

Monitoring and Modeling Total Phosphorus Contributions to a Freshwater Lake with Cage-Aquaculture

by

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ABSTRACT

MONITORING AND MODELING TOTAL PHOSPHORUS CONTRIBUTIONS TO A FRESHWATER LAKE WITH CAGE-AQUACULTURE

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A mass-balance modeling approach has been applied to gain an improved understanding of the relative contributions of phosphorus loading from various sources into a freshwater lake with cage-aquaculture in Ontario. All cage-aquaculture in Ontario is located within Lake Huron and Georgian Bay of the Great Lakes where concerns about potential environmental issues are constraining growth of the industry. Phosphorus is of particular concern for cage-aquaculture operations in Ontario since this nutrient may accelerate eutrophication of freshwater. Sound scientific information is needed for the Ministry of Environment lake managers to make defensible decisions to move the industry forward in a sustainable manner. Lake Wolsey is located on Manitoulin Island in Lake Huron, Ontario. The lake is connected to the North Channel by a small inlet where water exchanges periodically. A cage-aquaculture operation was established in 1986 and has an average annual production of approximately 250 metric tonnes of rainbow trout. We have estimated total phosphorus loadings from eight sources of inputs and three sinks from the lake. We then applied a sensitivity analysis to establish parameters that require empirical measurement and field validation. Results show non-point sources as the leading contributor of total phosphorus to Lake Wolsey (40%, 1120 kg) followed by the farm

(32%, 915 kg), groundwater (11%, 305 kg), and dwellings (8%, 219 kg), internal phosphorus load from the hypoxic hypolimnion mid-lake (7%, 186 kg), precipitation (3%, 79 kg) and leaf litter (0.3%, 8 kg). Results of the sensitivity analysis show non point sources to be the most significant parameter in terms of total phosphorus loading, followed by the lake sedimentation, then the contribution by the aquaculture operation. This is followed by the exchange via the inlet, groundwater, dwellings, the internal phosphorus, sportfishing, precipitation and leaf litter. Information from this project will provide water quality managers with scientific information to aid in decisions pertaining to policy and regulatory approaches for water quality risk assessment and management of cage-aquaculture in Ontario.

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Table of Contents

Acknowledgments	iv
List of Figures.....	viii
List of Tables	xi
List of Equations	xii
List of Appendices.....	xiii
List of Abbreviations	xiv
1.0 Introduction.....	1
1.1 What is aquaculture?.....	1
1.2 The importance of aquaculture	1
1.3 Environmental Concerns.....	2
1.3.1 Sediment and Water Quality.....	3
1.3.2 Current Sediment Quality Risk Management.....	6
1.3.3 Current Water Quality Risk Management.....	7
2.0 Literature Review	11
2.1 Modeling the Impacts of Aquaculture	11
2.1.1 Freshwater – Water Quality Modeling.....	11
2.1.1.1 Vollenweider (OECD) Model (1968, 1969, 1973).....	13
2.1.1.2 Dillon and Rigler (1974, 1975)	14
2.1.1.3 LEEDS (1998; 2008).....	19
2.1.1.5 Reid and Moccia 2007.....	20
2.1.1.4 Hamblin and Gale 2002.....	21
2.2 Objectives of the Study.....	22
3.0 Methods.....	24
3.1 Description of Study Lake.....	24
3.2 Description of fish farm.....	24

3.3 Modeling methodology	25
3.3.1 The Mass-Balance Model	25
3.3.2 Inlet Model (<i>inex</i> / <i>outex</i>)	27
3.3.3 Validation of Inlet Exchange Modeling.....	30
3.3.4 Farm Loads - <i>infarm</i> (Biomass net gain, feed conversion ratio, solid and dissolved waste phosphorus).....	31
3.4 Parameter Methodology	33
3.4.1 Non-Point sources (<i>inNP</i>).....	34
3.4.2 Precipitation (<i>inatm</i>)	35
3.4.3 Lake Evaporation.....	35
3.4.4 Evapotranspiration.....	36
3.4.5 Groundwater (<i>ingw</i>)	37
3.4.6 Dwellings (<i>indwell</i>).....	40
3.4.7 Internal P load (<i>inint</i>)	40
3.4.8 Leaf Litter (<i>inleaf</i>)	41
3.4.9 Lake sedimentation flux (<i>outsed</i>)	41
3.4.10 Sportfishing (<i>outsport</i>)	42
3.5 Observed Total Phosphorus Data.....	43
3.6 Uncertainty Analysis.....	45
3.7 Sensitivity Analysis	45
4.0 Results	47
4.1 Inlet/outlet exchange TP load (<i>inex</i>, <i>outex</i>).....	47
4.2 Validation of Inlet Hydrodynamic Model.....	48
4.3 Farm Load (<i>infarm</i> SWP/DWP/AWFP)	49
4.4 Tributaries: Observed Flows and Loads	50
4.5 Waterbalance: Non point source loads (<i>inNP</i>).....	51
4.6 Atmospheric Load (<i>inatm</i>)	52
4.7 Groundwater (<i>ingw</i>).....	52
4.8 Lake Sedimentation (<i>outsed</i>).....	52
4.9 Other minor TP contributions: Dwellings/Leaf Litter/Internal P/Sportfishing Load (<i>indwell</i>, <i>inleaf</i>, <i>inint</i>, <i>outsport</i>)	53

4.10 Contributions of TP loads to Lake Wolsey	53
4.11 Predicted Lake TP concentration and Uncertainty Analysis	53
4.12 Sensitivity Analysis	54
5.0 Discussion.....	56
5.1 Sensitive parameters.....	58
5.1.1 TP loading from Non point sources (<i>inNP</i>)	58
5.1.2. TP loads/sink in Inlet/outlet exchange (<i>inex/outex</i>).....	59
5.1.3 TP sink - Lake Sedimentation (<i>outsed</i>)	62
5.2. Non sensitive parameters	63
5.2.1 Farm Solid Waste Phosphorus (SWP)/ Farm Dissolved Waste Phosphorus (<i>DWP</i>)	63
5.2.2 TP loads in groundwater (<i>ingw</i>).....	65
5.2.3 TP loads in Atmospheric deposition (<i>inatm</i>).....	67
5.2.4 TP sinks in Sport fishing (<i>outsport</i>)	68
5.2.5 TP loads in Leaf Litter (<i>inleaf</i>).....	68
5.2.6 TP load from Internal Phosphorus (<i>inint</i>).....	69
5.2.7 TP load from Dwellings (<i>indwell</i>)	71
6.0 Strength and Weaknesses of the Mass-Balance Model	71
7.0 Recommendations for Model Improvement.....	73
8.0 Summary and Recommendations.....	75

List of Figures

Figure 1 A) Location of Lake Wolsey on Manitoulin Island, North Channel, Northern Lake Huron, Ontario, Canada B) causeway with small canal joining Lake Wolsey with Campbell Bay, North Channel.	85
Figure 2 The cage array of seasonal and permanent cages with location (star) of sediment trap in ice free season.	86
Figure 3 Conceptual schematic of the phosphorus mass-balance model with TP mass. ...	87
Figure 4 Acoustic Doppler Current Profiler mounted in a catamaran	88
Figure 5 Location of tributary TP samples and flows collected in 2009 and 2010.	89
Figure 6 Images showing the design of seepage meter (A) and seepage meter deployment (B).	90
Figure 7 Position of peizometers on the south shore of Lake Wolsey.	91
Figure 8 Image illustrates abundance of watercress (<i>Nasturtium officinale</i>) located at seepage areas on the east shore of Lake Wolsey	92
Figure 9 Location of stations monitored by Environment Canada with location of the fish farm.	93
Figure 10 TP concentration at 1m depth winter, spring, summer, and fall 2001 to 2010 measured in Lake Wolsey. Kendall Tau $p=0.22$	94
Figure 11 Estimated velocities entering (-ve) and exiting Lake Wolsey (+ve) May to November 2007.	95
Figure 12 Estimated daily velocities and wind gusts. Tidal fluctuations are shown by the bracket.	96

Figure 13 Wind speed and direction August 12 to August 17, 2011 Lake Wolsey.....	97
Figure 14 Wind speed and direction during the study the period, Lake Wolsey.....	98
Figure 15 Estimated inlet flows for May 2007 (Kendall Tau $p \leq 0.001$). Flows entering Lake Wolsey are +ve, flows exiting Lake Wolsey are -ve.	99
Figure 16 Frequency of hourly TP loads entering (-ve) and exiting (+ve) Lake Wolsey.	100
Figure 17 Validation of inlet hydrodynamic model. Observed (ADCP) vs estimated flows. ($r^2 = 0.79$).	101
Figure 18 Total monthly feed usage from May to November 2007. Total feed used May to November 2007 = 354820 kg.	102
Figure 19 Monthly dissolved and solid waste phosphorus (SWP +AFWP) (kg) released farm May to November 2007.....	103
Figure 20 Illustrates sheet flow during a precipitation event adjacent to Lake Wolsey.	104
Figure 21 Atmospheric monthly TP loads (kg) and precipitation (mm) May to November 2007.....	105
Figure 22 Daily estimated and observed non point sources of TP loads (kg). $r^2 = 0.49$, $p=0.3$	106
Figure 23 Groundwater discharge adjacent to the shoreline of Lake Wolsey	107
Figure 24 Total load (kg) and percent contribution of each source of TP to Lake Wolsey May to November, 2007.	108
Figure 25 Predicted and observed monthly TP lake concentration May to November, 2007 (NS = Nash-Sutcliffe). Thin solid line = uncertainty +/- 2 standard deviations. Dashed line = min and max of uncertainty analysis.	109

Figure 26 Histogram showing uncertainty range of the mean lake TP concentration (ug/L). The two vertical solid lines represent 2 standard deviations.	110
Figure 27 Lake TP concentration 1m June, July and August 2001 to 2010 (Kendall Tau $p=0.09$).	111
Figure 28 Shows the results of a sensitivity analysis where each parameter was increased 100% while the remaining parameter stayed constant. The error bars represent 2 standard deviations of uncertainty while the solid horizontal bar represents the estimated mean lake TP concentrations before application of sensitivity analysis (10.76 ug/L).	112
Figure 29 Shows the results of a sensitivity analysis where each parameter was removed (0%) while the remaining parameters stayed constant. The error bars represent 2 standard deviations of uncertainty while the solid horizontal bar represents the estimated mean lake TP concentrations before application of sensitivity analysis (10.76 ug/L).	113

List of Tables

Table 1 Lake and watershed dimensions.	114
Table 2 Biomass net gain of fish, May to November 2007.	115
Table 3 Tributaries, Seepage areas, peizometer locations and date sampled (✓).....	116
Table 4 Source of TP concentration data for calibrating mass-balance model on monthly time step. Int = integrated sample collection	117
Table 5 Stations monitored by Environment Canada for TP analyses.	118
Table 6 Bioenergetics phosphorus mass-balance: Estimated solid waste phosphorus (SWP) apparent feed wastage phosphorus (AWFP), dissolved waste phosphorus (DWP), biomass net gain phosphorus (BP) (kg). Percent P in feed = 0.95.....	119
Table 7 Monthly total phosphorus (TP) sources and sinks (kg) 2007.	120
Table 8 Mean lake TP concentration (ug/L) as a result of increasing and decreasing each model parameter by 100%. The estimated mean lake TP concentration before increase or decrease was 10.76 ug/L.	121

List of Equations

Equation 1	Mass–balance model from Chapra (1997).....	25
Equation 2	Phosphorus mass-balance model with identified sources and sinks.....	26
Equation 3	One dimensional inlet hydrodynamic model.....	28
Equation 4	Chezy or friction coefficient.....	29
Equation 5	Linear equation to determine farmed fish carcass phosphorus as a function of weight (Bureau, 2003).....	32
Equation 6	Bioenergetic model to estimate solid waste phosphorus from farmed fish (Bureau, 2003).....	33
Equation 7	Bioenergetic model to estimate dissolved waste phosphorus from farmed fish (Bureau, 2003).....	33
Equation 8	Bioenergetic phosphorus mass-balance.....	33
Equation 9	Water balance model.....	34
Equation 10	Water balance was used to estimate watershed runoff and non point phosphorus loads.....	34
Equation 11	Estimation of Potential Evapotranspiration.....	36
Equation 12	Estimation of internal phosphorus load.....	40
Equation 13	Estimation of lake and farm sedimentation.....	42
Equation 14	Estimation of phosphorus in perch (<i>Perca flavescens</i>).....	43

List of Appendices

Appendix 1 MatLab [®] V6.1 code to run the one dimensional inlet hydrodynamic model to estimate velocity in the inlet connecting Lake Wolsey with the North Channel.....	122
Appendix 2 Graph showing phosphorus vs live weight of rainbow trout (<i>Oncorhynchus mykiss</i>). From Bureau et al 2003).....	124
Appendix 3 Method summary for the analysis of Total Phosphorus in water. Excerpt from the NLET Schedule of Services, 2010.	125
Appendix 4 Method summary for the analysis of Total Phosphorus in feed and sediment. Excerpt from the NLET Schedule of Services, 2010.....	126
Appendix 5 Uncertainty range given each model parameter.....	128
Appendix 6 A brief summary of Recommendations For Operational Water Quality Monitoring at Cage Culture Aquaculture Operations (Boyd, 2001).....	129

List of Abbreviations

ΔV	change in lake volume (m^3)
A	total surface area of the sediment trap tubes (m^2)
ADC of P	Apparent Digestibility Coefficient of Phosphorus (%)
ADCP	Acoustic Doppler Current Profiler
ADJ	adjustment factor to correct for length of day
AE	Actual Evaporation
AFWP	Phosphorus in Apparent Feed Waste (kg)
AGNPS	Agricultural Non-Point Source Pollution Model
Al	Aluminum
<i>b</i>	width of channel (m)
BMP	Best Management Practices
BOD	Biological Oxygen Demand
BP	the mass of phosphorus in biomass (net gain) (kg)
C	Celcius
<i>c</i>	<i>P</i> concentration in the mass-balance model
<i>C</i>	Chezy coefficient
CANWET	Canadian Watershed Evaluation Tool
Chl <i>a</i>	Chlorophyll- <i>a</i>
cm	centimeter
CSTR	Completely Stirred Tank Reactor
CV	Coefficient of Variation
DO	dissolved oxygen
<i>dp</i>	difference in <i>P</i> concentration in the mass-balance model
<i>dt</i>	time step (10 min) in the mass-balance model
DWP	Dissolved Waste Phosphorus in fish feed (kg)
E	lake evaporation (m^3)
EST	Eastern Standard Time
Exp	exposure time (days)
FCR	Feed Conversion Ration
Fe	Iron
FeedP	the mass of phosphorus in the feed (kg)
g	grams
<i>g</i>	acceleration due to gravity (9.8 m/s/s)
<i>h</i>	depth of channel (m) in inlet hydrodynamic model
I	Thornwaite heat index
<i>I</i>	runoff from watershed (tributaries) (m^3)
ID	Identification
IN	Groundwater and inflow (m^3) in water balance model
<i>inatm</i>	atmospheric phosphorus load (kg)
<i>indwell</i>	dwelling phosphorus load (kg)
<i>inex</i>	inflow phosphorus load(kg)
<i>infarm</i>	farm load (solid waste phosphorus, apparent feed wastage, dissolved waste phosphorus) (kg)
<i>ingw</i>	groundwater phosphorus load (kg)
<i>inint</i>	internal phosphorus load (kg)
<i>inleaf</i>	leaf litter phosphorus load (kg)
<i>inNP</i>	non-point phosphorus load (kg)
<i>j</i>	mean monthly temperature for month <i>j</i> in evaporation model
kg	kilograms
<i>KsVc</i>	sedimentation load
L	Litre

L	length of channel (m)
L_{conc}	estimated lake TP concentration (ug/L)
L_{int}	Internal phosphorus load mg/m ² /day
m	meters
m ²	square meter
m ³	cubic meters
mg	milligrams
ml	milliliter
N	North
n	Manning's coefficient
NLET	National Laboratory for Environmental Testing
NS	Nash-Sutcliffe test
OMAFRA	Ontario Ministry of Agriculture, Food and Rural Affairs
OMOE	Ontario Ministry of Environment
Out	outflow via the inlet (m ³) in water balance model
<i>outex</i>	outflow phosphorus load (kg)
<i>outsed</i>	lake sedimentation load (kg)
<i>outsport</i>	sportfishing (kg)
P	phosphorus
P_{car}	whole body phosphorus in fish (g)
PE	Potential Evaporation
Precip	precipitation (m ³)
PWQO	Provincial Water Quality Objective
Q_c	Sum of all P loading out of Lake Wolsey as function of time
r	hydraulic radius in Chezy coefficient equation
r ²	coefficient of determination
s	seconds
S_{rate}	sedimentation rate (g/m ² /day)
SRP	Soluble Reactive Phosphorus
SWP	Solid Waste Phosphorus from farm (kg)
T	daily mean temperature (°C)
TKN	Total Kjeldahl Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
$T_{p_{fish}}$	mass of phosphorus in perch (kg)
TP_{int}	initial boundary condition of lake TP concentration or estimated TP from previous time step (kg)
u	velocity (m/s)
ug	micrograms
V	volume
V_{int}	initial boundary condition of Volume or estimated Volume from previous time step (m ³)
W	West
$W(t)$	Sum of all P loadings in to Lake Wolsey as function of time.
WAAS	Wide Area Augmentation System
W_{dry}	freeze dried weight of sediment (g)
W_{fish}	dry weight of perch (kg)
x	weight of fish (g)
ζ_{cb}	water level of Campbell Bay (m)
ζ_{hw}	water level of Lake Wolsey (m)

1.0 Introduction

1.1 What is aquaculture?

Aquaculture is defined as the farming of fish, mollusks, crustaceans and aquatic plants. “Farming implies some form of intervention in the rearing process which enhances biomass production, such as the control of reproduction and growth, management of health and disease or protection from predators. Farming also implies individual or corporate ownership of the stock being cultivated and distinguishes aquaculture from the wild harvest fishery” (Food and Agriculture Organization 2009).

1.2 The importance of aquaculture

Aquaculture is one of the fastest growing food producing industries in the world. Currently, the industry is growing at a rate of 8.8% annually and accounts for 43% or 55.1 million tons of total fish supplying 2.6 billion people with approximately 20% of animal protein (Food and Agriculture Organization 2010).

In Canada, there were 6598 licensed aquaculture operations in 2004 (Department of Fisheries and Oceans 2007). Every province, including the Yukon Territory, has an aquaculture industry and accounts for >8000 jobs (Canadian Aquaculture Industry Alliance 2011). Canada’s broad coastline, high quality products, food security, and close proximity to US markets allow for aquaculture growth. In Canada production of salmon accounted for 85% of the total finfish aquaculture production, generating over \$748 million in 2006 (Canadian Aquaculture Industry Alliance 2011).

Freshwater culture operations consist of both land-based and cage systems. In Canada, there are currently more than 700 operations amounting to approximately \$70 million annually. The largest freshwater producing provinces are Ontario and Quebec. The predominant species farmed is rainbow trout, followed by brook trout, arctic char, tilapia, lake trout and brown trout (Department of Fisheries and Oceans 2007). In Ontario there are 7 licensed farms, which produce 4060 tonnes of rainbow trout worth approximately \$17.1 million in 2010. The value added to the Ontario economy is worth approximately \$60 million (Moccia and Bevan 2011). It also provides direct and indirect employment of 144 and 150 person-years, respectively in Ontario. Cage-culture is a relatively new industry in Ontario where the first private operation was established in 1986. Cage-culture of rainbow trout accounts for 88% of Ontario production (Moccia and Bevan 2011). All cage-aquaculture in Ontario is located in the Georgian Bay/Lake Huron Region of the Great Lakes.

With an increase in market demand, the Great Lakes provide a potential opportunity for growth in the fresh water sector; however, at present, perceived environmental impacts are influencing growth of this industry (Moccia and Hynes 1998). Industry, regulators, academic and other government agencies are striving to create a sustainable ecosystem approach.

1.3 Environmental Concerns

Because the industry is relatively new in Ontario, and there are very few studies on the Great Lakes, there are a number of environmental and regulatory challenges facing Ontario regulators. The foremost regulatory issues are water and sediment quality, fish communities, fish health, navigable waters, aboriginal community considerations, and species at risk.

Cage-aquaculture in Ontario is located in the in Lake Huron/Georgian Bay. The Great Lakes are a public resource with a diversity of users. Concerns have been raised regarding the environmental effect cage farming will have on the Great Lakes in terms of eutrophication. Most aquaculturalists are well aware of these concerns and have implemented the most up to date sustainable best management practices (BMPs) to ensure fish and ecosystem health. Regardless, because the lakes are a multi user resource appropriate monitoring and regulations must be in place. A brief summary of the environmental concerns; specifically water and sediment quality, and regulatory issues will be discussed below with the associated challenges.

1.3.1 Sediment and Water Quality

One of the primary environmental concerns of freshwater cage-aquaculture operations is the potential increase in nutrients in the receiving water from the aquaculture waste discharge. Aquaculture waste consists of two components: solid waste and dissolved waste. Solid waste generally may have a greater impact on sediment whereas dissolved waste may impact water quality.

Deposition of solid waste on the sediment is a primary concern. Solid wastes are composed primarily of fecal material, consisting of the organic undigested component of feed consumed: proteins, lipids, starches and fiber as well as phosphorus and nitrogen compounds (Bureau and Hua 2010). A smaller proportion of the solid waste consists of waste feed. Accumulation of solid waste is generally localized to within 50 m of the cage site (Milne 2008). The degree of impact is largely dependent on depth, morphometry, currents, and assimilative capacity of the receiving water. Bacteria degrade organic matter while consuming oxygen and produce carbon dioxide and ammonia. When excess solid waste is added to the sediment in areas of limited mixing of the hypolimnion, hypoxic or anoxic conditions may occur (Axler et al. 1996; Axler et al. 1998; Abo and Yokoyama 2006). Further to this, increased hydrogen sulfide levels as result of hypoxic conditions may be toxic to the local benthos (Hargrave et al. 2008). Solid waste may also leach bio-available phosphorus or nitrogen contributing to eutrophication via remineralization, ligand exchange or reduced redox (Garcia-Ruiz and Hall 1996; Pettersson 1998). Eutrophication is defined as enriched nutrients (nitrogen or phosphorus) in the aquatic environment causing “excessive algal biomass resulting in a deterioration of water quality which interferes with most of the beneficial uses of water; it is causing, in many cases, significant economic losses” (Organisation for Economic Co-operation and Development 1982).

Recovery of the solid waste, in terms of benthos and sediment chemistry, to background levels is largely unknown in Ontario. Milne (2008), reported it takes approximately 9

years, or a decrease of 2 cm of solid waste per year, for the fish deposit to be dispersed/assimilated at a site in the North Channel, Ontario.

Dissolved wastes occur as a result of catabolism of absorbed nutrients that are not retained in the fish (Bureau and Hua 2010). Ammonia, orthophosphates and carbon dioxide are considered the primary dissolved wastes and are excreted through the gills or urine.

The ammonia waste is generated by the catabolism of amino acids required for energy. Ammonia naturally occurs in the environment as a byproduct of biological metabolism. Higher concentrations are generally associated with waste water treatment plants, agricultural sources such as runoff from livestock waste, and manufacturing of fertilizer (Burkhalter and Kaya 1977). Ammonia is present in both the ionized (NH_4^+) and un-ionized (NH_3) form in a more or less equilibrium state. If temperature and pH increase there is a shift to the more toxic un-ionized form of ammonia. It has been reported that long-term NH_3 has caused damage to gills, skin and internal organs of fish as well as slow growth and development in trout fry (Burkhalter and Kaya 1977).

The main concern of excessive orthophosphates in freshwater is that it is readily available for biouptake potentially increasing algal biomass causing eutrophication. Increased levels of orthophosphate are generally associated with anthropogenic sources such as agricultural and urban runoff, and discharge from waste water treatment plants.

Carbon dioxide is not considered to be an environmental concern in cage-aquaculture operations. Excess carbon dioxide in land-based aquaculture facilities with recirculation systems is a concern because of the potential risk of hypercapnia in fish (Bureau and Hua 2010). Hypercapnia in fish occurs when high levels of carbon dioxide are present in the blood causing acidosis which in turn reduces oxygen haemoglobin and transportation of oxygen to body tissues (Hosfeld et al. 2008). To mitigate this concern recirculation systems include procedures to ‘strip’ carbon dioxide to maintain good water quality (Bureau and Hua 2010).

1.3.2 Current Sediment Quality Risk Management

As part of the “Recommendations for Operational Water Quality Monitoring at Cage Aquaculture Operations” in 1999/2000 (Boyd et al. 2001) sediment chemistry analyses is required at the site tenure agreement boundary and a reference site. The sampling is to take place before renewal of, or revision of an existing aquaculture license. Samples are analyzed for particle size, total organic carbon (TOC), total phosphorus (TP), and total Kjeldahl nitrogen (TKN). The results from the site tenure boundary are then compared to reference sites. If the “nutrient concentrations in the sediment at the site tenure agreement significantly exceed those at reference stations then the operator will be required.....to submit abatement plan leading to a reduction in the site’s operational scale (feed quota) for the subsequent operational season”. The rationale being if the TOC, TP and TKN at the site tenure agreement is significantly greater than the reference site the current operational scale is exceeding the assimilative capacity (Boyd et al. 2001). See appendix 6 for a brief summary of recommendations.

A number of science gaps currently exists preventing sound regulatory risk management:

1) there is a need for predicting the assimilative capacity of solid waste from cage-aquaculture along with predicting temporal and spatial distribution of solid waste; 2) a need for a benthic community assessment in the depositional zone of solid waste, the assimilation rate of solid waste and the level of alteration of the benthic community by solid waste without effecting the assimilation and 3) development of tools to assess rates of recovery of sites to pre-operational conditions (Ontario Ministry of Natural Resource 2007).

1.3.3 Current Water Quality Risk Management

Water quality risk management is presently based on recommendations by the Ontario Ministry of Environment (OMOE). The report, entitled “Recommendations for Operational Water Quality Monitoring at Cage Aquaculture Operations” was created in 1999/2000 after OMOE discovered an anoxic area at an aquaculture farm in LaCloche Channel on Manitoulin Island, Ontario (Boyd et al. 2001). This area has an enclosed basin with a limited assimilative capacity due to low flushing with Georgian Bay. The farm closed in 1998.

Based on the water quality recommendations, the OMOE has classified farm sites into three categories to reduce potential water quality impairment caused by farming operations. Type 1 sites are enclosed (lake-like) basins with limited flushing. Type 2 sites are partially exposed, having good epilimnion/metalimnion flushing but limited to no

hypolimnion exchange. Type 3 sites are exposed locations where the hypolimnion is also well flushed (Boyd et al. 2001). Farms in Type 1 sites generally contribute to deleterious conditions in the receiving waters because excess nutrient input and low flushing rates may contribute to exceedance of the assimilative capacity of the receiving water thus resulting in eutrophication or hypereutrophication (Clerk et al. 2004; Axler et al. 1996; Axler et al. 1998) whereas Type 3 sites seem to support farming operations with very little measurable effects (Cromey et al. 2002). Type 2 sites may or may not be suitable for farming operations (i.e., addition of a farm or expansion of an existing farm) depending on, for example, the surrounding land use, basin morphometry and retention times as well as the existing nutrient loadings. For these sites, more extensive modeling is needed to determine assimilative capacities for the receiving waters. For example, Lake Wolsey on Manitoulin Island is considered a Type 2 site because 1) there is extensive flushing with the epilimnion but limited flushing with the hypolimnion and 2) prior to cage operation the mean TP lake concentration was ~11 ug/L.

Risk management of water quality for licenses is presently based on seven recommendations by the MOE (Boyd et al. 2001). The first three recommendations are based on sampling positions and total phosphorus concentrations while the latter four deal with water clarity (secchi measurements), dissolved oxygen and temperature (profiling), sediment assessments, and an annual review of findings in the scientific community to improve recommendations. Each of the monitoring requirements are recommended to be customized to a type 1, 2, or 3 site.

Establishment of permanent water quality stations are recommended at 30 m from each of the four corners of the cage operation or “mid-point of the three sides of cage complex if they are attached to shore” (Boyd et al. 2001). Water sample collection is to take place for analysis of total phosphorus during spring turnover (isothermic), summer (stratification) and fall turnover (isothermic). A total of 51 or 68 samples (depending if the cage structure is attached or unattached to shore) plus reference samples are collected annually.

The current recommendations state that if the interim Provincial Water Quality Objective (PWQO) of 10 ug/L of total phosphorus is exceeded anytime during spring through fall, mitigation of operations is required (reduced feed quotas and increased operational efficiency), sampling frequency is increased, and confirmation of the absence of algae must be made. Sites which experience TP background concentrations ≥ 10 ug/L such as Lake Wolsey will receive special consideration (Boyd et al. 2001).

The existing recommendations require sample collection near the farm to detect very localized effects, however, once greater than 30 m away from the cages TP concentrations become close to background lake concentration and are therefore not helpful in managing TP on a lake-wide basis. Also, assessments of operating and future farming sites are based on in situ samples, meaning results are based on phosphorus concentrations at the time of measurement. Because nutrient loadings can be dynamic, where non-point sources may be important, the need for a predictive tool is of utmost

importance to determine any current or future problems that may arise on a near-field and lake-wide basis.

The current PWQO's were established in 1984 as a revision of the Ontario Water Resources Commission's "Guidelines and Criteria for Water Quality Management in Ontario" by the Ontario Ministry of Environment. The PWQO's represent a set level of water quality "which is protective of all forms of aquatic life and all aspects of the aquatic life cycles"(Ontario Ministry of the Environment 1992; Ministry of Environment and Energy 1994). The total phosphorus objective is an "interim guideline" meaning the set value is "based on a search of the best information on hand" (Ontario Ministry of the Environment 1992). The interim PWQO of spring mean total phosphorus concentration of 20 ug/L was based on conclusions of a study by Dillon and Rigler (1975) where no algal blooms would occur if the chlorophyll-*a* (chl*a*) concentration did not exceed 5 ug/L and corresponding spring TP concentration of 20 ug/L. However, Dillon and Rigler (1975) also concluded that a set value of 20 ug/L may not protect the dissolved oxygen in bottom waters; therefore for as "a higher level of protection" they recommended a spring mean TP concentration of 10 ug/L (Ontario Ministry of Environment 1979).

The following chapter will review studies where modeling is used to assess water quality in freshwater and describe the successes and limitations as well as the hypothesis and objectives of the present study.

2.0 Literature Review

2.1 Modeling the Impacts of Aquaculture

Modeling can be a powerful tool in predicting current and future impacts on receiving waters. Models need to be quick and user-friendly in order to allow for a streamlined regulatory process. Assimilative capacity models provide an estimate of the ability of a body of water to absorb excess nutrients from farming activities without adverse impacts to the ecosystem or farm fish. Direct application of models is in use in only a few countries, whereas indirect use is more common within regulatory frameworks. Most models developed for use in mariculture generally focus on deposition and dispersion of particulate organic material and benthic impacts because they tend to be relatively easy to observe and sample (Silvert and Sowles 1996) whereas fewer models apply to broad scale nutrient management because it is believed dissolved nutrients are more transient in the marine environment. Most research on contribution of freshwater cage-aquaculture on eutrophication in Ontario is limited because the industry is relatively new. The majority of research impacts of nutrients from cage-aquaculture in freshwater is from Sweden and Scotland. The following chapter describes the evolution of TP models beginning with Organisation for Economic Co-operation and Development (OECD) nutrient models.

2.1.1 Freshwater – Water Quality Modeling

It has long been known that phosphorus (P) is typically the limiting nutrient in Ontario lakes (Schindler 1974; Schindler 1977; Organisation for Economic Co-operation and Development 1982). Total Phosphorus (TP) includes the solid and the dissolved form.

Soluble Reactive (SRP), the dissolved bio-available form of P, is of particular concern in P -limited lakes. Generally, in Ontario lakes, according to the MOE's PWQO, an increase in *Chla* levels may occur when TP exceeds 10 ug/L (Ministry of Environment and Energy 1994). When TP concentrations exceed 20 ug/L nuisance algal blooms may occur (Dillon and Rigler 1974a), where under certain conditions may result in toxic cyanobacteria outbreaks which in turn could compromise source water for human consumption (Watson et al. 2008; Davies and Mazumder 2003). This in turn, leads to subsequent decay of algal biomass and decreased oxygen affecting lake ecosystems. Dodds et al (2009), report that economic loss in the US due to eutrophication of freshwater is approximately \$2.2 billion annually. Many TP models have been developed to aid lake managers in predicting maximum TP loading without exceeding trigger points or to predict reductions in TP loadings in a lake experiencing eutrophication (Dillon and Rigler 1974b; Dillon and Molot 1996; Hakanson et al. 2003; Hakanson and Bryhn 2008; Vollenweider 1975), however, no water quality model geared specifically for freshwater lakes with cage-aquaculture is utilized within the regulatory framework in Ontario to date.

One of the key concerns of organic enrichment from fish farms in freshwater lakes is eutrophication. Marine systems tend to have higher flushing rates thereby constantly replenishing oxygen at farm sites. In contrast, freshwater lakes tend to have smaller volumes and lower flushing rates, therefore decreased dissolved oxygen replenishment and a possibility of eutrophication of the receiving water (Cornel and Whoriskey 1993). Low dissolved oxygen in the hypolimnion may result in an efflux of phosphorous into the hypolimnion via ligand exchange, reduction of iron (III) to iron (II) and the release of

phosphorus, enzymatic hydrolysis of organic esters by bacteria and desorption of particles (Kelly 1992; Kelly 1993; Nurnberg 1984; Nurnberg 1987; Nurnberg 1995; Pettersson 1998). Models have been developed to assist regulators in decision making processes regarding environmental policies for cage-aquaculture. One of the main goals of the models is a eutrophication risk assessment. Very few models have focused on fresh water impacts of farming operations (Hamblin and Gale 2002; Johansson et al. 1998; Johansson and Nordvarg 2002). Freshwater models are generally based on lake-wide mass-balance approaches of TP using variations of Vollenweider's OECD model or Dillon-Rigler models (Dillon and Rigler 1974b; Vollenweider 1975).

A brief outline of Vollenweider (OECD) and Dillon-Rigler models as well as current freshwater models are discussed below.

2.1.1.1 Vollenweider (OECD) Model (1968, 1969, 1973)

In 1968 Vollenweider was the first to predict “critical loading” of phosphorus in freshwater lakes. He plotted “specific surface loading” of phosphorus (P) of 30 lakes against mean depth and discovered that the lakes separated into three types according to the degree of eutrophy: eutrophic, mesotrophic and oligotrophic. A lower line separating oligotrophic lakes from mesotrophic lakes were coined “permissible loading”, where as an upper line above mesotrophic was termed “critical loading” or eutrophic (Vollenweider 1968). He also noted other parameters should be included in the model: length of shoreline, volume and flushing rate of lakes; retention time, internal loading and sedimentation (Vollenweider and Dillon 1974). In 1969, Vollenweider modified his original model to include sedimentation as sink in the budget calculation. The model was

validated on a number of Swiss lakes with reasonable success (Vollenweider 1969). In 1973, the model was further modified to include residence time (Vollenweider 1975) . He plotted loading vs. residence time and discovered a more accurate representation of eutrophy.

He assumed the lake represents a single open box at steady state. The model worked well when applied to oligotrophic lakes, however, less satisfactory for eutrophic lakes.

To predict lake phosphorus concentration he formulated the following model for steady state conditions:

$$[P] = \frac{L}{\bar{z}(\sigma + \rho)}$$

Where: $[P]$ = concentration of phosphorus (g/m^3)

L = phosphorus loading ($\text{g/m}^2/\text{year}$)

$\bar{z}$ = mean depth (m)

σ = sedimentation rate (year^{-1})

ρ = flushing rate (year^{-1})

To test the model he compared the predicted lake phosphorus concentration with observed concentrations in spring during isothermic conditions with reasonable results.

2.1.1.2 Dillon and Rigler (1974, 1975)

According to Dillon and Rigler (1974), Vollenweider's relationship (above) did not work well with lakes in Ontario. These lakes had elevated P loadings and according to the

model should have been eutrophic, however, from observed data they clearly were not (Vollenweider and Dillon 1974). It was concluded these lakes had a high flushing rate as a result of a large drainage basin to lake area ratio. Because the sedimentation coefficient is not easily measured, Dillon and Rigler (1974a) , recommended that the retention coefficient could be substituted for the sedimentation coefficient assuming steady state. By plotting the loading multiplied by one, minus the retention coefficient, divided by the flushing rate vs. the mean depth, a more satisfactory relationship exists.

To predict concentration, Dillon (1974), modified Vollenweider's 1969 model by substituting the sediment coefficient with a retention coefficient which is much easier to measure. The retention being the fraction of P lost through the outflow (Vollenweider and Dillon 1974; Dillon and Rigler 1974b):

$$R_{\text{exp}} = 1 - \frac{q_o[P]_o}{\sum q_i[P]_i}$$

Where:

R_{exp} = Retention coefficient (dimensionless)

q_o = volume of outflow (m^3/year)

$[P]_o$ = concentration in the outflow (g/m^3)

q_i = volume of inflow (m^3)

$[P]_i$ = concentration in the inflow (g/m^3)

Vollenweiders equation can then be rewritten as:

$$[P] = \frac{L(1 - R_{\text{exp}})}{z\rho}$$

Where: $[P]$ = concentration of phosphorus (g/m^3)

L = loading ($\text{g}/\text{m}^2/\text{year}$)

R_{exp} = retention coefficient (dimensionless)

$\bar{Z}$ = mean depth (m)

ρ = flushing rate (year^{-1})

When applied to 18 lakes in Ontario Dillon, 1974, found the model to predict concentration at spring turnover reasonably well. Furthermore, Kirchner and Dillon (1975), developed a model to predict retention in lakes, since retention is time consuming and expensive to directly measure. They found, by multiple regression, that R was highly correlated to the areal water loading:

$$R_p = 0.426^{(-0.271qs)} + 0.574^{(-0.00949qs)}$$

Where:

R_p = retention of phosphorus

qs = areal water load ($\text{g}/\text{m}^2/\text{year}$)

They concluded that the following models prediction of retention was in close agreement to measured retention.

On a broad scale, Vollenweider's 1968 phosphorous loading vs. mean depth relationship was instrumental in the International Joint Commissions (IJC) decisions in banning phosphates in detergents and implementing upgrades in waste water treatment plants to tertiary treatment (Dillon 1975). Dillon, 1974, and Dillon 1975; and later Dillon, 1994, Dillon, 1996, modified Vollenweider's model to respond to lake managers and regulators

requests for simple methods to predict eutrophication in light of increased development on the shores of Ontario lakes.

A limited number of studies have attempted to model nutrients in freshwater lakes with cage-aquaculture using the mass-balance approach, Dillon-Rigler and/or Vollenweider's models (Kelly 1995; Nordvang and Hakanson 2002; Johansson et al. 1998; Hakanson et al. 2003) each with varying success.

Kelly (1995), conducted a study to test the robustness of using the Dillon-Rigler model which is used as part of regulatory process in Scotland for predicting loading in new and existing sites. The authors found that modeled results closely resembled observed results where aquaculture was the only significant source of loading, otherwise loadings such as other non point sources were not accounted for in the Dillon-Rigler model. They concluded that all nutrient loadings should be considered to more effectively assess the lake.

Hakanson 1998, studied two lakes in Sweden to determine the effect of fish farming on total phosphorus concentrations by using the Vollenweider model. The authors found that Vollenweider's model overestimated the lake TP concentration in both lakes. They attributed the cause of low TP to primary or "Direct Uptake (DU)" of fecal material from farmed fish by wild fish, zooplankton and "Eliminated (EL)" by sedimentation, birds, fishing and secondary consumption by bacteria and benthic organisms. The authors modified Vollenweider's model to include an algorithm for uptake and elimination. They

conclude that uptake and elimination could be accounted for in the mass-balance equation, however, it was not possible to distinguish between the two.

Johansson (1998) confirmed Vollenweider's model was unsuitable for predicting trophic state of lakes with aquaculture. The authors recommended models to include seasonal variation, oxygen concentrations and phytoplankton biomass. In 2002, Johansson and Nordvarg produced TP mass-balance models based on Vollenweider's and Dillon-Rigler model that are specifically calibrated for lakes with fish farms. They too confirmed that the traditional Vollenweider and Dillon-Rigler models are unsuitable of predicting effects fish farms on the trophic state of lakes. They found the models specifically calibrated for lakes with farms predicted 50% lower TP than the traditional Vollenweider and Dillon-Rigler models. The new version of the Dillon-Rigler model was slightly better than new version of Vollenweider model compared to a calibrated dataset.

Johansson and Nordvarg (2002) tested the traditional Vollenweider and Dillon-Rigler models using the Vollenweider's original calibrations on predicting the effects of a fish farm emissions on lake TP concentration. They found the traditional models to overestimate the effects. They therefore used the traditional models with a farm-specific calibration and found small differences between the models, however, slightly better accuracy when compared to traditional models. Lake morphometry and drainage parameters; positioning of the farm in relation to the outlet; unequilibrated lakes; variation of TP concentrations between years; and non point loading that were not

included in the modeling may explain some discrepancies between modeled and observed data.

2.1.1.3 LEEDS (1998; 2008)

The LEEDS (Lake Eutrophication, Effect-Dose-Sensitivity) model essentially divides the ecosystem into sections including dissolved and particulate, epilimnion and hypolimnion, and active and inert sediments. The model was tested on Lake Southern Bullaren, Sweden where blooms of blue-green algae occurred. It was believed a cage-aquaculture farm with an approximate annual production of 500 tons, was responsible for the algal outbreak. The model includes four segments: 1) Tributary inflow a similar approach as Dillon and Kirchner, 1975; 2) Lake retention; 3) Internal loading; and 4) Sediment sub-model where P retention in surface and deep sediments are predicted (Hakanson and Carlsson 1998). The authors concluded that variation in climate could alone cause algal blooms and the fish farm was only a minor contributor to phosphorus concentration (Hakanson and Carlsson 1998).

Malmaeus, 2004 tested and improved the LEEDS model by incorporating fluxes in phytoplankton, mixing and diffusion, colloidal in surface and deep water, and organic content of fine sediments. The improved model was calibrated and validated against data from lakes in Balrus and Sweden. The authors concluded the model predicted TP concentration reasonably well against empirical data.

In 2008, Hakanson presented an advanced version of the LEEDS model. The most important improvement is the applicability of the model to most types of lakes. The fundamental structure of the model is based on LakeMab which was originally developed to trace radiocesium in the aquatic environment resulting from the Chernobyl nuclear accident in 1986. Forty-one lakes were selected to test the model. The lakes were highly variable in depths, size, and trophic status. The results were compared to empirical data and both Vollenweider and OECD models. The authors concluded that the model may provide realistic estimations on the lake eutrophy based on changing, adding or excluding variables and would be very practical in lake management.

2.1.1.5 Reid and Moccia 2007

Reid and Moccia (2007) attempted to estimate mean daily TP concentrations 30 m down stream of a cage-aquaculture farm in Lake Huron, Ontario. A lake-wide loading modeling approach is not always necessary in large lakes where eutrophication from a cage-aquaculture farm is negligible (Reid and Moccia 2007). Such information could assist in fish production decisions and allow for a more cost effective and efficient TP assessment as necessary for current regulatory compliance. Non-settleable and dissolved waste phosphorus estimates were modeled and divided by an uncorrected daily flushing volume to estimate down stream TP concentrations, where flushing volumes through a net cage are uncorrected for drag and turbulence (Reid and Moccia 2007) . Despite a number of uncertainties, the estimated downstream TP concentrations compared reasonably well to observed TP concentrations ($r^2=0.75$) and the authors concluded that it is possible to estimate near-field TP concentrations.

2.1.1.4 Hamblin and Gale 2002

Hamblin and Gale (2002) attempted to estimate TP lake concentration in Lake Wolsey Ontario. According to the authors the location of farms may be an issue where enclosed areas such as LaCloche Channel on Manitoulin Island have limited circulation and smaller volume of water where nutrients from a fish farm may cause eutrophication.

The objectives of the study were to better understand “flushing characteristics and assimilative capacities of various embayments” including Lake Wolsey (Hamblin and Gale 2002). The authors applied a one box mass-balance model, where the only input parameters were the cage-aquaculture farm TP load (kg/day) and the inflow inlet load. No distinction was made between solid waste or the dissolved waste released from the farm. Other important sources of TP loading were ignored such as non point sources, groundwater and dwellings. Despite a number of shortcomings in the study they concluded that Lake Wolsey may experience seasonal algal blooms but because the lake retention time is less than a year, long term cumulative effects of water quality should be negligible.

2.2 Application of a Phosphorus Mass-Balance Model to a Freshwater Lake in Ontario with Cage-Aquaculture

The present study attempts to address current challenges of water quality risk management of cage-aquaculture in Ontario. Arguments can be made both advocating and questioning modeling to assess water quality because some may consider black box models to be over simplistic and not address underlying complex processes; some models may not be applied to other systems; or they may not consider other potential nutrient sources (Kelly 1995). On the other hand, simplistic models allow for economically feasible and rapid lake assessments without intensive sample collection (Kelly 1995) and lake managers and regulators can easily apply models to establish baselines, regardless of the simplistic assumptions.

In this study, we have attempted to apply a mass-balance approach to a type 2 site. The mass-balance model used in this study may be a suitable tool, because this study lake has defined bathymetric boundaries, there are no complicated bays or inlets; the inlet exchange, tributaries (non-point sources) and the farm are potentially the only important sources of TP loading; we have a strong data set, and we have real time lake level observations to provide a better estimate of inlet exchange loading.

2.2 Objectives of the Study

A simple phosphorus mass-balance equation was used as a tool to test the hypothesis: Cage-aquaculture is not an important contributor of TP load to a type 2 lake in Ontario. The mass-balance model was also be used to answer the following research questions: 1) what is the TP load contribution of a cage-aquaculture farm to a type 2 lake? 2) if or when a farm expands, new farm begins, or farming ceases what will be the estimated lake

TP concentration? The findings from this study will provide defensible information to guide regulatory policy towards appropriate monitoring and modeling programs.

Water quality and particulate assimilative capacity models in concurrence with sound monitoring programs can be powerful tools to predict environmental impacts where a sustainability approach is the key for success of the aquaculture industry. It is important for industry, government agencies and universities to work together towards a common goal. A research approach provides improved understanding of processes which constantly provides updated information for regulators to develop improved policies. Research in fish health and communities, farm management practices, ecosystem interactions and feed formulations allow for more accurate model predictions which enable managers and regulators to make more informed science-based decisions.

This project focused on Lake Wolsey on Manitoulin Island, Ontario. The objectives of the study were to:

1. Determine the relative importance of phosphorus contributions from various sources.
2. Conduct a sensitivity analysis to rank phosphorus contributors.
3. Determine if the lake can support the expansion or addition of a fish farm using the sensitivity analysis.
4. Provide practical information to regulators to develop improved monitoring programs.

3.0 Methods

3.1 Description of Study Lake

Lake Wolsey is located on Manitoulin Island, Ontario (45° 49.41' N 82° 31.731' W) (fig. 1A). The lake is connected to the North Channel of Lake Huron by a 30 X 30 m inlet where surface water exchanges periodically (fig. 1B). Maximum depth of the inlet is approximately 4.0 M. The surface area of Lake Wolsey is close to 2,315 ha with the maximum depth of ~22 m and volume of $\sim 260 \times 10^6 \text{ m}^3$. The watershed is ~8,100 ha where forest and agriculture dominate the land use (table 1). Forests are dominated by dense coniferous and mixed coniferous stands. Agriculture consists of hay, pasture and livestock (cattle). There are no major tributaries flowing into the lake, however, there are ephemeral runoff points and groundwater discharges.

Manitoulin Island is located in Northern Ontario; however, is not part of the Canadian Shield rather an extension of the Niagara Escarpment. The island is dominated by dolomite, limestone, alvars and karst geology where very little topsoil exists (Karrow 1993). This may be important in terms of phosphorus loadings to Lake Wolsey via direct runoff from the watershed and groundwater seeps via karst formations.

3.2 Description of fish farm

The cage-aquaculture operation, established in 1986, is located on the west shore (45° 48' 09" N 82° 32' 47" W) (fig. 1A). The fish farm has an average annual production of approximately 250 metric tonnes. The farm consists of eight permanent 15 X 15 m cages and 8 seasonal cages (fig. 2) where rainbow trout (*Oncorhynchus mykiss*) are raised from

about 50 g to a market weight of about 1000 g in approximately 12 to 18 months. Total depth under the farm ranges from 15 to 17 m. Water temperatures adjacent to the cages ranges between 4.0 and 25.9 °C.

3.3 Modeling methodology

The following section will describe model theory and assumptions used in this study.

3.3.1 The Mass-Balance Model

Phosphorus mass-balance models are generally used in eutrophication risk assessment. The mass-balance approach is based on the Law of Conservation of Mass. A simple one-box mass-balance model has been implemented in this study to estimate lake phosphorus concentration in Lake Wolsey (fig. 3) and assess loads from other important sources. Mechanistic models may not be required at this time because: 1) the complexity of such models and 2) availability of data. As data becomes available, more complex models may be implemented to address specific managerial questions.

In this study, Lake Wolsey is considered to be a homogenous waterbody where the following assumptions apply: (1) constant density; (2) isothermic conditions; (3) steady state conditions; (4) single reaction and (5) first order reaction (Chapra 1997). We also assume the lake is a “completely stirred tank reactor” (CSTR) where a chemical is dispersed instantly and evenly throughout the water column (Chapra 1997). To predict lake phosphorus concentration the following equation was applied:

$$V(dp/dt) = W(t) - (Qc) - KsVc \quad (1)$$

Where,
 V = volume

dp = difference in P concentration

dt = time step

c = P concentration

$W(t)$ = Sum of all P loadings in to Lake Wolsey as function of time.

Qc = Sum of all P loading out of Lake Wolsey as function of time.

$KsVc$ = Sedimentation load

The TP mass-balance model with sources and sinks considered in this study thus becomes:

$$L_{conc} = ((TP_{int} + (inex + infarm + inNP + inatm + ingw + indwell + inint + inleaf) - (outex + outsed + outsport)) / V_{int}) * 1000000 \quad (2)$$

Where:

L_{conc} = estimated lake TP concentration (ug/L)

TP_{int} = initial boundary condition of lake TP concentration or estimated TP from previous time step (kg)

$inex$ = inflow (kg)

$ingw$ = groundwater load (kg)

$inNP$ = non-point load (kg)

$inatm$ = atmospheric load (kg)

$indwell$ = dwelling load (kg)

$infarm$ = farm load (solid waste phosphorus, apparent feed wastage, dissolved waste phosphorus) (kg)

$inleaf$ = leaf litter load (kg)

$inint$ = internal load (kg)

$outex$ = outflow (kg)

$outsed$ = lake sedimentation (kg)

$outsport$ = sportfishing (kg)

V_{int} = initial boundary condition of Volume or estimated Volume from previous time step (m^3)

The TP mass-balance model was coded in Microsoft® Office Excel 2003 and can be transferable to other lakes. Methods for estimating each source and sink parameter are discussed below.

The model was calibrated against monthly observed TP values obtained from Environment Canada and the Ontario Ministry of Environment as described below under “Observed Data”.

Volume (V_{int})

The cut and fill method in Surfer[®] V7.2, geospatial software was used to estimate the starting lake volume. For lake volumes, the depth value is considered the cut portion (lower surface) and a constant of “0” is used as the fill portion (upper surface). The volume of the lower surface under the upper surface volume is calculated and thus is the volume of the lake. Lake depths, digitized from hydrographic chart # 6030, were the only required input parameter to estimate volume. Modeled inlet velocities, as described below, were used to estimate changes in lake volume.

3.3.2 Inlet Model (*inex* / *outex*)

Inlets provide access to bays and oceans or large bodies of water where they consist of short, narrow structured channels or unstructured sand barriers or shoals. Inlet hydrodynamics is based on Bernoulli’s Principle where as velocity of the water increases, the pressure exerted by the water decreases (van de Kreeke 1988; Mehta and Joshi 1988). This is important for natural flushing of bays and improving water quality. If an inlet is sufficiently wide and deep, oscillation of water between the bay and a lake will be equivalent. If the channel size becomes significantly smaller compared to the bay area, movement of water through the inlet will become pressure driven where forcing from astronomical tides, pressure changes (weather) or horizontal density gradient between

two bodies of water will cause surface water to have a hydraulic head creating a slope and therefore velocity (U.S.Army Corps of Engineers 2006).

The exchange channel in this study consists of a simple inlet connecting the North Channel to Lake Wolsey (fig.1B). In some cases, because the depth of inlets is greater than the depth of stratification between the adjacent bodies of water, a bi-directional flow occurs within the inlet. However, the inlet depth in this study is ~4.0 m which is much shallower than the thermocline in Lake Wolsey (~19 m) and Campbell Bay (~10 m), therefore during the stratified season a simple, a one-dimensional equation of motion can be applied as the inlet is unstratified. The governing one-dimensional shallow water equation below (2) is applied to estimate velocity from depth and the cross-sectionally averaged area (U.S.Army Corps of Engineers 2006):

$$\frac{dbhu}{dt} = -gbh \frac{\zeta_{cb} - \zeta_{lw}}{L} - \frac{g}{C} b |u|^p |u|^p \quad (3)$$

Where:

b = width of channel (m)
 h =depth of channel (m)
 u =velocity (m/s)
 dt = time step (10 min)
 g =acceleration due to gravity (9.8 m/s/s)
 ζ_{cb} =water level of Campbell Bay (m)
 ζ_{lw} = water level of Lake Wolsey (m)
 L = length of channel (m)
 C = Chezy coefficient

The first term on the right hand side of the equation accounts for inertial forcing and the second term accounts for velocity lost to friction.

The friction term was estimated using a Chezy coefficient:

$$C = \frac{1}{n} (r_h)^{1/6} \quad (4)$$

Where:

C= Chezy coefficient

n= Manning's coefficient

r= hydraulic radius

The assumptions of equation 1 are: 1) velocities in Lake Wolsey and Campbell Bay are negligible compared to the inlet; 2) the difference in the wave height in the channel is small compared to the mean depth; 3) the difference of volume in the channel is negligible compared to the mean volume.

To determine flow velocities using the governing equations, one level logger (Solinst® Level Logger model 3001, Georgetown, Ontario, Canada) was installed adjacent to the inlet in Lake Wolsey (SOUTH) (N45° 49' 43" W82° 33' 6") and another logger installed in Campbell Bay (NORTH) (N45° 49' 43" W82° 33' 11") near the inlet on May 3, 2007. Depth of both loggers was approximately 0.5 m. The loggers were set to collect level data (m) at 10 min intervals. A Barologger was placed in close proximity to the level loggers on land to measure barometric pressure. Relative elevations of the level loggers were determined at the time of deployment with surveying equipment using a relative bench mark of 100 m by Environment Canada. Positions of level loggers were determined by a Magellan Meridian Marine WAAS enabled GPS (MiTAC International Corporation, Santa Clara, California, USA). The loggers were retrieved and downloaded on August 16, 2007 and redeployed within 2 hours. The level loggers and barologger were retrieved on December 3, 2007 and transported to Environment Canada to be cleaned and downloaded. All data was barometric pressure compensated. As discussed above the one dimensional shallow water differential equation (1) was solved numerically to estimate

velocity in the inlet. The model was coded in MatLab[®] V6.1 (MathWorks[®], Natick, Massachusetts, USA). The code is located in appendix (1). Manning's roughness coefficient of 0.02 was applied to this model representing a stone/gravel substrate of the canal (Ward and Trimble 2004). A Chezy coefficient of 55 was chosen after calibration of the model. The initial condition used for u (velocity) and elevation was 0.

The resulting velocities were averaged hourly and multiplied by the area of the inlet to give the flow rate (m^3/s). A positive flow rate signifies surface water exiting Lake Wolsey, whereas a negative flow rate indicates surface water entering Lake Wolsey from Campbell Bay. The mean modeled hourly flow rates were multiplied by the observed mean TP concentration of Campbell Bay ($6.83 \pm 1.28 \text{ ug/L}$) or Lake Wolsey ($10.4 \pm 2.5 \text{ ug/L}$). The results were averaged once again to give monthly flow rates and phosphorus loads in kg.

3.3.3 Validation of Inlet Exchange Modeling

Validation of the modeled velocities was completed on September 16, 2010. Level loggers were deployed on June 24, 2010 and retrieved September 16, 2010 using the same positions and logging intervals as the previous deployment. A Workhorse Sentinel 600 Acoustic Doppler Current Profiler (ADCP) (Teledyne RD Instruments[®], San Diego, California, USA) was installed in a small catamaran (fig. 4). The ADCP was towed across the width of the inlet recording velocity, volume, cross-sectional area and temperature. Transects were repeated until five velocities were within 5%. This process was repeated at three separate intervals during the day: between 10:00 and 10:30 (water

exiting Lake Wolsey); 12:45 and 13:15 (water entering Lake Wolsey); 15:00 and 15:30 (water exiting Lake Wolsey). The modeled velocities were estimated using the governing equations as previously described (section 3.3.2). The modeled velocities from the level loggers were then compared to the velocities from the ADCP. A correlation of determination (r^2) was applied to determine goodness of fit.

3.3.4 Farm Loads - *infarm* (Biomass net gain, feed conversion ratio, solid and dissolved waste phosphorus)

In 2007, a total of ~400, 000 fingerlings were added to the farm May, June, August and October, 2007. The individual fingerling weight ranged between 28 and 138 g with most between 50 and 80 g. Two batches of fingerlings, added in May were harvested in the beginning of December of the same year with a final weight of 926g. No farm records of final mass of fingerlings at the end Nov 2007 were recorded; therefore estimates of final weights were based on expert opinion of the owner/operator of the farm. All fingerlings added between May 1 and 24 were assumed to grow to a final weight of 926 g by the end of November of the same year. Fingerlings added in May 28 and June 1 were assumed to have a final weight of 850 g by the end of November; fingerlings added August 15 and October 1 were assumed a final weight of 300g and 100 g by the end of November, respectively. The net gain of fingerlings was estimated by multiplying the initial number of fish in each batch by the initial body weight to give the initial biomass. The final biomass was estimated by multiplying the final number of fish by the final body weight. We assumed a mortality of 6% for each batch of fingerlings (Bristow et al. 2008). The final body mass was subtracted from the initial biomass to give the net gain from May to November for each batch of fingerlings (Table 2).

Farm records indicated the number of fish harvested, the date of harvest, and individual weight. Because the initial weights of the harvested fish were not recorded in May 2007, estimates of initial weights were based on expert opinion of the owner/operator of the farm. Fish harvested in May were assumed to have the same initial weights as final weights. Fish harvested in June, August, September, and October were assumed to have an initial weight in May of 700g, 500g, 450 and 350g respectively. No harvesting took place in July, or November. The net gain of harvested fish was estimated by multiplying the initial number of fish by the estimated initial weight to give the initial biomass. The final number of fish was multiplied by the final harvest weight to give the final biomass. The final biomass was subtracted from the initial biomass to give the net gain. The net gain of harvested fish was then added to the net gain of the fingerlings to give a total net gain of all fish between May and November 2007. The total feed mass May to November 2007 was divided by the net gain to give the Feed Conversion Ratio (FCR) (table 2).

Bureau et al (2003) found that carcass P increased linearly with live weight Appendix (2), therefore carcass P was estimated using:

$$P_{car}=0.0036x + 0.0173 \quad (5)$$

Where:

P_{car} = whole body phosphorus (g)
 x = weight of fish (g)

Estimation of solid and dissolved waste phosphorus (*infarm*) from the fish and feed is derived from a bioenergetics mass-balance model presented by Bureau et al (2003).

Solid portion of P waste is estimated as follows:

$$\text{SWP} = (\text{P consumed} * (1 - \text{ADC of P of feed})) + \text{AFWP} \quad (6)$$

The AFW was assumed to be 2% of the total feed.

The dissolved P is calculated by the difference between digestible P and the retained P.

$$\text{DWP} = (\text{P consumed} * \text{ADC of P}) - (\text{P retained by fish}) \quad (7)$$

Where:

SWP = Solid Waste Phosphorus (kg)

ADC of P = Apparent Digestibility Coefficient of Phosphorus (%)

AFWP = Phosphorus in Apparent Feed Waste (kg)

DWP = Dissolved Waste Phosphorus (kg)

The bioenergetic mass-balance model thus becomes:

$$\text{FeedP} = \text{SWP} + \text{AFWP} + \text{DWP} + \text{BP} \quad (8)$$

Where:

FeedP = the mass of phosphorus in the feed (kg)

BP = the mass of phosphorus in biomass (net gain) (kg)

The ADC of P was estimated as 0.55 by Bureau (2010) using a model to estimate digestible P of trout feed (Hua and Bureau 2006). Two feed subsamples (adult and fingerling feed), were freeze dried and analyzed by the National Laboratory for Environmental Testing (NLET) (certified by the Canadian Association for Laboratory Accreditation) in Burlington, Ontario. The commercial feed used in this study (mean of adult and fingerling feed) had a phosphorus concentration of 0.95%. A summary of the analysis method is located in Appendix 4

3.4 Parameter Methodology

3.4.1 Non-Point sources (*inNP*)

Total Phosphorus loads from the Lake Wolsey Watershed were difficult to estimate due to generally small and ephemeral tributaries coupled with very thin topsoil overlying limestone. Instantaneous current velocities, using a Price mini current meter (Rickly Hydrological Company, Columbus, Ohio, USA) stream depth and width (m), were recorded at four tributaries April 28, 29 and 30; six tributaries May 6, 2009; ten tributaries Nov 4, 2009; twelve tributaries June 23, 2010 and two tributaries September 15, 2010. Table 2 shows tributary stations, positions and date sampled; figure 5 illustrates station positions. At the time of measurement, a water sample (with blanks and duplicates) was collected to analyze for total phosphorus. The flow was then multiplied by the TP concentration to give a load in kg. The observed TP loads were used to compare modeled non-point loads as estimated by the water balance.

A water balance describes the flow of water in and out of a lake. In the simplest terms, the change in volume of a lake equals the inflows minus the outflows:

$$\Delta V = \text{Precip} - E + ET + I - (\text{In} - \text{Out}) \quad (9)$$

Where:

ΔV = change in lake volume (m^3)
Precip = precipitation (m^3)
E = lake evaporation (m^3)
ET = evapotranspiration (m^3)
I = runoff from watershed (tributaries) (m^3)
In = groundwater and inflow via inlet (m^3)
Out = outflow via the inlet (m^3)

We estimated the non-point flow by calculating I on a daily time step:

$$I = -\text{Precip} + E + ET - \text{In} + \text{out} + \Delta V \quad (10)$$

I was then multiplied by the mean observed TP concentration to give the monthly load in kg. Methods for estimating precipitation (Precip), lake evaporation (E) and evapotranspiration (ET) are discussed below.

3.4.2 Precipitation (*in atm*)

No climate station is located within the Lake Wolsey watershed. Daily precipitation (mm) for 2007 was downloaded from the National Climate Data and Information Archive of Environment Canada for Gore Bay, Ontario, Climate ID 6092930 (N45.88 W82.57) which is located 3.5 km north of the watershed boundary. Volume of daily precipitation was calculated by multiplying the precipitation (mm) by the surface area of Lake Wolsey (mm²). Phosphorus concentration in the precipitation was collected and analyzed by Environment Canada's Ontario Fresh Water Quality Monitoring Section. Environment Canada's station is located on Burnt Island approximately 28 km from the western watershed boundary (45° 49' 42" N / 82° 56' 53" W). The monthly P load in kg was estimated by multiplying volume of precipitation falling on Lake Wolsey by the P concentration in precipitation as reported by Environment Canada.

3.4.3 Lake Evaporation

Evaporation rates were not available for Lake Wolsey; therefore monthly lake pan evaporation normals from 1971 to 2000 were downloaded from the National Climate Data and Information Archive of Environment Canada for Ottawa, Ontario, Climate ID 6105976. Volume of lake evaporation (mm³) was estimated by multiplying the surface area of Lake Wolsey (ha) by the monthly evaporation rate (mm).

3.4.4 Evapotranspiration

Evapotranspiration (ET) is difficult to measure; therefore models of varying complexity have been developed to estimate ET. We have chosen to estimate actual ET by implementing the Water Balance Tabulation Model from the Hydrometeorology Division of the Canadian Climate Centre, Environment Canada (Johnstone and Louie 1983). This model estimates Potential Evapotranspiration (PE) and Actual Evapotranspiration (AE) from a water balance where water gains and losses and changes in water storage are estimated at a certain location.

Air temperature, precipitation, water holding capacity of the soil and the geographical coordinates of the climate station are the only required inputs. Potential evaporation is estimated based on Thornwaite and Mather's (1955) revised water balance. The potential evaporation (PE) is the amount of water that could be possibly lost to the atmosphere given sufficient water in the soil at all times for use by vegetation (Johnstone and Louie 1983). The PE is estimated as:

$$PE = ADJ * 0.533 (10 T/I)^A \quad (11)$$

Where:

ADJ = adjustment factor to correct for length of day

T = daily mean temperature (°C)

$$A = 6.75 * 10^{-7} * I^3 - 7.7 * 10^{-5} * I^2 + 1.79 * 10^{-2} * I + 0.49$$

I = Thornwaite heat index:

$$I = \sum_{j=1}^{12} I_j = \sum_{j=1}^{12} (T_j/5)^{1.514}$$

T_j = mean monthly temperature for month j

The actual evapotranspiration (AE) is the amount of water lost to the atmosphere through evaporation and transpiration on a vegetated surface. “When the total available free water equals or exceeds the PE for the period, the AE is set to equal the PE. When the free water is less than the PE for the period, water is drawn from the soil storage to satisfy the evaporative demand” (Johnstone and Louie 1983).

The air temperature and precipitation were downloaded from the National Climate Data and Information Archive of Environment Canada for Gore Bay, Ontario, Climate ID 6092930 (N45.88 W82.57). A geospatial soil survey shape file was downloaded from the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA). The shape file was based on a report “Soil Survey of Manitoulin Island” by Hoffman et al (1959). Farmington loam was the dominant soil type covering approximately 56% of the Lake Wolsey watershed. The Farmington loam in this watershed is derived from the underlying limestone and is very shallow (<30cm) in most areas (Hoffman et al. 1959). Because of the calcareous nature of the soil we chose a water holding capacity of 180 mm (Kirkwood et al. 1983). Once the water balance tabulation model was implemented, the resulting weekly AE was multiplied by the watershed area to give a monthly total evapotranspiration.

3.4.5 Groundwater (*ingw*)

Although Lake Wolsey has no major tributaries, groundwater discharge has the potential for a significant P loading to the lake. SRP (the bio-available form of phosphorus) in groundwater can range from ≤ 20 to 80 ug/L (Flores-Lopez et al. 2005; Young and Briggs 2008) in agricultural areas.

Groundwater flows were estimated using seepage meters and peizometers. Seepage meters are used to quantify groundwater flow at the sediment-water interface. Seepage meters in this study were constructed using five gallon plastic pails cut in half (fig. 6A). A series of tubes and valves connect a plastic bag to a pail. The plastic bag was filled with 500 ml of distilled water, placed in a protective metal container, anchored to the sediment and then connected to the pail via tubes (fig. 6B). The pail was inverted and pushed to approx 2 cm above the sediment. A valve was opened to release the lake water then time and date recorded. A hydraulic head gradient (positive or negative) in the sediment forces groundwater into the bag or water out of the bag indicating a discharge or recharge zone. The assumption being that the hydraulic head inside the seepage meter is equivalent to outside the seepage meter. Upon retrieval, date, time and volume of water in each bag were recorded.

Three seepage meters were deployed on the south shore of Lake Wolsey on May 5, 2009 (45° 46' 51" 82° 32' 33") while two seepage meters were installed on the North West shore on May 6, 2009 (45° 49' 16" 82° 33' 08"). All meters were placed at depth of 0.2 m. All meters were retrieved on May 7, 2009. Flow rates were calculated as m³/day. We assumed the entire shoreline of Lake Wolsey was a discharge zone for groundwater, therefore we multiplied the flow results from the seepage meters by the perimeter of the lake to estimate total groundwater flow.

A peizometer is a plastic pipe with a series of small slits that is inserted into soil and or sediment to the depth of the water table to collect groundwater samples and measure the hydraulic head. We installed 3 peizometers approximately 5 m apart from the lake to land (fig. 7) (Table 1). Samples were collected using a peristaltic pump at peizometer #1 and #3 (fig. 7). Samples were then sent to the NLET in Burlington, Ontario to analyze for TP.

TP loads from groundwater were then estimated by multiplying TP concentrations from the peizometers by the total flow of groundwater to the lake as estimated by the seepage meters. Groundwater loads were then estimated on a monthly time step.

A seepage area, approximately 700 m long, on the east shore of Lake Wolsey appears to discharge from adjacent karst formations. The seepages were identified by sheet flow exiting the rock and discharging at the shoreline and an abundance of water cress (*Nasturtium officinale*) (fig. 8) which are known to grow in flowing water such as springs (Michaelis 1976; Kaskey and Tindall 1979). Water samples were collected at three sites along the seepage area (Table 2). The samples were analyzed for TP by NLET. Flows of the seep were difficult to quantify, therefore we assumed the flow to be equivalent to the observed flow of a small tributary located 3 km North West of the seep. The estimated flow was multiplied by the length of the seepage area to determine the total discharge (m^3). The volume was multiplied by the mean TP concentration of the three seepage samples (mg/m^3) to give the load (kg). Seepage loads were then added to the seepage meter loads to give a total monthly groundwater load.

3.4.6 Dwellings (*indwell*)

ESRI® ArcMap™ 9.3 and 10.0 was used to analyze topographic CanVec shapefiles from Natural Resources Canada (2008) to determine the number of dwellings within the Lake Wolsey Watershed. Dwellings were classified as “buildings”, “campgrounds”, or “resorts”. No distinction in the shapefiles was made between permanent and seasonal residents. Campgrounds were assumed to have nine campsites. The seasonal summer months were assumed to be July and August. We used an estimated TP loading of 0.66 kg/Person/year based on Paterson et al (2006) and 2.6 people per household in 2006 (Statistics Canada 2010). TP loads were calculated by multiplying the number of dwellings, camps or campsites by the number of people per household by the TP load/person/year to give a total load. Loads from camps and campsites were estimated for July and August only.

3.4.7 Internal P load (*inint*)

Internal P loading may be significant in lakes which experience periods of anoxia in the hypolimnion. Lake Wolsey generally experiences anoxia anywhere from mid July to October. Internal P was calculated using the following equation (Nurnberg 1984):

$$L_{int} = \text{anoxic area} * \text{anoxic period} * \text{P release rate} / \text{lake area} \quad (12)$$

Where:

L_{int} = internal P load mg/m²/day

Anoxic area= m²

Anoxic period = days

P release rate= mg/m²/day

Lake area = m²

The area of the anoxic zone was estimated by determining the depth contour at which dissolved oxygen (DO) levels decreased to below 1 mg/L from oxygen profiles collected

by a YSI 6600 in July 2006 and August 2007 on a longitudinal transect of Lake Wolsey. The anoxic period was estimated to be from mid July to mid Oct. The anoxic period can vary from year to year depending on primary driving forces such as precipitation, temperature and wind. The P release rate was taken as 12 mg/m²/day as was estimated from a median of 14 lakes reported by Nurnberg (1984).

3.4.8 Leaf Litter (*inleaf*)

Leaf litter was assumed to be 260g/m of shoreline based on Dillon et al (1986) and Hanlon (1981). We used a phosphorus content of 0.146% reported by Polyakovac and Billor (2007). We assumed leaf litter for the entire shoreline of Lake Wolsey; therefore leaf litter weight (g/m) was multiplied by the perimeter of Lake Wolsey (m) then multiplied by the %P to give a total load. Leaf litter loading was assumed to begin on September 1 and end on October 31. Loading was calculated on a monthly time step.

3.4.9 Lake sedimentation flux (*outsed*)

A sediment trap was deployed to estimate the sedimentation rate in Lake Wolsey. One trap was deployed at station 226 (45°48' 31" 82° 31' 59") at a depth of 20 m (fig. 9) on Aug 14, 2007. The trap was refurbished and re-deployed on Aug 18, 2007 with final retrieval on Aug 23, 2007. Each sediment trap consisted of 4 one meter cellulose acetate butyrate tubes with 250 ml plastic cups tightly fitted over the outside of the base of the tube. All 4 tubes had a drain hole approximately 20 cm from the base where the holes were taped before deployment. All 4 tubes with cups were placed in a metal bracket and taped together. The base of the bracket was attached to two pieces of railroad track for

weight and the top of the bracket was attached to a length of ¼ inch stainless steel air craft cable to a subsurface float to keep the entire mooring up right. A surface float was attached to the subsurface float with 5/8 inch rope to allow for easy refurbishing. Upon refurbishing, the tubes were taken from the brackets, drained; cups with sediment were removed and capped. Clean tubes and cups were attached to the bracket and re-deployed. After sediment settled in the cups the water was decanted and remaining sediment was combined into a pre-weighed 150 ml plastic container. The samples were freeze dried, and re-weighed. The following equation was then used to estimate sediment rates:

$$S_{\text{rate}} = W_{\text{dry}} / A / \text{Exp} \quad (13)$$

Where:

S_{rate} = sedimentation rate (g/m²/day)

W_{dry} = freeze dried weight of sediment (g)

A = total surface area of the tubes (m²)

Exp = exposure time (days)

The remaining sediment was submitted to NLET for TP analysis (mg/kg). The sediment TP load was estimated by multiplying the TP in sediment (mg/kg) by the sedimentation flux (mg/m²/day) then by the area of the depositional zone (m²). The sedimentation load was used to as sink and to calibrate the mass-balance model.

3.4.10 Sportfishing (*outsport*)

No creel census has been completed on Lake Wolsey. Alternatively, perch (*Perca flavescens*) tournaments periodically take place during the year. It was estimated that 200 perch per day were taken from Lake Wolsey between January 15 and March 15 for a total of 11800 (C. Whitty, September 10, 2009, personal communication). The perch caught

ranged between 18 and 33 cm. This was multiplied by an average weight of 181 g (Lorantas et al. 2006). The TP content in adult yellow perch is estimated as 2% of dry weight. The dry weight to wet weight ratio is 0.25 (Kraft 1992). The TP in perch is thus estimated as:

$$TP_{\text{fish}} = W_{\text{dry}} * 2\% \quad (14)$$

Where:

TP_{fish} = mass phosphorus in perch (kg)

W_{fish} = dry weight of perch (kg)

The TP_{fish} was then divided by 59 days to give the daily TP in kg. We assumed the equivalent fishing effort for May to November, therefore monthly TP was estimated by adding the daily TP as estimated for fish caught in the winter.

3.5 Observed Total Phosphorus Data

Validation of the mass-balance model was achieved using observed monthly TP lake concentrations from Environment Canada and the Ontario Ministry of Environment. Figure 9 shows all lake stations sampled in 2007. TP lake concentrations were not available for every month in 2007, therefore mean concentrations from 2006 were used for July, October and November and mean concentrations from 2010 were used for June and September. Despite some inter-annual variability, there has been no significant change in TP lake concentration between 2001 and 2010 (fig. 10), therefore substituting alternate years will not introduce significant error in model calibration. Integrated sampling is generally the preferred method for collecting composite water samples to give an accurate representation of the water body at that particular location and time.

Unfortunately, Environment Canada and the MOE did not sample integrated TP in July, August or September 2006 or 2007 and integrated TP data collected by the farm operator was unreliable and therefore was not included in the analyses (Linguist, 2009, personal communication). The only available data was 1m and b-2m from Environment Canada. TP data from 1m and b-2m was not averaged July, August or September because internal TP load would introduce significant bias, therefore 1m only was used. Table 3 describes months, depths samples, year and source used to estimate monthly TP observed means. Table 4 shows station numbers, positions and total depths. Stn 229 has a total depth of <5 m and periodically subjected to resuspension events, therefore was not included in calculations of TP means.

Environment Canada sampled ten stations at the surface and bottom-2 m in Lake Wolsey and one station in Campbell Bay approximately 1 km north of the causeway in May and November 2007 (fig. 9). A 3 litre VanDorn (a water sampler) was used to collect discrete water samples. Water samples were collected into 4 l acid washed plastic jugs. Water samples were prepared following the NLET protocols (Operational Analytical Laboratories and Research Support Water Science and Technology Directorate Environment Canada 2010). An aliquot of unfiltered sample water was fixed with 30% sulphuric acid. The prepared samples and duplicates were analyzed by the NLET in Burlington, Ontario. A summary of the methods are located in appendix 3.

3.6 Uncertainty Analysis

An uncertainty analysis is used to account for error (or uncertainty) in model parameters. Because ecological systems are complex, calibrated model parameters do not necessarily represent reality, therefore a range (or error) for each parameter must be included in model estimations. A Monte Carlo simulation was implemented in this study to determine uncertainty range of the estimated lake TP concentrations. Each model parameter (sources and sinks) was assigned a range based on a literature review or mathematically by running various scenarios on uncertain parameters. For example, apparent feed wastage (taken as 2%) is not explicitly known therefore the bioenergetic model was run using 1% and 3% to establish an uncertainty range for that parameter. A random distribution of 10,000 values was created for each parameter based on the uncertainty range. The mass-balance model, equation (2), was run using MatLab[®] V6.1 10,000 times to establish the uncertainty distribution. Appendix 5 shows the TP range (kg) assigned to each model parameter.

3.7 Sensitivity Analysis

A sensitivity analysis was applied to determine how important each source and sink of TP is to the overall TP concentrations of Lake Wolsey. Each estimate of source and sink of TP load was increased and decreased 100%, then run using equation (2). The resulting mean concentration of the increased or decreased parameter was compared to the estimated mean lake TP concentrations (no change of parameters) to determine which sources and sinks were most sensitive to the overall concentrations of TP in Lake Wolsey. Microsoft[®] Office Excel 2003 (Redmond, Washington, United States) was used

to conduct the sensitivity analyses. Statistical significance was determined using the Mann-Whitney Test for non-parametric data using statistical software OriginPro[®] v8.6 (OriginLab©, Northampton, Massachusetts, United States).

4.0 Results

4.1 Inlet/outlet exchange TP load (*inex*, *outex*)

Flows in and out of Lake Wolsey were highly variable as shown in figure 11. Flow reversals can occur up to 5 times per day. During quiescent conditions tidal fluctuations of a period of 12.5 hours were evident co-oscillating between Lake Wolsey and Campbell Bay as shown by the modeled velocities in figure 12. During wind driven events, flows become extremely variable. For example, a wind event began on August 12, 2007 with maximum gusts of 61 km/h and continued after August 15, 2007 blowing predominantly from the southwest (fig. 13). The velocities during this time increased from ~ 0.38 m/s (-0.51 m/s) to ~ 1.5 m/s (-1.4 m/s) (fig. 12). Figure 14 shows the wind direction during the study period predominantly blew from the North for 25% of the time with an average wind speed of 10 to 20 km/h and maximum 30 to 40 km/h followed by Southwest at 14.5% of the time with a wind speed of 10 to 20 km/h and maximum >40 km/h.

Maximum modeled inflow and outflow from mean hourly velocities during the study period was -1.6 m/s and 3.14 m/s, respectively while the mean modeled volume inflow and outflow was -19.34 m³/s (± 14.6 m³/s) and 30.19 m³/s (± 32.3 m³/s) with maximum flows reaching -111.5 m³/s and 212.8 m³/s. According to a Kendall Tau test, significant flows rates leaving Lake Wolsey were evident in May (mean of 30.05 m³/s ± 14.5 , $p < 0.001$) inferring the discharge of the spring freshet (fig. 15).

Figure 16 shows a histogram of modeled hourly TP loads flowing into (negative) and out of Lake Wolsey (positive). Normally distributed flow rates are evident with the highest

frequency inflow load of -0.25 kg occurring 19% of the time whereas the highest frequency outflow load was 0.5 kg occurring 7% of the time with notable loads of >2 kg occurring 6% of the time. Mean daily TP loads entering Lake Wolsey from Campbell Bay ranged between 0 and -18.5 kg/day with a total load entering Lake Wolsey of 539 kg during the study period. Mean daily TP loads discharging from Lake Wolsey to Campbell Bay ranged from 0 to 39.18 kg/day with a total load of 802 kg exiting Lake Wolsey to Campbell Bay. The estimated net TP was 262 kg exiting Lake Wolsey from May to November 2007.

The mean modeled outflow was 30.2 m³/s. As this occurs half the time, the flushing rate was estimated using an outflow of 15.1 m³/s. Using a volume of 260,680,125 m³, the flushing rate is 199 days. This estimate should be used with caution as the outflow represents the epilimnion water only.

4.2 Validation of Inlet Hydrodynamic Model

Validation of the modeled flow rates were compared with observed mean flow rates from an Acoustic Doppler Current Profiler (ADCP) (fig. 17). The modeled flow rates were 72.87, -48.90 and 77.15 m³/s at 10:30, 13:10, and 15:10 EST on September 16, 2010. The mean flow rates from the ADCP during the same time period were 50.80, -31.02 and 62.55 m³/s. The modeled and observed flow rates compared reasonably well with an r² of 0.79. Generally, three data points are not sufficient for validation, however, ADCP and field personnel were only available September 16, 2010. Despite this, the ADCP data

does show that the model performed as it should showing the flow reversals. Future studies should include a longer validation period.

4.3 Farm Load (*infarm SWP/DWP/AFWP*)

The estimated total biomass net gain (adult and fingerlings) between May and November was 280,448 kg (table 2). Total feed applied to the farm from May to November 2007 was 354820 kg. The resulting FCR was ~1:1.3 (354820 kg / 280,448 kg) (Table 2). Daily feed records show intensive feeding occurred during the month of July (64300 kg) and October, 2007 (57090 kg) (fig. 18)

The FCR was then used to estimate the solid waste phosphorus (SWP) and dissolved waste phosphorus (DWP) using the existing bioenergetics nutrient mass-balance model by Bureau et al (2003) (equation 6 and 7). The TP in fingerling and adult feed was 9 g/ kg and 10 g/kg, respectively. A mean of 9.5 g/kg TP was used to estimate the SWP and DWP. The monthly SWP, apparent feed waste phosphorus (AFWP), and DWP are shown in figure 19 and Table 6. The SWP ranged between 113 and 222 kg per month with highest load in July. The DWP was approximately 50% less than the SWP where the load ranged between 59 and 129 kg/month.

The total estimated SWP that is discharged to the receiving water was 1383 kg (table 6). This estimate is not realistic for the mass-balance model, as not all the SWP is available for bio uptake in the water column. Most the SWP is deposited on the sediment where it is assimilated by the benthos or is inert as apatite P from bone in fish meal. A smaller

portion of the SWP is leached as soluble reactive P available for biouptake. Garcia-Ruiz and Hall (1996) reported that 0.216 kg of soluble reactive phosphorus per ton rainbow trout produced leached from solid fecal waste and Phillips (1993) concluded 4 to 10% of the total solid waste phosphorus produced from Atlantic Salmon was leached as dissolved phosphorus. Further to this, Kelly (1993) reported 1.5 to 57.6 mg/m²/day of dissolved phosphorus leached from solid waste phosphorus beneath cages of Atlantic Salmon. We have assumed that 10% of the solid waste phosphorus is leached as dissolved waste phosphorus. Therefore the total measurable phosphorus load from the farm becomes 915 kg: leached dissolved P portion of SWP (138 kg) + AFWP (63 kg) + DWP (714 kg). These estimates were used in the mass-balance model to predict mean lake TP concentration and also used in the sensitivity analyses.

The SWP derived from the bioenergetic model was an order of magnitude more than TP loads derived from a sediment trap centered beneath the cages (6.8 and 0.71 kg/day, respectively). The discrepancy may result because solid waste beneath the cages does not fall vertically due the generally flocculent nature of fish feces, density characteristics of fish feed and the hydraulic regimens under the cages (Cromey and Black 2005).

Table 6 shows the total monthly load of TP (kg) from the farm (SWP + AFWP + DWP). July has the highest load TP (391 kg,) while May had the lowest load (177 kg).

4.4 Tributaries: Observed Flows and Loads

Instantaneous current readings and TP samples were collected from up to 10 tributaries April/May and November 2009, June and September 2010. The minimum and maximum flow recorded was $<124 \text{ m}^3/\text{s}$ and $163,578 \text{ m}^3/\text{day}$. The tributaries in this watershed are ephemeral where no flows were recorded in three tributaries in June 2010 and 8 tributaries in September 2010. TP concentrations were highly variable ranging between 2.7 and 166 ug/L with a mean of 26.6 ug/L ($\pm 40.7 \text{ ug/L}$). TP loads ranged between 0 and 5 kg/day with a mean of 0.33 kg/day ($\pm 0.94 \text{ kg/day}$)

4.5 Waterbalance: Non point source loads (*inNP*)

Because of the unique geology of the Lake Wolsey watershed, which is dominated by dolomite, limestone, alvars, karst geology and thin top soil (Karrow 1993), and coupled with the small and ephemeral tributaries, it is extremely difficult to measure and estimate non point source TP loading. Precipitation events result in simultaneous runoff causing sheet flow in some areas adjacent to the lake (fig. 20). Therefore, we chose to use a water balance to more accurately estimate the non point TP loads.

Figure 21 shows the precipitation during the study period. Total annual precipitation was 600 mm. The monthly volume of rainfall on Lake Wolsey ranged between $6.2 \times 10^5 \text{ m}^3$ (May) to $> 2.5 \times 10^6 \text{ m}^3$ (October). Lake evaporation ranged between $8.3 \times 10^5 \text{ m}^3$ (November) to $>3.0 \times 10^6 \text{ m}^3$ (July) while evapotranspiration ranged between $1.8 \times 10^6 \text{ m}^3$ to $11.1 \times 10^6 \text{ m}^3$. Observed groundwater discharge flow was assumed to be constant at $7.0 \times 10^3 \text{ m}^3$ per month.

Figure 22 shows the daily non point source and observed tributary TP loads. Generally the highest loads occurred in May ($11.3 \text{ kg} \pm 3.4$) while lowest loading occurred in November ($0.8 \text{ kg} \pm 1.1$). Observed tributary TP estimates were lower than modeled estimates with an r^2 of 0.5 primarily because most of the runoff was not captured in tributary flow. Monthly non point source TP load ranged between 328 kg ($\pm 3.3 \text{ kg}$) in May and steadily decreased to 20 kg ($\pm 1.8 \text{ kg}$) in November.

4.6 Atmospheric Load (*inatm*)

Total precipitation during the study period was 600 mm with maximum precipitation occurring in October (117 mm). Monthly atmospheric TP loads ranged between 2.3 kg in June to 48.0 kg in September (fig. 21).

4.7 Groundwater (*ingw*)

Estimated groundwater/seepage TP load was estimated as 1.45 kg/day. This result may be a severe underestimation, as based on observations in this study, numerous surface and subsurface groundwater discharges were noted around the perimeter of Lake Wolsey which are difficult to quantify. Figure 23 shows an example of one such groundwater discharge near the northeast shoreline of Lake Wolsey.

4.8 Lake Sedimentation (*outsed*)

The estimated sedimentation rate was $2.0 \text{ g/m}^2/\text{day}$. The TP in sediment was analyzed as 1100 mg/kg resulting in a sink of $2.22 \text{ mg/m}^2/\text{day}$. The sedimentation load was used to

calibrate the mass-balance model because the actual depositional area was unknown. The total load from May to November was 963 kg (table 7).

4.9 Other minor TP contributions: Dwellings/Leaf Litter/Internal P/Sportfishing

Load (*indwell, inleaf, inint, outsport*)

Leaf litter and dwelling load was 0.13 kg/day and 1.0 kg/day, respectively whereas internal TP load was 1.9 kg/day. Sportfishing of primarily Yellow Perch (*Perca flavescens*) resulted in an extraction of TP of 0.14 kg/day. Monthly loads are shown in table 7.

4.10 Contributions of TP loads to Lake Wolsey

Figure 24 (and fig. 3) is a particularly important graph showing that non point sources (watershed) are the leading contributors of TP loading to Lake Wolsey at 40% or 1120 kg (May to November). This was followed by the farm (32%, 915 kg), groundwater (11%, 305 kg), dwellings (8%, 219 kg, internal P (7%, 186 kg), atmospheric P (3%, 79 kg) and leaf litter (0.3%, 8 kg) (fig. 24). Results of the high level modeling performed in this study, give a first indication to lake managers and regulators that the farm is not the leading contributor of TP in this Type 2 lake. Future management decisions may need to engage watershed stewardship as part of a management strategy.

4.11 Predicted Lake TP concentration and Uncertainty Analysis

Figure 25 illustrates the mass-balance estimates of monthly lake TP concentrations, 2007, and uncertainty analysis. The estimated TP concentrations correlated well with observed values ($r^2=0.85$, Nash-Sutcliffe = 0.85). A steady increase in TP concentration (~ 0.8 ug/L/month) was evident from May until September. The estimated mean TP concentration for the study period was 10.76 ug/L (± 1.8 ug/L). Figure 26 shows a frequency distribution curve of the mean TP lake concentration as result of the uncertainty analysis where the observed mean seasonal TP lake concentration of 10.3 ug/L falls within 2 standard deviations of the estimated mean (10 and 11.5 ug/L).

4.12 Sensitivity Analysis

An increase in each source and sink by 100% resulted in the non-point TP load to be the most sensitive parameter and marginally significant ($p=0.05$) to Lake Wolsey (14 ug/L) compared to the estimated mean lake TP concentration (10.76 ug/L) (fig. 28). This was followed by the lake sedimentation (8.6 ug/L) then the contribution from the farm. When the total farm load was increased 100% (SWP + AFWP + DWP), simulating the addition of a farm, the mean lake TP concentration increased from 10.76 ug/L to 12.60 ug/L but was not significantly different from the mean lake TP concentration before the increase ($p=0.22$). This was followed by the inlet load out and into Lake Wolsey, groundwater, dwellings, internal phosphorus, atmospheric load, sport fishing, leaf litter and (fig. 28 and table 8).

The estimated SWP (138 kg) from the farm was increased and decreased by 100% while keeping the remaining parameters constant to simulate the best and worst case scenario of

farm loads. The worst and best case scenario resulted in a mean lake TP concentration of 10.9 ug/L and 10.4 ug/L respectively and was not significantly different from the estimated mean lake TP concentration of 10.76 ug/L.

Similarly, when parameters were removed from the model (0%), the non point sources were the most sensitive with a mean lake TP concentration of 7.5 ug/L ($p=0.007$) followed by lake sedimentation (12.8 ug/l), then the farm (8.7 ug/L) (fig. 29). Again, the farm loads were not significantly sensitive to Lake Wolsey ($p=0.02$). This was followed by inflow and outflow, groundwater, dwellings, internal phosphorus, sportfishing, leaf litter and atmospheric load (fig. 29 table 8).

5.0 Discussion

Modeled monthly TP concentration in this study closely resembled observed results ($r^2 = 0.85$) indicating no other significant loss of TP other than the considered parameters. As opposed to other studies, modeling in this study had better agreement because 1) a reasonable assumption can be made that the lake has reached steady state because the farm has been in operation since 1986 with generally consistent production and solid waste deposition; no significant changes to the watershed have been evident for the last three decades; and there has been no significant change in mean lake TP concentration 2) we included all feasible estimated sources of loading to make the modeling more robust; 3) observed lake TP concentrations have smaller inter annual variation ($CV = 14$ to 30%). Therefore this modeling can be used as tool for lake managers and regulators to predict TP in type 2 sites for existing and future farms as long as it is calibrated to the lake in question.

Current water quality guidelines for monitoring TP, has recommend that samples be collected at 30 m from each corner of the cage operation or “mid-point of the three sides of cage complex if they are attached to shore..... and two local reference stations” periodically through the ice-free season (Boyd et al. 2001). If a trigger of 10 ug/L TP is exceeded, further mitigation of farming operations is required, such as reduced feed quotas and increased operational efficiency (Boyd et al. 2001). Type 2 sites such as Lake Wolsey may receive “special consideration and consultation” since the pre farm mean spring lake TP concentration was 11 ug/L. In fact the MOE stated Lake Wolsey had “good water quality and development capacity to sustain the cage operation” (Gale 2000).

The above recommendations work well if the farm is the only source of TP in the lake. However, to accurately assess the mean lake TP concentrations, a holistic approach must be considered. For example, if a sample is collected mid-lake (reference station) and analyzed for TP from a Type 2 site what proportion of TP is from the non point sources, the dwellings, the farm etc? This is an important question managers and regulators need to ask in order to effectively assess and monitor type 2 sites. For example, this study shows the non point source (watershed) to be the leading loading source of TP contributing 40% and was significantly sensitive ($p=0.05$) (fig. 24, fig. 28). This is not surprising given that approximately 34% of the watershed land use is agriculture. Salazar-Hermoso (2007) attempted to use unique isotopic signature in feed to identify aquaculture waste in Lake Wolsey. While the laboratory evaluation was successful, the field studies in Lake Wolsey failed to show the link of farm waste in the sediment. The author concluded that “ecological and methodical effects” may have been responsible for the lack of isotopic signature from feed in the waste sedimented adjacent to the farm. Farm waste is potentially assimilated by benthic invertebrates, zooplankton and fish, therefore unavailable for phytoplankton uptake.

Studies have shown that most of the TP wastes from farm sites in the solid phase are rapidly delivered to the hypolimnion (Azevedo et al. 2011; Bristow et al. 2008). Azevedo (2011) reported lower observed dissolved P than expected near the farm, as estimated by mass-balance, as a result of possible adsorption to particles and uptake by bacteria and algae with removal by sedimentation. Further to this, TP concentrations in fresh water >30 m from the cages are near background lake concentrations (Reid et al. 2006; Milne

and Charlton 2007; Kelly 1995; Azevedo et al. 2011). This coupled with the relatively short epilimnion retention time in Lake Wolsey, is evidence that the recommended sampling may be redundant during the stratified season. Under the current recommendations, if a significant difference between the reference and near farm TP is evident, regulators may erroneously assume that the farm emissions are affecting the entire lake. This may lead to unnecessary economic losses for the farmer through farm mitigative procedures including lowering stocks and feed quotas. Monitoring efforts should be shifted to include a holistic approach by increasing spatial monitoring of the lake, in addition to monitoring non point sources for example, agriculture.

Sensitive and non sensitive sources and sinks of TP will be discussed below.

5.1 Sensitive parameters

5.1.1 TP loading from Non point sources (*inNP*)

Transportation of phosphorus from the watershed to the lake is typically by adsorption of phosphorus to soil particles (Wang et al. 2010). Sources of P originating from fertilizer, animal and human waste are generally bioavailable and are of particular concern for eutrophication issues and lake management. Soil type, pH and infiltration rate will affect the load of TP reaching the lake. Lake Wolsey has an agricultural dominated watershed comprised of alvars (flat limestone) overlaid with a thin layer of topsoil with small ephemeral tributaries. Not all flow during a heavy precipitation event reaches the lake via tributaries rather the flow discharges directly from the land as sheet flow over the alvars around the perimeter of the lake. This may allow for a direct rout and increase of

phosphorus loads from the watershed to the lake where it would otherwise be taken up by biota or wetlands at the mouth of tributaries. This is potentially validated by the much lower TP loads observed in the tributaries vs modeled TP loads of non point sources (fig. 22). The non point sources contributed an estimated 44% (16.0 kg/km²) of the total load to Lake Wolsey during the study period. This is similar to loads reported by Winters (2002) where TP loads in the Lake Simcoe watershed, with similar agriculture land cover, ranged between 5 and 23 kg/km²/year. Observed tributary TP concentrations ranged between 2.7 and 166 ug/L. Four of 10 tributaries consistently exceeded the PWQO of 30 ug/l for Ontario streams during the course of the study. Of these tributaries, land use consisted of 28 to 75% agriculture (crop or pasture). Future management strategies should include increased spatial monitoring in Lake Wolsey, tributary monitoring, septic system inspections, erosion control, and ensure agricultural BMP's are in place.

5.1.2. TP loads/sink in Inlet/outlet exchange (*inex/outex*)

Despite a proportionally large load from the non-point sources (watershed), the long term mean lake TP concentration has not changed significantly over the last decade (fig. 10). This may be due in part to the importance of a vigorous exchange flow between Lake Wolsey and Campbell Bay. Forcing from wind events, primarily from the north, can cause counter clockwise flows of > 3.0 m/s and TP >2 kg/h exiting Lake Wolsey. Hamblin and Gale (2002) conducted a study on Lake Wolsey in 1999 to determine flushing rate to apply to a one box model for TP concentrations in Lake Wolsey. Inherent weaknesses in the modeling were 1) No real-time lake levels were observed in Lake

Wolsey; 2) No independent velocities in the inlet were collected to validate the modeled results; 3) No tributary, sedimentation or internal phosphorus loads were included in the model; 4) No distinction was made between SWP or DWP from the farm. The authors results showed that the resulting farm load would increase the mean lake concentration by 5 ug/L from a background concentration of 8 ug/L. Regardless, the authors concluded that at a production of 250×10^3 kg of fish and p load of 1352 kg the long-term effects of cage-aquaculture were probably minimal due to the relatively short epilimnion flushing time.

Modeled velocities in this study are higher than results reported by Hamblin and Gale (2002). They determined maximum modeled inflow/outflow velocities to be -1.6 and 1.3 m/s, respectively. In this study, modeled maximum inflow/outflow velocities were -1.6 m/s and 3.1 m/s. The mean inflow/outflow volumes of -19.3 m³/s and 30.1 m³/s were higher compared to Hamblin and Gale (2002) of 28.6 and 30.0 m³/s. The flushing time of Lake Wolsey in this study was 199 days, whereas Hamblin and Gale (2002) estimated 215 days. The discrepancies may be due to three differences in the studies: 1) a difference in the initial lake volume where in this study, lake volume estimate was 260,680,125 m³ and Hamblin and Gale (2002) used 263,910,000 m³. 2) The observed length and width of inlet was 30 m by 30 m and cross-sectional area of 67.7, whereas Hamblin and Gale used inlet dimensions of 45 by 45 m and cross-sectional area of 67.5; and 3) the authors used the inflow rate of 14.3 m³/s to estimate the flushing time due to lack of observed lake levels in Lake Wolsey whereas we used an outflow rate of 15.0 m³/s based on modeled results after calibration. They concluded that the farm was responsible for increasing the

mean lake TP concentration 5 ug/L above the background of 8 ug/L. This study shows the SWP + DWP + AFWP discharged to the receiving water was 915 kg which equates to ~2 ug/L increase.

It is widely documented that flushing time and retention rates are important in overall lake health (Dillon 1975; Osborne 1980; Vollenweider 1975). A lake with a high areal phosphorus load, high flushing rate and a low retention rate may have significantly less phosphorus concentration compared to a lake with low areal phosphorus loading and low flushing rate and a high sedimentation rate (Dillon 1975). Flushing rate may be important for Lake Wolsey. It is widely reported that cage-aquaculture sites located in well flushed areas tend to have better water quality (Silvert and Sowles 1996; Hargrave et al. 2005; Habitat Advisory Board of the Great Lakes Fishery Commission and Great Lakes Water Quality Board of the International Joint Commission 1999; Boyd et al. 2001; Johansson and Nordvarg 2002).

The flushing rate in Lake Wolsey is 199 days. Caution should be used as the flushing time of Lake Wolsey represents the epilimnion water only. However, most productivity occurs in the epilimnion due to optimum light and nutrient conditions for phytoplankton growth. The high flow rates through the inlet may provide a pathway for TP to leave the lake much faster than if a wider, or no inlet, existed. This is similar to Hamilton Harbour which is connected to western Lake Ontario by an 840 m ship canal. The estimated residence time of Hamilton Harbour before the construction of the canal ranged between 180 and 200 days (Barica 1989). After the construction of the canal the residence time

decreased to 73 to 107 days improving water quality. Since the residence time of Lake Wolsey is shorter than the growing season of rainbow trout, and the lake has assumed to have reached steady state, thus in theory, no long term increases in TP should occur. The lake may experience temporary seasonal algal blooms and hypoxic conditions during the ice-free season; however, no cumulative effects should be noticed. This is potentially validated by observed water quality data in this study, where no significant change in the lake TP concentration has occurred over the last decade (fig. 10). In fact, mean spring (May) observed TP concentrations post farm, 2004 to 2009, in this study (7.9 ug/L) was less than the pre farm concentration (11 ug/L) and the summer lake TP concentration between 2001 and 2010 (June, July and August) shows a declining trend, however, not significant ($p=0.09$) (fig. 27) (Gale 2000).

5.1.3 TP sink - Lake Sedimentation (*outsed*)

TP flux to the sediment is important for mass-balances, determining hypolimnion hypoxia/anoxia and discriminating changes in biological activity (Dillon and Evans 1993). Regardless of its importance, it is typically very difficult to quantify. Sediment traps, mass-balance approaches and core dating are generally used to estimate TP flux. In this study sediment traps were used; however, there is some uncertainty with sediment traps that may compromise estimations, those being: resuspension of bottom sediments, mineralization within the trap, or inadvertent increase or decrease of trapping particles through a reduction in horizontal velocity at the mouth of the trap (Dillon and Evans 1993). Because the actual depositional area was unknown we used the sedimentation

load to calibrate the mass-balance. We estimated the total loss of TP to be 963 kg and marginally sensitive to Lake Wolsey. In the future to eliminate some uncertainty, sediment core dating may be more applicable to accurately assess TP down flux.

5.2. Non sensitive parameters

5.2.1 Farm Solid Waste Phosphorus (SWP)/ Farm Dissolved Waste Phosphorus (DWP)

The FCR (feed use / weight gain) is needed to mathematically estimate the SWP and DWP using the bioenergetic mass-balance approach used by (Bureau et al. 2003). A general FCR was used in this study because the mass of feed given to specific masses of fish was unknown. Farm records show mass of feed fed per cage on a daily basis; however there is no record of periodic weights of fish in each cage to determine a more accurate net weight gain. Further to this, fish are frequently separated and moved from cage to cage with no records of which batch of fish are in each cage. Despite this, the estimated FCR of 1:1.3 was similar to that reported by Reid and Moccia (2007) and Azevedo (2011) where the FCRs at a freshwater aquaculture farm in the North Channel and Northern Ontario (Experimental Lake Area) ranged from 0.79 to 1.49 and 1.10 and 1.16, respectively.

The total estimated total solid waste phosphorus (SWP+AFWP) was twice the dissolved waste phosphorus (DWP) (1445.16 and 714.47 kg, respectively). This is in agreement with Azevedo et al (2011). The load of SWP and DWP typically follow a sinusoidal trend because trout are poikilothermic where temperature regulates metabolic rate, feed intake and growth (Moccia et al. 2010). The optimum temperature range of rainbow trout

growth is 9 to 14°C (Food and Agriculture Organization 2005) with significant growth occurring up to 22°C (Myrick and Cech 1996). Below the minimum temperature, the digestive system slows down and the fish eat less, therefore a maintenance diet only is required (Hinshaw 2012). In warmer water the fish tend to eat more. The excess nutrients and lower dissolved oxygen in warmer water may cause respiratory stress for fish therefore feeding is rationed to maintain optimum water quality and conserve feed (Hinshaw 2012). In this study, mean temperature for June and July (5 m depth, maximum loading) was 17°C and 20°C, respectively and May, August and November (min loading) was 10 °C, 22°C, and 8°C, respectively. Despite the optimum temperature in May the minimum loading resulted because the farm had the lowest standing biomass where most of the fingerlings were added to the farm after mid May. The estimated SWP and DWP load (kg) was proportionally less than that reported by Bristow et al. (2008) and Azevedo et al. (2011). The feed used in this study is a custom diet with low animal and high plant protein content. The feed formulation had less phosphorus ($0.95\% \pm 0.06$) and a higher ADC (55%) compared to Azevedo (1.1% P and 54%, respectively) and Bureau (2003) (~1.0% and 50%, respectively). However, despite the higher ADC and lower feed P, the comparatively higher FCR in this study resulted in higher SWP and DWP loading.

Feed formulations and feed wastage have greatly improved over the last four decades resulting in less P entering the environment per unit of fish produced. This is especially important for the DWP which may be more readily available for biouptake in the receiving environment whereas SWP is rapidly delivered to the hypolimnion. The DWP is excreted primarily in the urine. In the past, feed formulas had low protein, fat and

digestible energy but were elevated in starch, fiber and phosphorus (Bureau and Hua 2010). Feed formulation and feed production improvements have resulted in increased nutrient digestibility resulting improved FCR's. According to Bureau and Cho (1999), fish fed a diet with increasing digestible P resulted in a significant increase in DWP. Further feed and growth studies have resulted in much greater efficiency feeds. In the past, feed wastage was as much as 20% of total feed. More recently, feeding techniques have improved and feed wastage is ≤ 2 % (Meeker, MTM Aquaculture, 2010, personal communication). Improved feed formulations result in both improvements to the environment and lower costs to farmers.

Given current farming practices on Lake Wolsey and if all other sources of TP remain constant, it is unlikely that farm expansion or even the addition of a farm of similar production, will contribute to an overall decline in lake health. In fact, the farm SWP, AFW, and DWP could be increased three times before a significant increase in mean lake TP concentration occurs ($p=0.05$). An increase in non point sources by two times ($p=0.05$), for instance a growth in agriculture, could result in immediate and long term water quality impairment.

5.2.2 TP loads in groundwater (*ingw*)

Generally mass-balance studies consider groundwater to be a minor source of P loading, therefore are not included in nutrient estimations (Hakanson et al. 2003; Hakanson et al. 2004; Hakanson et al. 2004; Johansson and Nordvarg 2002; Dillon and Evans 1993).

Because groundwater was difficult to measure given the limited resources, and the observation of numerous groundwater discharges, we believe groundwater P loading may be more important to Lake Wolsey than what the results in this study show despite the load being less important than other parameters to the lake. Using a conservative estimate we found that groundwater was the third largest loading source 12% or 305 kg of the total P to Lake Wolsey. This may be due in part to the geologic landscape of the watershed. Groundwater discharges (springs) were observed along the east shore of Lake Wolsey. This area consists of karst formations formed by the dissolution of weakly soluble rock such as limestone by mildly acidic water (Karst Waters Institute 2011; Thornbury 1969). Formations can include, pitted limestone, fissures, caves, sinkholes, subsurface streams and springs. “Disappearing streams” or “swallow holes” are common in these areas where tributaries flow overland and descend into fissures in limestone to reemerge at a point some distance from the entry (Fetter 2001). In this study, we found a larger loading of TP in the spring water from the karst formations compared to samples taken from the peizometers in the south end of lake, despite the fact we found much higher concentrations of TP in the latter (554 and 188 ug/L, respectively) . This is because flows through karst formations tend to be higher due to higher hydraulic conductivity than deeper groundwater (Fetter 2001). The land use adjacent to the springs is 10% agriculture/87% forested, whereas the southern portion of Lake Wolsey is dominated by 63% agriculture/36% forested which may account for the increased TP concentration values.

Agricultural land use located near karst formations can have negative environmental impact where the formation allows for rapid flow of nutrients, bacteria and other toxins between surface and subsurface (within minutes) where typically the materials would be stripped by infiltration through soil (White et al. 1995). Further groundwater studies are needed to assess the nutrient implications of agriculture and karst and alvar formations on Lake Wolsey.

5.2.3 TP loads in Atmospheric deposition (*inatm*)

TP in atmospheric deposition occurs as dust particles and transported via wind where TP originates from various sources such as pollen, coal fired plants, forest fires, and agriculture (Winters et al. 2002). TP in atmospheric deposition can be a significant source of loading. For example, in an agricultural area, Winters et al. (2002) reported that atmospheric deposition accounted for 23 to 56% of the total TP budget for Lake Simcoe or 56 to 80 kg/m²/year. Dillon et al. (1986) reported much lower mean deposition loads of 37 mg/m²/year in small lakes on the Canadian shield. In this study, TP load in atmospheric deposition was quite variable with the highest loads occurring in September. The load accounted for 3 % of the total or a mean of 4.7 mg/m²/year (± 7.5 mg/m²/year) much less than Dillon et al and Winters et al. We may have underestimated the loading as the bulk collector was located on the north shore of Lake Huron ~25 km from western boundary of the Lake Wolsey watershed where much of the site is exposed to Lake Huron. Samples collected may not accurately represent the TP deposition over land. Despite this, atmospheric loading was not sensitive even when increased by 100%.

5.2.4 TP sinks in Sport fishing (*outsport*)

The primary sport fish in Lake Wolsey is Yellow Perch (*Perca flavescens*) (W. Salinger, Ontario Ministry of Natural Resources, Espanola, Ontario, May 6, 2011, personal communication). In 2007, estimates of ~12000 fish caught between January 15 and March 15 in a perch tournament (C. Whitty, September 10, 2009, personal communication) were similar to the Ontario Ministry of Natural Resources estimates of 19,000 fish caught between January 1 March 31 (W. Salinger, Ontario Ministry of Natural Resources, Espanola, Ontario, May 6, 2011, personal communication). These estimates were extrapolated to the entire study period. This is equal to ~32 kg of TP removed from the lake. No information was available for other species of sportfish or escaped trout. Despite this, sportfishing was not sensitive to Lake Wolsey even when the mass of TP was doubled.

5.2.5 TP loads in Leaf Litter (*inleaf*)

Leaf litter can be an important source of nutrients to oligotrophic lakes with small watersheds (Dillon et al. 1986). In large eutrophic lakes nutrients from leaf litter is generally not significant contributing 0.2 to 0.3% of primary production (Hanlon 1981). Nutrient input from leaf litter may also be important for benthic communities in lakes (Hanlon 1981). Deposition of leaf litter is largely dependent on the time of year, species of tree, distance from the shoreline and wind direction (Hanlon 1981). Hanlon (1981) found no significant leaf litter in the nearshore >10 m from the shoreline. Because it was

unknown in this study what species of trees were within 10 m of the shoreline, it was assumed the shoreline consisted of all deciduous trees and all leaf litter fell in the nearshore area. In this study, a predominant north wind will likely limit leaf litter fall into the lake along some areas of the shoreline. Despite the estimation in this study, TP from leaf litter was the least important contributor of all sources of TP (8 kg or 0.3%). Dillon (1986) reported similar results where leaf litter accounted for 3% of the total estimated TP in a small lake in central Ontario.

5.2.6 TP load from Internal Phosphorus (*inint*)

Generally sediments are known to act as a sink for phosphorus. Under anoxic conditions phosphorus may be released to the hypolimnion in several ways. The dominant process for P release is through reduced redox where iron-bound phosphorus is released when Fe (III) is reduced to Fe (II) and is the most bioavailable (Pettersson 1998). Increasing pH levels can decrease the phosphorus binding ability to Al and Fe where hydroxide ions replace P due to ligand exchange (Pettersson 1998). An increase in temperature may promote greater bacterial activity resulting in phosphate-mobilizing enzyme which increases oxygen consumption and decreases redox potential (Pettersson 1998). Implications for internal P loading can include entire lake water quality impairment where mixing of the water column due to weather conditions and internal seiche, cycles phosphorus to shallower depths, therefore making it available for bio uptake. This may have implications for restoration management, where despite reductions in external loads, lake water quality may fail to improve because of the greater internal loading.

Lake Wolsey experiences annual hypolimnetic hypoxia where dissolved oxygen is typically <2 mg/L and TP > 50 ug/L. The estimated internal P load accounted for 186 kg or 7% of the total load and was not sensitive to the lake implying that the internal loading is less important than external loading. This estimate was excessive, as the beginning and the end of the hypoxic period was explicitly unknown. The hypoxic area of Lake Wolsey was approximately 1.7% of the total surface area resulting in a load of 1.9 kg/day. Because of the shallow hypolimnion (<5 m) and relatively small area (Milne 2007), less of the hypolimnion is in contact with the sediment, therefore minimizing the internal P load (Nurnberg 1995). During fall turnover the internal phosphorus is mixed through the water column, however given the relatively short retention time no long term effects should be noticeable.

From a management perspective questions may arise on what drives the internal P load process. Generally, organic material sinks to the bottom sediments where bacterial decomposition occurs. Hypoxia/anoxia occurs when the BOD equals or exceeds the available dissolved oxygen and this in turn, provides a suitable environment for release of P. In Lake Wolsey it has been implied that solid waste loading from the farm may be a source of BOD contributing to the hypoxia/anoxia in the hypolimnion mid-lake. From this study and others this is likely not the case given that 40% of the nutrient load is from non point sources rather than the farm (32%). Further to this Salazar-Hermoso, 2007, found no link between isotopic signature in feed and fecal material adjacent to cages inferring other ecological processes may be responsible for minimal waste deposits such as microbial degradation or direct uptake by higher fauna. Even if waste deposition does

accumulate, most SWP from the farm will discharge directly under cages and bound in the sediment under aerobic conditions (Azevedo et al. 2011; Bristow et al. 2008) and is not transported to mid-lake.

5.2.7 TP load from Dwellings (*indwell*)

Human sources of TP loading include septic systems, lawn and garden fertilizer, addition/removal of fill, and agitation of nearshore sediments (Dillon, 1986). Retention of TP is generally achieved with septic systems, the popular types being a holding tank and holding tanks with tile beds. Theoretically, holding tanks have 100% TP retention while holding tanks with tile beds range from 5 to 99% ((Dillon et al. 1986). There are no urban areas in the Lake Wolsey watershed and many of the dwellings are seasonal. In this study, we found TP input from dwellings accounted for 8 % or 219 kg of total load. We assumed all dwellings in the watershed were occupied by 2.6 people for the entire study period, because it was unknown which dwellings were seasonal. We also assumed that all TP did in fact reach the lake, where generally movement of TP from septic systems via groundwater to surface water tends to be very slow (years to move 10's of meters) (A. Crowe, Environment Canada, 2009, personal communication). Regardless, the TP load from dwellings was not sensitive to Lake Wolsey when increased or decreased by 100%.

6.0 Strength and Weaknesses of the Mass-Balance Model

Models are generally used to create simplifications of reality in order to study systems.

Every model has strengths and weaknesses. The mass-balance model used in this study is a one box model to address higher level lake management concerns. Strengths include:

1) simplistic use: this model could easily be used by lake managers and regulators as a tool to make rapid decisions; 2) not data intensive: input data can be gathered relatively easily given the availability of data. For example, farm data including detailed feed usage and fish mass is generally available from the farmer, whereas other inputs such as land use (non-point sources), lake TP concentrations, tributary data etc, can be found in the literature or through collaborations with government agencies and universities. 3) inexpensive: simplistic models generally are more cost effective in terms of data acquisition, personnel and software to run the model: for example this model can be implemented using Microsoft Excel; 4) can be transferable to other Type 2 lakes: this type of modeling could be used to assess new sites before a farm begins operation. For example the sensitivity analysis could be used to determine an optimum production for that particular lake given other TP loads; or predict TP lake concentration if /when changes occur in the watershed (ie urbanization, increase/decrease in agriculture production, etc). The model could also be used to assess lakes with existing aquaculture which have not been assessed in the past or assess lakes where farming has ceased.

Weaknesses include: 1) the model will not estimate daily loads in order to understand processes that happen at a smaller temporal scale; 2) the model will not answer specific ecosystem questions such as ‘what is the assimilative capacity of Lake Wolsey?’ To address these weaknesses, a more complex multi box model is needed which is far more

data intensive, costly and requires more time and personnel. 3) only selected sources and sinks TP loads were used in this study. We did not consider, for example, the effect of invasive species such as gobies and zebra mussels and implications of TP cycling. 4) Best estimates of input parameters are needed to improve model uncertainty. Model results can be misleading if the best available data is not used or if quantification of a processes was achieved in the lab and may not represent a lake system.

7.0 Recommendations for Model Improvement

Hamblin and Gale (2002) conducted a similar study on Lake Wolsey using a very simplified one box model to predict increases in lake TP concentration due to the farm above back ground level. This study improved on Hamblin and Gales' modeling by including additional sources of TP; better estimates of the inlet exchange between Lake Wolsey and Campbell Bay; better estimates of farm sources, for example Hamblin and Gale did not partition the SWP and DWP farm waste. With the improvements to the modeling, increases in mean lake TP concentration as a result of the farm, was estimated to be much lower than that recorded by Hamblin and Gale (5 ug/L vs 2 ug/L, respectively). Despite this, further improvements to this model are recommended:

- 1) Groundwater - Estimates of groundwater TP loads in this study are vastly underestimated. In the future, implementation of a full scale groundwater survey would be needed to assess flow rates and TP concentrations discharging adjacent to Lake Wolsey.

- 2) Non point sources and tributaries – High level modeling used in this study showed non point sources to be the leading contributor of TP to Lake Wolsey. Further studies are needed to better quantify TP loads to support this finding. Because tributaries in this watershed are very small and ephemeral and very little top soil exists over flat limestone, perhaps a modeling approach such as (Canadian Watershed Evaluation Tool (CANWET) or Agricultural Non-Point Source model (AGNPS) would be best suited to give better estimates of loads.
- 3) Lake sedimentation flux – An alternative, more accurate, means to assessing sedimentation flux is sediment core dating. A spatial collection of sediment cores and dating would improve TP sink estimates
- 4) Improved farm records – Periodic recording of fish mass would improve estimates of biomass gain throughout the study period to improve FCR estimates. Using a nutrient mass-balance approach, SWP and DWP can be easily and confidently estimated using the FCR.
- 5) Extended ADCP validation of inlet modeling – An ADCP should be deployed for the same time span and intervals as level loggers to give an independent comparison for validation of inlet modeling.
- 6) Extend the sampling period through the winter season. Very few studies using mass-balance modeling in lakes with aquaculture include the ice covered period. Characterizing TP loads in the winter would improve modeling estimates.
- 7) Statistical analysis, for example, bootstrapping may be implemented to determine the number of spatial lake stations to account for TP variability in Lake Wolsey.

This information would aid lake managers in creating improved monitoring programs.

8.0 Summary and Recommendations

Current water quality monitoring programs on lakes with cage aquaculture, specifically type 2 sites, may not target the appropriate source of impact. Because lake managers favour simplistic models for rapid assessment, we have utilized a phosphorus mass-balance model and sensitivity analysis on a type 2 site to predict lake TP concentration and determine important TP contributors to Lake Wolsey.

Referring to the objectives in section 2.4 we found that:

- 1) the non point loads of TP were the most important source (40%, 1120 kg) whereas the farm was second most important (32%, 915 kg). This was followed by groundwater (11%, 305 kg), dwellings (8%, 219 kg), internal phosphorus load from the hypoxic hypolimnion mid-lake (7%, 186 kg), precipitation (3%, 79 kg) and leaf litter (0.3%, 8 kg). Further studies on non point sources are needed to support these results;
- 2) The non point TP loads were the most sensitive to Lake Wolsey followed by the lake sedimentation, the farm, the inlet load out and into Lake Wolsey, groundwater, dwellings, internal phosphorus load from the hypoxic hypolimnion mid-lake, precipitation, sportfishing and leaf litter.

- 3) Farm expansion of 100% would not significantly change the mean lake TP concentration of Lake Wolsey during the ice free season provided all parameters remain constant. However, caution should be taken as we considered 90% of the SWP to be inert in the sediment. Further studies are needed to determine the fate of the SWP once it enters the lake;
- 4) We determined from this study that future management strategies should include
- a) a public outreach component to educate residents of the watershed on implications of increased TP given the geology of the catchment;
 - b) spatial monitoring of lake water to increase statistical integrity and account for TP variability.;
 - c) increase frequency of sampling during the ice free season;
 - d) tributary monitoring, specifically during peak pulses of precipitation events to better characterize and to isolate areas of excessive TP loading within the watershed;
 - e) septic system inspections;
 - f) ensure agricultural BMP's are in place. Given the geology of the watershed and sensitivity of non point sources, BMP's are of utmost importance not only to farmers but also residences (ie lawn fertilizing);
 - g) ensure farm, both agriculture and aquaculture, adapt the most up to date sustainable management strategies available.

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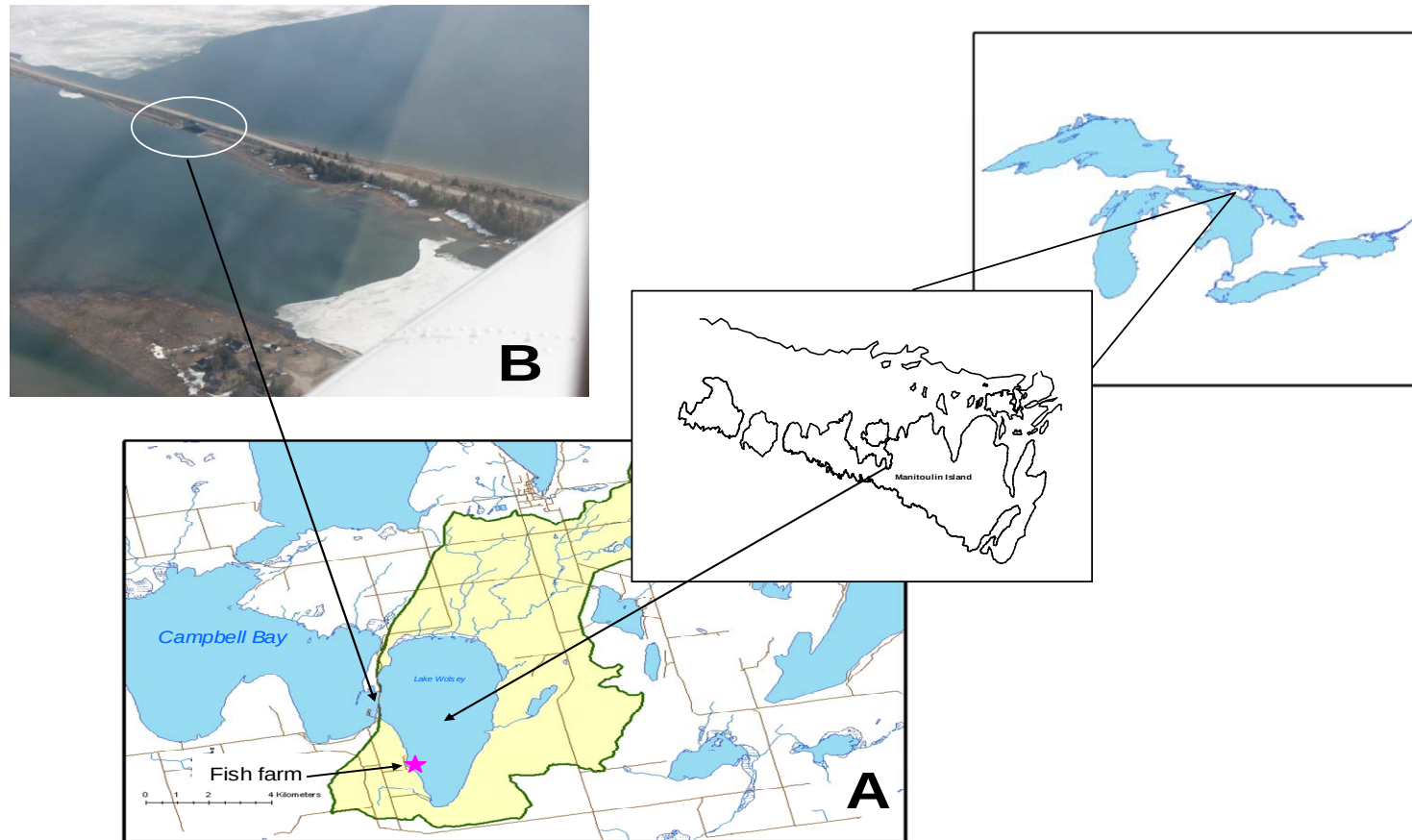


Figure 1 A) Location of Lake Wolsey on Manitoulin Island, North Channel, Northern Lake Huron, Ontario, Canada B) causeway with small canal joining Lake Wolsey with Campbell Bay, North Channel.



Figure 2 The cage array of seasonal and permanent cages with location (star) of sediment trap in ice free season.

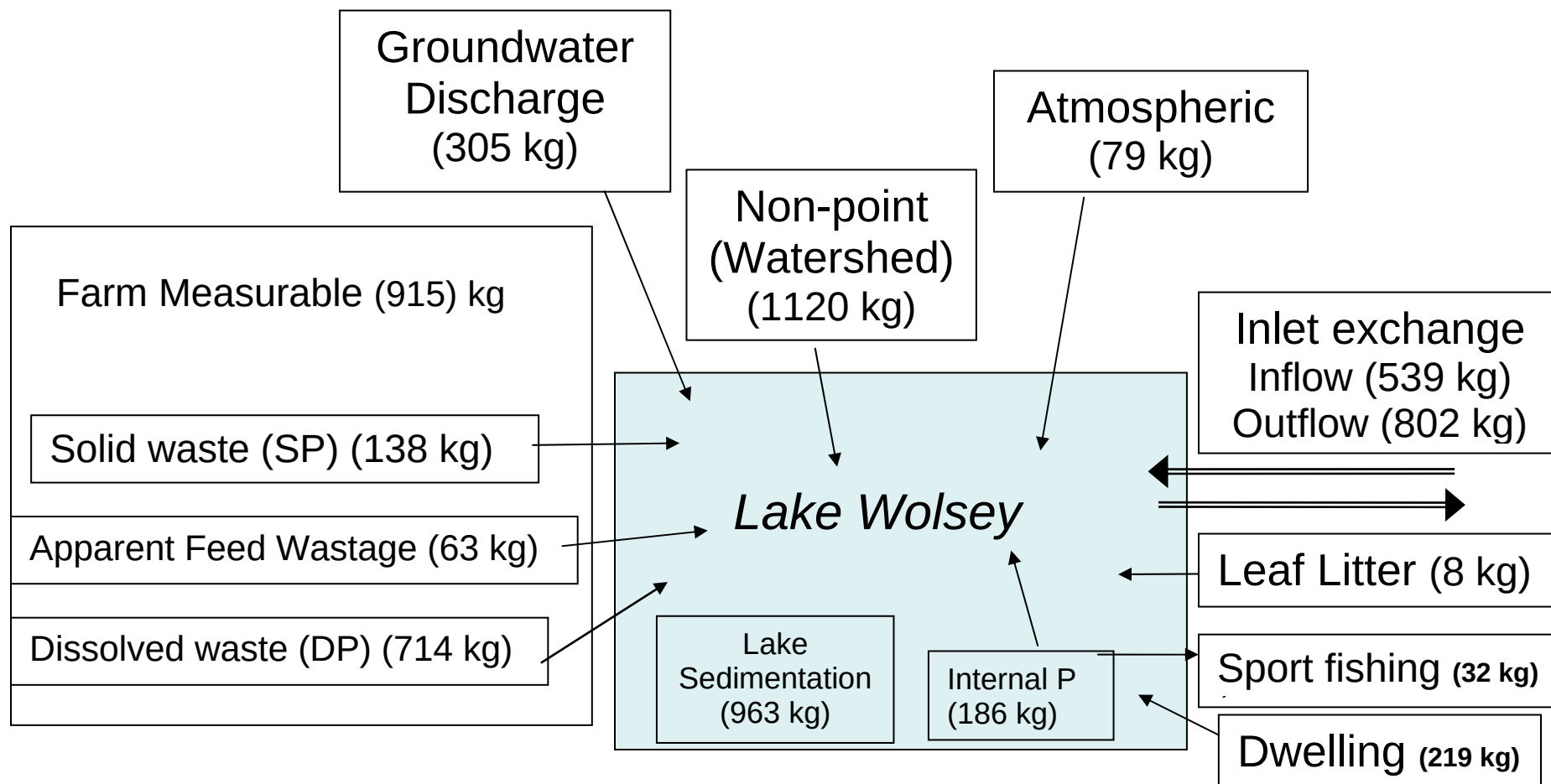


Figure 3 Conceptual schematic of the phosphorus mass-balance model with TP mass.



Figure 4 Acoustic Doppler Current Profiler mounted in a catamaran

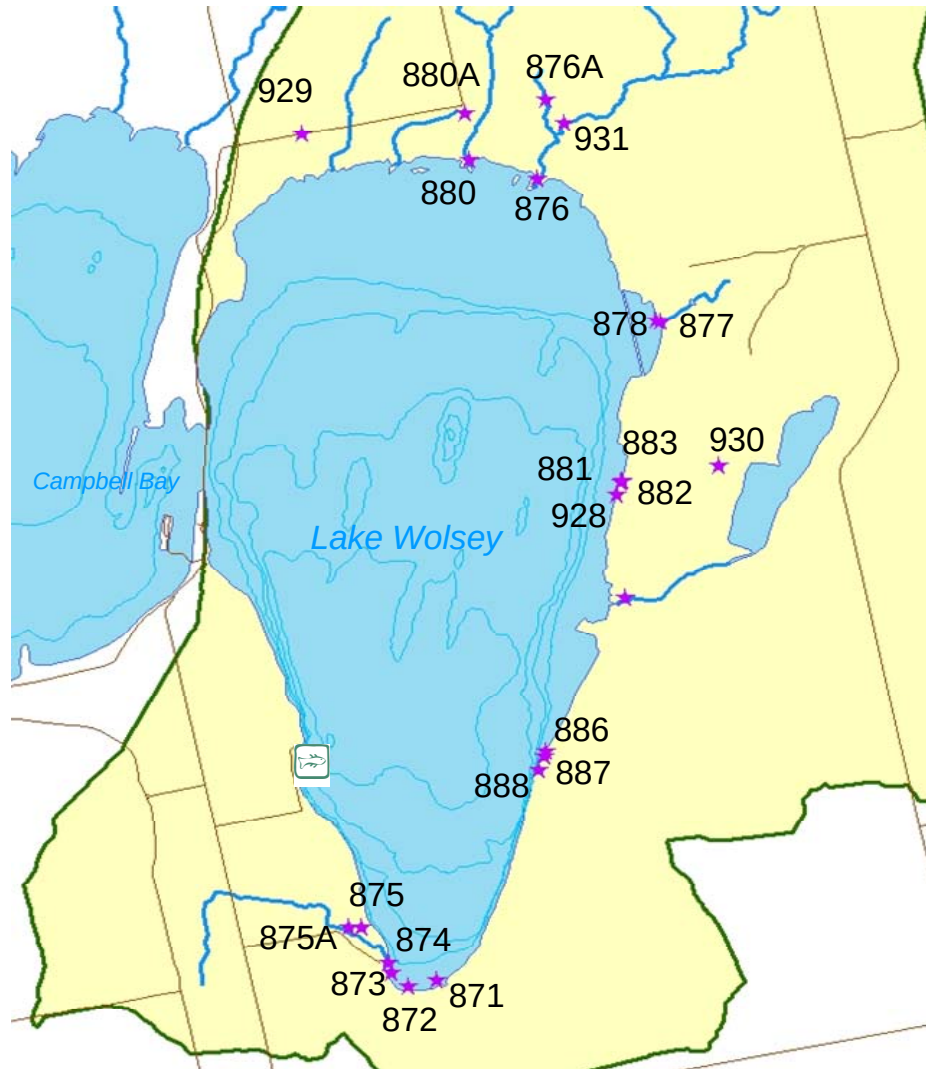


Figure 5 Location of tributary TP samples and flows collected in 2009 and 2010.



A



B

Figure 6 Images showing the design of seepage meter (A) and seepage meter deployment (B).

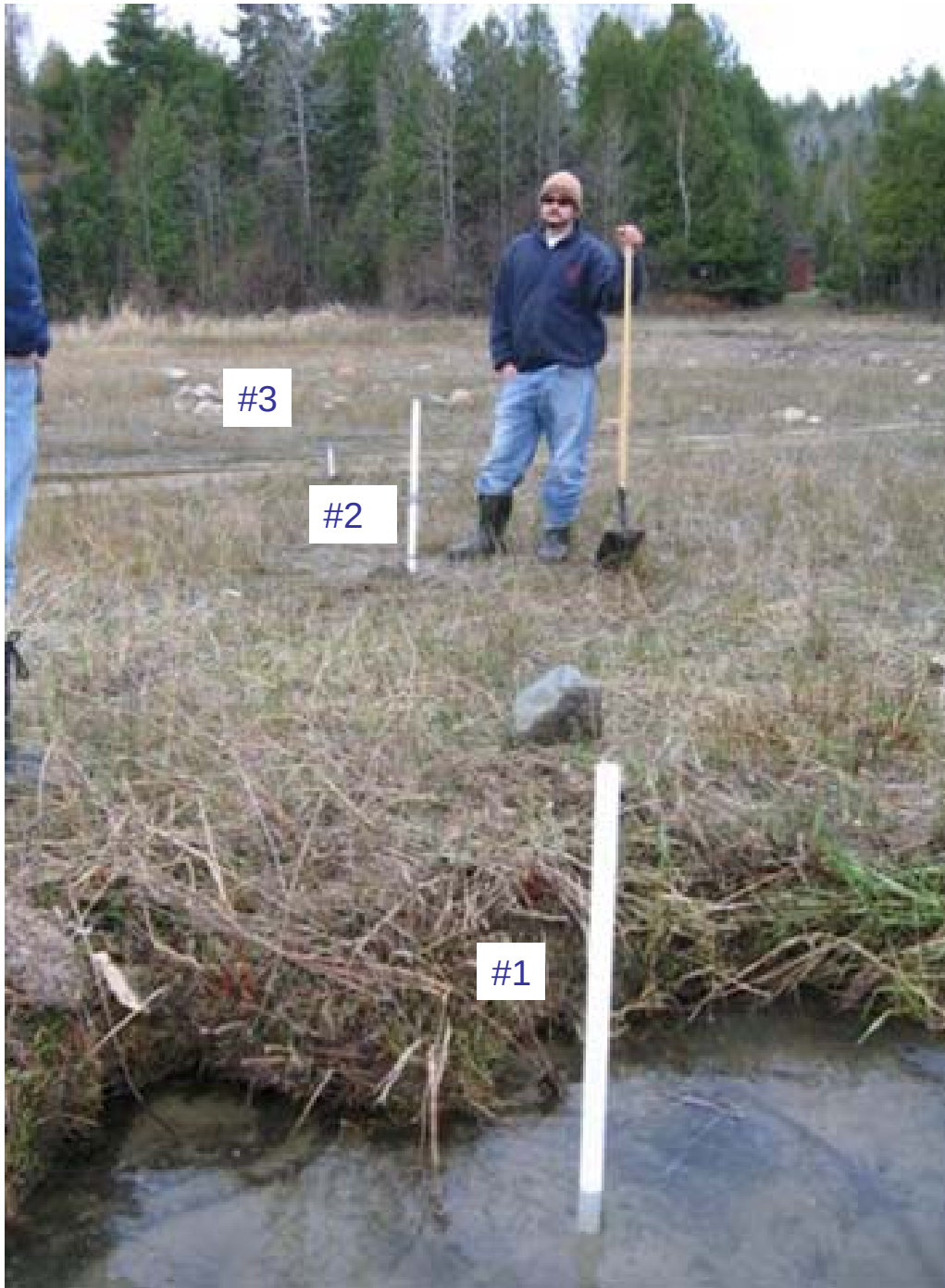


Figure 7 Position of piezometers on the south shore of Lake Wolsey.



Figure 8 Image illustrates abundance of watercress (Nasturtium *officinale*) located at seepage areas on the east shore of Lake Wolsey

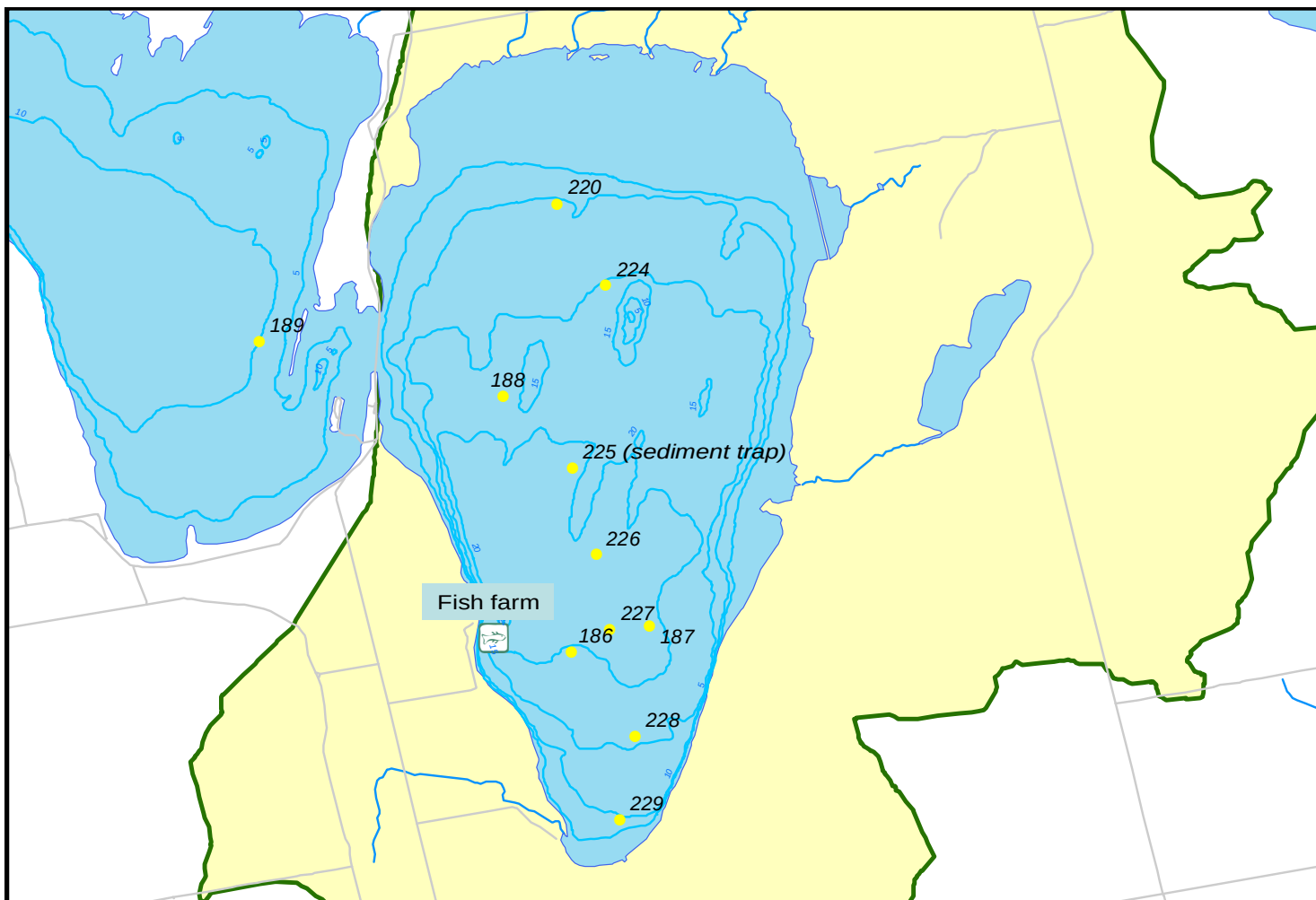


Figure 9 Location of stations monitored by Environment Canada with location of the fish farm.

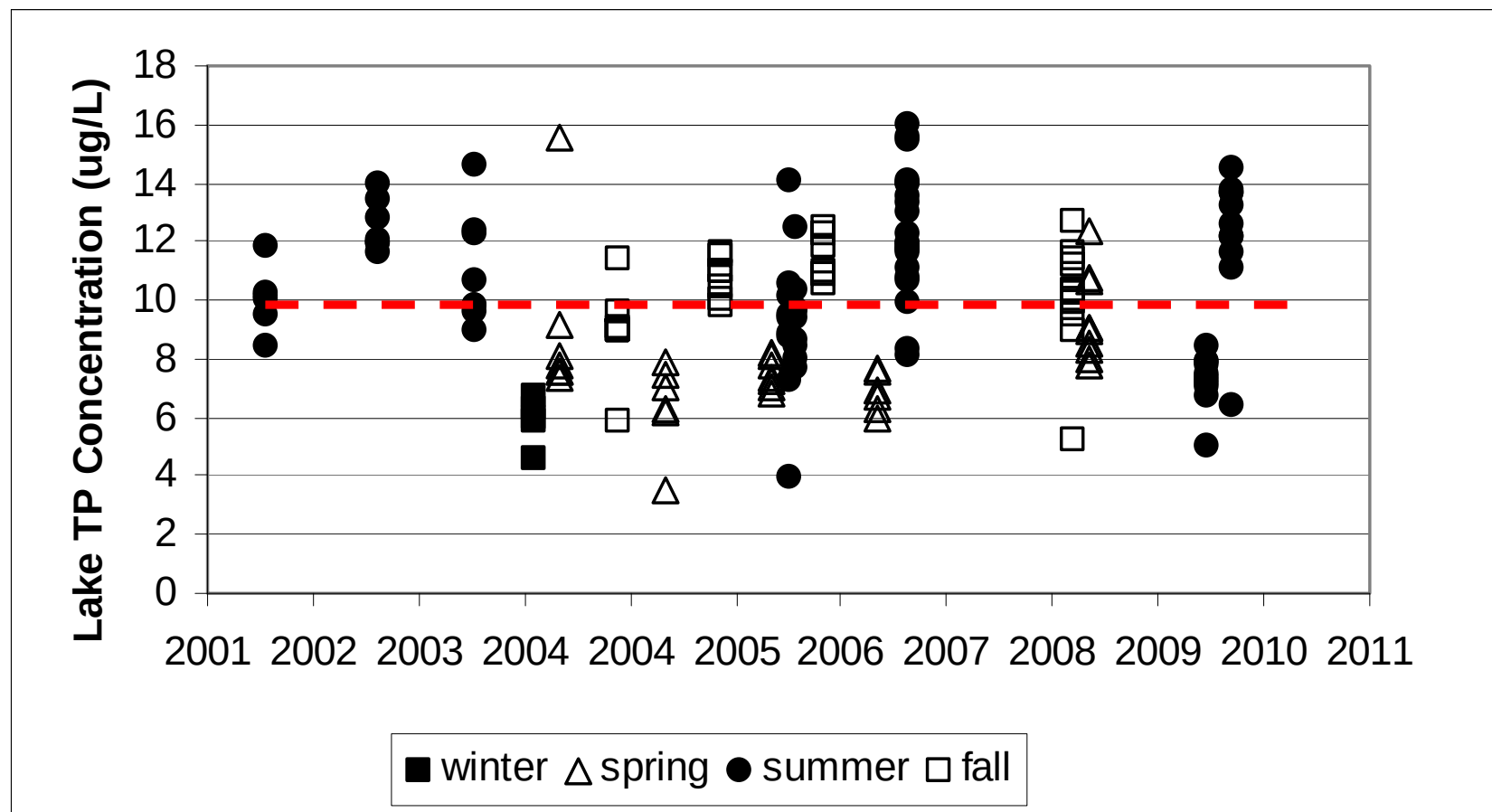


Figure 10 TP concentration at 1m depth winter, spring, summer, and fall 2001 to 2010 measured in Lake Wolsey. Kendall Tau $p=0.22$

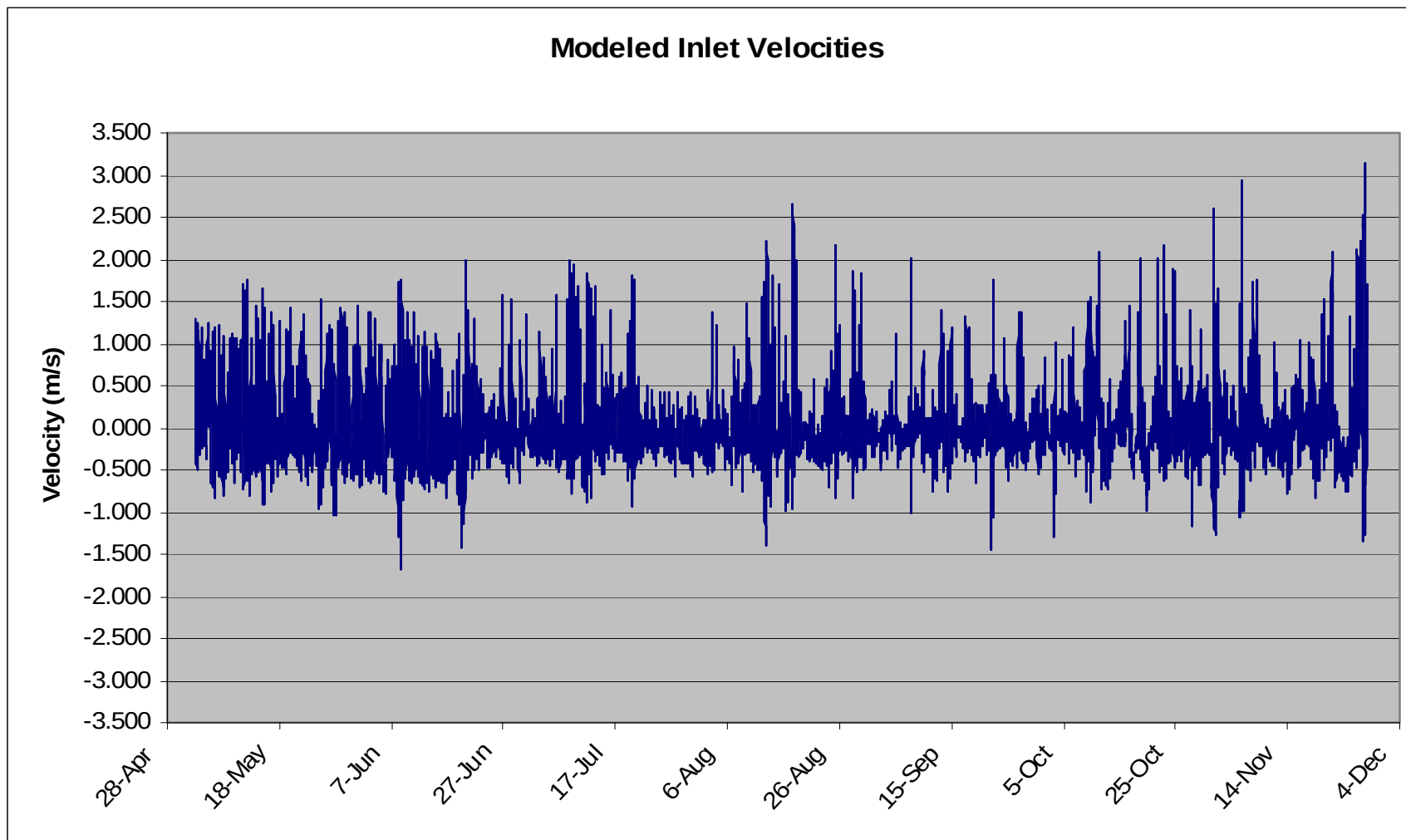


Figure 11 Estimated velocities entering (-ve) and exiting Lake Wolsey (+ve) May to November 2007.

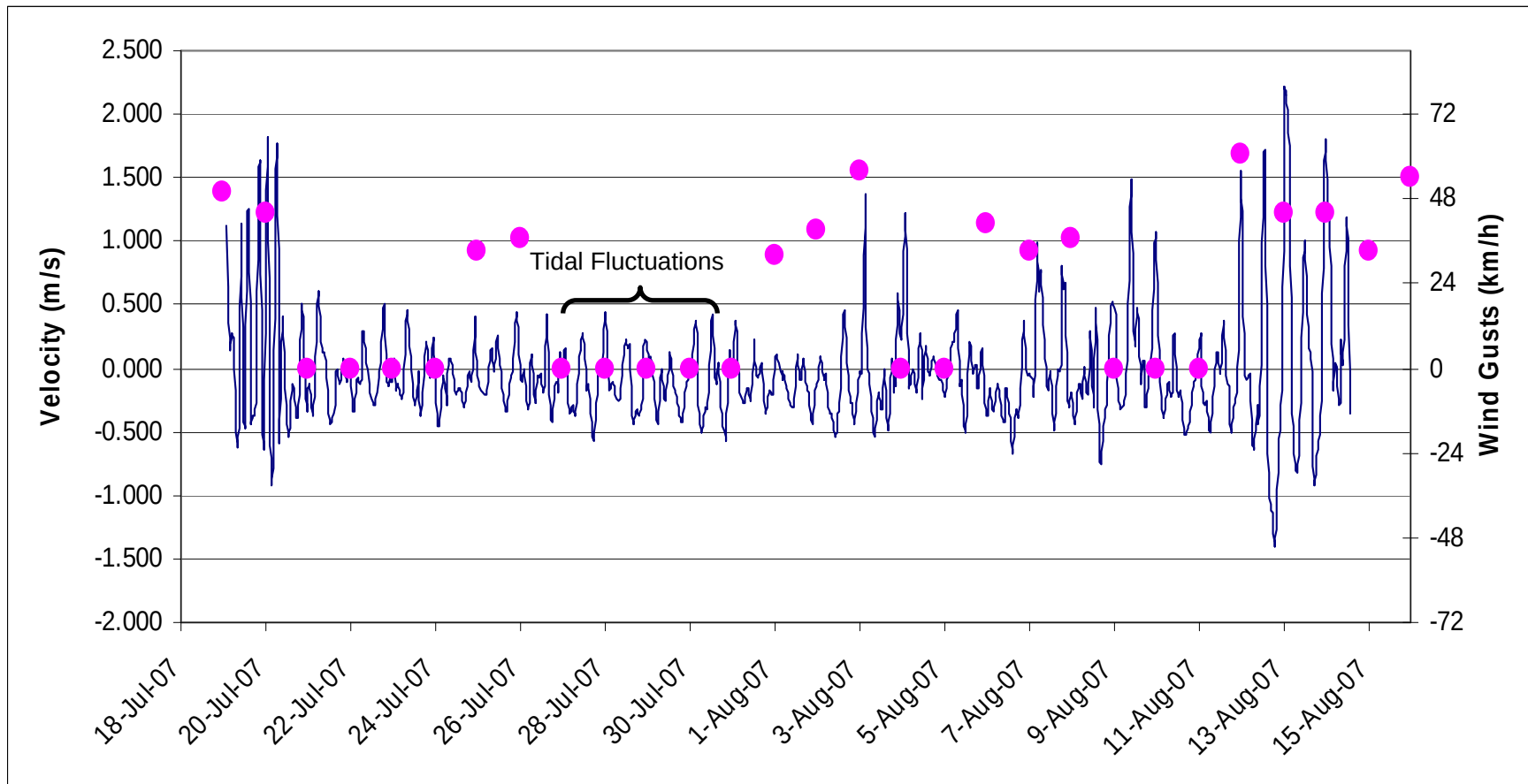


Figure 12 Estimated daily velocities and wind gusts. Tidal fluctuations are shown by the bracket.

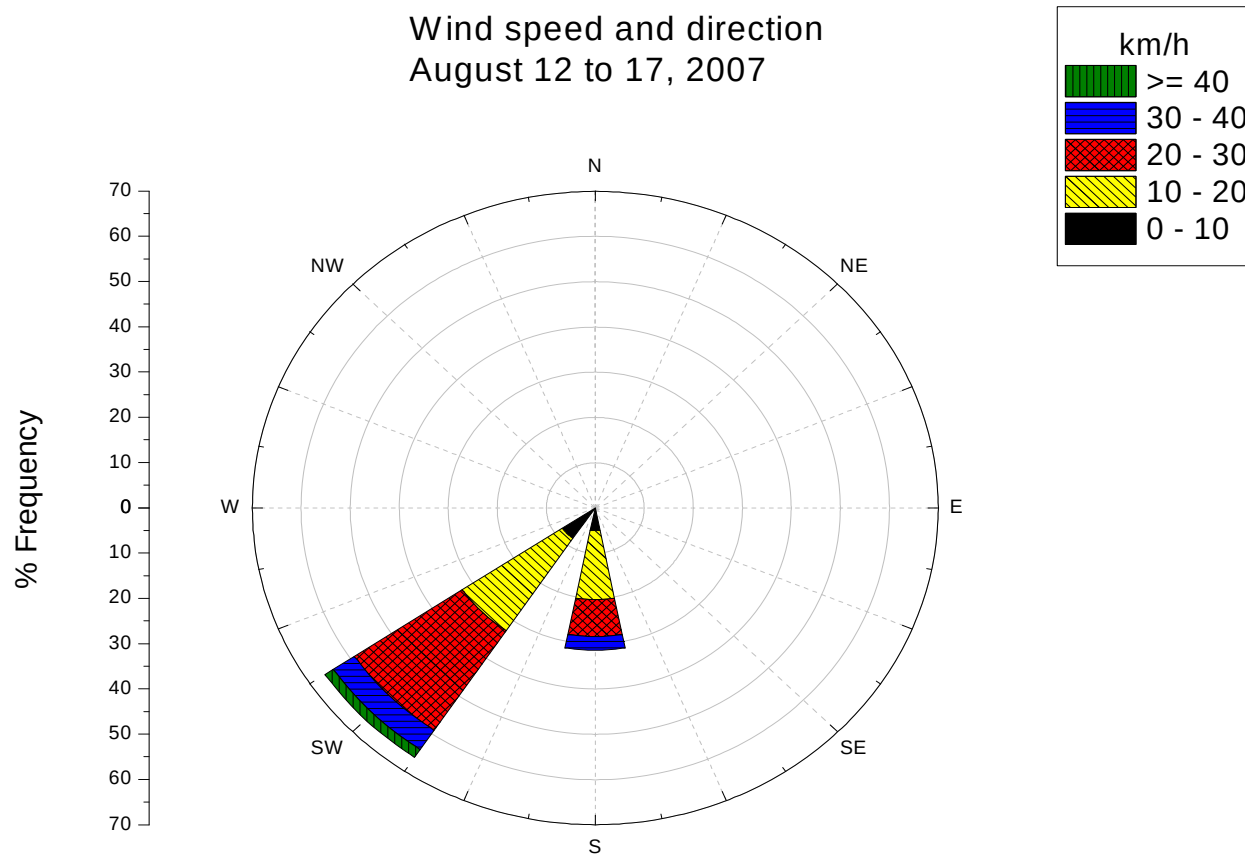


Figure 13 Wind speed and direction August 12 to August 17, 2011 Lake Wolsey.

Wind Speed and Direction May 1 to November 30, 2007

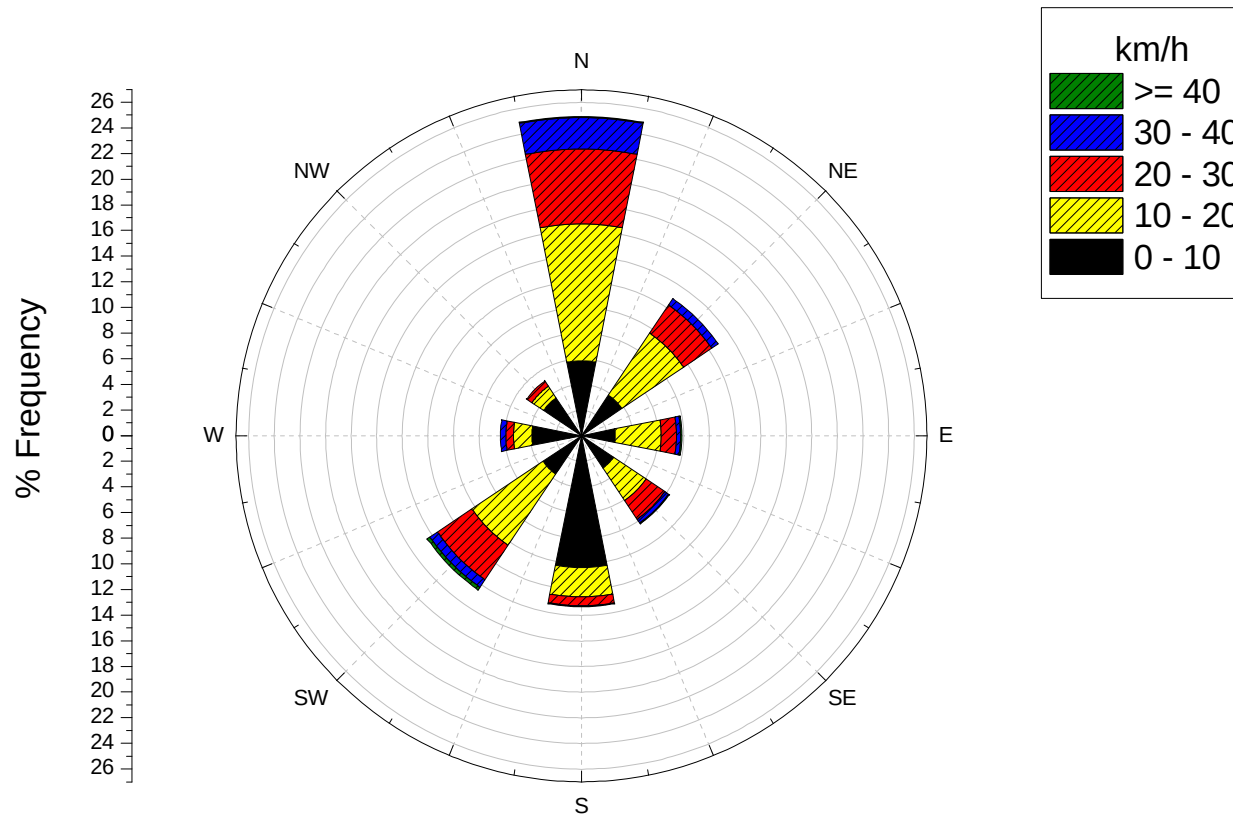


Figure 14 Wind speed and direction during the study the period, Lake Wolsey.

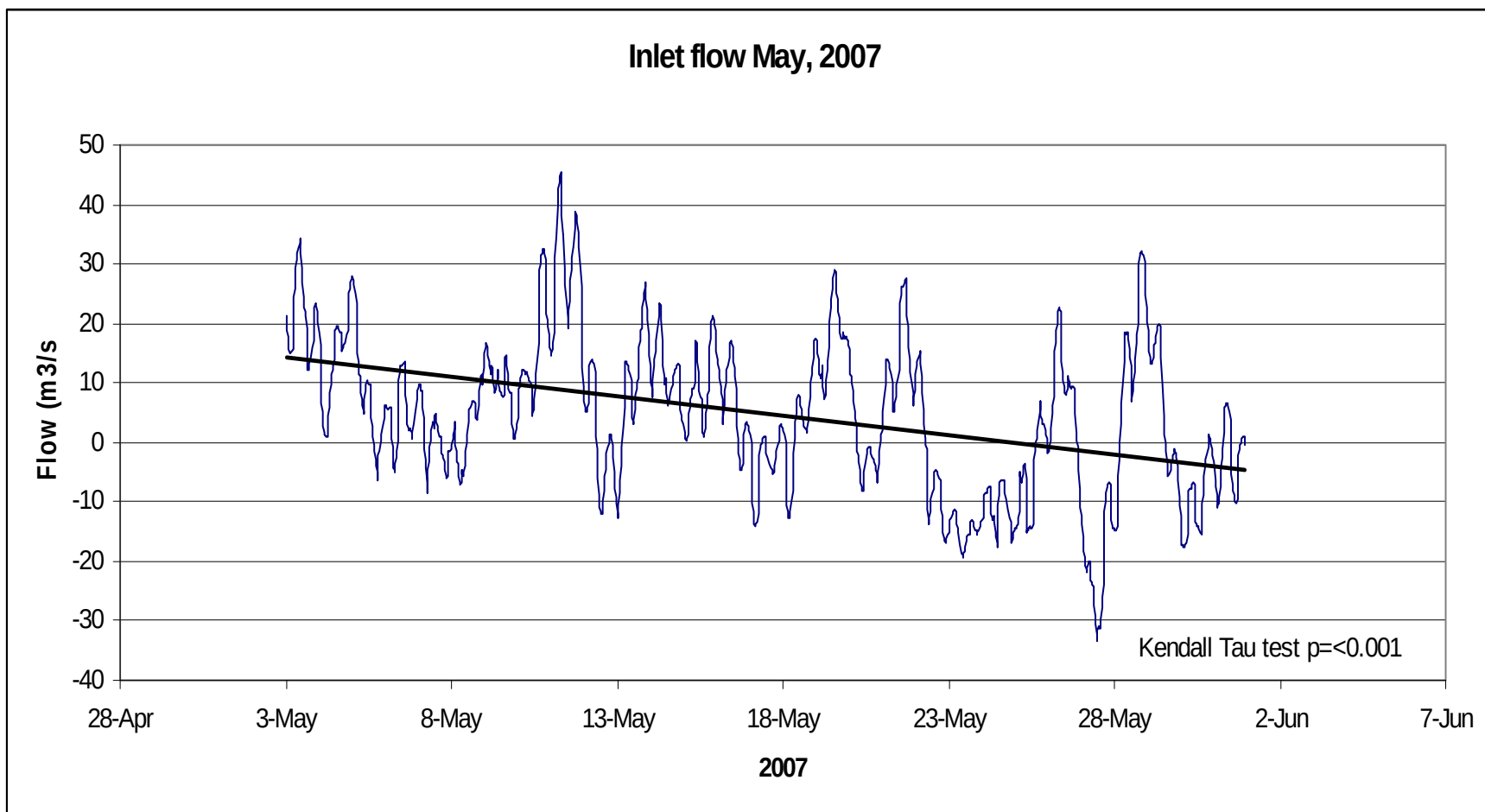


Figure 15 Estimated inlet flows for May 2007 (Kendall Tau $p < 0.001$). Flows entering Lake Wolsey are +ve, flows exiting Lake Wolsey are -ve.

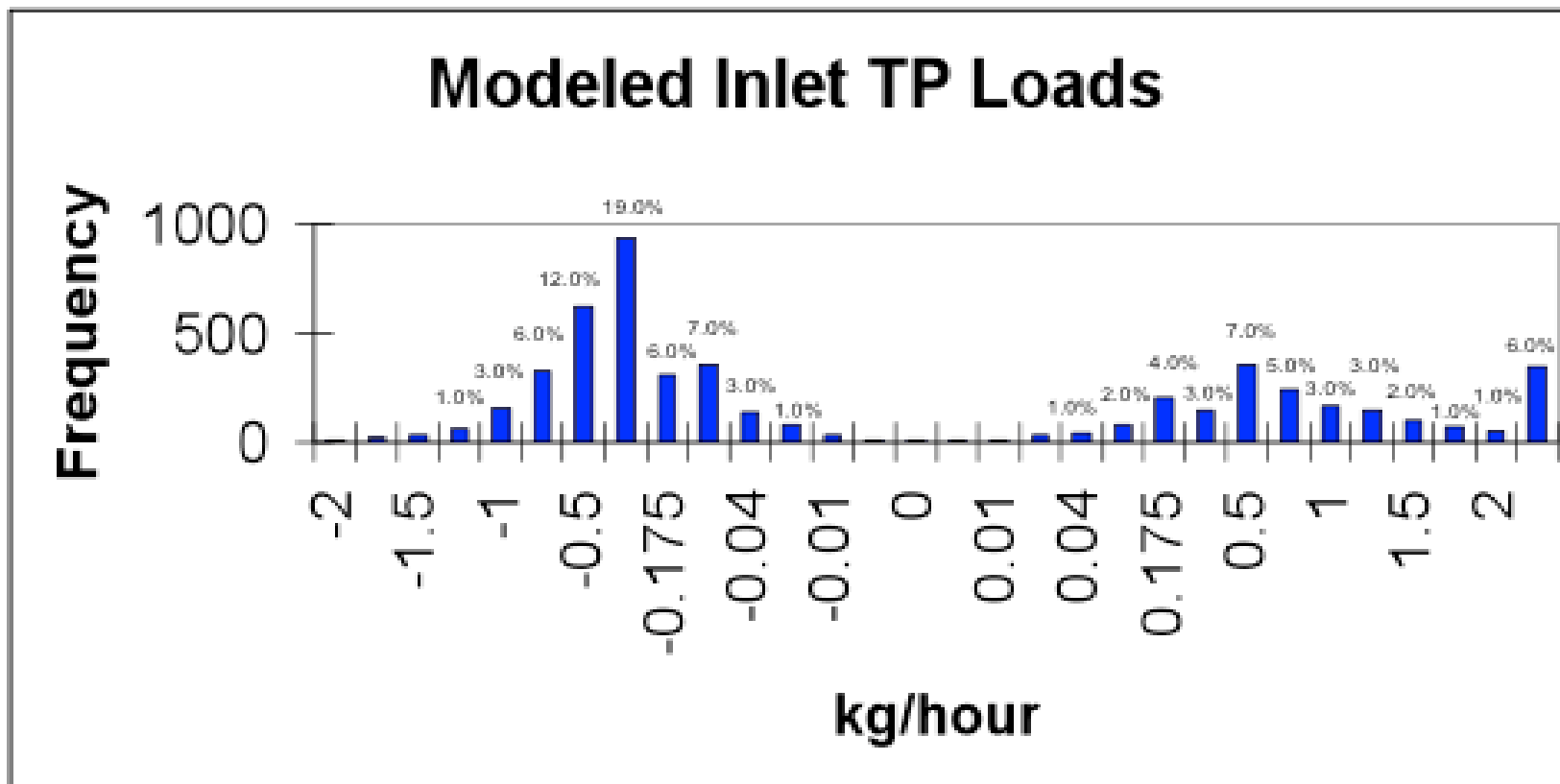


Figure 16 Frequency of hourly TP loads entering (-ve) and exiting (+ve) Lake Wolsey.

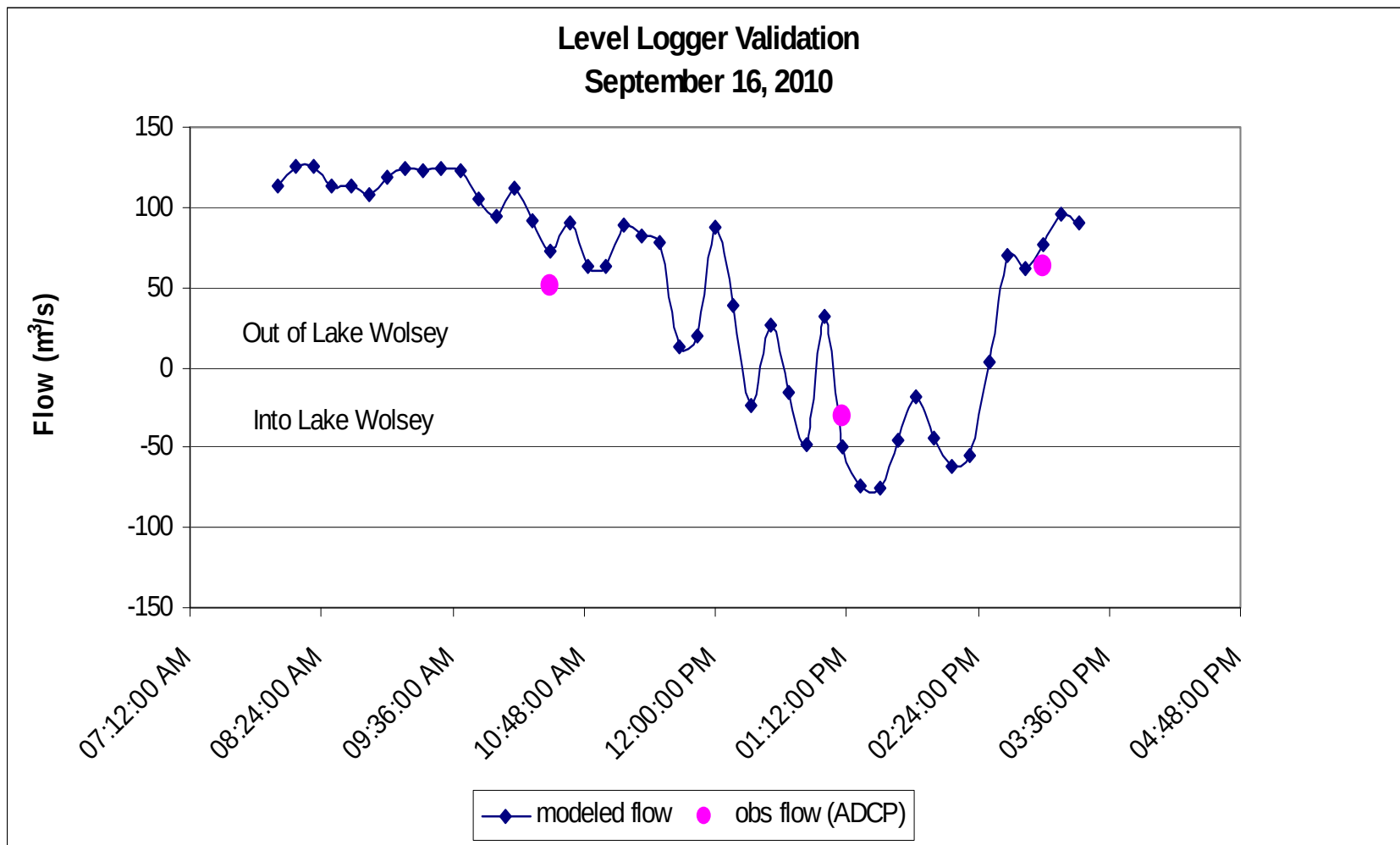


Figure 17 Validation of inlet hydrodynamic model. Observed (ADCP) vs estimated flows. ($r^2 = 0.79$).

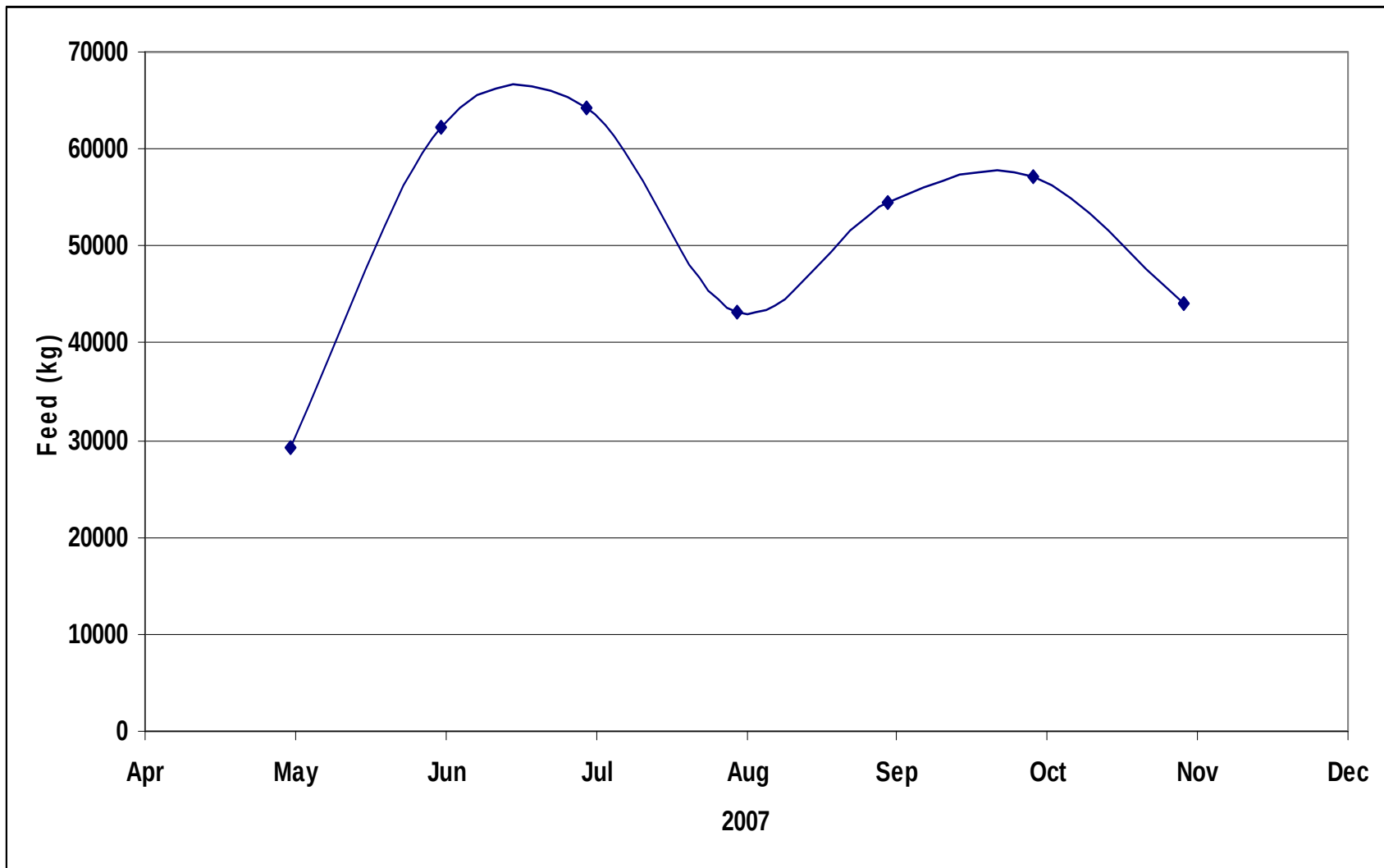


Figure 18 Total monthly feed usage from May to November 2007. Total feed used May to November 2007 = 354820 kg.

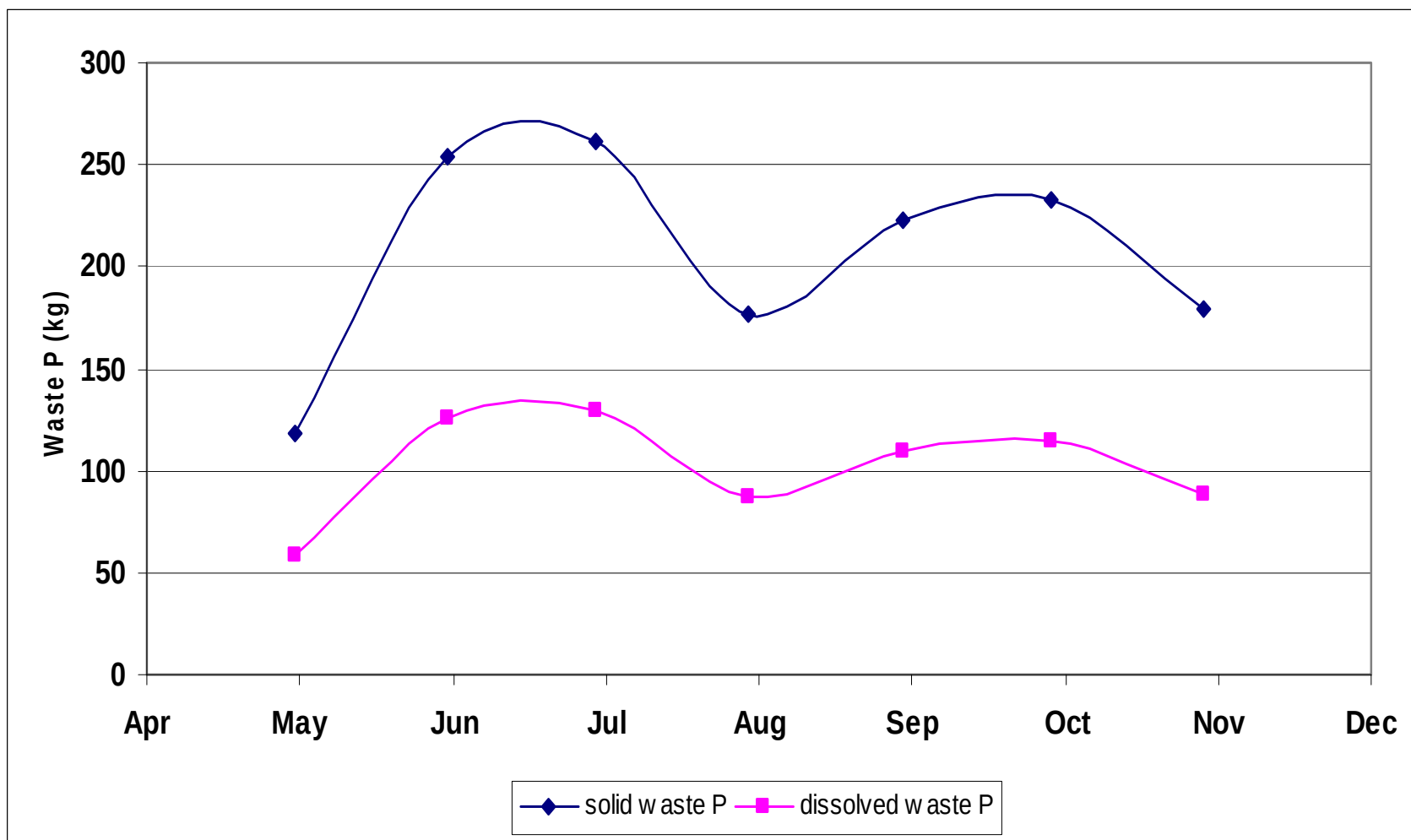


Figure 19 Monthly dissolved and solid waste phosphorus (SWP +AFWP) (kg) released farm May to November 2007.



Figure 20 Illustrates sheet flow during a precipitation event adjacent to Lake Wolsey.

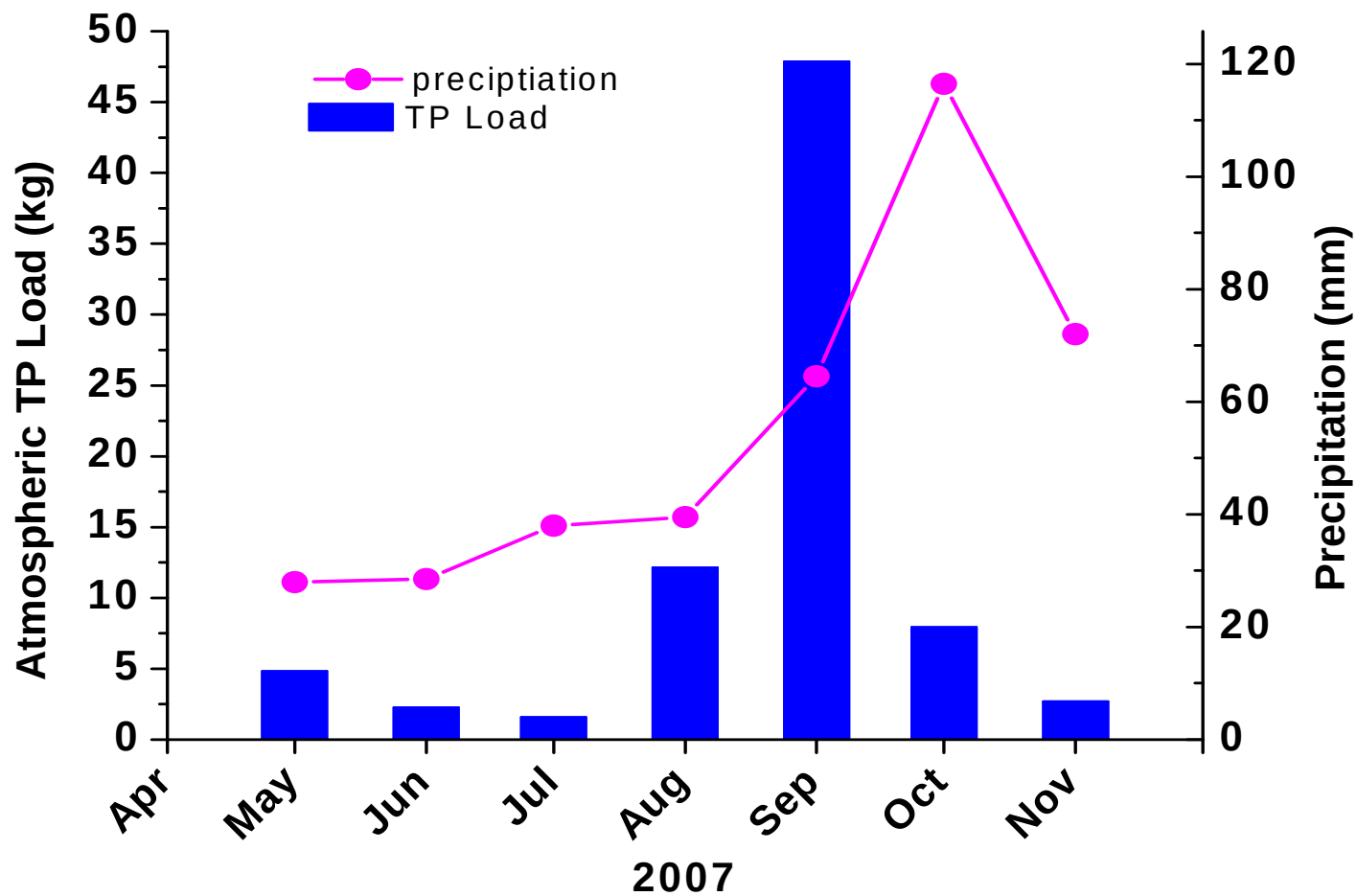


Figure 21 Atmospheric monthly TP loads (kg) and precipitation (mm) May to November 2007.

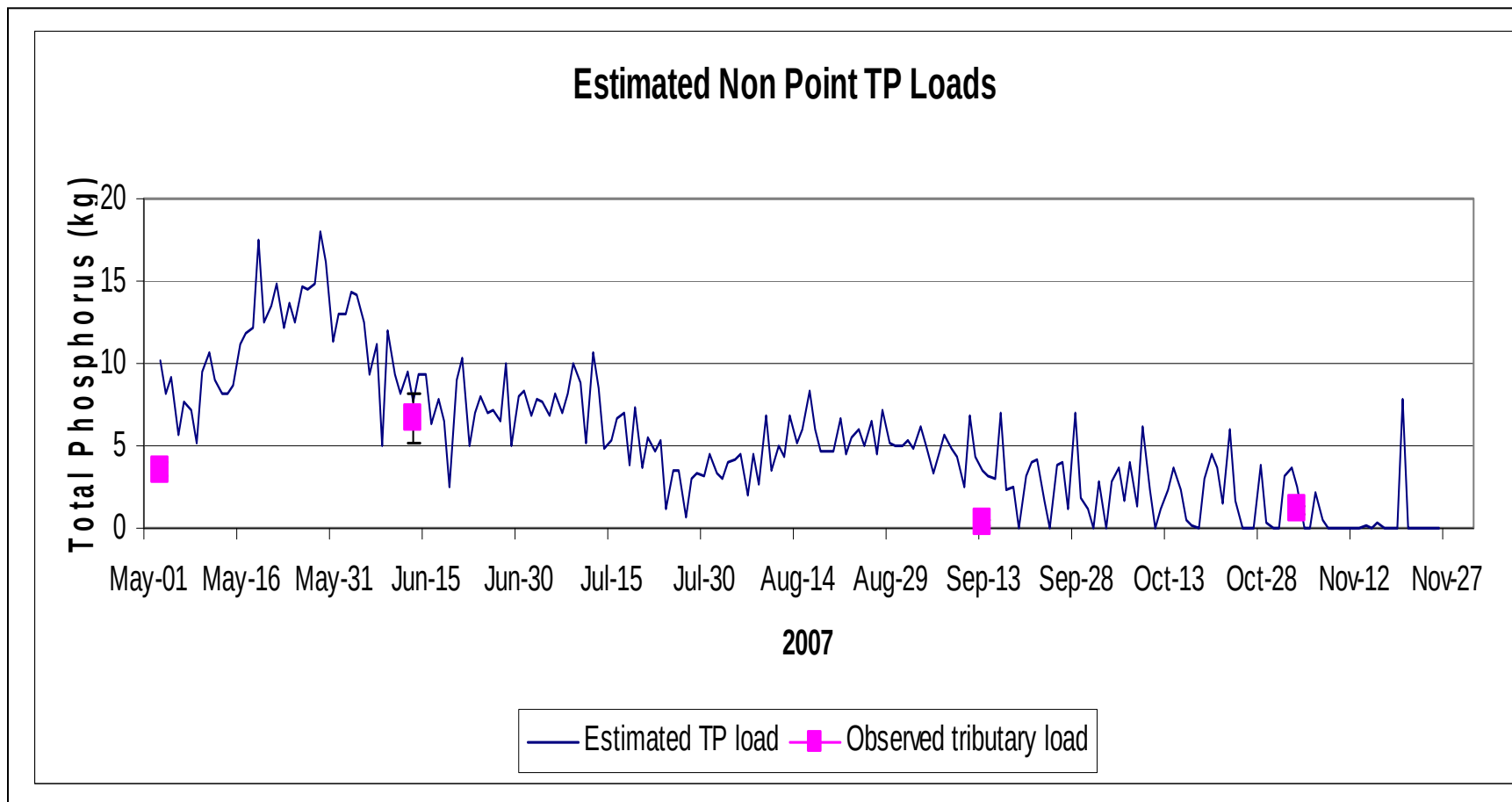


Figure 22 Daily estimated and observed non point sources of TP loads (kg). $r^2 = 0.49$, $p=0.3$



Figure 23 Groundwater discharge adjacent to the shoreline of Lake Wolsey

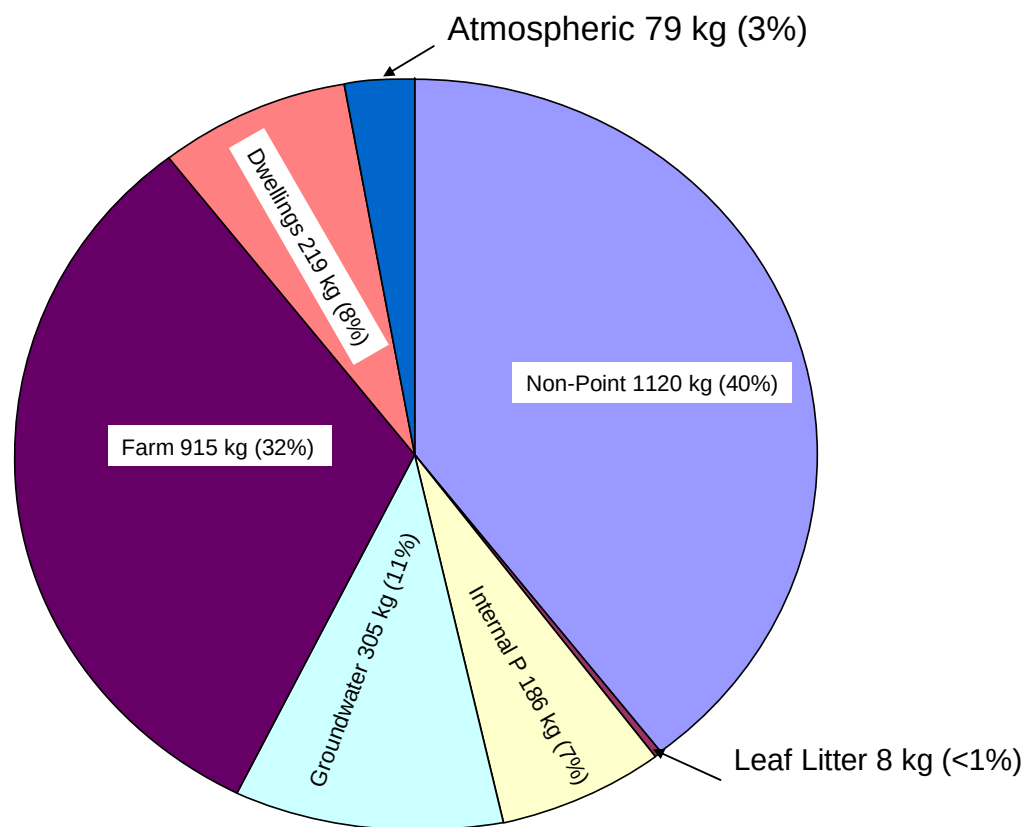


Figure 24 Total load (kg) and percent contribution of each source of TP to Lake Wolsey May to November, 2007.

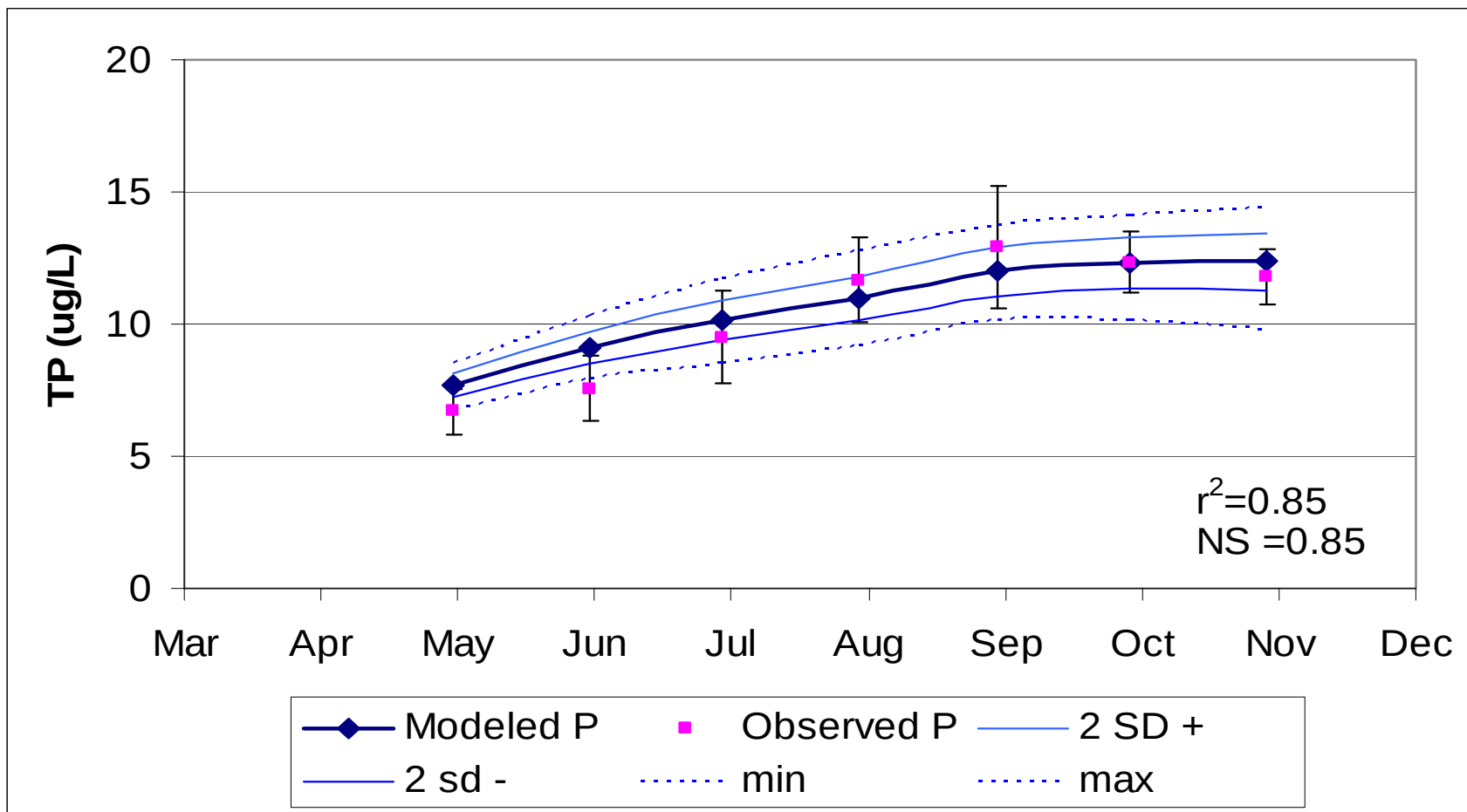


Figure 25 Predicted and observed monthly TP lake concentration May to November, 2007 (NS = Nash-Sutcliffe). Thin solid line = uncertainty ± 2 standard deviations. Dashed line = min and max of uncertainty analysis.

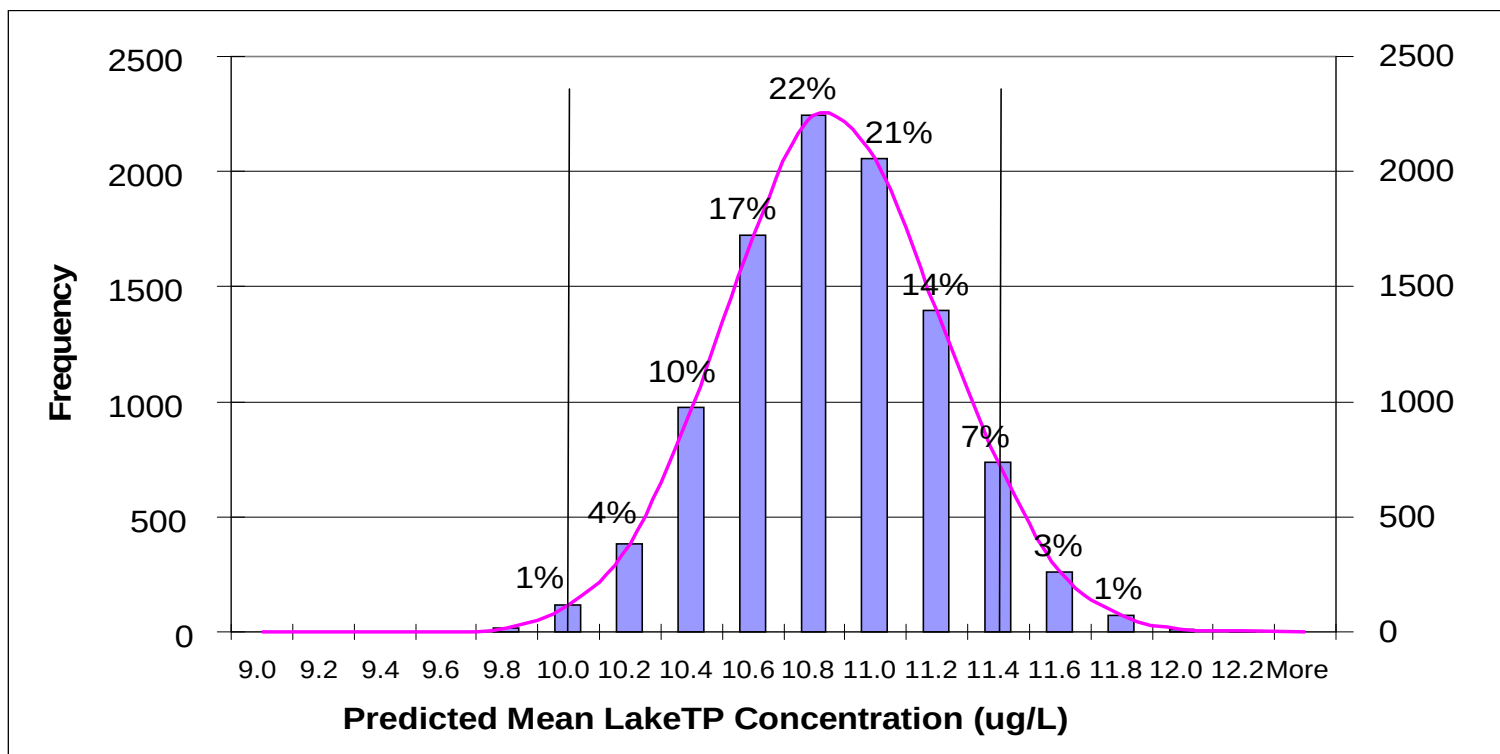


Figure 26 Histogram showing uncertainty range of the mean lake TP concentration (ug/L). The two vertical solid lines represent 2 standard deviations.

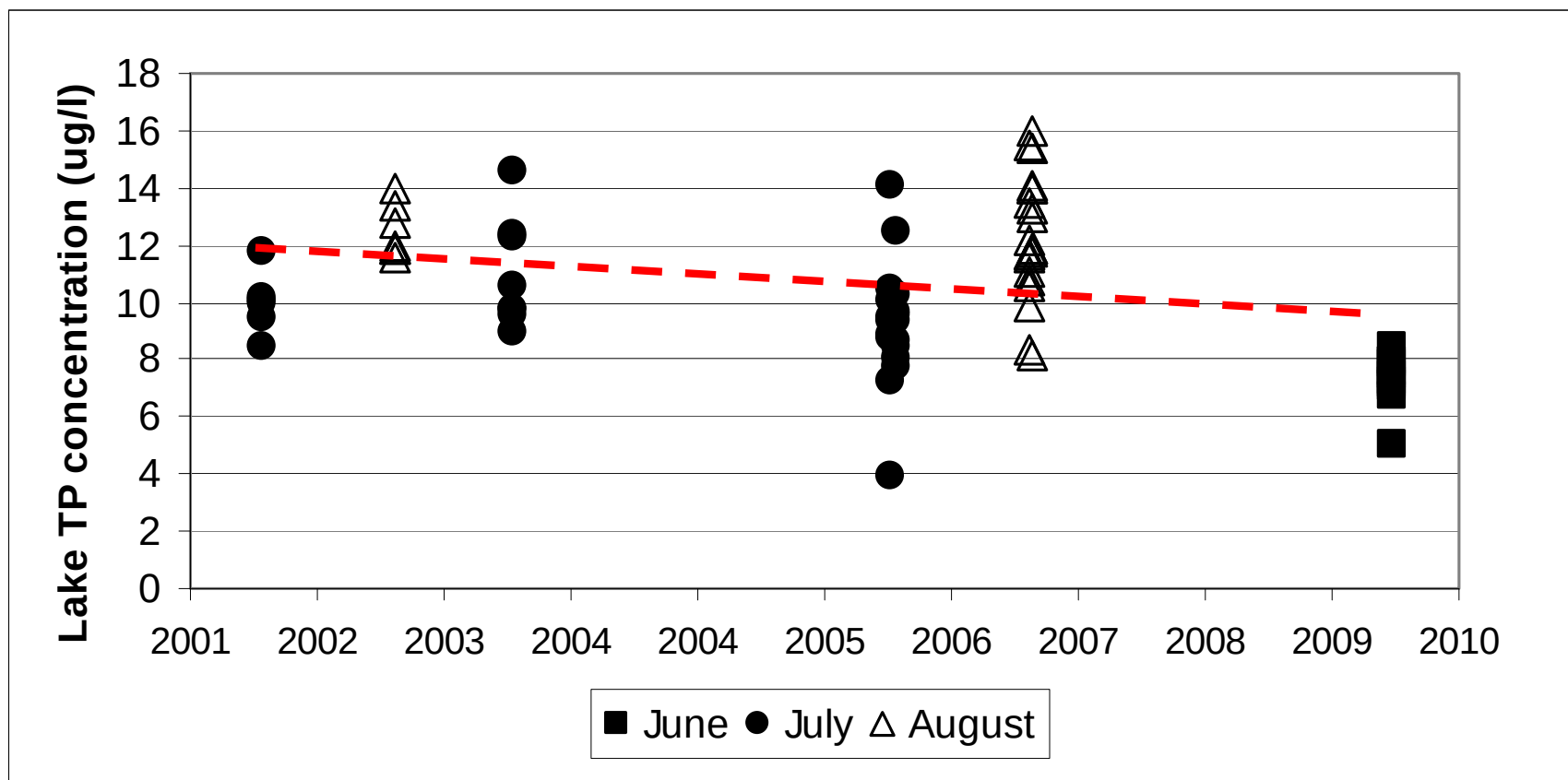


Figure 27 Lake TP concentration 1m June, July and August 2001 to 2010 (Kendall Tau $p=0.09$).

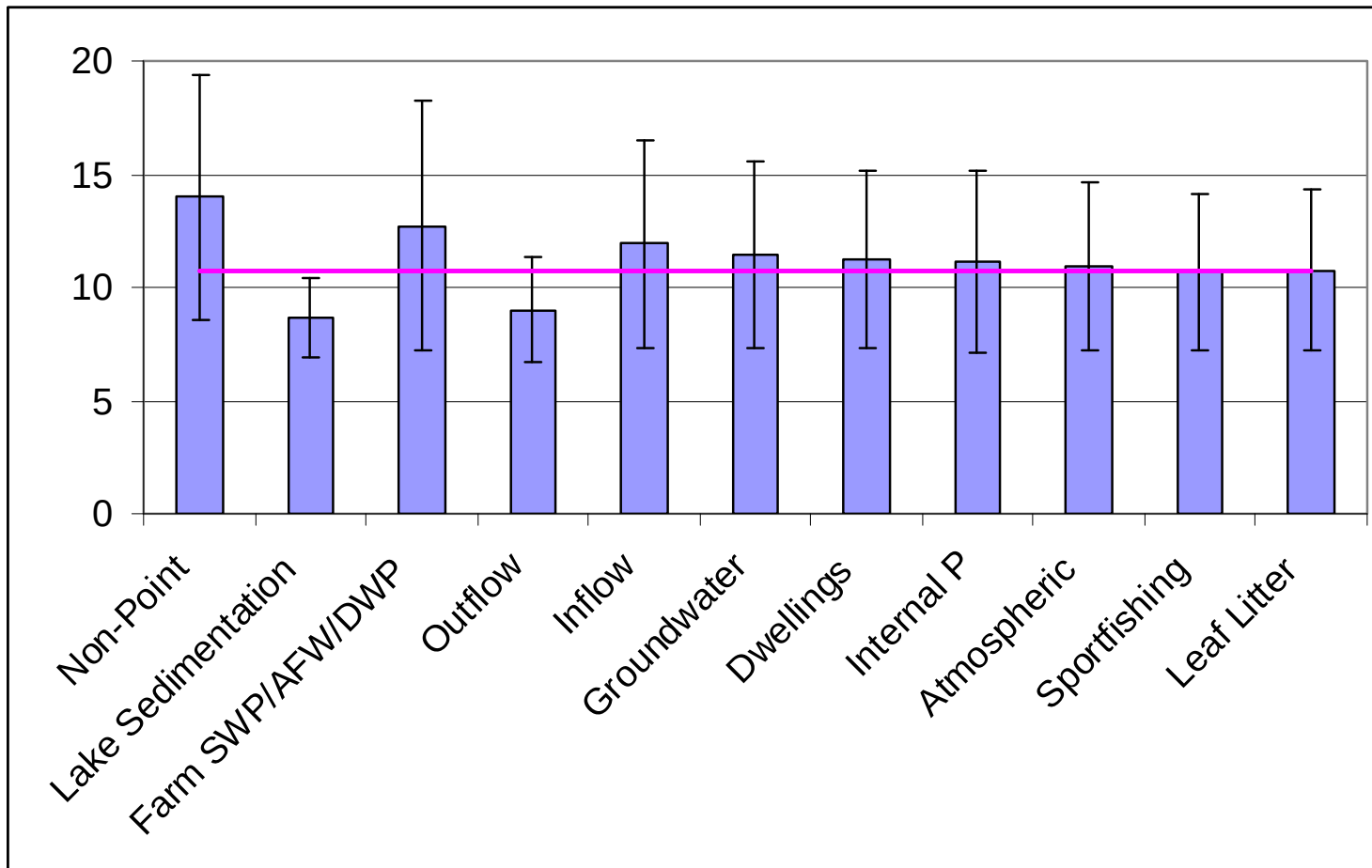


Figure 28 Shows the results of a sensitivity analysis where each parameter was increased 100% while the remaining parameter stayed constant. The error bars represent 2 standard deviations of uncertainty while the solid horizontal bar represents the estimated mean lake TP concentrations before application of sensitivity analysis (10.76 ug/L).

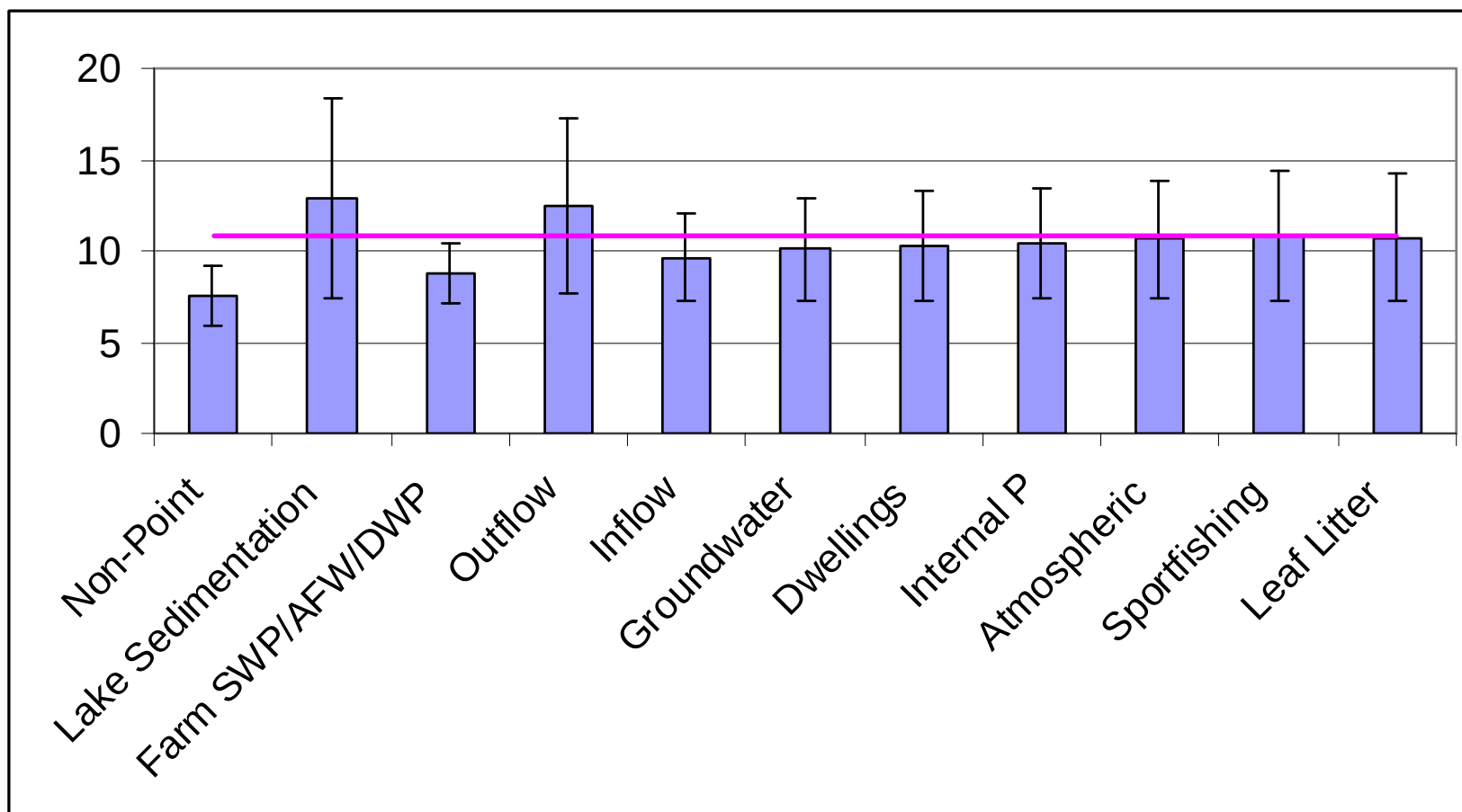


Figure 29 Shows the results of a sensitivity analysis where each parameter was removed (0%) while the remaining parameters stayed constant. The error bars represent 2 standard deviations of uncertainty while the solid horizontal bar represents the estimated mean lake TP concentrations before application of sensitivity analysis (10.76 ug/L).

Table 1 Lake and watershed dimensions.

A_{lake} (ha)	A_w (ha)	Z_{max} (m)	Z_{mean} (m)	V_{lake} (m ³)	LU (%)			
					Dense Forest	Sparse Forest	Agriculture	Alvars/ Bedrock
2203.2	8104.3	22	8.5	260.7 X 10 ⁶	50%	12%	33%	5%

Note: A_{lake} , total area of lake; A_w Area of watershed including tributaries; Z_{max} , maximum depth; Z_{mean} , mean depth; V_{lake} , lake volume; LU, Land Use of entire watershed.

Table 2 Biomass net gain of fish, May to November 2007.

	Batch #										
Fingerlings	1	2	3	4	5	6	7	8	9	10	11
Month added	May	May	May	May	May	May	May	May	June	Aug	Oct
Initial number of fish	25000	25000	25000	25000	25000	25000	50000	25000	25000	75000	75000
Initial wet body* (g)	50	80	68	138	72	80	80	65	65	28	30
Initial wet biomass (kg)	1250	2000	1700	34500	1800	2000	4000	1625	1625	2100	2250
Final number of fish^	23250	23250	23250	23250	23250	23250	46500	23250	23250	69750	69750
Final body wt^^ (g)	926	926	926	926	926	926	926	926	850	300	100
Final wet biomass (kg)	21761	21761	21761	21761	21761	21761	43522	21761	19975	21150	7050
Wet biomass gain (kg)	20511	19761	20061	18311	19961	19761	39522	20136	18350	19050	4800
Adult	1	2	3	4							
Harvest month	June	August	Sept	October							
Initial number of fish*	10373	25887	25972	42123							
Initial wet body~ (g)	700	500	450	350							
Initial wet biomass (kg)	7261	12944	11688	14743							
Final number of fish	10373	25887	25927	42123							
Final body wt (g)	745	1004	1071	1076							
Final wet biomass (kg)	7728	25991	27817	45325							
Wet biomass gain (kg)	467	13047	16129	30581							
Total net gain of adult fish		60224									
Total net gain of fingerlings		220224									
Adult + fingerling net gain		280448									
Mass of feed		354820									
FCR		1:1.3									

* based on farm records

^ assumed 6% mortality (Bristow et al. 2008)

^^ Fingerlings added to the farm in May were assumed a final wt of 926 g in November. Fingerling final weights for June, Aug and Oct were based on expert opinion of owner/operator of the farm.

~ Initial weights in May of fish harvested in June, August, September and October were based on expert opinion of owner/operator of the farm

Table 3 Tributaries, Seepage areas, peizometer locations and date sampled (✓).

Stn	Lat	Long	Location	April 28-30	6-May-09	4-Nov-09	23-Jun-10	15-Sep-10
871	45° 50' 46"	82° 32' 83"	Tributary					
872	45° 46' 49"	82° 32' 31"	Tributary			✓	✓	no flow
873	45° 50' 46"	82° 33' 83"	Tributary					
874	45° 46' 58"	82° 32' 38"	Tributary			✓	✓	dry
875	45° 47' 11"	82° 32' 45"	Tributary	✓		✓	✓	dry
875A	45° 47' 11"	82° 32' 45"	Tributary					
876	45° 51' 02"	82° 30' 01"	Tributary	✓			✓	
876A	45° 51' 36"	82° 30' 13"	Tributary			✓	✓	no flow
877	45° 50' 46"	82° 30' 82"	Tributary	✓				
878	45° 50' 46"	82° 30' 82"	Tributary	✓				
879	45° 49' 46"	82° 30' 83"	Tributary					
880	45° 51' 18"	82° 30' 50"	Tributary		✓			
880A	45° 51' 36"	82° 30' 47"	Tributary			✓	✓	no flow
881	45° 49' 20"	82° 30' 19"	Tributary		✓	✓	✓	dry
882	45° 49' 23"	82° 30' 18"	Tributary		✓	✓	✓	no flow
883	45° 49' 23"	82° 32' 17"	Tributary		✓	✓	✓	dry
884	45° 50' 32"	82° 30' 13"	Tributary		✓			
885	45° 47' 49"	82° 33' 31"	Tributary		✓			
886	45° 48' 58"	82° 31' 14"	Seepage		✓			
887	45° 48' 57"	82° 31' 16"	Seepage		✓			
888	45° 48' 52"	82° 31' 19"	Seepage		✓			
889A	45° 48' 43"	82° 31' 26"	peizometers		✓			
889B	45° 48' 43"	82° 31' 26"	peizometers		✓			
928	45° 49' 18"	82° 30' 21"	Tributary			✓	✓	no flow
929	45° 51' 35"	82° 31' 47"	Tributary			✓	✓	
930	45° 49' 22"	82° 29' 38"	Tributary				✓	✓
931	45° 51' 24"	82° 30' 08"	Tributary				✓	✓

Table 4 Source of TP concentration data for calibrating mass-balance model on monthly time step. Int = integrated sample collection

Month	Averaged Depths	Year	Data Source
May	1m and b-2m	2007	EC
June	1m and b-2m	2010	EC
July	1m	2006	EC
Aug	1m	2007	EC
Sept	1m	2010	EC
Oct	int (Stn 86)	2006/2007	MOE
Nov	1m and b-2m	2006	EC

Table 5 Stations monitored by Environment Canada for TP analyses.

Station	Latitude	Longitude	Total Depth (m)
186	45° 48' 00"	82° 32' 00"	18.7
187	45° 48' 04"	82° 31' 45"	21.6
188	45° 49' 29"	82° 32' 20"	16.3
189	45° 50' 01"	82° 33' 50"	10.3
220	45° 50' 30"	82° 31' 41"	9.8
224	45° 50' 00"	82° 31' 30"	14.3
225	45° 49' 01"	82° 32' 00"	20.4
226	45° 48' 31"	82° 31' 59"	21.8
227	45° 48' 05"	82° 32' 01"	21.5
228	45° 47' 28"	82° 32' 01"	16.3

Table 6 Bioenergetics phosphorus mass-balance: Estimated solid waste phosphorus (SWP) apparent feed wastage phosphorus (AWFP), dissolved waste phosphorus (DWP), biomass net gain phosphorus (BP) (kg). Percent P in feed = 0.95.

Month	Dry feed applied	P in Feed		SWP	AWFP	DWP	BP	Sum
May	27119	258	=	114	5	59	80	258
June	57948	551	=	243	11	125	172	551
July	59799	568	=	250	11	129	178	568
August	40269	383	=	169	8	87	120	384
September	50741	482	=	212	10	110	151	483
October	53094	504	=	222	10	115	158	505
November	41013	390	=	171	8	89	122	390
Sum		3136		1383	63	714	983	3139

Table 7 Monthly total phosphorus (TP) sources and sinks (kg) 2007.

Month	Sources						Sinks				
	<i>inex</i>	<i>ingw</i>	<i>inNP</i>	<i>inAtm</i>	<i>indwell</i>	<i>inFarm</i>	<i>inleaf</i>	<i>inint</i>	<i>outex</i>	<i>outsed</i>	<i>outsport</i>
May	47.0	42.3	328.6	4.8	30.7	75.2	0	0	202.0	139.5	5
June	119.6	43.8	264.2	2.3	29.8	160.8	0	0	63.8	135.0	4
July	66.9	45.3	181.6	1.6	34.1	165.9	0	34	98.7	139.5	5
August	76.0	45.3	155.2	12.1	34.1	111.7	0	62	83.6	139.5	5
September	68.8	43.8	111.1	47.9	29.8	140.8	4.0	60	49.7	135.0	4
October	79.5	45.3	59.5	8.0	30.7	147.3	4.1	30	152.3	139.5	5
November	81.6	39.4	20.2	2.7	29.8	113.8	0	0	151.5	135.0	4

Note: *inex*, inlet load entering Lake Wolsey; *ingw*, groundwater; *inNP*, non point source; *inAtm*, Atmospheric; *indwell*, dwellings; *inFarm*, farm load in (SWP+AFWP+DWP); *inleaf*, leaf litter; *inint*, internal phosphorus; *outex*, inlet load exiting Lake Wolsey, *outsed*, lake sedimentation; *outsport*, sportfishing

Table 8 Mean lake TP concentration (ug/L) as a result of increasing and decreasing each model parameter by 100%. The estimated mean lake TP concentration before increase or decrease was 10.76 ug/L.

Parameter	100% increase Mean Lake TP concentration	Removal of parameter Mean Lake TP concentration
Non-Point	14.0	7.5
Lake Sedimentation	8.7	12.9
Farm (SWP/AFWP/DWP)	12.7	8.8
Outflow	9.0	12.5
Inflow	11.9	9.6
Groundwater	11.4	10.1
Dwellings	11.2	10.3
Internal P	11.1	10.4
Atmospheric	10.9	10.6
Sportfising	10.7	10.8
Leaf Litter	10.8	10.8

Appendixes

Appendix 1 MatLab® V6.1 code to run the one dimensional inlet hydrodynamic model to estimate velocity in the inlet connecting Lake Wolsey with the North Channel.

```
close all  
clear all
```

```
w=30; % this is the width of the channel  
L=30; % this is the length of the channel  
g=9.8; % this is gravity  
C=55;  
dt=6; %this the time interval in seconds because we are interpolating to 600 sec between the 10 min readings the xi = 0.01 600 X  
0.01=6 sec timestep  
AREA=67.7;% obs from ADCP
```

```
load MaytoAug.txt; % this the data file
```

```
north=MaytoAug(:,1); % is the water level data north Lake Wolsey in column 1  
south=MaytoAug(:,2); % is the water level data south of Lake Wolsey in column 2  
number=size(MaytoAug); % this gives the number of rows and columns of files  
Lin=[15181]; % this is the number of readings for the whole time period - 144 readings/day
```

```
x=1:length(north);  
xi=1:0.01:length(north);
```

```
north1=interp1(x,north,xi);  
south1=interp1(x,south,xi);
```

```

U(1)=0;
nbegin=2;
nend=50;
for i=nbegin:nend;

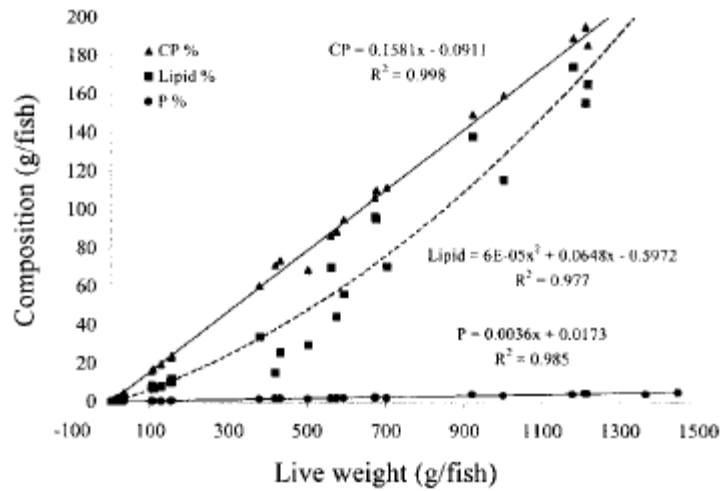
U(i)=(dt/AREA)*((-1)*g*AREA*(north1(i)-south1(i)))/L-(g/(C*C))*w*abs(U(i-1))*U(i-1)+U(i-1); % this is the equation to calculate
velocity
end

for i=1:15180;
Umean(i)=mean(U((i-1)*100+1:100*i)); %this calculates 10 min mean of data - 6 seconds X 100 seconds = 10 min averages every
100 readings.


U=U'; % this converts values from rows to columns.
Umean=Umean';
%north1=north1';
south1=south1'
%save Vel.dat U -ascii; % saves velocities
%save MaytoAug.dat Umean -ascii;
%save north.dat north1 -ascii;
save south.dat south1 -ascii;
%plot(U);
End

```

Appendix 2 Graph showing phosphorus vs live weight of rainbow trout (*Oncorhynchus mykiss*). From Bureau et al 2003).



Appendix 3 Method summary for the analysis of Total Phosphorus in water. Excerpt from the NLET Schedule of Services, 2010.

Acidic Persulfate Digestion, Automated Continuous Segmented Flow Analyzer (CFA), Stannous Chloride-Molybdate Complex, Photometric Method: A mixture of sulphuric acid/persulfate is added to an aliquot of a well mixed whole water sample which is then digested in an autoclave for 30 minutes at 121°C. The oxidative-acid digestion releases organically bound phosphorous as orthophosphate, PO₄. Orthophosphate is also formed in the digestion from the hydrolysis of inorganic condensed phosphorous (pyro-, meta-, and other polyphosphates). The digest is analyzed in a continuous segmented flow analyzer (CFA) where the orthophosphate is reacted in an acid medium with ammonium molybdate to form heteropoly molybdophosphoric acid, H₃(PMo₃O₁₀)₄. This is then reduced with stannous chloride to form the molybdenum blue complex. The absorption of radiation by the complex represents the total phosphorous concentration in the sample and is determined by the CFA photometer at 660 nm wavelength.

Detection limit = 0.5 ug/L

Appendix 4 Method summary for the analysis of Total Phosphorus in feed and sediment.
Excerpt from the NLET Schedule of Services, 2010.

Total metals analysis requires the complete destruction of the sample in order to release all forms of metal. These include metals adsorbed on the sediment particles, metals in the form of insoluble salts and organic complexes and metals in the mineral phase of the sediment. Decomposition of the sample is achieved by treating it with a mixture of strong acids. Under high temperature, high pressure microwave conditions a combination of nitric acid, hydrochloric acid and hydrogen peroxide are successful in dissolving salts and destroying organic matter. For total analysis, hydrofluoric acid is required to decompose mineral silicates by the reaction of fluoride with silica to form H_2SiF_6 . This reaction breaks up the matrix and liberates all entrapped metals. The data from elements analyzed in this procedure may be used to characterize the geochemical composition of the sediment. The procedure is, as follows:

One-half gram of freeze dried sediment is weighed into a TFM Teflon® vessel and a mixture of nitric acid, hydrochloric acid and hydrogen peroxide (9:2:1 mL) is added. The vessel is assembled, sealed and fitted onto a microwave rotor which is placed in a high pressure microwave oven. The sample is digested and the temperature is maintained at 200°C for 15 minutes. The vessel is removed from the microwave oven and the cooled digest is transferred to a Teflon® digestion tube. The sample volume is then reduced on a hot block at 130°C to near dryness (~ 1 mL), followed by treatment with 10 mL hydrofluoric acid, overnight at 90°C, to ensure destruction of silicates. The excess hydrofluoric acid is boiled off and the salts are dissolved in a solution of 5% hydrochloric

acid. The 100 mL resultant solution is analyzed for Al, Ba, Ca, Cr, Fe, Mg, Mn, P, K, Na, Sr, V and Zn by ICP-OES. As, Sb, Be, Bi, Cd, Co, Cu, Ga, La, Pb, Li, Mo, Ni, Rb, Tl and U are analyzed by ICP-MS.

Detection Limit = 10 mg/kg

Appendix 5 Uncertainty range given each model parameter.

Model Parameter	Uncertainty Range (kg) +/-
Sources of TP	
Inex (inflow) (kg)	5
ingw (groundwater load) (kg)	40
inNP (non-point load) (kg)	20
inatm (atmospheric load) (kg)	1.5
indwell (dwelling load) (kg)	5
inleaf (leaf litter load) (kg)	2
inint (internal load) (kg)	5
infarm (farm load)	
SWP	10
AFWP	5
DWP	5
Sinks of TP	
outex (outflow) (kg)	15
outsed (lake sedimentation) (kg)	14
outsport (sportfishing) (kg)	1.5

Appendix 6 A brief summary of Recommendations For Operational Water Quality Monitoring at Cage Culture Aquaculture Operations (Boyd, 2001).

#	Recommendation	Explanation
1	Regional background water quality	Routinely monitored index/reference sites in the North Channel and Georgian Bay by the MOE will be used as regional background water quality data (wq).
2	Location of water quality sampling stations	Water quality monitoring stations will be located at 4 corners of off-shore cages or mid points on 3 sides if cages anchored nearshore. Stations should be positioned 30 m from the cages. Two reference sites will be established “upstream” and “downstream” of the cages, respectively.
3	Phosphorus sample collection and data analysis	A minimum of eleven wq samples should be collected at 3 or 4 stations near the cage and 2 reference stations: 3 collected during spring turnover, 5 during summer stratification, 3 during fall turnover
4	Water clarity sample collection and data analysis	A minimum of weekly secchi depths and colour should recorded at the 3 or 4 near cage sites and 2 reference stations during the ice-free season.
5	Temperature/Dissolved oxygen sample collection	Temperature and dissolved oxygen (DO) recorded at 1m intervals at depths ≤ 20 m and 2m intervals for depths >20 m monthly at type 3

		sites. At type 1 and type 2 sites temperature profiles should be collected weekly at the deepest near cage stations commencing after ice out. DO sampling should be done monthly until stratification then weekly.
6	Sediment collection and data analysis	Sediment sampling should be done at the upstream and downstream boundary of the LUP one year minimum before renewal of or revision to existing license or LUP.
7	Evaluation and modification of monitoring requirements	Monitoring requirements may change with additional scientific information.
