22.90	0.51
20-Apr-15	71.27	0.50	362.66	0.83	22.75	0.04

Table J.1 continued

Date	North Side (TP)	+/- SE	Mike Webb (TP)	+/- SE	Outflow (TP)	+/- SE
21-Apr-15	83.42	0.40	410.26	1.30	46.04	0.03
22-Apr-15	96.11	0.20	324.60	0.27	30.10	0.18
23-Apr-15	86.88	0.73	332.02	2.68	30.58	
24-Apr-15	110.03	0.05	466.39	0.19	31.06	
25-Apr-15	79.92	0.29	398.73	0.78	31.54	
26-Apr-15	109.58	0.38	387.10	0.93	32.02	
27-Apr-15	89.18	0.16	328.28	0.03	32.49	
28-Apr-15	96.08	0.09	466.84	0.98	32.97	
29-Apr-15	67.96	0.12	359.16	0.13	33.45	

Appendix K: Daily total residue concentrations for the two inflows and outflow of Fernan Lake

Table K.1: Total residue concentrations (TR) (mg·L^{-1}) as measured for daily samples for all three sample locations at Fernan Lake during the 2014- 2015 sampling period. Samples designated as "--"represent days where the location was dry.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
29-Apr-14	45.07	705.88	50.75
30-Apr-14	108.33	247.83	63.83
1-May-14	110.48	248.66	52.63
2-May-14	68.87	231.77	89.29
3-May-14	19.94	231.17	93.37
4-May-14	34.48	140.16	103.03
5-May-14	86.11	94.85	65.57
6-May-14	88.83	140.96	94.51
7-May-14	109.55	84.93	104.29
8-May-14	84.51	133.15	85.11
9-May-14	56.82	81.79	114.46
10-May-14	80.78	132.98	71.01
11-May-14	83.10	136.13	57.06
12-May-14	86.96	337.02	158.68
13-May-14	105.57	256.91	155.22
14-May-14	113.04	187.83	121.21
15-May-14	93.48	139.34	124.61
16-May-14	84.06	192.21	185.76
17-May-14	162.32	133.33	136.51
18-May-14	103.06	196.89	148.04
19-May-14	149.30	210.39	134.73
20-May-14	123.89	234.67	62.69
21-May-14	127.66	272.49	89.82
22-May-14	308.41	189.47	73.96
23-May-14	545.45	168.87	94.67
24-May-14	211.54	185.48	80.36
25-May-14	182.20	151.52	81.25
26-May-14	150.77	136.13	80.23
27-May-14	160.49	275.77	71.23
28-May-14	158.50	110.82	45.16
29-May-14	137.84	279.68	65.16
30-May-14	134.62	276.92	45.71
31-May-14	113.26	207.69	62.86
1-Jun-14	96.97	204.66	54.91
2-Jun-14	146.03	161.73	40.35
3-Jun-14	109.51	169.27	779.82
4-Jun-14	114.22	145.21	350.43

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
5-Jun-14	154.97	188.98	61.58
6-Jun-14	151.52	143.96	110.14
7-Jun-14	128.57	136.60	89.60
8-Jun-14	121.05	141.75	8.80
9-Jun-14	43.90	155.00	48.43
10-Jun-14	243.24	201.61	128.03
11-Jun-14	197.01	211.89	132.95
12-Jun-14	180.64	172.13	221.26
13-Jun-14	164.27	93.26	334.34
14-Jun-14	179.71	77.52	247.73
15-Jun-14	153.61	109.29	260.45
16-Jun-14	246.91	51.08	68.77
17-Jun-14	216.87	163.49	70.80
18-Jun-14	243.90	121.13	79.03
19-Jun-14	177.71	106.33	147.15
20-Jun-14	158.52	119.05	124.63
21-Jun-14	139.33	186.67	128.74
22-Jun-14	120.14	85.33	129.41
23-Jun-14	100.95	123.99	541.67
24-Jun-14	81.76	108.57	250.00
25-Jun-14	75.76	151.43	219.02
26-Jun-14	82.84	139.28	171.17
27-Jun-14	107.25	147.14	166.67
28-Jun-14	97.77	111.72	37.04
29-Jun-14	111.75	103.64	31.06
30-Jun-14	48.57	79.55	57.23
1-Jul-14	47.62	84.75	56.05
2-Jul-14	53.73	83.58	391.81
3-Jul-14	46.78	550.43	320.70
4-Jul-14	100.00	594.67	207.60
5-Jul-14	95.10	1756.52	117.99
6-Jul-14	135.38	396.60	345.93
7-Jul-14	113.70	351.43	633.93
8-Jul-14	106.99	118.13	98.84
9-Jul-14	100.28	92.59	147.69
10-Jul-14	93.57	64.61	147.83
11-Jul-14	54.20	75.27	363.08
12-Jul-14	14.84	51.35	378.95
13-Jul-14	83.60	71.05	753.01
14-Jul-14	78.49	48.52	1149.25

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
15-Jul-14	67.96	59.90	284.90
16-Jul-14	79.37	61.97	280.35
17-Jul-14	112.36	82.35	171.92
18-Jul-14	320.00	81.01	301.72
19-Jul-14	253.77	85.49	304.35
20-Jul-14	187.54	88.57	71.84
21-Jul-14	121.30	106.74	75.36
22-Jul-14	55.07	151.19	39.33
23-Jul-14	64.31	152.63	48.30
24-Jul-14	45.05	148.25	42.62
25-Jul-14	51.87	135.20	42.25
26-Jul-14	56.05	109.04	50.30
27-Jul-14	61.22	129.20	58.48
28-Jul-14	87.09	115.18	38.01
29-Jul-14	382.35	166.23	45.63
30-Jul-14	98.55	151.26	53.25
31-Jul-14	439.44	155.56	60.87
1-Aug-14	207.10	276.06	59.25
2-Aug-14	--	123.04	57.64
3-Aug-14	--	124.68	56.02
4-Aug-14	--	178.67	51.72
5-Aug-14	--	155.56	101.12
6-Aug-14	--	147.45	120.48
7-Aug-14	--	95.24	131.43
8-Aug-14	--	102.49	126.80
9-Aug-14	--	209.94	149.43
10-Aug-14	--	95.74	136.51
11-Aug-14	--	187.67	99.72
12-Aug-14	--	131.72	104.82
13-Aug-14	--	194.99	112.39
14-Aug-14	--	55.56	108.64
15-Aug-14	--	121.13	93.75
16-Aug-14	--	108.47	78.87
17-Aug-14	--	120.22	63.99
18-Aug-14	--	102.78	49.10
19-Aug-14	--	132.39	34.22
20-Aug-14	--	19.39	19.34
21-Aug-14	--	5.28	44.32
22-Aug-14	--	20.25	27.93
23-Aug-14	--	53.05	19.89

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
24-Aug-14	--	26.32	31.43
25-Aug-14	--	21.18	27.86
26-Aug-14	--	16.04	19.61
27-Aug-14	--	32.09	62.15
28-Aug-14	--	12.95	46.11
29-Aug-14	--	18.92	52.02
30-Aug-14	--	24.00	26.87
31-Aug-14	--	21.92	34.58
1-Sep-14	--	2.62	67.04
2-Sep-14	--	153.85	28.25
3-Sep-14	--	128.14	131.15
4-Sep-14	--	108.31	118.98
5-Sep-14	--	139.41	65.16
6-Sep-14	--	115.59	92.26
7-Sep-14	--	131.65	96.69
8-Sep-14	--	85.33	121.47
9-Sep-14	--	116.53	106.02
10-Sep-14	--	77.14	98.59
11-Sep-14	--	69.06	151.43
12-Sep-14	--	81.79	100.00
13-Sep-14	--	69.59	89.19
14-Sep-14	--	68.24	106.44
15-Sep-14	--	86.49	111.11
16-Sep-14	--	114.36	101.12
17-Sep-14	--	90.67	98.27
18-Sep-14	--	68.42	60.00
19-Sep-14	--	78.95	74.50
20-Sep-14	--	70.10	69.16
21-Sep-14	--	53.94	134.29
22-Sep-14	--	69.44	34.09
23-Sep-14	--	65.04	40.58
24-Sep-14	--	63.58	97.92
25-Sep-14	--	50.00	95.24
26-Sep-14	--	52.34	136.36
27-Sep-14	--	77.54	171.83
28-Sep-14	--	74.47	144.44
29-Sep-14	--	56.34	134.50
30-Sep-14	--	46.58	117.61
1-Oct-14	--	50.13	100.71
2-Oct-14	--	83.33	83.82

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
3-Oct-14	--	26.74	55.88
4-Oct-14	--	46.45	52.79
5-Oct-14	--	44.82	61.76
6-Oct-14	--	42.13	68.97
7-Oct-14	--	62.86	72.83
8-Oct-14	--	54.64	56.72
9-Oct-14	--	42.25	67.99
10-Oct-14	--	42.67	37.36
11-Oct-14	--	52.92	44.94
12-Oct-14	--	67.75	40.11
13-Oct-14	--	63.89	42.74
14-Oct-14	--	25.97	20.06
15-Oct-14	--	10.36	29.59
16-Oct-14	--	15.63	28.99
17-Oct-14	--	13.09	39.39
18-Oct-14	--	2.72	68.45
19-Oct-14	--	2.69	51.58
20-Oct-14	--	21.80	71.43
21-Oct-14	--	17.19	44.12
22-Oct-14	--	15.02	43.86
23-Oct-14	--	90.91	64.02
24-Oct-14	--	31.34	50.60
25-Oct-14	--	46.07	49.42
26-Oct-14	--	13.40	53.57
27-Oct-14	--	44.74	59.52
28-Oct-14	--	26.20	21.36
29-Oct-14	--	7.67	2.11
30-Oct-14	--	5.01	16.48
31-Oct-14	--	16.71	11.83
1-Nov-14	--	14.66	23.88
2-Nov-14	--	12.62	6.01
3-Nov-14	--	10.58	15.24
4-Nov-14	--	51.00	8.88
5-Nov-14	--	35.05	3.98
6-Nov-14	--	44.28	8.81
7-Nov-14	--	53.51	33.33
8-Nov-14	--	29.14	31.01
9-Nov-14	--	4.76	59.26
10-Nov-14	--	7.11	49.79
11-Nov-14	--	7.25	35.40

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
12-Nov-14	--	8.65	54.15
13-Nov-14	--	15.38	72.90
14-Nov-14	--	22.12	91.65
15-Nov-14	--	31.75	110.40
16-Nov-14	--	14.34	129.16
17-Nov-14	--	40.82	147.91
18-Nov-14	--	102.34	166.66
19-Nov-14	--	123.49	185.41
20-Nov-14	--	169.59	204.16
21-Nov-14	--	146.63	222.91
22-Nov-14	--	132.74	241.67
23-Nov-14	--	171.17	157.36
24-Nov-14	--	160.71	128.57
25-Nov-14	--	174.42	116.16
26-Nov-14	--	171.01	91.27
27-Nov-14	--	164.18	92.68
28-Nov-14	--	207.21	138.12
29-Nov-14	96.05	130.43	126.21
30-Nov-14	79.77	175.10	99.53
1-Dec-14	74.59	124.26	90.56
2-Dec-14	57.47	132.72	81.59
3-Dec-14	77.81	129.79	72.63
4-Dec-14	56.38	57.47	63.66
5-Dec-14	68.77	73.10	54.70
6-Dec-14	80.43	107.95	45.73
7-Dec-14	64.14	101.98	36.76
8-Dec-14	73.53	118.16	27.78
9-Dec-14	71.43	105.26	6.85
10-Dec-14	49.84	147.93	20.98
11-Dec-14	62.50	127.54	29.63
12-Dec-14	64.33	175.60	22.99
13-Dec-14	54.44	111.76	17.34
14-Dec-14	48.99	111.44	16.22
15-Dec-14	12.12	114.11	20.00
16-Dec-14	67.18	39.66	39.68
17-Dec-14	12.53	44.64	70.31
18-Dec-14	26.90	43.96	81.17
19-Dec-14	41.26	55.70	119.12
20-Dec-14	24.69	53.37	92.65
21-Dec-14	64.20	65.71	99.68

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
22-Dec-14	28.06	57.74	102.89
23-Dec-14	34.78	64.33	107.78
24-Dec-14	28.80	65.04	93.55
25-Dec-14	35.04	44.38	90.03
26-Dec-14	31.01	35.81	93.02
27-Dec-14	19.61	53.05	94.56
28-Dec-14	26.18	76.09	96.09
29-Dec-14	40.10	78.35	97.63
30-Dec-14	47.27	80.61	99.85
31-Dec-14	54.44	82.87	102.07
1-Jan-15	61.61	85.13	104.30
2-Jan-15	68.78	87.40	106.52
3-Jan-15	82.20	89.66	108.75
4-Jan-15	95.62	91.92	110.97
5-Jan-15	109.03	94.18	113.20
6-Jan-15	122.45	64.50	115.42
7-Jan-15	132.02	34.81	117.65
8-Jan-15	155.31	65.09	96.15
9-Jan-15	104.82	95.38	59.94
10-Jan-15	153.85	104.24	71.17
11-Jan-15	131.58	113.10	82.40
12-Jan-15	27.10	101.65	93.64
13-Jan-15	47.89	27.25	104.87
14-Jan-15	102.27	53.41	116.10
15-Jan-15	75.00	73.95	103.30
16-Jan-15	114.21	67.65	90.50
17-Jan-15	86.35	58.82	77.70
18-Jan-15	177.33	113.89	50.60
19-Jan-15	147.14	200.00	41.94
20-Jan-15	120.45	295.26	31.65
21-Jan-15	104.68	141.33	6.54
22-Jan-15	150.27	138.46	15.43
23-Jan-15	186.73	101.30	116.42
24-Jan-15	142.86	96.20	110.12
25-Jan-15	271.23	131.23	83.33
26-Jan-15	273.65	147.87	91.69
27-Jan-15	505.15	265.66	72.05
28-Jan-15	181.01	151.90	146.20
29-Jan-15	204.48	207.65	156.52
30-Jan-15	156.98	110.50	128.28

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
31-Jan-15	164.18	83.33	152.05
1-Feb-15	142.01	144.39	134.90
2-Feb-15	203.13	105.69	107.14
3-Feb-15	238.81	131.65	95.81
4-Feb-15	118.50	147.37	56.55
5-Feb-15	106.38	109.63	72.67
6-Feb-15	105.88	72.19	69.85
7-Feb-15	164.79	97.11	104.29
8-Feb-15	120.97	475.68	87.84
9-Feb-15	147.68	195.12	86.21
10-Feb-15	124.03	228.95	76.70
11-Feb-15	144.88	146.28	95.98
12-Feb-15	134.02	288.52	93.33
13-Feb-15	96.77	215.47	90.69
14-Feb-15	90.53	155.74	88.05
15-Feb-15	126.70	129.83	85.41
16-Feb-15	151.39	120.69	82.76
17-Feb-15	130.84	103.45	80.12
18-Feb-15	163.27	111.43	77.48
19-Feb-15	151.01	320.86	74.84
20-Feb-15	120.48	103.72	72.19
21-Feb-15	139.61	154.47	71.81
22-Feb-15	143.44	90.66	73.17
23-Feb-15	133.33	113.40	96.15
24-Feb-15	99.53	102.43	90.21
25-Feb-15	165.52	127.73	78.17
26-Feb-15	117.02	103.26	99.69
27-Feb-15	141.73	127.84	92.39
28-Feb-15	116.07	104.23	99.43
1-Mar-15	407.41	94.74	107.04
2-Mar-15	222.22	121.62	65.79
3-Mar-15	270.83	114.97	75.68
4-Mar-15	236.36	151.76	64.17
5-Mar-15	226.41	445.10	81.30
6-Mar-15	216.45	135.95	109.14
7-Mar-15	206.49	112.43	95.24
8-Mar-15	196.54	164.18	68.05
9-Mar-15	186.58	116.07	84.51
10-Mar-15	176.62	123.92	85.29
11-Mar-15	166.67	136.36	67.23

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
12-Mar-15	125.97	148.15	70.18
13-Mar-15	85.27	148.69	60.69
14-Mar-15	137.78	137.14	83.57
15-Mar-15	121.21	204.48	78.55
16-Mar-15	157.30	290.50	83.33
17-Mar-15	133.57	204.66	83.10
18-Mar-15	130.29	187.34	124.65
19-Mar-15	102.34	333.33	100.00
20-Mar-15	104.35	304.35	80.36
21-Mar-15	121.74	164.86	32.74
22-Mar-15	143.27	129.48	31.06
23-Mar-15	123.60	68.29	63.40
24-Mar-15	125.00	117.33	47.90
25-Mar-15	115.09	151.67	54.38
26-Mar-15	128.46	151.86	47.34
27-Mar-15	126.68	138.64	57.56
28-Mar-15	118.60	468.49	67.78
29-Mar-15	111.41	299.09	78.01
30-Mar-15	109.63	431.37	88.23
31-Mar-15	109.55	358.60	98.45
1-Apr-15	105.56	445.05	108.67
2-Apr-15	130.06	391.85	118.90
3-Apr-15	173.79	365.50	129.12
4-Apr-15	152.65	489.80	112.39
5-Apr-15	131.25	541.42	154.08
6-Apr-15	138.73	556.47	95.65
7-Apr-15	120.45	497.02	150.60
8-Apr-15	144.12	500.00	118.16
9-Apr-15	116.81	414.63	120.34
10-Apr-15	113.37	520.20	117.99
11-Apr-15	122.70	430.27	115.94
12-Apr-15	105.42	336.28	114.37
13-Apr-15	141.18	278.79	110.78
14-Apr-15	107.14	389.68	96.26
15-Apr-15	119.50	304.60	107.87
16-Apr-15	185.07	311.80	104.65
17-Apr-15	118.97	322.13	112.04
18-Apr-15	144.33	260.74	115.38
19-Apr-15	99.04	393.39	104.05
20-Apr-15	94.28	378.45	88.89

Table K.1 continued.

Date	North Side (TP)	Mike Webb (TP)	Outflow (TP)
21-Apr-15	108.84	428.98	107.73
22-Apr-15	183.33	319.65	199.41
23-Apr-15	118.47	337.54	156.05
24-Apr-15	98.64	414.63	90.12
25-Apr-15	145.91	334.32	
26-Apr-15	140.07	411.08	
27-Apr-15	108.02	252.34	
28-Apr-15	127.66	416.18	
29-Apr-15	103.77	326.15	

Appendix L: Bi-weekly total and dissolved phosphorus concentrations for the two inflows and outflow at Fernan Lake

Table L.1: Total phosphorus concentrations [TP] ($\mu g \cdot L^{-1}$) and total dissolved phosphorous [TD] ($\mu g \cdot L^{-1}$) with associated standard errors (SE) as measured for bi-weekly grab samples for the two inflows and one outflow location at Fernan Lake. Days with a "-- "represent dates where the location was dry (NS), or could not be sampled due to ice (OF).

Date	NS (TP)	+/- SE	NS (DP)	+/- SE	MW (TP)	+/- SE	MW (DP)	+/- SE	OF (TP)	+/- SE	OF (DP)	+/- SE
12-May-14	47.69	0.13	27.73	0.39	25.96	0.15	21.57	0.14	18.45	0.66	11.97	0.38
27-May-14	45.46	0.03	37.79	0.11	46.15	0.14	7.18	0.42	27.51	0.06	12.12	1.12
10-Jun-14	40.45	0.12	7.11	0.02	56.56	0.20	18.37	0.20	30.10	0.09	18.18	1.90
24-Jun-14	35.18	0.11	6.58	0.55	52.84	0.45	20.81	0.15	58.66	0.52	13.62	0.19
8-Jul-14	39.94	0.18	18.73	0.06	39.99	0.07	23.02	0.07	38.32	0.48	13.88	0.03
22-Jul-14	42.19	0.12	30.20	0.13	39.05	0.39	28.61	0.07	58.32	0.30	22.61	0.07
5-Aug-14	63.84	0.00	33.48	0.14	44.42	0.07	26.61	0.01	45.14	0.09	18.92	0.11
20-Aug-14	--	--	--	--	41.57	0.39	23.08	0.09	39.60	0.11	19.64	0.52
2-Sep-14	--	--	--	--	39.11	0.00	23.18	0.45	56.48	0.08	27.57	0.04
16-Sep-14	--	--	--	--	26.94	0.16	10.81	0.48	203.39	0.06	25.90	0.12
30-Sep-14	--	--	--	--	32.39	0.05	25.49	0.09	72.34	0.17	23.86	0.02
14-Oct-14	--	--	--	--	34.61	0.20	20.56	0.22	68.24	0.18	59.13	0.15
28-Oct-14	--	--	--	--	43.89	0.07	25.58	0.12	54.63	0.10	20.32	0.03
18-Nov-14	46.31	0.17	31.63	0.09	39.94	0.27	29.52	0.13	--	--	--	--
2-Dec-14	35.63	0.00	30.41	0.09	46.88	0.09	23.53	0.18	35.63	0.00	30.41	0.09
16-Dec-14	33.37	0.16	27.79	0.04	50.24	0.73	25.92	0.04	34.93	0.11	14.45	0.11
6-Jan-15	54.93	0.11	38.42	0.70	49.58	0.27	28.64	0.12	38.17	0.09	17.82	0.04
22-Jan-15	70.05	0.19	42.29	0.12	78.21	0.21	28.05	0.03	30.15	0.18	17.89	0.22
5-Feb-15	48.00	0.05	31.65	0.01	43.61	0.10	21.02	0.24	27.39	0.06	15.34	0.04
19-Feb-15	62.02	0.08	29.84	0.02	64.99	0.14	20.71	0.01	34.77	0.31	12.90	0.19
5-Mar-15	61.66	0.31	25.21	0.06	43.05	0.06	13.14	0.08	40.79	0.04	17.36	0.01
19-Mar-15	77.03	0.20	47.93	0.17	73.24	0.11	26.57	0.21	51.50	0.28	27.66	0.34
2-Apr-15	59.74	0.25	32.71	0.17	187.53	0.44	196.70	0.30	29.91	0.22	16.43	0.15
16-Apr-15	67.87	0.20	33.92	0.19	189.16	0.48	15.26	0.12	29.99	0.01	11.18	0.16

Table L.1 continued.

Date	NS (TP)	+/- SE	NS (DP)	+/- SE	MW (TP)	+/- SE	MW (DP)	+/- SE	OF (TP)	+/- SE	OF (DP)	+/- SE
30-Apr-15	59.36	0.18	30.33	0.03	309.25	2.17	43.87	0.13	33.93	0.40	13.39	0.01

Appendix M: Bi-weekly nitrogen data

The Idaho Department of Environmental Quality provided funding for the analysis of total nitrogen, ammonia and nitrate/nitrite during the summer of 2014. Samples were taken from 10 June 2014 to 18 November 2014. Samples from the inflow (Table M.1), the outflow (Table M.2) and the lake (Table M.3) were taken very two weeks and submitted to SVL Analytical Laboratories in Coeur d'Alene where they were analyzed using method 353.2 for nitrate and nitrite, method 350.1 for ammonia, and ASTMD-5176 for total nitrogen (USEPA). Total nitrogen values were compared to total phosphorus values to get Bi-weekly TN:TP ratios of the lake.

References

USEPA. 1996a. Environmental indicators of water quality in the United States. EPA 841-R- 96-002. USEPA, Office of Water (4503F), U.S. Gov. Print. Office, Washington, DC.

Table M.1: Bi-weekly total nitrogen, ammonia and nitrate/nitrite values for the MW location during the sampling period 10-June 2014 to 18 November 2014.

Date	Total Nitrogen ($mg \cdot L^{-1}$)	Ammonia ($mg \cdot L^{-1}$)	Nitrate/Nitrite ($mg \cdot L^{-1}$)
10-Jun-14	0.325	<0.030	<0.050
24-Jun-14	0.176	0.04	0.051
8-Jul-14	0.196	0.081	<0.050
22-Jul-14	0.282	<0.030	<0.050
5-Aug-14	0.235	<0.030	<0.050
20-Aug-14	0.193	<0.030	<0.050
2-Sep-14	0.168	0.042	0.068
16-Sep-14	0.151	0.041	0.084
30-Sep-14	0.145	<0.030	0.089
14-Oct-14	0.152	<0.030	0.102
28-Oct-14	0.197	0.031	0.119
18-Nov-14	0.175	<0.030	0.084

Table M.2: Bi-weekly total nitrogen, ammonia and nitrate/nitrite values for the OF location during the sampling period 10-June 2014 to 18 November 2014.

Date	Total Nitrogen ($mg \cdot L^{-1}$)	Ammonia ($mg \cdot L^{-1}$)	Nitrate/Nitrite ($mg \cdot L^{-1}$)
10-Jun-14	0.392	<0.030	<0.050
24-Jun-14	0.386	<0.030	<0.050
8-Jul-14	0.404	0.044	<0.050
22-Jul-14	0.458	<0.030	<0.050
5-Aug-14	0.378	<0.030	<0.050
20-Aug-14	0.528	0.031	<0.050
2-Sep-14	0.559	0.042	<0.050
16-Sep-14	2.37	0.059	<0.050
30-Sep-14	0.428	0.037	<0.050
14-Oct-14	0.404	<0.030	<0.050
28-Oct-14	0.555	0.053	<0.050
18-Nov-14	0.631	0.094	0.084

Table M.3: Bi-weekly total nitrogen, ammonia and nitrate/nitrite values for Fernan Lake during the sampling period 10-June 2014 to 18 November 2014. Note the last sampling date is 28 October 2014 due to the lake ice.

Date	Total Nitrogen ($mg \cdot L^{-1}$)	Ammonia ($mg \cdot L^{-1}$)	Nitrate/Nitrite ($mg \cdot L^{-1}$)
10-Jun-14	0.223	<0.030	<0.050
24-Jun-14	0.267	<0.030	<0.050
8-Jul-14	0.498	0.051	<0.050
22-Jul-14	0.252	0.141	<0.050
5-Aug-14	0.302	<0.030	<0.050
20-Aug-14	0.38	<0.030	<0.050
2-Sep-14	0.36	0.046	<0.050
16-Sep-14	0.302	<0.030	<0.050
30-Sep-14	0.277	<0.030	<0.050
14-Oct-14	0.491	0.046	<0.050
28-Oct-14	0.324	0.043	<0.050