IDENTIFICATION OF WETLANDS ON THE CANADIAN BOREAL PLAIN AND THEIR CONTRIBUTIONS TO STREAM WATER CHEMISTRY.

by

Kendra Couling

A graduate thesis submitted in partial fulfillment
of the requirements for the degree of
Master of Science in Forestry

Faculty of Forestry and the Forest Environment

Lakehead University

November 2006

Your file Votre référence
ISBN: 978-0-494-31190-5
Our file Notre référence
ISBN: 978-0-494-31190-5

ABSTRACT

Couling, K. 2006. Identification of wetlands on the Canadian Boreal Plain and their contributions to stream water chemistry. 87 p.

Key Words: forest management, peatlands, wetland identification, aerial photography, Alberta Wetland Inventory, Wetland Inventory and Identification Tool, Canadian Boreal Plain, phosphorus, nutrient retention, phosphorus budget, water budget.

The Alberta Wetland Inventory (AWI), which is used in a variety of applications across the province to estimate wetland cover from aerial photographs, detected only 34% of confirmed wetland field plots in boreal forest watersheds in the Swan Hills of Alberta. Given the association between wetland cover and runoff and surface water chemistry in western Canadian boreal forest (Boreal Plain) watersheds, accurate quantification of wetland cover is critical to efforts to model hydrologic processes and water quality. Therefore, as a component of the Forest Watershed and Riparian Disturbance (FORWARD) Project, the Wetland Inventory and Identification Tool (WIIT) was developed and successfully detected 81% of the wetland field plots. Application of both models across a variety of landscapes in the boreal forest of Alberta demonstrated that wetland cover estimates were 1.5 times higher with the new WIIT model than with the AWI. Also, the WIIT identified polygons that were both smaller and contained taller trees than those identified by the AWI, indicating that this computer model may be more effective than wetland identification methods that use only aerial photography. Results of this study show that careful interpretation of aerial photography at the 1:15 000 scale, coupled with ground truthing and computer models, can provide an accurate means of identifying wetlands on Boreal Plain landscapes.

A preliminary annual (November through October water year) water and phosphorus (P) budget was also constructed for a 3-ha peatland in the Swan Hills, to quantify some aspects of peatland water and P cycling. Understanding the relationship between wetlands, and water and nutrient (P) inputs and outputs from watersheds is central to models being developed for stream water quality and quantity. The study wetland in the FORWARD Willow watershed retained 27% of the water collected through rainfall and runoff, and evapotranspiration represented the dominant route for water loss from the wetland, constituting 63% of rainfall inputs. The wetland retained (within soils, vegetation and microbial pools) approximately 77% of P entering the wetland via wet and dry atmospheric deposition and runoff.

CONTENTS

TABLES

CHAPTER II:

CHAPTER III:

FIGURES

CHAPTER II:

CHAPTER III:

ACKNOWLEDGEMENTS

If it were not for the endless love, patience, support, motivation and encouragement provided by my partner in life, my husband Jamie Fligg, I would not have been able to take on this endeavour.

Dr. Ellie E. Prepas afforded me the opportunity to research this fascinating topic, which allowed me to become more confident in my abilities as a field and laboratory researcher. Committee members, Dr. Daniel W. Smith (University of Alberta) and Dr. James J. Germida (University of Saskatchewan), provided extensive input and assistance with this entire endeavour. I would also like to thank Dr. Brian McLaren (Lakehead University) for helping with the internal examination process and Dr. Brian Palik (USDA Forest Service) for agreeing to assist with the external examination process. The comments and critiques by both the internal and external examiners were extremely helpful.

Janice Burke's insight and suggestions made this thesis a better piece of work. She was my guide to learning to write for a scientific audience. Over the past two years, I have been motivated by her confidence in my abilities.

Dr. Ivan Whitson was of great support both in the field and during the academic year, including his invaluable knowledge of the study area.

Many FORWARD project team members provided their expertise in project development and implementation.

Scott Eberwein, Jennifer Bond, Shannon Higgins, Stefanie-Anne van Huystee, Roshan Gervais and Jacquie Truemner provided many hours of field and laboratory assistance. Fellow grad students were very helpful in project formulation, problem solving, as well as for motivation and support.

Olenka Bakowsky and The Forestry Corp provided expertise on data analysis and many hours of assistance both in house and via e-mail.

Daryl D'Amico and Blue Ridge Lumber Inc. (a Division of West Fraser Timber Company Ltd.) directly supported my work through the NSERC Industrial Post-Graduate Scholarship program. Their financial support made my graduate studies possible.

Lakehead University was very generous with supplementary funding in form of the Ontario Graduate Summer Fellowship and the research incentive bonus.

Mr. Jonathan Russell and Millar Western Forest Products Ltd. took great interest in this project and provided logistical support.

This work was carried out with grants from the Natural Sciences and Engineering Research Council of Canada Discovery and CRD programs (Millar Western Forest Products Ltd., Blue Ridge Lumber Inc., Vanderwell Contractors (1971) Ltd., Alberta Newsprint Company (ANC Timber), Louisiana-Pacific Canada Ltd.) and the Canada Foundation for Innovation, matched with funding from the Government of Ontario, to the FORWARD project and to Dr. E. E. Prepas.

Last, but certainly not least, I thank my parents. They constantly encouraged me to challenge myself. Without their confidence in me over the years, I never would have made it to this point.

CHAPTER I: GENERAL INTRODUCTION

It is expected that the southern boundary of the boreal forest may move northwards up to hundreds of kilometres in the next century due to global warming (Vitt et al. 2000, Environment Canada 2004). In the boreal and subarctic forests alone, the thawing of peatlands could potentially release 440 billion m^3 of water (currently stored as ice), as well as the nutrients and contaminants stored therein, causing drastic changes to local water quality and hydrology (Tarnocai 2006).

Eutrophication, the nutrient enrichment of freshwater ecosystems (Wetzel 2001), is associated with deterioration of water quality, such as the formation of toxic algal blooms and the reduction of dissolved oxygen concentrations (Schindler 1974, Chambers et al. 2006). Due to the documented nutrient retention abilities of natural (Devito et al. 1989, Kellogg and Bridgham 2003) and constructed wetlands (Moustafa 1999, White et al. 2004) the detrimental impacts of eutrophication highlight the need for wetland identification, as well as an understanding of their nutrient retention capabilities. Wetland nutrient cycling and retention processes are poorly documented for the nutrient-rich Boreal Plain of western Canada. Efforts to link landscape changes with dynamic modeling of surface water quality and quantity are currently hindered by the lack of accurate tools to remotely predict wetland presence and a weak understanding of vital wetland processes for sites on the Boreal Plain. Given the decreases in water storage in wetlands that are observed and predicted for the Boreal Plain, and the widespread distribution of wetlands therein, further investigation of patterns and processes in these wetlands was warranted.

CHAPTER II: IMPROVED ESTIMATION OF WETLAND COVER IN WESTERN CANADIAN BOREAL FOREST WATERSHEDS[1]

Introduction

Wetlands cover 14 to 18% of Canada's land area (National Wetlands Working Group 1988, Conference on Canadian Wetlands Stewardship 2003), yet they are often ignored in resource management planning in the Canadian boreal forest. In addition to serving as wildlife habitat, wetlands sequester carbon and may moderate climate warming (Gorham 1991, Vitt et al. 2000, Wilson et al. 2001) and associated impacts on streams and lakes. Wetlands (i.e. rich fens) sequester phosphorus, reducing phosphorus inputs from watersheds to lakes and streams (Prepas et al. 2001b). On the western Canadian Boreal Plain, peatland cover was positively associated with runoff (Gibson et al. 2002) and phosphorus and ammonium exports from small watersheds to streams (Prepas et al. 2006). On the central Canadian Boreal Shield, wetlands also had a positive relationship with phosphorus export to lakes (Paterson et al. 2006) and in general, surface waters were found to have elevated DOC concentrations in catchments with wetlands compared to those without wetlands (Marin et al. 1990, Dalva and Moore 1991). This association has been attributed to prolonged periods of time for water in wetlands to interact with shallow organic-rich soils before this water is released to receiving waters (Schiff at al. 1998). Human activities such as peat harvesting, mining and agriculture have reduced the number of wetlands in Canada (Wilson et al. 2001,

[1] A version of this manuscript, co-authored by E.E. Prepas and D.W. Smith, has been accepted by the journal, *Lake and Reservoir Management*.

Conference on Canadian Wetlands Stewardship 2003). For this reason, an effective wetland classification and identification system is essential to quantify wetlands within managed boreal forest watersheds.

Accurate wetland identification is required for application of the Soil and Water Assessment Tool (SWAT) (Arnold et al. 1998) to boreal watersheds. This tool was originally developed to model water and nutrient movement in agricultural settings. It has been modified for use by researchers in the Forest Watershed and Riparian Disturbance (FORWARD) Project, a watershed-based experimental disturbance project on the Boreal Plain (Smith et al. 2003), to predict runoff and surface water quality in forested watersheds before and after harvest (Putz et al. 2003, McKeown et al. 2005). Application of the SWAT model to the boreal forest requires accurate estimates of wetland area within a watershed. More specifically, model adaptation requires accurate peatland projections, because peatlands influence runoff nutrient exports from FORWARD project watersheds (Prepas et al. 2006). Ground truthing is not feasible across large landscape areas (e.g., the FORWARD study area, in the Swan Hill in the province of Alberta, includes an area of 12 000 km^2); therefore a wetland identification method that uses remotely accessed information was required.

Although remote sensing techniques (e.g., satellite imagery, light detection and ranging (LiDar) and aerial photograph interpretation) facilitate resource identification, their application can be difficult. For example, the optical sensors (e.g., Landsat MSS) used in satellite remote sensing have proven of limited usefulness in vegetated wetlands, because they cannot penetrate dense foliage (Sader et al. 1995). Radar sensors are useful at low and high frequencies for detecting wet areas under forest and shrub vegetation, respectively. A combination of these techniques, in association with digital elevation model (DEM) data was applied to three boreal forest wetland areas in Ontario, Quebec and Labrador (Li and Chen 2005). Similar to optical sensors, the low-frequency C-band Radarsat-1/SAR sensor applied in this study could not penetrate dense forest cover. Application of LiDar to produce DEMs of the ground and canopy in Boreal Plain wetlands tended to overestimate the ground surface where conditions were wet, therefore application of DEM data requires ground truthing to correct biases introduced by local vegetation characteristics (Hopkinson et al. 2005). Aerial photographs, though more costly to obtain than satellite-based data, provide information at a much higher spatial resolution (Franklin et al. 2002).

On the Boreal Plain, the Alberta Wetland Inventory (AWI) has been widely used to identify and classify wetlands based on aerial photograph interpretation. The AWI is ideally suited for the identification of large wetlands, including areas specified as caribou habitat (Anderson 1999). This focus on the large scale resulted in a system that missed information on the fine spatial scale that is mandatory for the development of hydrologic modeling. More recently, the AWI has been used to identify and classify wetlands within industrial footprints, identify vegetation patterns and thus wetland potential (e.g., Prepas et al. 2001a, Smith et al. 2003). However, ground truthing in the small (median 5 km^2) FORWARD watersheds demonstrated that the AWI missed many wetlands important for hydrologic modeling (Prepas, unpubl. data) and a new improved wetland identification system was thought to be achievable. The goals of this project were to: 1) develop an alternative method to quantify and classify wetlands on the Boreal Plain with the same original database as the AWI (1:15 000 aerial photography), but revised based on field data and adjusted parameter selection and 2) compare the accuracy of this alternative method to the AWI.

Methods

Study Area

The FORWARD study area (UTM coordinates: X 540697 to 575474; Y 6009112 to 6057501) in the Swan Hills of Alberta, Canada, occurs within the Forest Management Areas (FMAs) of Millar Western Forest Products Ltd., the FORWARD major industry partner, and Blue Ridge Lumber Inc., a Division of West Fraser Timber Company Ltd., a FORWARD industry partner (Figure 2.1).

Twelve small FORWARD project watersheds were used for this study. Nine inventory watersheds occur in a western portion of the Millar Western FMA (FMA-W) (976 km^2) and 3 test watersheds occur in the Blue Ridge FMA (2700 km^2) (Figure 2.1 and Table 2.1). Part of the Millar Western FMA (1757 km^2) to east of the Swan Hills (Millar Western FMA-E), although used for testing in this project, is not shown in Figure 2.1 for reasons of scale.

The study area lies within the Boreal Plain ecozone, which constitutes approximately 20% of the Canadian boreal forest and includes central portions of the provinces of Manitoba, Saskatchewan and most of northern and central Alberta (Figure 2.1). Elevations across the Boreal Plain are typically 300 to 600 m above sea level and mean annual precipitation is 450 mm (Canadian Council on Ecological Areas 2006). Compared to the Boreal Plain as whole, the Swan Hills contains special features including more topographic relief, higher elevations (775 to 1225 m), and greater annual precipitation (1975-2005 mean 551 mm; Environment Canada 2006). Internal drainage is slow within the fine-textured Luvisolic soils that dominate the study area (Ecological Stratification Working Group 1996), therefore peatland formation may occur in poorly drained areas. Peatlands with peat depths of several metres are common in the region (Ecological Stratification Working Group 1996). For example, using soil based methods, peatland cover was estimated to be as much as 29% of the area among the small FORWARD watersheds (Prepas et al. 2006).

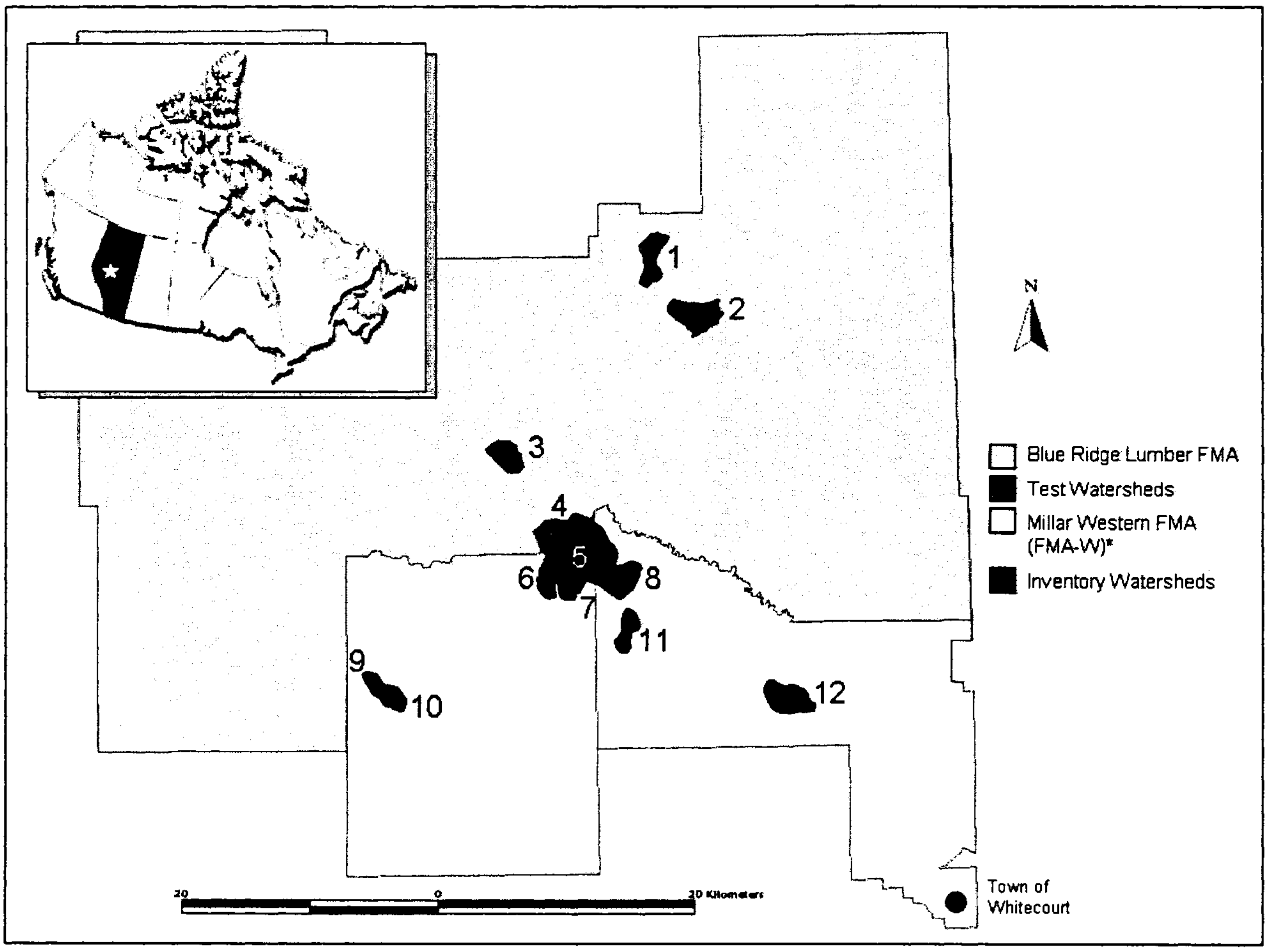

Figure 2.1. The FORWARD study area in the Swan Hills, in the province of Alberta, Canada (inset), including the western portion of the Millar Western Forest Management Area (FMA-W) and Blue Ridge Forest Management Area. Project watersheds:1 Fireweed, 2 Burnt Pine, 3 1A, 4 Thistle, 5 Willow, 6 Millions, 7 Mosquito, 8 Cassidy, 9 Toby, 10 Pierre, 11 Kashka, and 12 Sakwatamau B. * The eastern section of the Millar Western Forest Management Area (FMA-E), although used for testing, is not shown.

In the study area, tree species such as black spruce (*Picea mariana* (Mill.) BSP) and larch *(Larix laricina* (Du Roi) K. Koch) dominate wet areas, whereas trembling aspen (*Populus tremuloides* Michx.), balsam poplar (*P. balsamifera* L.), white spruce (*Picea glauca* (Moench) Voss), and lodgepole pine (*Pinus contorta* Dougl. ex Loud. var. *latifolia* Engelm.) dominate in uplands. Common shrub species include willow (*Salix* spp.), alder (*Alnus* spp.), common Labrador tea (*Ledum groenlandicum*), and bracted honeysuckle (*Lonicera involucrata*). In wetlands, common Labrador tea, peat moss (*Sphagnum* spp.), bog cranberry (*Vaccinium vitis-idaea*), cloudberry (*Rubus chamaemorus*), three-leaved false Solomon's seal (*Smilacina trifolia*) and sedges (*Carex* spp.) are the dominant cover species.

Alberta Vegetation and Wetland Inventories (AVI and AWI)

The Alberta Vegetation Inventory (AVI) was designed for use by forest managers in the 1990s (Nesby 1997). The AVI permits delineation of homogeneous areas of vegetation (polygons) using 1:15 000 to 1:20 000 aerial photographs and it can be used to gain information regarding topography, vegetation and anthropogenic features such as right-of-ways. AVI Version 2.1 is used in government and industry applications in Alberta, where it is applied to aerial photographs to generate polygonal shapefiles, including data on dominant tree species, anthropogenic alteration and year of origin of the forest.

In Alberta, Halsey and Vitt (1997) developed the AWI based upon work by the National Wetlands Working Group (1988). The AWI was designed for application in tandem with the AVI to reduce overlap and interpreter error. The AWI also uses features visible on aerial photographs to quantify and classify wetlands. The AWI divides wetlands into peatlands (fens, bogs) or non-peat forming wetlands (swamps, marshes and shallow open-water) based on water level, underlying substrate, vegetation species and distribution, and vegetation and frost patterns. In peatlands, organic matter accumulates to > 400 mm depth, whereas non-peat forming wetlands accumulate less organic matter (National Wetlands Working Group 1997).

In this study, the AWI was initially applied to the nine inventory watersheds covering an area of 49.6 km^2 within the Millar Western FMA-W (Figure 2.1 and Table 2.1). Subsequently, the AWI was applied to all of Millar Western FMA-W. The AWI rules were applied to an AVI shapefile using 1:15 000 aerial photographs taken in 1994 and ESRI ArcView Geographic Information Systems (GIS) (v. 3.2., Environmental Systems Research Institute Inc., Redlands, CA). Once wet (hygric and subhydric soils) and mesic (submesic, mesic and subhygric soils) polygons were identified within the AVI, individual wetlands were classified using the AWI rules, which use vegetation structure (horizontal and vertical) and species composition as key features to identify wetlands and to distinguish among wetland classes. All wet areas were given wetland specific codes (e.g., FONS for open, non-patterned, shrub dominated fen), as specified by the AWI guidelines, whereas non-wetlands were given the code “Z”.

Wetland Inventory and Identification Tool Development

The Wetland Inventory and Identification Tool (WIIT) was developed to improve on wetland identification efforts in the FORWARD project area. The WIIT was created from a combination of the AVI and field confirmed plot data uploaded to ArcView. First, more detailed maps of the inventory watersheds indicated in Figure 2.1 were used to select 138 field plots (Table 2.1 and Figure 2.1). Field plots were then visited between

July 2004 and August 2005 and marked with a Garmin eTrex Legend 360° Global Positioning System (GPS) unit. Once the plot center point was determined, a 2 m diameter lower vegetation plot, 6 m diameter shrub plot and 10 m diameter tree plot was established in concentric circles. Within the lower vegetation plot, all non-woody plant species were identified and percent cover values determined. The sum of these values sometimes exceeded 100%, because mosses can cover 100% of the area and vegetation occupying other strata can also cover a percentage of the area. In this case, bog cranberry and small bog cranberry (*Oxycoccus microcarpus*) were classified as a lower vegetation species because they do not attain vertical dominance, although they are considered woody shrubs (Johnson et al. 1995). The shrub plots were similarly quantified and again, percent cover values could exceed 100%. For tree plots, percent cover never exceeded 100%. For the three tallest trees, species was identified and height (m) was determined using a Suunto PM-5/1520 clinometer.

Within each 10 m diameter plot boundary, a peat auger was used to core into the peat, peat depth (mm) was measured and soil colour was determined using the Munsell Soil Colour Chart (Kollmorgen Corp., Baltimore, MD). Soil colour was only used to determine that the site was indeed a wetland site, as the areas sampled had already been evaluated for soil characteristics (Prepas et al. 2006). If the field plot was more representative of an upland site, litter depth was recorded rather than the peat depth. Also, a portable well was placed at a depth of 0.3 m in the peat or soil to collect soil pore water. If no water seeped into the well after 20 minutes, the field plot was deemed un-wetted. Since all measurements were taken during the summer months and within a 10-year dry period for Alberta, false positive identifications of wetlands due to seasonably wet conditions in the field were reduced. Porewater pH was measured with an Accumet Model 1001 meter (Fisher Scientific, Nepean, ON). Photographs were taken at each field plot to illustrate the ground and canopy cover, as well as in the cardinal directions to give an accurate representation of the site.

Table 2.1. Watershed area, wetland cover, and number of field plots in twelve FORWARD watersheds, including the inventory watersheds in the western portion of the Millar Western Forest Management Area (FMA-W) and test watersheds in Blue Ridge Lumber Forest Management Area. Watershed areas were redigitized after Prepas et al. (2006).

Watershed Name	Watershed area (ha)	Wetland cover (ha)	No. field plots
Inventory Watersheds (Millar Western FMA-W)			
Pierre	258	0.9	2
Cassidy	593	28.4	4
Mosquito	311	13.4	4
Kashka	398	21.1	8
Millions	335	40.1	13
Toby	263	58.3	15
Sakwatamau B	704	219.7	31
Thistle[a]	512	111.5	25
Willow	1562	201.4	36
Subtotal	4936	694.8	138
Test Watersheds (Blue Ridge Lumber FMA)			
Thistle[a]	336	29.7	10
1A	510	128.6	24
Burnt Pine	766	121.7	9
Fireweed	569	157.7	13
Subtotal	2,181	437.7	56
Total	**7117**	**1133**	**194**

[a] Watershed straddles the two FMAs.

A field plot was deemed a wetland if water entered the portable wells and the site possessed appropriate wetland vegetation such as three leaved false Solomon's seal, cloudberry, bog cranberry and *Sphagnum* species (Johnson et al. 1995). A wetland field plot was deemed a peatland if the peat depth was > 400 mm. Peatlands were further characterized by pH: if the water that seeped into the wells was ≥ pH 4.5, then the peatland was labelled a fen and if it was < pH 4.5, it was labelled a bog (Clymo 1987, Price et al. 2005). Following the field investigation, a computer model was created using ArcView. The WIIT classifies land areas based on tree species and percent species composition data from the AVI, in conjunction with peat depth and water presence data collected at each field plot. Field plot coordinates were labelled with a code: confirmed wetland, possible wetland or confirmed non-wetland. For the inventory watersheds, a query was written in ArcView to select as many polygons as possible containing 'wetland' waypoints, and to avoid the capture of 'non-wetland' waypoints (Figure 2.2).

Waypoints designated as 'possible wetland' were only used as confirmation for other points in the development of the WIIT. Once a suitable query was written using overstory and understory tree species and non-tree vegetation, it was expanded to include the Swan Hills. Upon completion, the flowchart could be used with GIS software to identify wetlands at both the watershed and FMA level. In its final form, WIIT functions as a hierarchical decision-based rule set that identifies wetlands by utilizing polygon attribute information contained within the AVI, in conjunction with field data.

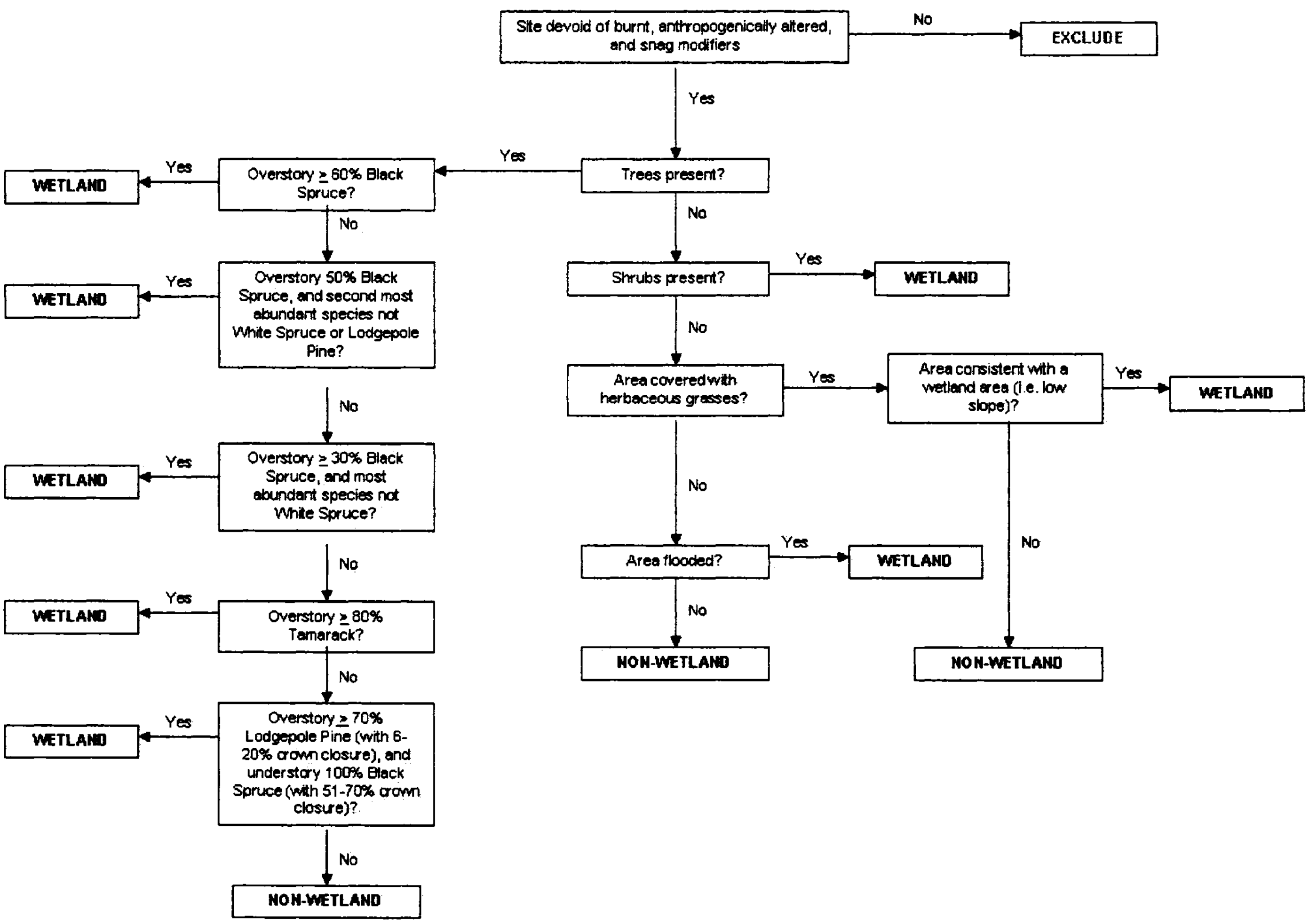

Figure 2.2. Flow chart describing the ArcView GIS v. 3.2 query processes used to create the Wetland Inventory and Identification Tool (WIIT).

AWI and WIIT Accuracy Assessment

The AWI and WIIT were tested against three datasets. First, both the AWI and WIIT were compared in their abilities to identify the 138 field plots established in the nine inventory watersheds. An independent set of wetland plot data in the inventory watersheds was not available for further testing. Second, the wetland area predicted by the two systems was tested against wetland area estimates created using an independent soil dataset compiled from intensive field sampling in all twelve (both inventory and test) small FORWARD watersheds (71 km^2) and Millar Western FMA-W (redigitized data from Prepas et al. 2006) (Figure 2.1). This soil dataset was initiated at the beginning of the FORWARD project in 2001 and new soil investigations are included in the database every field season. Soil plots have been established in the FORWARD project watersheds, as well as in the areas immediately surrounding the watersheds within Millar Western FMA-W and Blue Ridge FMA. Furthermore, a few plots were established in Millar Western FMA-E. While not as ideal as using wetland plots, the tests using the soil plots could still confirm wetland presence. Third, the AWI and WIIT were compared against the individual data plots from the soil survey mentioned previously. Organic soils generally accumulate in wetlands due to poorly drained conditions (Mitsch and Gosselink 2000), and gleysolic soils indicate long term or periodic flooding (Agriculture and Agri-Food Canada 1998). For these reasons, soil plots possessing organic and gleysolic soils were used as confirmation of wetland presence.

In addition to their abilities to identify wetlands, the AWI and WIIT models were compared in terms of their abilities to: 1) detect wetland polygons in Millar Western FMA-W in five size classes (<1 ha, 1 to 30 ha, 31 to 60 ha, 61 to 90 ha and 91 ha +) and 2) detect wetland polygons in Millar Western FMA-W belonging to four tree height classes (no trees, 1 to 5 m, 6 to 10 m and 11 m +). The chi-square goodness of fit test was used to compare modeled and observed wetland site distributions and distributions created by the two models based upon size class, tree height and soil type (Zar 1998).

The WIIT was further tested with two additional datasets. The WIIT was tested against the 56 ground truthed field plots in the Swan Hills test watersheds that are managed by Blue Ridge Lumber Inc. Different interpreters apply the AVI to the Blue Ridge FMA than in the Millar Western FMA-W, therefore an independent test of the WIIT could be conducted. The WIIT was also compared to the independent soil dataset (78 soil plots), after application to Millar Western FMA-E, to determine if the WIIT behaved similarly on a landscape with less topographic relief. The inventory watersheds occur in an area of greater relief (6% slope) whereas the remainder of the Boreal Plain possesses less relief (Prepas unpubl. data, Putz et al. 2003). Surface analysis with ESRI ArcGIS (v. 8.3) confirms that the three-dimensional area in Millar Western FMA-E exceeded two-dimensional area by only 0.06%, compared to 0.31% for the Swan Hills.

Results

Among the initial 138 field plots established in the inventory watersheds, ground truthing demonstrated that 106 were in confirmed wetland areas; 8 were within a potential wetland areas and the remaining field plots were non-wetlands (Table 2.2).

Table 2.2. Number of field plots confirmed as wetland sites by ground truthing and percentage predicted by the Wetland Inventory and Identification Tool (WIIT) and Alberta Wetland Inventory (AWI) in the inventory watersheds in the western portion of the Millar Western Forest Management Area (FMA-W).

Type of Wetland	Confirmed	% Predicted by WIIT	% Predicted by AWI
Fen	69	91%	39%
Swamp	19	47%	5%
Bog	7	86%	71%
Un-wetted	4	75%	25%
Marsh	3	67%	67%
Peatland	3	67%	0%
Open Water	1	100%	0%
Total	**106**	**81%**	**34%**

The WIIT and AWI detected 81 and 34% of the 138 inventory watershed wetland plots, respectively (Table 2.2). Of all sites (wetland and non-wetland), the omission error rate (confirmed wetlands that were not identified by the model) for the WIIT within the watersheds was only 15%, compared to 55% for the AWI (Figure 2.3).

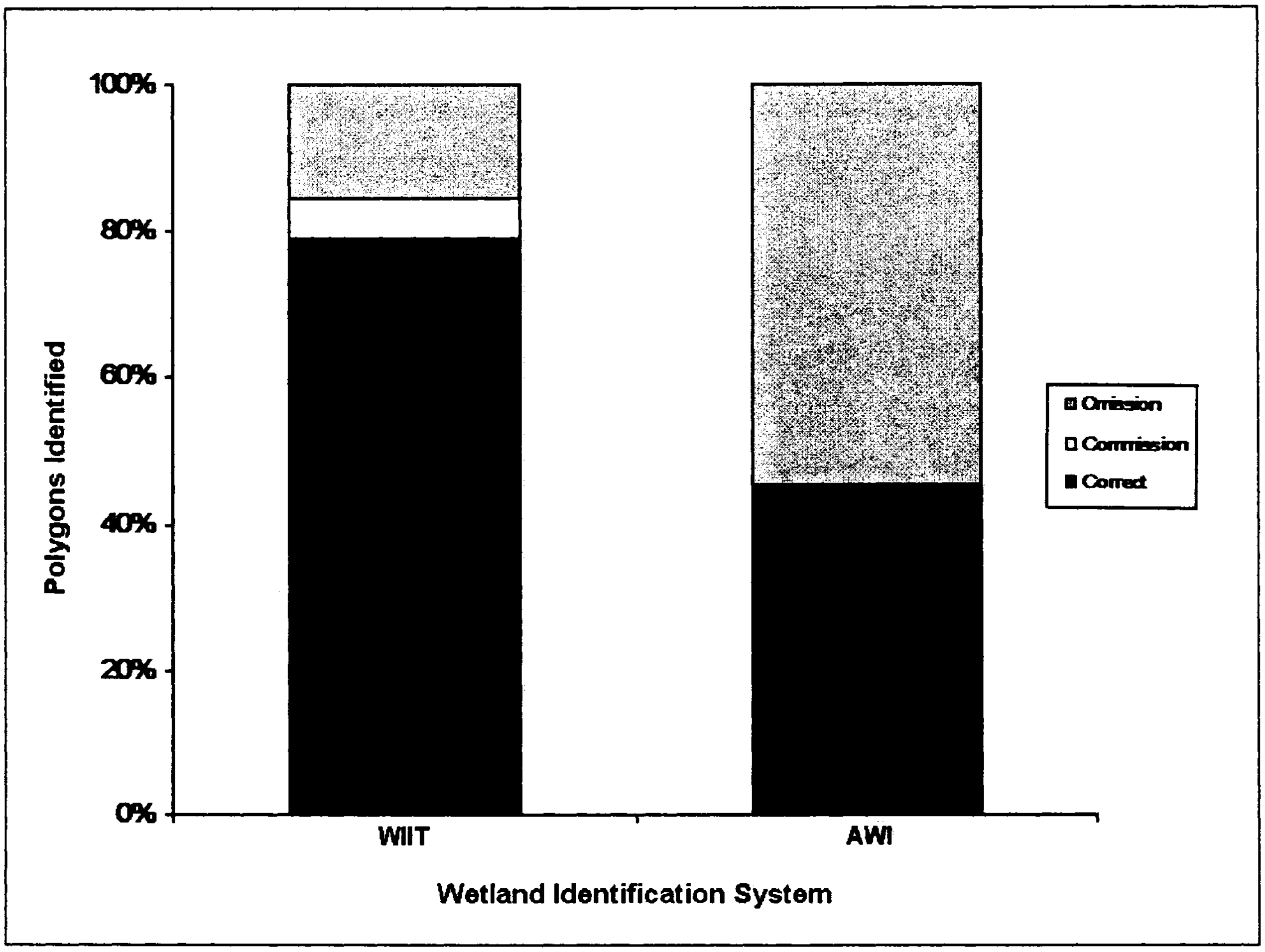

Figure 2.3. Comparison of the Wetland Inventory and Identification Tool (WIIT) and Alberta Wetland Inventory (AWI) methods to correctly identify wetland and non-wetland sites in the western portion of the Millar Western Forest Management Area (FMA-W), including omission and commission errors.

The overall correct identification rate for the AWI was 45% (including correct upland identification) (Figure 2.3). The improved ability of the WIIT to detect wetland plots was accompanied by a slightly higher commission error (5% of non-wetland plots were identified as wetlands) compared to the AWI (1%) (Figure 2.3). The WIIT captured approximately 1.5 times the wetland area as the AWI (e.g., Figure 2.4).

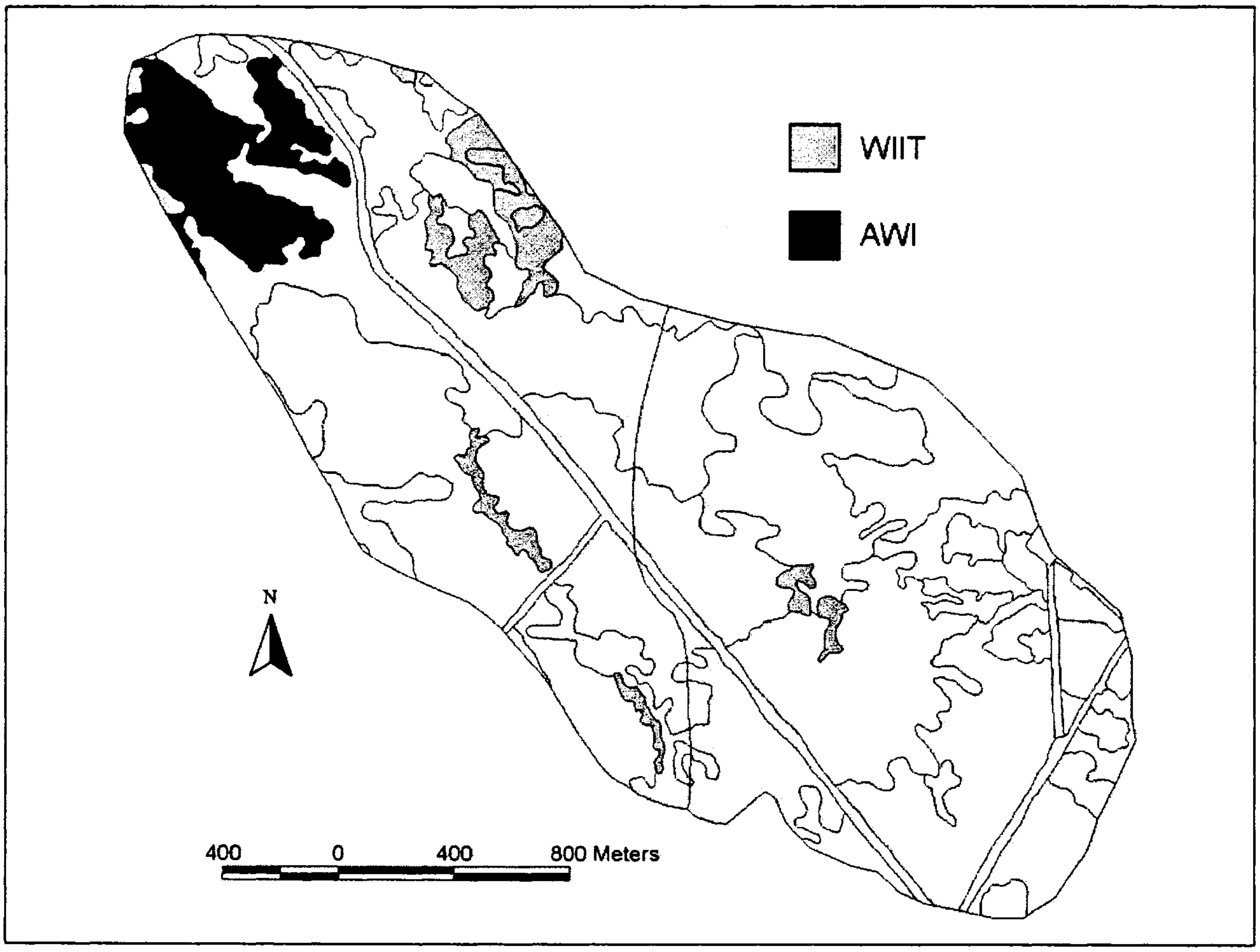

Figure 2.4. Wetlands identified using the Wetland Inventory and Identification Tool (WIIT) and Alberta Wetland Inventory (AWI) in the Toby and Pierre watersheds in the western portion of the Millar Western Forest Management Area (FMA-W). All wetlands identified by the AWI were also identified by the WIIT.

Compared to soil-based wetland areal estimates in the inventory watersheds, the WIIT captured 71% (492 ha) of the wetland area within the inventory watersheds. By comparison, the AWI identified only 46% of the wetland area (322 ha) within the inventory watersheds. The WIIT predicted 10% wetland cover for the test watersheds, while the AWI predicted 7% wetland cover. Application of the WIIT to Millar Western

FMA-W predicted 24% wetland cover, whereas application of the AWI rules predicted 12% wetland cover.

In the inventory watersheds, the WIIT and AWI differed in terms of their ability to identify wetland soil plots. The AWI did not identify as many wetland soils as the WIIT. It detected only 27% of confirmed organic soil plots and 3% of confirmed gleysolic soil plots, whereas the WIIT detected 73% of organic soil plots and 27% of gleysolic soil plots. The relative proportions of the soils identified differed between the two methods as well (χ^2=4.7, $P < 0.10$; $df = 2$) (Appendix A). Of all the wetland plots identified by the AWI, 91% were organic soils while 6% were gleysolic soils. These values were 63% and 27% for the WIIT. As opposed to the soil identification within the inventory watersheds, the number of wetland soil plots identified by the WIIT and the AWI did not differ when tested across Millar Western FMA-W (χ^2=1.3, $P > 0.25$; $df = 2$).

However, across Millar Western FMA-W, the wetland prediction models differed in terms of the size of wetland polygons detected. The WIIT identified more (12%) polygons of a smaller size than the AWI (7%) (χ^2=45, $P << 0.001$, $df = 4$) (Figure 2.5). Although the frequency distribution suggests that the two wetland identification systems detect a similar number of wetland polygons across the five size categories, there was sufficient weight in the < 1 ha category to allow the the chi square analysis to detect a difference.

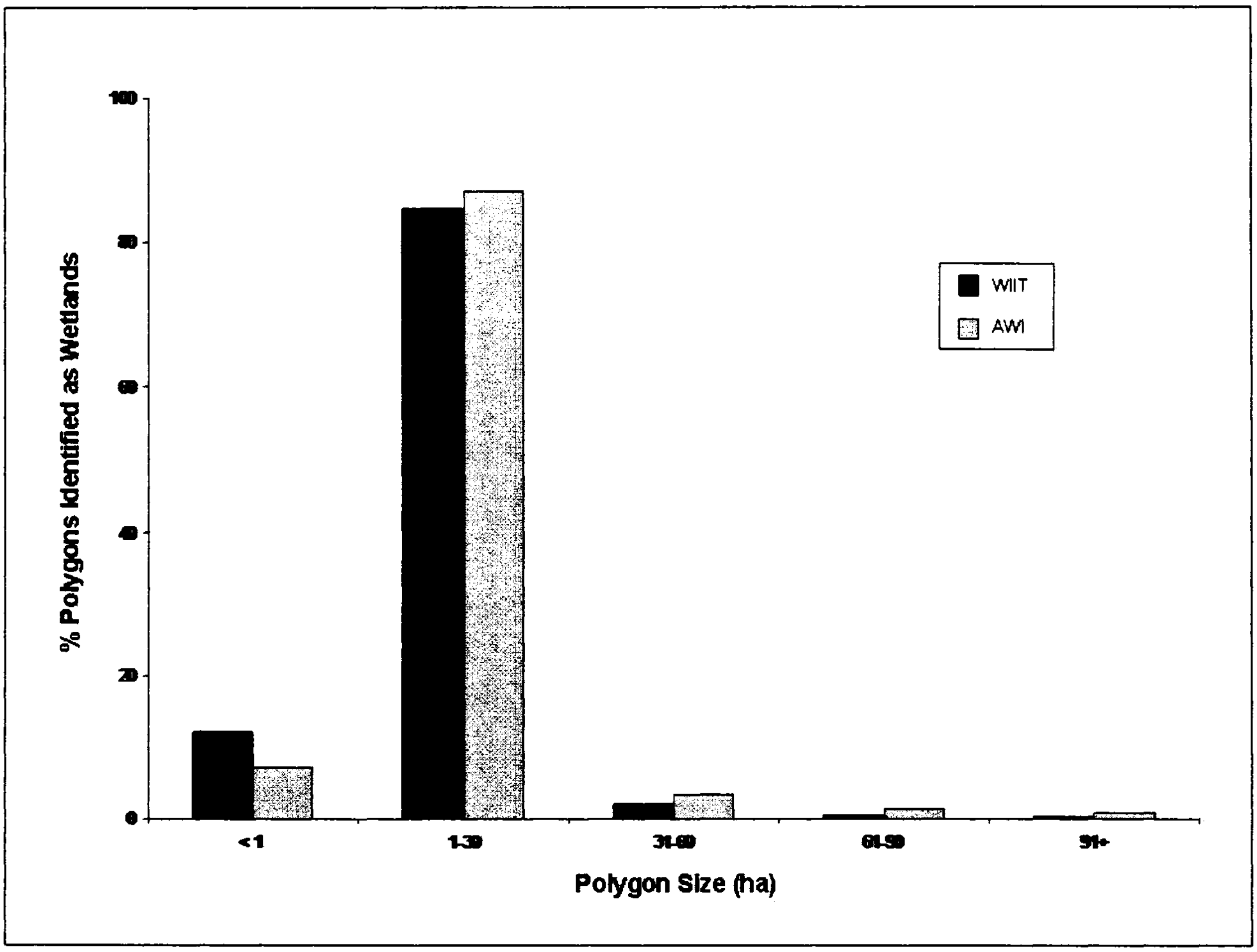

Figure 2.5. Percentage of polygons of five size classes identified by the Wetland Inventory and Identification Tool (WIIT) and Alberta Wetland Inventory (AWI) in the western portion of the Millar Western Forest Management Area (FMA-W).

Tree heights within the wetland polygons identified by the WIIT were taller than those identified by the AWI (χ^2=75, P << 0.001, df = 3) (Figure 2.6). Using the WIIT, 36% of the wetland polygons identified had a tree height > 11 m, compared to 24% for the AWI. Conversely, the WIIT only identified 28% of the wetland polygons as having tree heights between 1 and 5 m, while the AWI placed 36% of the polygons in this category.

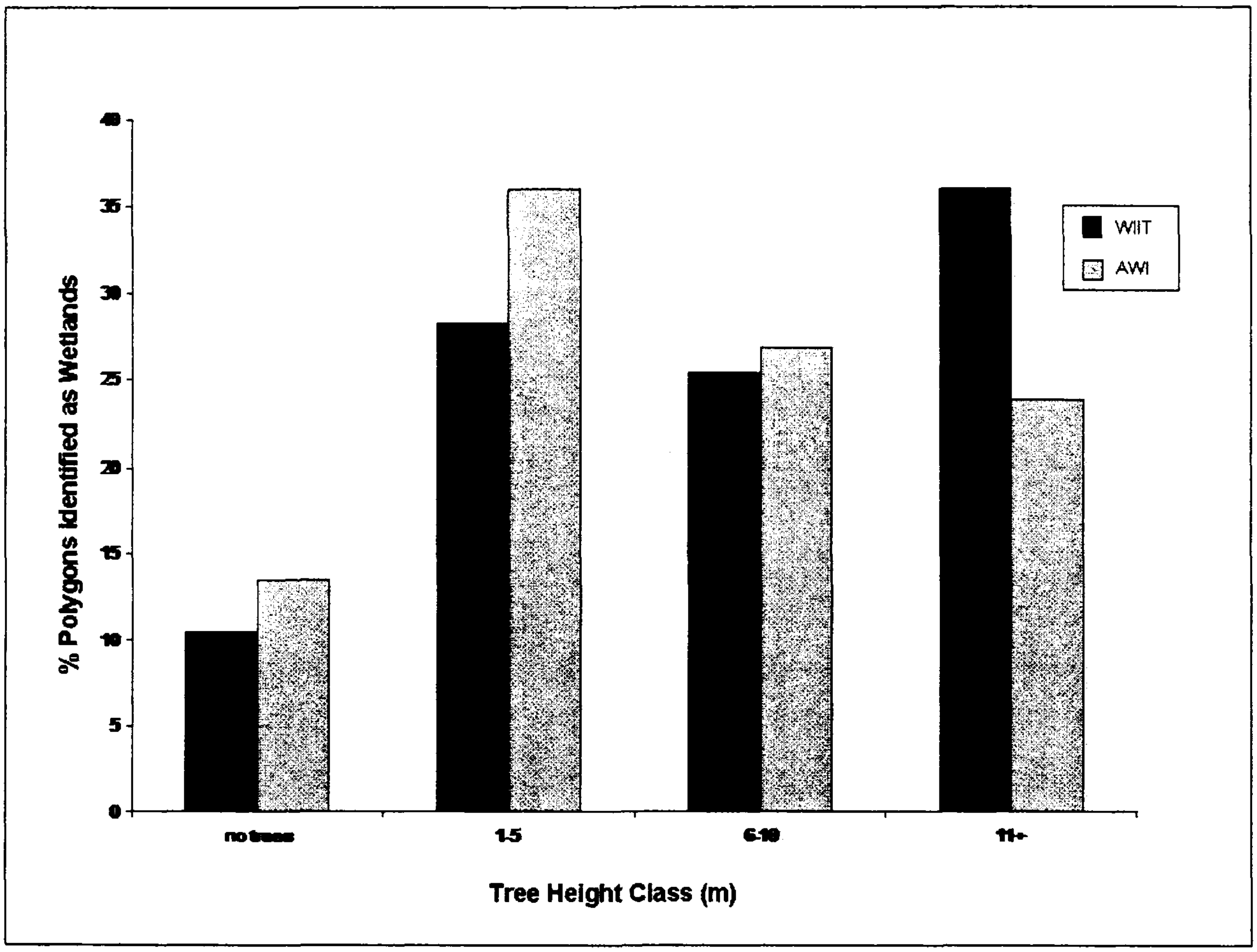

Figure 2.6. Percentage of polygons of four tree height classes identified by the Wetland Inventory and Identification Tool (WIIT) and Alberta Wetland Inventory (AWI) in the western portion of the Millar Western Forest Management Area (FMA-W).

Of the 56 plots ground truthed in the test watersheds, 31 were in confirmed wetland areas, 13 were in potential wetland areas and 12 were in confirmed upland areas. When applied to these test watersheds, the WIIT correctly identified 84% of the field plots as wetlands, or correctly excluded upland field plots as non-wetlands. Commission and omission field plot identifications were 7% and 9%, respectively. Of the 78 soil sites established in Millar Western FMA-E, 22 plots had organic soils and 15 plots had gleysolic soils. The WIIT identified 77% of the organic soils and 20% of the gleysolic soils.

Discussion

The newly developed WIIT was better able to detect ground truthed wetland field plots than the AWI, largely due to its ability to identify wetland areas with taller trees. The Swan Hills appears to contain many treed wetlands that were not detected using the AWI, possibly because the trees were considered too tall to occur in wetlands by the AWI rules. Conversely, the WIIT lacks strict rules relating to tree height. Given the greater relief in the Swan Hills, and thus greater drainage, trees can attain greater heights. The AWI may be better suited to identify bogs and marshes in areas outside of the Swan Hills, which possess short trees, and grasses and sedges, respectively.

In the future, trees in wetland areas in the Swan Hills could possibly be slated for harvest, having implications for the amount of runoff and quality of water draining these watersheds. In the boreal forest of Alberta, wet and fine-textured soils are prone to soil compaction during harvesting operations (McNabb et al. 2001, Whitson et al. 2003) and after tree removal, soils are vulnerable to raindrop impact. These factors are associated with lower infiltration rates, higher runoff and more overland flow, which can exacerbate erosion. One year after forest harvest, runoff from small watersheds in the Swan Hills increased by approximately 60% (Prepas unpubl. data). In Alberta, soils are phosphorus-rich, therefore harvesting in treed wetlands could enhance eutrophication in receiving waters. Knowledge of the location of treed wetlands in this region can enhance planning and management that try to limit hydrologic impacts of forest harvest.

The newly developed WIIT was also better able to identify smaller wetland polygons. A potential source of error with the AWI method could stem from its dependence on aerial photograph interpretation of larger areas and local features.

Application of the AWI rules to aerial photographs at a larger scale (e.g., 1:5000) could potentially detect smaller wetlands on the landscape, but the general landscape features, on which some of the AWI rules are based, could be missed (O. Bakowsky, 21 Nov 2005, Edmonton, AB., pers. comm.). Conversely, the WIIT is entirely based on the AVI and field confirmation, and selects polygons regardless of polygon area. The AWI did not identify some wetland polygons because of their small size, though this was not a consistent error in the method. The FORWARD watersheds may fall in areas where the general landscape characteristics suggest uplands rather than lowlands. It appears that the AWI rules are too exclusive for use in this part of Alberta; the WIIT is a more encompassing wetland identification model than the AWI.

The AWI was created and tested in portions of Alberta possessing lower relief, which could explain why it does not work well in areas of higher relief, such as the Swan Hills. Furthermore, while AWI rules result in a high degree of precision of wetland identification, in this situation the process was too exclusionary and confirmed wetlands were not identified. A process using more relaxed rules is therefore ideal. While the WIIT is an improvement on the AWI wetland identification in this area, it may not be applicable elsewhere in the province. It is understood that wetland hydrology faces a challenge when using generalized site-specific studies and scaling up to apply the model to larger landscape (Sophocleous 2002). Although the WIIT is most suitable to the Swan Hills, it may be successfully applied in other high-relief areas on the Boreal Plain, such as the Caribou and Birch Mountains. It must be emphasized, however, that while the WIIT was developed for and within areas of greater relief, the process did identify 77% of organic soil points established in areas of lesser relief.

The resolution of aerial photographs is more detailed relative to other remote sensing techniques and smaller wetlands are captured, as was the case with the WIIT. Whereas optical and radar-based remote sensing offers advancement towards landscape-scale wetland identification, particularly in areas where aerial photographs have not been taken (Wulder et al. 2003), the WIIT or other field based predictions could be integrated with remote sensing to provide a more detailed landscape-based confirmation for satellite based estimations.

The WIIT was applied successfully to a variety of landscapes in and adjacent to the FORWARD study watersheds. With improved wetland identification, the FORWARD project can now more accurately model the hydrology of the individual watersheds. At this point, the limitations of the WIIT method relate to potential inaccuracies within the AVI, upon which the WIIT is based, as well as the resolution of the mapping procedure. Some non-wetland polygons were captured and some wetland polygons were not captured, mostly due to projection errors (i.e. inaccuracies when GPS reading was taken) or potential AVI inaccuracies. Existing methods may have also underestimated very small wetland areas, because there appear to be numerous very small wetlands in the FORWARD study area. Some wetted areas may be too small to be assigned a polygon, based on aerial photograph interpretation, but they could be important hydrologically. Logistically, it may be difficult for field operators to avoid these small wetlands during harvest operations. The WIIT has been tested most extensively in the Swan Hills, and to some degree in Millar Western FMA-E. Although there is confirmation of the abilities of the WIIT within the project watersheds, the wetland coverage estimates outside of the watersheds is, at this time, based on fewer tests. The

Swan Hills and related area tests (12 000 km^2) were successful based on ground truthed soil plots. With testing on more landscapes in the boreal forest, this simplified process should be widely applicable.

The simplified techniques used to test the WIIT, using tree species and soil types, are easily applied. Furthermore, testing of the process can be conducted elsewhere in the boreal forest. Given that black spruce is widespread across the boreal forest, and that jack pine (*Pinus banksiana* Lamb.) can be supplemented for lodgepole pine in the WIIT rules, testing of the WIIT could easily be conducted in other boreal forest regions in Canada. Testing of the WIIT is slated for Boreal Shield forests in northwestern Ontario; at this time the challenge of upland black spruce will be addressed.

The WIIT identified 81% of wetland field plots in inventory watersheds, whereas the AWI identified 34% of the same plots. Similarly, the WIIT outperformed the AWI by identifying 73 versus 27% of organic soil plots within the inventory watersheds. While inaccuracies are still present with both systems, the WIIT is easier to apply, less susceptible to interpreter error and more economically feasible, because it is applied by a computer system rather than an interpreter. The improved process used in the WIIT can be applied within the context of resource extraction, road and municipal planning and watershed management. More specifically, the WIIT could be used by conservation authorities, other watershed managers and lake associations to identify wetlands and the hydrological and chemical storage capacity benefit stored therein. In addition, the WIIT could be used by conservation groups and provincial governments to identify natural areas of interest to establish priorities for protection.

CHAPTER III: PEATLAND PHOSPHORUS AND WATER BUDGET IN THE BOREAL PLAIN OF ALBERTA, CANADA, AND THE IMPLICATIONS FOR SMALL STREAM CHEMISTRY[2].

Introduction

Wetlands cover approximately 20% of the boreal forest across the globe (Zoltai and Pollett 1983), and approximately 14 to 18% of the landbase in Canada (National Wetlands Working Group 1988, Conference on Canadian Wetlands Stewardship 2003). In Canada, 85% of wetlands are peatlands, which are wetlands with at least 400 mm of accumulated peat. On the western Canadian Boreal Plain, fens and bogs are the most dominant wetland types (National Wetlands Working Group 1988). Fens are peatlands with water at pH greater than or equal to 4.5, and bogs are peatlands with water at pH less than 4.5 (National Wetlands Working Group 1997). Wetlands in Canada develop based on the north-south temperature gradient and the east-west precipitation gradient. These gradients were used to establish 20 wetland regions across the country (National Wetlands Working Group 1986).

The role of wetlands in hydrological networks in boreal forests depends upon wetland position in the network and wetland type. For example, the residence time of water in riparian wetlands (wetlands adjacent to streams) is shorter than in wetlands surrounded by upland, containing a single outlet (Mitsch and Gosselink 2000). Water table levels and outflow rates remain relatively uniform in fens and swamps because

[2] Specifics detailed in this chapter will be integrated into a manuscript, co-authored by D. Pelster, S. Luke, J. Burke and E. Prepas, to be published in a special issue of *Journal of Environmental Engineering and Science* in 2008.

they are connected to groundwater. Bogs, however, are not connected to groundwater networks, and therefore water tables fluctuate in response to precipitation events (Devito et al. 1996). In one central Ontario study, sedge fens (peatland), beaver ponds (non-peatland) and treed swamps (non-peatland) in close proximity to each other retained 2, 5 and 8% of yearly water inputs (2 year study mean), respectively (Devito et al. 1989).

In addition to influencing local hydrology, wetlands influence phosphorus (P) movement through watersheds to varying degrees, based upon the amount of contact between water and the mineral soil (Richardson 1985). As an essential nutrient for all living organisms, P is a component of ATP and DNA (Emsley and Hall 1976). In aquatic systems, P takes on additional importance because P is often the limiting nutrient, having no gaseous form and thus unable to be replenished from the atmosphere like nitrogen and carbon. Because P is also expensive to remove from many point sources (e. g. sewage, industrial effluents) it is often a cause of anthropogenic eutrophication, or nutrient enrichment resulting in increased primary productivity. A lake is considered eutrophic if the P concentration is 75 µg/L or greater (Wetzel 2001). In a Boreal Shield lake in northern Ontario, carbon and nitrogen amendments to one half of an experimentally divided lake did not cause eutrophication, but augmented with P, these treatments resulted in rapid growth of cyanobacteria (Schindler 1974). Recovery of the lake was swift following the removal of the P source. In another classic North American study, eutrophication was also associated with P inputs from the drainage basin to Lake Washington, including significant phytoplankton population changes and hypolimnetic oxygen deficits (Edmondson et al. 1956).

Among wetlands, peatlands appear to retain P, that is, they remove P from the water column through biological, physical and chemical means, while not releasing it under unexceptional hydrological conditions (e.g., mean rainfall) (Reddy et al. 1999). The retention of P results from uptake by plants and microbes (Richardson 1985), adsorption to metals such as aluminum and iron (Hansen et al. 2003), adsorption to non-metal ions like calcium, and adsorption to sediments and organic matter, which are buried and become essentially unavailable. In a three-year study, peatlands in Minnesota retained 56% of P (Verry and Timmons 1982), most likely due to geochemical sorption of P from the water column (Bridgham et al. 1998). In northern Michigan, peatlands retained 90 to 100% of a radioactive P tracer within the first 24 hours after application (Kellogg and Bridgham 2003). On the basis of this ability to retain P, treatment wetlands have been established in places such as Florida to remove P from agricultural wastewater with high P concentrations (Moustafa 1999). The understanding of wetland functioning on the Boreal Plain is of importance given that the wetland loss in Alberta can potentially accelerate due to increased economic growth (Wilson et al. 2001)

The perception that peatlands function as P sinks may not apply across all regions. For example, peatland cover and P export in streams draining experimental watersheds in the Swan Hills, Alberta, on the Canadian Boreal Plain were positively correlated (Prepas et al. 2006). Soil parent materials in this region are P-rich (Mitchell and Prepas 1990, Cooke and Prepas 1998). Wetland cover ranged from 1 to >30% of the landbase in the 12 small (258 to 1562 ha) Forest Watershed and Riparian Disturbance (FORWARD) project watersheds. Although fens, bogs, swamps, marshes and shallow open water are all represented among wetlands in the FORWARD watersheds, fens comprised the majority of established field plots (Chapter 1, this thesis, Table 2.2). As part of the FORWARD project, key watershed features that influence streamflow and water quality are being quantified in an effort to model these variables, so that water and associated nutrients can be included as a constraint in forest management planning (Smith et al. 2003). Given the importance of wetlands in terms of runoff and water quality, the importance of fens among the wetland types in the Swan Hills and the importance of P in defining surface water quality (Schindler 1974, Prepas and Trew 1983), knowledge of fen water and P budgets is essential to FORWARD modelling efforts. The goal of this project was to construct an annual (November 2004 to October 2005) water and P budget for a small boreal peatland, to relate wetland cycling of water and P to the surrounding upland.

Methods

Project Area

The study wetland, in the Swan Hills of Alberta, is located at UTM coordinates X 555283 to 554795 and Y 6024962 to 6025329, and at a mean elevation of 1025 m above sea level. The long-term (1975-2005) mean annual precipitation measured at Whitecourt, Alberta (35 km to the southeast of the wetland) is 551 mm, of which 76% is rainfall, and the long-term mean annual temperature is 2.7°C, a degree cooler than the annual mean temperature for 2005 (Environment Canada 2006). The study wetland is within the small (1562 ha), relatively undisturbed Willow watershed (Figure 3.1).

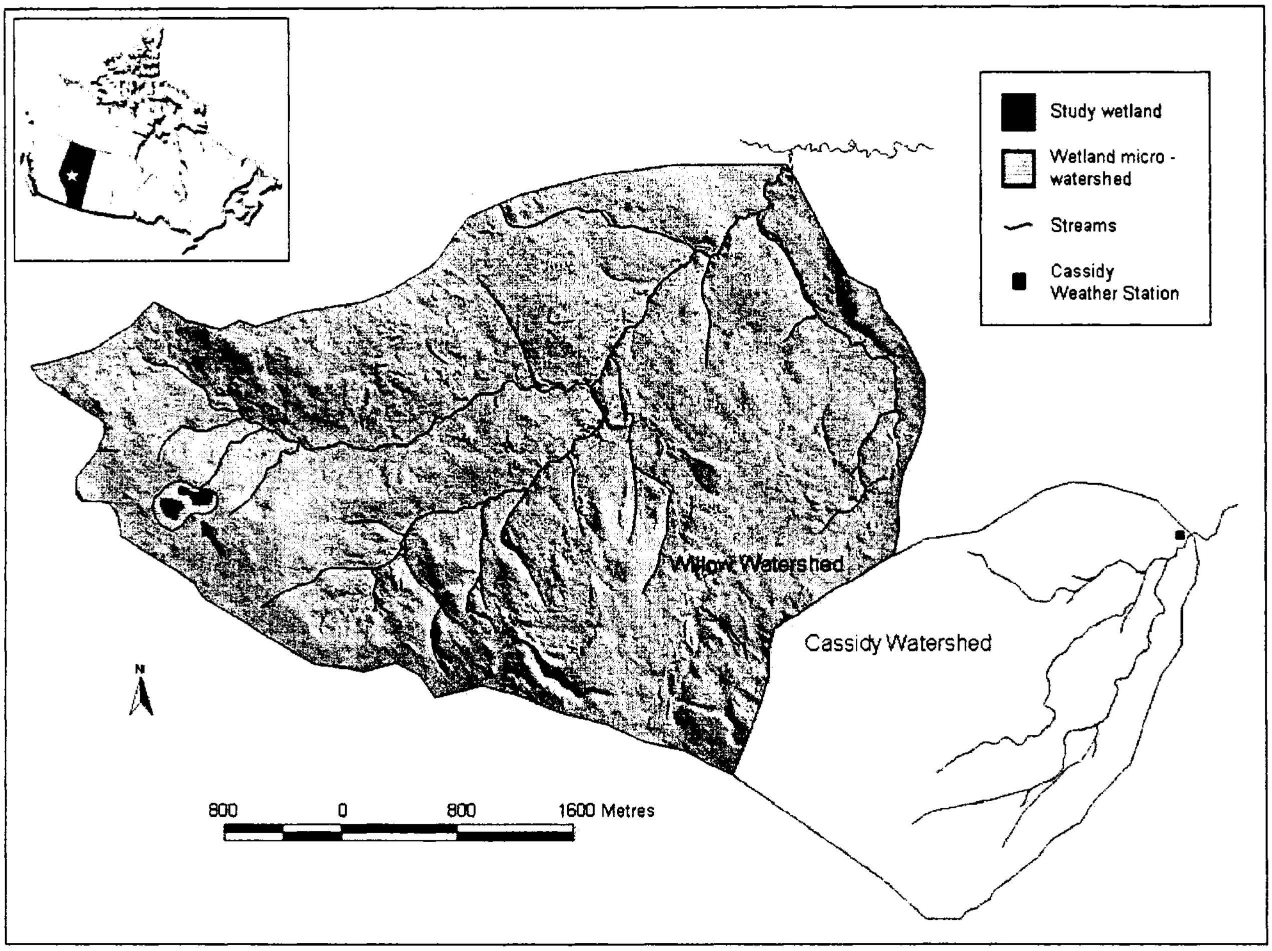

Figure 3.1. Study wetland location (indicated by the arrow) within FORWARD Willow watershed, Swan Hills, Alberta, Canada (inset). Cassidy watershed is shown adjacent to Willow watershed.

The position, size and other characteristics of the wetland and the land area draining directly into the wetland were determined from ground surveys, aerial photograph interpretation and topographic data from a recently flown LiDar layer (11 May 2005 at 800 m, Airborne Imaging Inc., Calgary, AB) (Brooks et al. 2003) (Figure 3.2). The 3 ha wetland is surrounded by a 6.2 ha micro-watershed, which based upon surface topography, contributes runoff to the wetland (Figure 3.3). The wetland was predominantly populated by black spruce (*Picea mariana* (Mill.) BSP), but also present were a few lodgepole pine (*Pinus contorta* Dougl. ex Loud. var. *latifolia* Engelm.) trees.

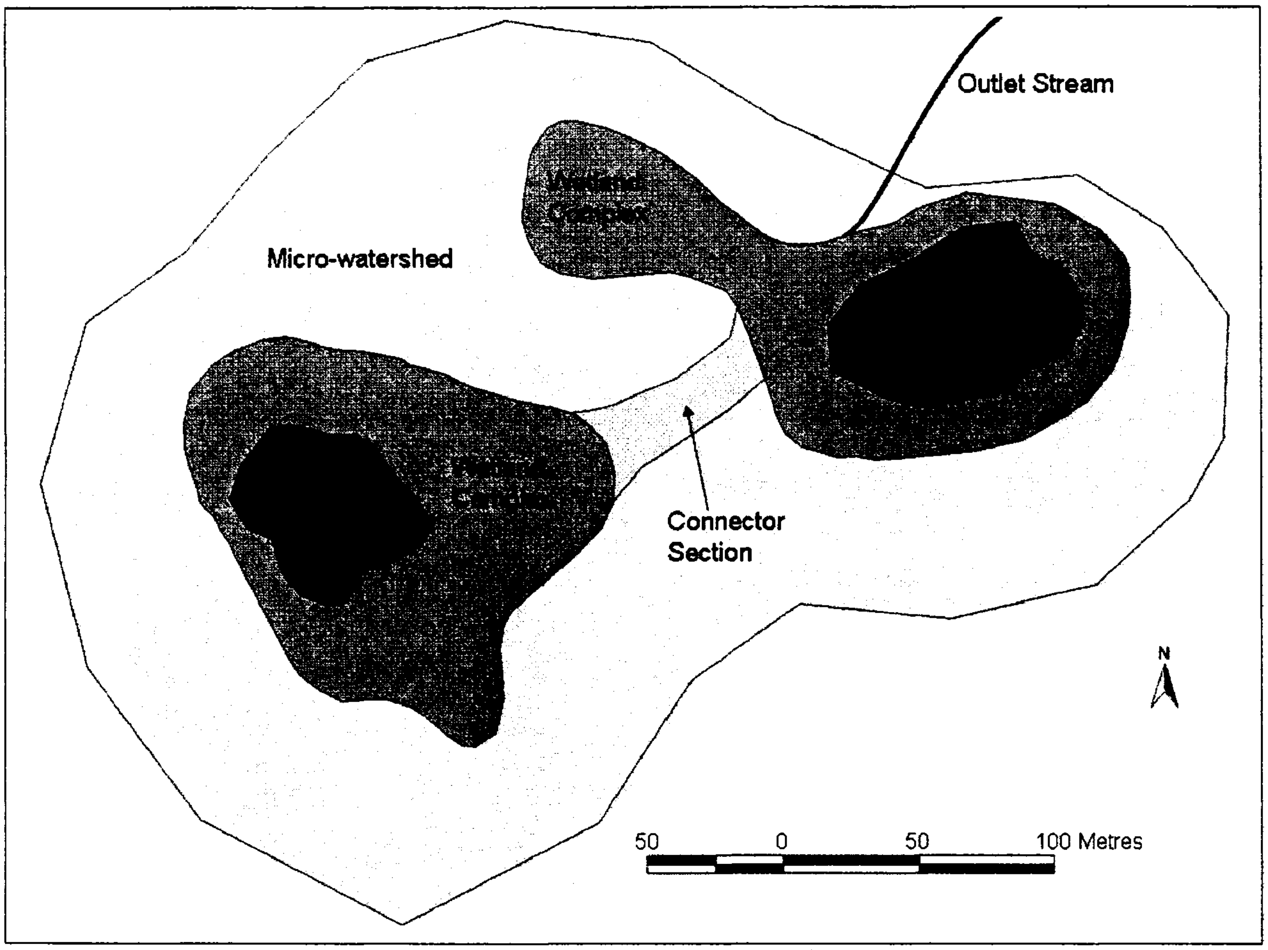

Figure 3.2. Study wetland including peatland, treed peatland margin (wetland complex) and surrounding micro-watershed, situated within the Willow watershed.

Peat depth ranges from 100 mm at the peatland margin to 810 mm (mean = 316 mm), creating a connected two-lobed wetland complex. The dominant tree species in the micro-watershed surrounding the project wetland is trembling aspen (*Populus tremuloides* Michx.). The surface topography is flat to very slightly northeast sloping. A single, clearly defined outlet was identified northeast of the wetland, and there was no evident obstacle to water passage.

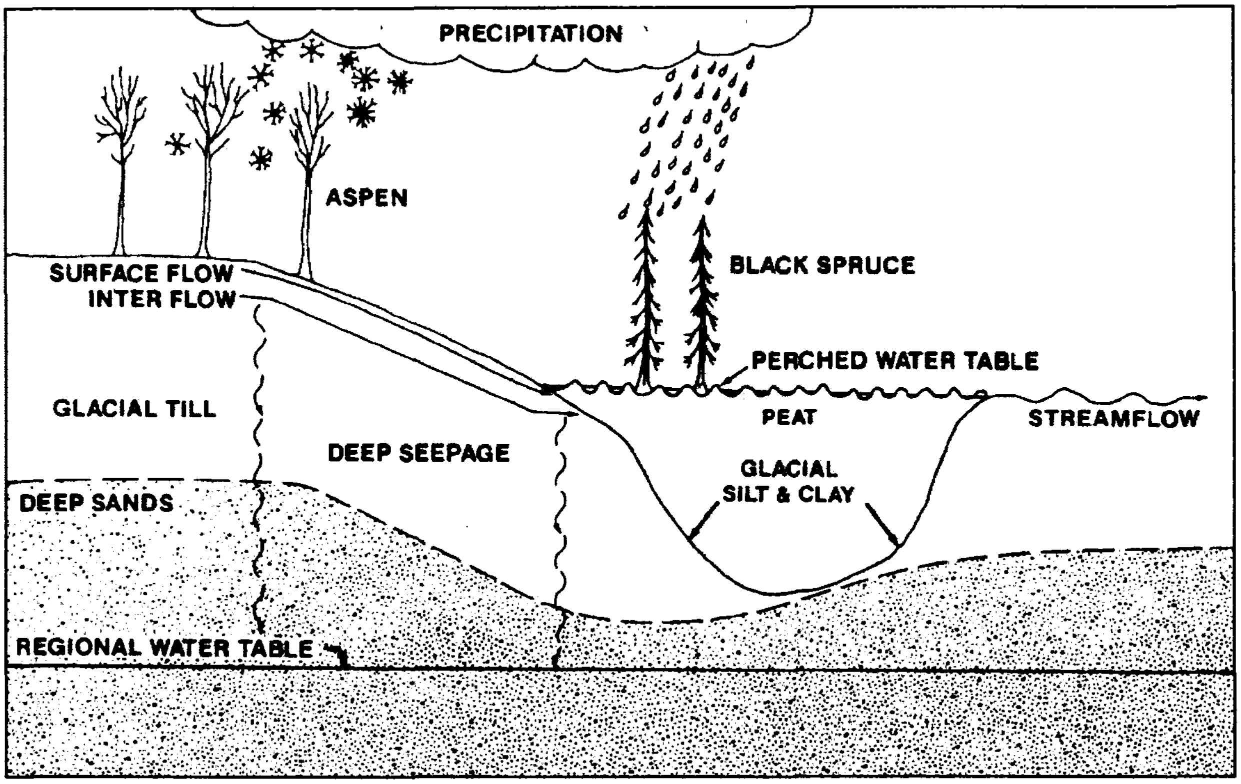

Figure 3.3. Cross section of a wetland and micro-watershed showing water routes (Verry and Timmons 1982). Used by permission from Ecological Society of America.

Field Methods

Inputs

Atmospheric Deposition and Throughfall

Precipitation input volumes (Figure 3.3) were estimated for the November 2004 to October 2005 water year using three sources. First, a rain gauge was erected in the study wetland to collect rainwater for the month of August. For the remainder of the 01 April to 31 October 2005 period, data from a rain gauge located 6.5 km east of the wetland in Cassidy watershed (Figure 3.1) were used to estimate rainfall over the project wetland. The Cassidy watershed is approximately one third of the area of the

Willow watershed, and it is similar to the Willow watershed in that it drains towards the northeast and into the Sakwatamau River. To estimate 01 November 2004 to 31 March 2005 rain and snow fall, a regression equation was developed (Microsoft Excel, Microsoft Corporation, Redmond, WA) based upon weekly total rainfall data from the Cassidy rain gauge (dependent variable) and from the Whitecourt A station (independent variable), a year-round meteorological station located 35 km southeast of the wetland (Environment Canada 2006) (Appendix B).

Bulk (wet plus dry) deposition samples were collected from 02 July to 22 August 2005 to determine atmospheric P deposition rates. Four atmospheric deposition and rain collectors were erected in natural clearings within the wetland. Each collector was constructed of two 3 m lengths of Schedule 40 polyvinylchloride (PVC) pipe (Rice Engineering, Edmonton, AB), a 210 mm inner diameter plastic funnel, glass wool (in the neck of the funnel to prevent debris from entering containment vessel), clear PVC tubing (Greenline Tubing, Edmonton, AB) and a 3.8 L brown opaque polypropylene collection bottle (Figure 3.4).

Deposition samples were collected every third day. If it had rained since the last collection date, the rainwater volume was measured and poured into a 300 mL acid-washed clear polypropylene bottle, which had been rinsed with a small amount of the sample. Any P remaining in the funnel was assumed negligible, and was washed into the collection bottle with distilled deionized water (DDW) and discarded. If no rain was collected, 300 mL of DDW was washed down the funnel to collect any dry-only deposition that had accumulated. The 3.8 L brown opaque polypropylene collection bottles, funnels, tubing and graduated cylinder were rinsed with DDW between collection periods, and glass wool was replaced to prevent contamination. Some P may have sorbed into the plastic and onto glassware, and therefore our final P estimates may be underestimates.

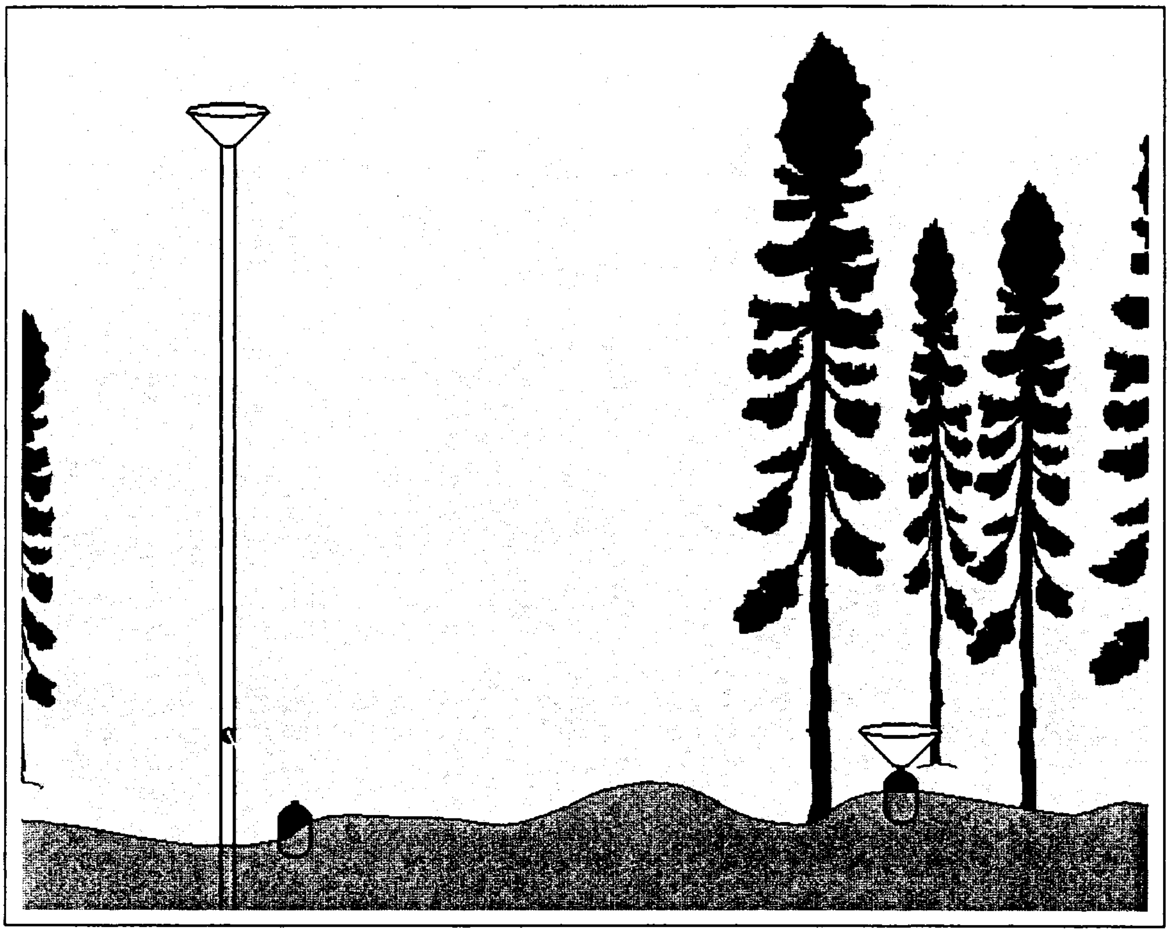

Figure 3.4. Schematic diagram of atmospheric and throughfall collectors used in the study wetland and its micro-watershed.

Throughfall samplers were constructed in the same manner as atmospheric collectors, but they did not sit atop PVC pipe (Figure 3.4). Five collectors were placed under black spruce canopy within the wetland and two were placed under trembling aspen in the wetland micro-watershed. Bulk and dry-only deposition samples were collected and analyzed with the same frequency and methods as atmospheric collectors.

Runoff

Runoff volume and P export for the micro-watershed surrounding the wetland were calculated using areal runoff and P export values from the Willow watershed, measured with the same protocols employed in this study (FORWARD core data). The amount of water and P contributed to the wetland through runoff includes both surface and subsurface (interflow) sources (Figure 3.3). Deep seepage from upland areas to the regional water table was assumed to bypass the wetland and was ignored in the budget (Figure 3.3).

Groundwater

Groundwater wells were installed in the peatland (n = 5), peatland margin (n = 5) and upland soil (n = 2). These wells were constructed from PVC pipe (Schedule 40, inner diameter 63 mm) with 1 mm slots 6 mm apart, washed with DDW and wrapped in spunbond polyester fabric (NILEX MD7407, opening size <100 μm) to prevent large particulates from entering them. The base of the well rested on mineral soil in wetlands, whereas in the upland soils, wells were installed deep enough into the mineral soil so that the water table was reached. Groundwater samples were collected every six days by emptying the wells using a Nalgene hand pump, allowing them to recharge and then pumping water into acid-washed sample bottles using the hand pump, an Erlenmeyer flask and two lengths of clear PVC tubing (Greenline Tubing, Edmonton, AB). Upon collection, the samples were analyzed for pH.

Outflow

Water from the wetland flowed through a well defined, well sloping ravine toward the Willow stream channel (Figure 3.3). A 90° v-notch weir and staff gauge were established and surveyed before and after sample periods to ensure that they did not shift. Outflow measurements and water samples were taken every three days during the July through August sampling period.

Phosphorus Analyses

All water samples were kept cool until analysis could take place (within one week). Analysis for total P (TP) and total dissolved P (TDP) (< 0.45 µm) concentration in water were conducted using the potassium persulfate digestion method (Menzel and Corwin 1965, Prepas and Rigler 1982) at the University of Alberta's Meanook Biological Research Station. The detection limits of this analysis are less than 1 µg/L (Stainton et al. 1977). Particulate P (PP) concentration was calculated as the difference between TP and TDP concentration.

Data Analysis

Inputs

Atmospheric Deposition and Throughfall

For April to October 2005, measured rain gauge values were used for precipitation calculations. For November 2004 to March 2005, the following relationship between the rain gauge in Cassidy watershed and the Whitecourt A meteorological station was used ($r^2 = 0.66$, $P < 0.0001$, $n = 23$; Equation 1):

$$R_{wet} = 0.664(R_{E.Can}) + 3.890 \quad (1)$$

where:
R_{wet} = rainwater depth estimated at Cassidy watershed rain gauge in mm
$R_{E.Can}$ = rainwater depth at Whitecourt A meteorological station in mm

Spruce throughfall, aspen throughfall and atmospheric deposition water volumes and P concentrations were compared with a one-way Analysis of Variance (ANOVA). All statistical analyses were conducted using Data Desk 6.0 (Data Description, Inc. Ithaca, NY). Where differences were found, a Tukey's honest significant difference (HSD) test was performed.

Atmospheric and throughfall P loading rates were calculated for July and August from the bulk and dry-only deposition atmospheric and throughfall collectors, using the following equation from Shaw et al. (1989) (Equation 2):

$$P_a = \frac{[P] \times V_{sample}}{SA_{funnel} \times D} \quad (2)$$

where:
P_a = atmospheric or throughfall P loading rate in $mg/m^2/d$
[P] = P concentration in sample water in mg/m^3
V_{sample} = volume of sample in m^3
SA_{funnel} = surface area of funnel in m^2
D = period of time exposed in days

Mean monthly TP and TDP loading rates were calculated by determining the deposition totals for the exposure periods and summing the values. For the months of

July and August, the monthly mean was used as a substitute for periods when sample collection did not occur. Throughfall estimations were only made for July and August, the period of sample collection. Given that the investigation of P concentrations and amounts deposited via throughfall were preliminary, their contributions were not included in the annual P budget.

For atmospheric bulk deposition alone, the remaining 10 months of the year, when sample collection did not take place, supplemental deposition values from Shaw et al. (1989) were used, the only published regional atmospheric study to date,

Groundwater

The P concentrations measured among the groundwater wells in the peatland margin, peatland and upland soils were compared using a single factor ANOVA. Where differences were found, a Tukey's HSD test was performed.

Output

Outflow

The outflow rate over the weir was measured from 02 July 2005 to 22 August 2005 and determined using equations from Grant and Dawson (1997) (Equation 3):

$$Q = \frac{4969H^{2.5}}{3600} \qquad (3)$$

where:
Q = discharge in m^3/s
H = height of water flowing over weir in m

For the remainder of the open-water season (01 April to 31 October), when outflow was not monitored, outflow volume was estimated based upon the positive relationship between daily wetland outflow (Q) and daily outflow for the Willow watershed, which was monitored during the entire open water season ($r^2 = 0.58$, $P = 0.006$, $n = 11$; Equation 4):

$$\log Q_{wet} = 0.573(\log Q_W) - 7.124 \qquad \textbf{(4)}$$

where:
Q_{wet} = discharge (m^3/s) from wetland
Q_W = discharge (m^3/s) from the Willow watershed

For months when no outflow water was collected, P concentrations were derived from negative relationships for TP ($r^2 = 0.37$, $P = 0.02$) and TDP concentration ($r^2 = 0.42$, $P = 0.009$) ($n = 15$; Equations 5 and 6) *versus* wetland discharge, using Microsoft Excel:

$$\log[TP_{wetland}] = -0.217(\log Q_w) + 2.126 \qquad \textbf{(5)}$$

$$\log[TDP_{wetland}] = -0.210(\log Q_w) + 1.761 \qquad \textbf{(6)}$$

where:
$TP_{wetland}$ = total P concentration (µg/L) in water leaving wetland
$TDP_{wetland}$ = total dissolved P concentration (µg/L) in water leaving wetland
Q_w = flow leaving wetland

Seepage and Evapotranspiration

Shallow seepage from the wetland to the stream channel and evapotranspiration (ET) from the wetland soil, water and vegetation surfaces are potentially important output routes (Brooks 2003) that were not directly measured in the field.

To include seepage and ET in the wetland budget, values from a study conducted in north-central Minnesota by Verry and Timmons (1982) were used as surrogates. The wetland and surrounding micro-watershed areas in that study are similar (3.24 ha and 6.48 ha, respectively) to this study. Tree species included black spruce in the peatland and trembling aspen and paper birch (*Betula papyrifera* Marsh.) in the uplands. The underlying bedrock is Precambrian Ely Greenstone and is overlain by Warba mineral soils in the upland and glacial sediments in the wetland. The annual mean temperature was 4°C and the area received an average of 762 mm of precipitation each year, of which 75% occurred as rainfall (Verry and Timmons 1982). Although mean annual precipitation and temperature were 38 and 5% greater, respectively, in the Minnesota study area, this was the most comparable study to use as a surrogate for this study.

Budget Calculation

Water and P budgets were completed for the study wetland for the November 2004 through October 2005 water year by using field values for the months of July and August and supplemental data from Verry and Timmons (1982) and Shaw et al. (1989). The P budget was conducted as per Likens et al. (1977) (Equations 7 and 8):

$$W_{bal} = W_{input} - W_{outflow} - W_{seepage} - W_{ET} \tag{7}$$

where:

W_{bal} = amount of water retained or released by wetland mm
W_{input} = amount of water entering the wetland through runoff and direct rainfall in mm
$W_{outflow}$ = amount of water leaving the wetland through outflow in mm
$W_{seepage}$ = amount of water leaving the wetland through shallow seepage in mm
W_{ET} = amount of water leaving the wetland through evapotranspiration in mm

$$P_{bal} = P_{input} - P_{outflow} - P_{seepage} \qquad \textbf{(8)}$$

where:
P_{bal} = mass of P accumulated or released during the year in g
P_{input} = mass of phosphorus accumulated by the wetland through rainfall and runoff in g
$P_{outflow}$ = mass of phosphorus released by the wetland through outflow in g
$P_{seepage}$ = mass of phosphorus released by the wetland through shallow seepage in g

A negative W_{bal} or P_{bal} value incomeates that the wetland lost water or P (wetland as a water or P source to the stream network) and a positive balance indicate that the wetland gained water or P (wetland as a water or P sink), during the November 2004 to October 2005 water year.

Results

Inputs

Atmospheric Deposition and Throughfall

The annual precipitation received by the 3 ha project wetland in Willow watershed was 403 mm. It was estimated that 45 mm of precipitation was deposited as rain or snow during the November 2004 through March 2005 period; 127 mm was measured directly as rain during July and August 2005. For the remainder of the April-October period, it was estimated that 231 mm of rain fell on the wetland surface. In total, precipitation represents approximately two thirds of the total water input to the wetland.

During July and August, 13 and 16% of the annual atmospheric bulk TP deposition occurred, respectively, based on an annual TP bulk deposition loading rate of 19.6 $mg/m^2/y$, determined from these data and data from Shaw et al. (1989) (Table

3.1). Bulk atmospheric TP contributions to the wetland were 80 g for July, of which 60% occurred as wet deposition at a rate of 2.61 mg/m^2/mo, and 100 g for August, of which 84% occurred as wet deposition at a rate of 3.19 mg/m^2/mo (Table 3.1). For the months of July and August, TDP constituted 44.7 and 58.7% to the TP deposition, respectively. The TDP fraction constituted 41.7 ± 5.5% of wet deposition and 62.2± 7.0% of dry-only deposition. The TP and TDP mass did not differ detectably in bulk and dry-only deposition (*P* = 0.23 and 0.32, respectively). During one storm on 06 August 2005, a large amount of hail was deposited onto the wetland. Bulk atmospheric samples collected the following day indicate that 39 g or approximately 6% of the annual atmospheric TP load occurred during this event.

Table 3.1. Atmospheric phosphorus contributions for the study wetland in the Willow watershed, Swan Hills, Alberta.

Period*	Atmospheric loading rate (mg P/m^2)	P deposited on wetland (g)
November - April	0.8	25
May	7.4	228
June	4.3	133
July	2.6	80
August	3.2	98
September	1.1	34
October	0.2	7
Total Annual	**19.6**	**605**

*data from periods other than July and August from Shaw et al. (1989).

Aspen throughfall volume measured during July and August 2005 was similar to rainfall volume, and these values exceeded spruce throughfall volume by 100 and 159%, respectively (P = 0.001) (Table 3.2) (Appendix C).

Table 3.2. Mean July though August water volume, total phosphorus (TP) and total dissolved phosphorus (TDP) concentrations and phosphorus mass in bulk atmospheric deposition and throughfall from two different canopy types in the study wetland micro-watershed (July – August 2005).

Collector Type	Volume (mL)	TP concentration (μg/L)	TDP concentration (μg/L)	TP mass (mg)	TDP mass (mg)
Bulk	74.3	165.3	65.7	0.014	0.008
Spruce	28.7	668.4	378.6	0.016	0.008
Aspen	57.6	265.5	165.8	0.012	0.007

The mean bulk TP and TDP concentration in spruce throughfall was 4 and 6 times higher, respectively, than in rainfall (P = 0.01 and 0.04, respectively). The mean bulk TP concentration in aspen throughfall was similar to rainfall, but only 40% of bulk TP concentration in spruce throughfall (P = 0.56 and 0.03, respectively), and the bulk TDP concentration did not differ from either spruce throughfall or atmospheric deposition (P = 0.12 and 0.46, respectively) (Table 3.2.).

The net effect of differences in water volume and P concentration was that the total mass of TP and TDP deposited was the same for aspen and spruce throughfall, and bulk deposition (P = 0.67 and 0.80, respectively). The concentration and mass of dry-only TP and TDP deposition was also similar (TP: P = 0.70 for both; TDP: P = 0.37 and 0.41, respectively).

Runoff

The 6.2 ha micro-watershed contributed 105 mm of water and 389 g of P (6.3 mg/m^2) to the 3 ha wetland for the November 2004 to October 2005 water year. The micro-watershed contributed approximately 33 and 40%, respectively, of the total annual water and P input to the wetland.

Groundwater

The TP concentrations in groundwater in the peatland margin wells and peat wells differed from each other and were 8.4 and 3.3 times greater than in upland wells, respectively ($P < 0.001$) (Table 3.3). The TDP concentrations in groundwater were similar between the peatland and peatland margin wells, however, they were 4.7 and 3.4 times greater than in the upland wells, respectively. ($P = 0.07$; $P < 0.001$).

Table 3.3. Mean (± Standard Error) particulate phosphorus (PP) and total dissolved phosphorus (TDP) concentrations for peatland, peatland margin and micro-watershed groundwater wells in the study wetland and micro-watershed.

	Peatland		**Peatland margin**		**Upland**	
	PP (µg/L)	**TDP (µg/L)**	**PP (µg/L)**	**TDP (µg/L)**	**PP (µg/L)**	**TDP (µg/L)**
July	98 ± 20	69 ± 4	288 ±46	80 ± 8	27 ± 3	13 ± 4
August	78 ± 17	117 ± 8	370 ± 68	53 ± 15	24 ± 2	19 ± 3

As the summer progressed, there was a trend for the proportion of TDP in TP to increase in the peatland section of the wetland and in the wells in the upland section of the micro-watershed (Figure 3.5). Conversely, the proportion of TDP in TP decreased in the peatland margin sections of the wetland during progression of the summer.

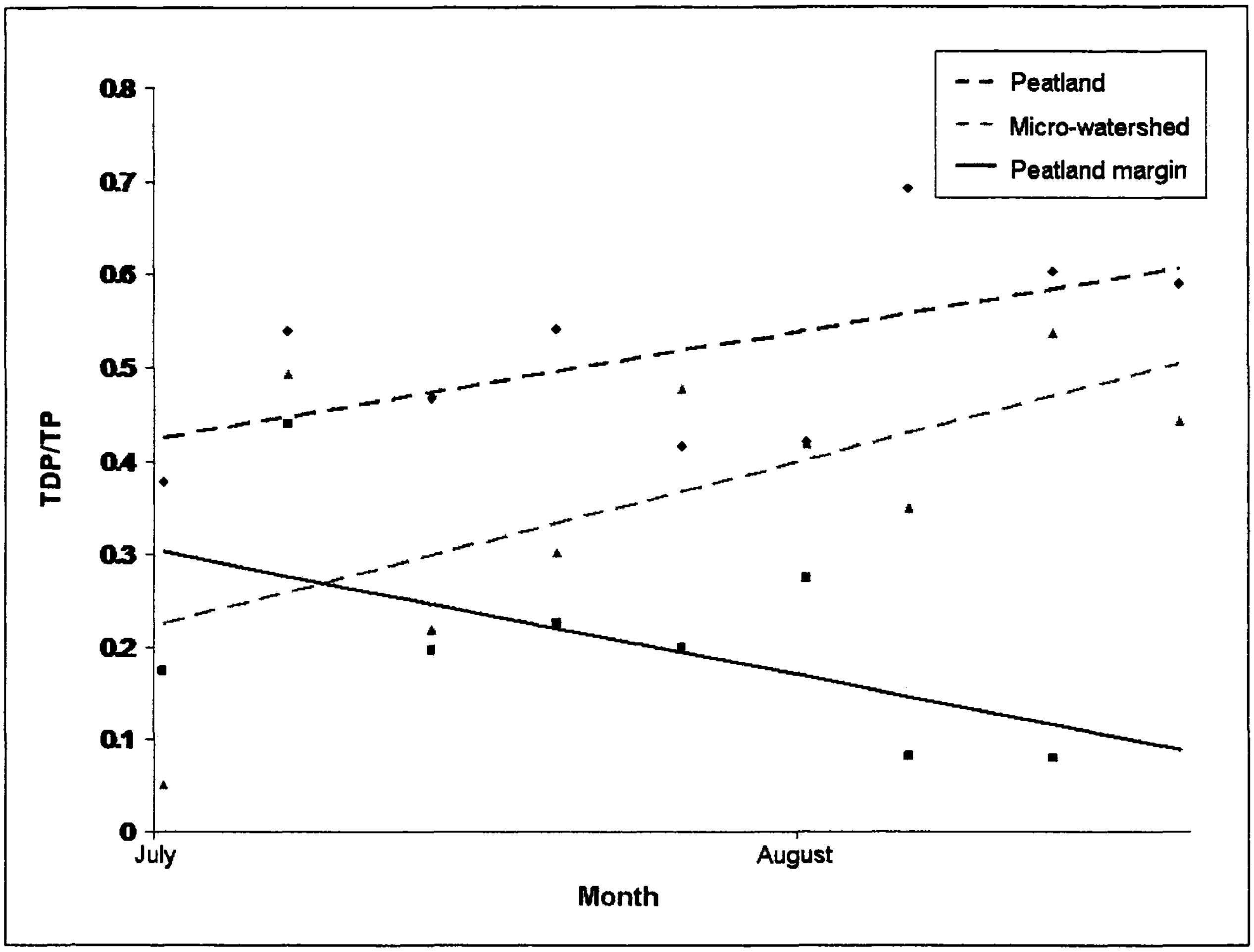

Figure 3.5. Total dissolved phosphorus (TDP):total phosphorus (TP) relationship in peatland, peatland margin and upland locations in the study wetland for the months of July and August 2005.

The mean pH for the four peatland wells for the duration of the study period was 4.4 ± 0.05.

<u>Output</u>

Outflow

A total of 98 mm of water (Figure 3.6) and 158 g of P were exported from the wetland from 01 November 2004 to 31 October 2005). These values represent 84 and 68%, respectively, of the total outflow of water and P from the wetland. Given that the majority of the analyses used to derive water and P exports were based on regression relationships, the values are estimates.

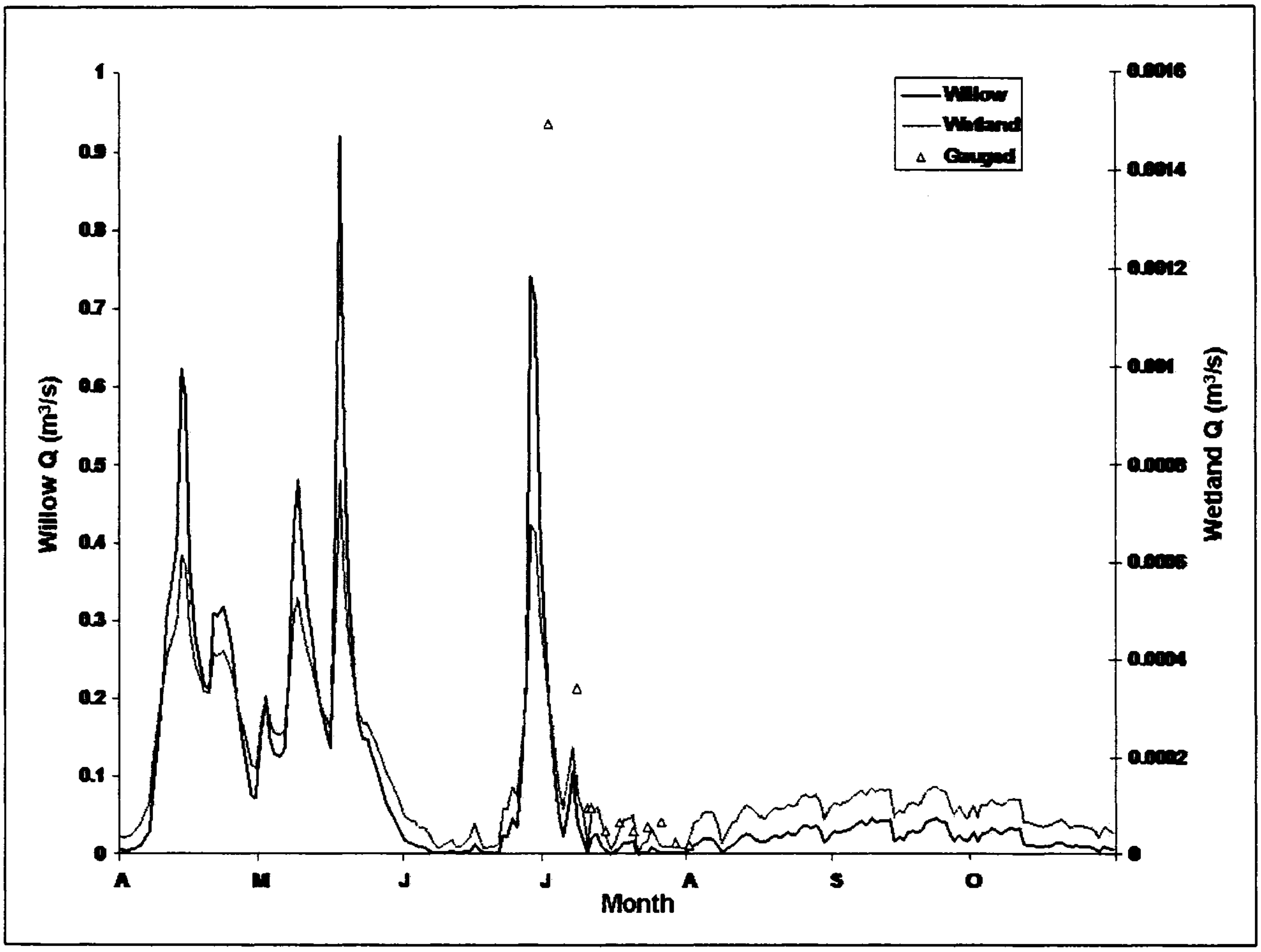

Figure 3.6. The Willow watershed and gauged and predicted wetland outflow (Q).

The export of P during the months of April and May was almost half of the annual outflow export (Table 3.4). Also, both TP and TDP concentrations decreased in water as the flow leaving the wetland increased. The proportion of TDP in TP ranged from 32 to 91% among individual samples.

Table 3.4. Monthly stream outflow export and flow-weighted phosphorus (total phosphorus (TP), total dissolved phosphorus (TDP) and particulate phosphorus (PP)) concentrations for the study wetland in the Willow watershed.

		Concentration (μg/L)			Export (g)		
Month	**Water export (mm)**	**TP**	**TDP**	**PP**	**TP**	**TDP**	**PP**
April	25	52.8	34.6	18.2	36.2	23.8	12.4
May	28	49.1	32.3	16.8	41.0	27.0	14.0
June	10	73.6	47.4	26.2	17.5	11.1	6.4
July	12	71.3	46.4	24.9	18.7	12.8	5.9
August	7	65.4	42.6	22.8	14.4	9.4	5.0
September	10	60.6	39.6	21.0	17.5	11.4	6.1
October	6	67.1	43.7	23.4	12.9	8.4	4.5
Total	98				**158**	**104**	**54**

Seepage

Water loss to shallow seepage from the wetland was estimated at 19% of the total volume of water lost to outflow and the mass of P lost to seepage was approximated at half of the P lost to outflow (Verry and Timmons 1982). Using these proportions in the project area resulted in a loss of 18 mm of water and 74 g of P to seepage from the wetland.

Evapotranspiration

Similar to seepage, ET values from the water and nutrient budget conducted in Minnesota were used to estimate ET from the study wetland. In the Verry and Timmons (1982) study, ET from the wetland was calculated using the water balance method (water input less water output) and estimated at approximately 63% of the inputs directly to the wetland (Verry and Timmons 1982). Applying this ET value to the study peatland, the wetland lost 255 mm of water, or 41% of the direct inputs to the wetland.

Water Budget

Of the 508 mm that entered the wetland during the water year, it was estimated that 255 mm was lost to ET from the wetland surface, 98 mm was lost to surface outflow and 18 mm was lost to wetland seepage. The wetland retained 27% of the input water.

Phosphorus Budget

Given the bulk atmospheric P loading rate of 19.6 $mg/m^2/y$, and the runoff P loading rate of 6 $mg/m^2/y$, the TP input to the wetland was 994 g for the November 2004 to October 2005 water year (Table 3.5).

The TP export from the wetland via streamflow and from seepage is estimated at 158 g and 74 g, respectively, resulting in a total P export of 232 g during the study year. The small wetland in the Willow watershed retained 762 g of P, or 77% of the P deposited from the atmosphere and through runoff and seepage (Table 3.5).

Table 3.5. Monthly total phosphorus (TP) inputs and outputs for the study wetland.

Period	Inputs (g P)			Outputs (g P)		
	Atmosphere	Runoff	Total	Outflow	Seepage*	Total
Nov - April	25	-	25	-	-	-
April**	-	31	31	36	17	53
May	228	192	420	41	19	60
June	133	101	234	18	8	26
July	80	30	110	19	9	28
August	98	13	111	14	7	21
September	34	14	48	17	8	25
October	7	8	15	13	6	19
Total	605	389	**994**	158	74	**232**

* Yearly total calculated from Verry and Timmons (1982)
** Atmospheric contributions are considered a part of the winter contributions

Discussion

This 3 ha study wetland on the Canadian Boreal Plain was a water and P sink during the November 2004 to October 2005 water year. It retained a maximum of 27% water and a maximum of 77% of the TP from atmospheric and terrestrial sources.

Inputs

Atmospheric Deposition and Throughfall

For the November 2004 to October 2005 water year, the measurement of precipitation for the wetland was 403 mm, 73% of the 25 year average, confirming that the study occurred within a dry period. For the months of July and August respectively, atmospheric P deposition rates were 23% lower and 6% higher than those reported by Shaw et al. (1989) for a forest 170 km east of the study wetland. The annual atmospheric P deposition rate of 19.6 $mg/m^2/y$ is less than values reported for the boreal forest on the Boreal Shield of northwestern Ontario (24 to 53 $mg/m^2/y$) (Schindler et al. 1976) and only 53% of rates estimated for the Great Lakes – St. Lawrence forest (Gomolka 1975). Similarly, atmospheric P loading rates in south Florida were approximately 31.3 $mg/m^2/yr$, of which 13% was estimated to originate from anthropogenic sources (Ahn and James 1999). The western Boreal Plain may be less impacted by anthropogenic atmospheric input of P than more eastern and southern forests (Shaw et al. 1989).

The proportion of PP in TP in both the bulk and dry-only atmospheric samples varied greatly, and only microscopic analysis of the samples would indicate the source of the PP. High variability in P concentration in rain water is common, even between replicate collectors (Welch and Legault 1986, Shaw et al. 1989, Pollman et al. 2002). The weathering of rock, smoke release from forest fires (which can travel several thousand kilometres) and release of plant biomass, such as pollen (Newman 1995), can all potentially contribute to particulates to the atmosphere, and they all vary spatially and with season. In central Ontario, pollen contributions to particulate P in dry-only fallout was high in the spring, and was replaced by silica and humic plus organic matter later in the growing season (Gomolka 1975).

In both Alberta and north-western Ontario, the general pattern of atmospheric P deposition in rainfall is for maxima to occur in the spring, followed by a decrease from May to October, and minima to occur in the winter months (Linsey et al.1987, Shaw et al. 1989). Without the single storm event on 06 August 2005, which increased the mean August deposition rate by 70%, the pattern observed in this study during the 2 month data collection period followed the July to August decrease. In Florida, summer P deposition rates were also found to be higher than in winter months (Pollman et al. 2002). It has been suggested that the decrease in P deposition throughout the summer could be due to changes in meteorological processes (less rainfall) or in biological processes that alter the release of mineral dust (Shaw et al. 1989). In a two-year study in different locations in New Jersey, however, there were no seasonal P deposition trends (Koelliker et al. 2004). It is possible that anthropogenic deposition confounded detection of seasonal patterns in this urban and industrial area.

Because bulk and dry-only sample collections were not collected simultaneously (i.e. dry deposition collected only during dry periods), it was not possible to determine the contribution made solely by dry or wet deposition. Due to the collection design, any dry fallout occurring earlier in the period would have been washed down into the collector with any rainfall, incorporating the value of the dry fallout into the bulk total. Although the amounts deposited per collection period were similar for both TP and TDP, only through using wet-only and dry-only collectors would the true proportion of wet and dry deposition be known. In central Alberta, dry deposition contributed approximately 50% of the yearly total atmospheric P deposition (Shaw et al. 1989).

Aspen throughfall volume exceeded spruce throughfall volume, but the P concentration in aspen throughfall was found to be similar to rainfall. The net effect of these differences was that the mass of P reaching the ground did not differ for these two canopy types, nor did these values differ from bulk atmospheric deposition. The spruce canopy retained water, increased the P concentration in throughfall and did not affect the mass of P reaching the forest floor, relative to bulk deposition. The aspen canopy did not appear to affect the volume nor P concentration in throughfall, therefore also did not affect the mass of P reaching the forest floor, relative to bulk deposition. These findings contradict studies in central and north-western Ontario that found that black spruce reduced the P loading to the forest floor to 75 and 35% of direct atmospheric deposition, respectively (Gordon et al. 2000, Morris et al. 2002). Conversely, in north-central Minnesota, black spruce increased the amount of P reaching the forest floor by 1.3 times compared to open areas (Verry and Timmons 1977). Aspen throughfall exceeded bulk deposition by 3 times in north-central Minnesota and by 1.4 times in Saskatchewan (Verry and Timmons 1977, Huang and Schoenau 1997). It is possible that the atmospheric collectors in this study were not tall enough to sample only rain water. Although the collectors were taller than the surrounding wetland trees, contamination from taller upland trees could have occurred. Also, the variation in P concentration among all collector types was high, which combined with a relatively small sample size may have prevented detection of treatment effects. A high degree of variation among sites was noted by Shaw et al. (1989) and was attributed to greater amounts of particulate fallout in sites on land versus sites on the water.

Runoff

Runoff from the micro-watershed surrounding the wetland contributed almost 40% of the P to the wetland, of which 73% was PP. The P export values were twice the greatest values found in central Ontario (9.5 to 29.5 g/ha/y, 1980-1992 mean) (Dillon and Molot 1997). Due to the P-rich nature of the soils of the micro-watershed (Cooke and Prepas 1998), the runoff P contribution results could have been elevated. Water pulses during storm events could have mobilized large amounts of particulates from the forest floor or stream channel, although the slope of the micro-watershed is gentle (Munn and Prepas 1986, Prepas et al. 2003). Low infiltration rates in the Gray Luvisolic soils that dominate the study area (Whitson et al. 2003) could have enhanced overland flow and erosion.

Groundwater

For the most part, the proportion of TDP in TP increased in groundwater as the sampling period progressed. Because the wetland received less rain in July and August than it did earlier in the season, the wetland water level decreased. In a boreal bog in Alberta, water levels were correlated negatively with TDP concentration, which was attributed to concentration of TDP when groundwater inputs decreased during dry periods (Thormann et al. 1998).

During the sampling period, the relative proportion of TDP increased in the peatland and micro-watershed groundwater, but decreased in peatland margin groundwater. This may be attributable to low water levels in the peatland margin areas of the wetland that made it difficult to exclude particulates from samples. Another possible cause could be that towards the end of the sampling season, on the two occasions when the TDP values were lower, rain had fallen within two days prior to sampling. One such event is the 06 August 2005 hail storm, during which increased P concentrations were noted. Since the peatland margin wells were shallower than the peatland and upland wells, it is possible that the new precipitation contribution could have had a stronger effect on the peatland margin TDP concentration.

Output

Outflow

The TP concentration in the outflow from the peatland complex decreased with increasing discharge rate, which is attributable to dilution and also largely attributable to decreases in the TDP fraction. As the summer progresses, normal daily temperatures decrease from approximately 15.5 to 14.5 °C between July and August (Environment Canada 2006) causing a decrease in decomposition rates in wetlands (Thormann et al. 2001), thus slowing the release of P bound to organic material. Furthermore, the limited oxygen diffusion into the peat causes anoxic conditions, minimizing decomposition as a result of reducing conditions (Clymo 1992). Dissolved P that was flushed from the wetland with rainfall is not replaced by decomposition within the wetland under these relatively cool, reducing and acidic conditions.

A second factor that could account for the negative association between TDP concentrations in wetland outflow and discharge rate is the relative increase in PP concentration with increasing discharge rate. In streams draining Boreal Plain watersheds, P concentration increased with discharge rate, a phenomenon that was attributed to PP loading caused by erosion of P-rich watershed soils and channel banks (Prepas et al. 2003). This pattern has also been noted in streams in the Hubbard Brook where particulate concentrations increased exponentially as flow increased (Hobbie and Likens 1973). When water enters the peatland, the slowing in velocity allows P-rich particulates to settle out. During high flow events, some of this PP may be mobilized and carried to the wetland outflow.

Seepage

Seepage P losses from the wetland were estimated at 74 g, approximately 7% of the amount deposited on the wetland, indicating that water loss to seepage was only half of the results found in north-central Minnesota (Verry and Timmons 1982). Given that nearly every wetland differs, based on physical, chemical and biological attributes, this seepage estimate is simply a guideline. In a forested watershed in central New England, water and P loss to seepage was deemed negligible, while in a wetland nutrient budget in central Ontario, loss to seepage was simply addressed as a balance to water and P inputs from seepage (Devito et al. 1989, Hornbeck et al. 1997). That water pathways through the wetland include water exchanges with upland systems (Branfireun and Roulet 1998) is challenged by statements that wetland recharge by groundwater or to groundwater is limited (Brooks et al. 2003). Although seepage was not directly measured in this study, and although loss of P to seepage is estimated as being lower than other investigations, seepage to groundwater is most likely occurring and worthy of further investigation.

Evapotranspiration

The wetland surface released 41% of the water it received through rainfall and runoff, to ET. Conversely, it can be expressed that total annual precipitation exceeded ET by 1.58 times. In a bog in southeastern Ontario, it was found that mean total annual precipitation exceeded ET from the wetland surface by 1.55 to 1.94 times (Lafleur et al. 2005). Although the amount of water lost to ET was estimated for this study, the amount is within the range found by others.

Water Budget

The results suggest that the wetland retained 27% of the water it received through precipitation and runoff during the November 2004 to October 2005 water year. Since seepage and ET were not measured in this study and because of the difference in annual average precipitation and annual average temperature between the two study areas, the proportions from the investigation in Minnesota may not translate exactly to this study. Also, because of the *in situ* investigation of seepage, the P and water budget results obtained by Verry and Timmons (1982) may be more representative of typical peatland water retention.

Phosphorus Budget

The relationships between precipitation chemistry and outflow indicates that transformation of P occurred while the rainwater resided in the wetland. At times TDP concentrations in precipitation accounted for as little as 20% of TP, and as much as 60% of the TP in outflow, supporting the concept of piston flow. Piston flow, or translatory flow, occurs when the resident water in a wetland is replaced by precipitation and/or runoff, and is therefore forced out via the outflow (Hewlett and Hibbert 1967). In subarctic peatlands, rainwater chemistry was only similar to outflow water chemistry when the water table was at the surface (McEachern et al. 2006). When the water table is not at or near the surface, rainwater comes in contact with soils and organics for a longer period of time, altering the water chemistry. Peatlands have also been shown to demonstrate quick response times for spring runoff when the water table is above the storage capacity elevation of the basin, while during drier seasons they can become partially disconnected from nearby hydrological systems (Glenn and Woo 1997, Quinton and Roulet 1998). Furthermore, the path of water through in wetlands is complex, and does not necessarily follow a horizontal path, but can move vertically or as sheetflow and channel flow on the surface (Taylor 1997). The response to a single rain event can be extremely varied (McKillop et al. 1999) among wetlands and within a given wetland.

The study wetland in the Willow watershed retained 77% of the P in atmospheric deposition and runoff, 350% greater than retention in a Boreal Shield rich fen (Devito et al. 1989), and 26% greater than the three year mean for a peatland in north-central Minnesota (Verry and Timmons 1982). The retention values obtained by Devito et al. (1989), however, were lower than expected. Alternately, another Boreal Shield micro-watershed (12.4 ha, wetland comprising 30% of the basin) retained 17% more P than the study wetland (Bayley et al. 1992).

If 762 g of P was retained by the wetland, how and where was it retained? This question cannot be immediately answered because of the limited data presented here. However, several processes occur within wetlands that are known to reduce the amount of P in groundwater. Firstly, aquatic plants and microbial activity remove P from wetland water (Richardson 1985). This removal of P by wetland plants is significant enough that emergent and submergent plants are being used in constructed wetlands to remove P and other nutrients from highly P enriched water (White et al. 2005). Furthermore, Chapin et al. (1987) found that wetland mosses had a greater ability to absorb P than the fine roots of black spruce and that the overstory is actually competing with the underlying mosses for nutrients, like P. Trees in treed and forested wetlands, however, provide longer term storage for P than other vegetation (Reddy and DeBrusk 1987). Understandably, any P retained by plant matter or microbial biomass is released upon the death of that plant via decomposition, and made available to other processes, most likely adsorption to soil particles (Reddy et al. 1999). Plants, therefore, provide only temporary P storage (Richardson 1985). Secondly, P adsorbs to elements such as aluminum, causing a precipitate to develop, and for this reason the addition of alum to lakes has been used to internally reduce the P concentration in lake water columns (Hansen et al. 2003). The success of removing P from watercourses by providing it with an adsorptive surface has furthered the development of a lightweight aggregate which causes the precipitation of calcium, aluminum and iron phosphates, depending on the pH (Jenssen and Krogstad 2003). Lastly, P adsorbs to sediments and organic matter, which settle out and become compressed peat (catotelm). Because of the compression, any water flowing through the wetland will flow above these layers (within the acrotelm), meaning that any nutrients stored within the catotelm become essentially unavailable (Brooks et al. 2003). Wetland soils provide a better and longer term storage location compared to P storage in plant biomass (Richardson and Marshall 1986, Walbridge and Struthers 1993).

There has been much discussion regarding how wetlands can be used reduce the amount of nutrients in surface waters. It would seem that naturally, wetlands provide such a service, but only to a point. Wetlands have a finite ability to retain nutrients such as P. At a point when a wetland becomes saturated, it then becomes a source of nutrients to connected water bodies (Richardson 1985, Brooks et al. 2003). In Florida, top sediments and plants were removed from treatments wetlands to ensure efficient P removal (Wang et al. 2005). Without this intervention the carrying capacity appeared to be reached, meaning that the wetland was physically, chemically and biologically unable to store any more P due to the increased P inputs. For this reason, the relationship between precipitation and wetland outflow should be investigated under natural conditions over lengthy periods of time. While a wetland may retain P now, through physical, chemical and biological means, it may possibly be approaching its carrying capacity and may release more P than it receives in the near future. This transition could go unnoticed and a large flush of P and other nutrients could enter watercourses, causing eutrophication. This study in particular, was conducted during a dry year within a drought, relative to the long term mean. Although year to year variation is expected, this prolonged dry period most likely influenced the P and water retention of peatlands, and therefore lead to non-indicative results.

It has been documented that the P export in the FORWARD project watersheds is positively related to peatland cover (Prepas et al. 2006); however, my results differ from these findings. In this study, 77% of the P input into the wetland was retained; the wetland retained more P relative to the surrounding landscape. A possible explanation for the differing results could be that the study wetland was a bog, while it has been reported that fens dominate in the FORWARD project area (Prepas et al. 2006). Bogs in north-central Minnesota are nutrient sinks (Verry and Timmons 1982). In the Netherlands, bogs were found to mineralize P much more quickly than fens, indicating that there is less P in the water column to export. Furthermore, because of the reduced decomposition rates in bogs, any P sequestered through plant uptake will be retained for a longer period of time (Verhoeven et al. 1990).

Future Work

Because this work was a pilot study, the sample design was simple. Certainly compounded errors could have occurred due to the reliance on data from other sources, not related to this study. Although values and relationships were used from other sources (e.g., Verry and Timmons 1982 and Shaw et al. 1989), the results remain unexceptional when compared to other wetland P and water budgets. It is important that long term wetland water and nutrient studies be conducted in order to understand wetland water and P cycling. Prolonged wet or dry periods will undoubtedly skew results, and only by observing long term trends will the true relationships be understood. In this study, several processes were not examined. Firstly, stemflow was not investigated. Gordon et al. (2000) note that although the volume of stemflow was only 5% of the volume for throughfall, it was still an important part of the nutrient cycle. Secondly, ET is a very important route of water loss, and the process should be considered in wetland water budgets. In Minnesota, 65% of all water inputs were lost to ET, a value which is very close to potential ET estimates (Verry and Timmons 1982). Thirdly, sublimation of snow was not investigated during winter months. In central Saskatchewan, loss of water due to the sublimation of snow captured by spruce canopies was up to 40% of the annual accumulation (Pomeroy et al. 1998), indicating that the input volumes estimated for this study may be much greater than in actuality. Lastly, although seepage values from other works have been cited as replacement for actual experimentation, field investigation is superior and this likely source of water loss from the wetland should be monitored. A lack of investigation of these water loss and water cycling processes are causing inaccuracies in the budgets.

A change in atmospheric deposition collection methods could also be investigated seeing as high variation between collectors has been observed in this project as well as in other investigations (Welch and Legault 1986, Shaw et al. 1989). It has also been proposed that local P sources contribute more to atmospheric P loading than actual atmospheric sources (Tsukuda et al. 2004) and that these sources should be considered as sources of contamination (Ahn and James 1999).

P speciation was also not investigated during this study. Future work could include studies to estimate P fractionations (e.g., soluble, insoluble, organic, inorganic) in the wetland. These investigations would be beneficial as the type and origins of the P would lead to a greater understanding of wetland P cycling.

CHAPTER IV: GENERAL CONCLUSIONS

Identifying wetlands on the landscape is a crucial step in determining their relationship to nearby waterbodies and local hydrology. By creating wetland identification systems with improved accuracy, such as the Wetland Inventory and Identification Tool (WIIT), wetlands can be linked with small scale landscape modeling tools. By pairing the WIIT with knowledge regarding wetland impacts on streamwater quantity and quality, planning for landscape changes has improved possibilities.

A wetland water and phosphorus budget was constructed to improve understanding of wetland functioning on the nutrient-rich and water-short Boreal Plain and to contribute to hydrologic modeling. Although my wetland budget results are preliminary, a 3 ha wetland in the Canadian Boreal Plain retained 762 g of P inputs for the time period of one year. This wetland retention of P is important in gaining knowledge on eutrophication management. The results from my wetland budget will be used directly, and as input information, to further studies linking wetlands, and peatlands in particular, to stream water quality on the Canadian Boreal Plain. Follow-up from my study includes construction of a nitrogen budget for the 16 km^2 Willow watershed (including the study wetland).

As suggested by Environment Canada (2004), most Canadians associate the word "wetland" with urbanized wetlands that function as stormwater retention pools. Given that global warming will impact all surface waters, including peatlands, which comprise 85% of all wetlands in Canada, it is time to focus more attention and resources on these vast boreal peatlands.

Literature Cited

Agriculture and Agri-Food Canada. 1998. The Canadian system of soil classification 3rd Ed. NRC Press, Ottawa, ON. 187 p.

Ahn H. and R. T. James. 1999. Outlier detection in phosphorus dry deposition rates measured in South Florida. Atmos. Environ. 33: 5123–5131.

Anderson, R. B. 1999. Peatland habitat use and selection by woodland caribou (*Rangifer tarandus caribou*) in northern Alberta. M.Sc. Thesis. University of Alberta. 49 p.

Arnold, J. G., R. Srinivasan, R. S. Muttiah and J. R. Williams. 1998. Large area hydrologic modeling and assessment. Part I: Model development. J. Amer. Water Resour. Assoc. 34:73-89.

Bayley, S. E., D. W. Schindler, K. G. Beaty, B. R. Parker and M. P. Stainton. 1992. Effects of multiple fires on nutrient yields from streams draining boreal forest and fen watersheds: nitrogen and phosphorus. Can. J. Fish. Aquat. Sci. 49:584-596.

Branfireun, B. A. and Roulet, N. T. 1998. The baseflow and storm flow hydrology of a Precambrian shield headwater peatland. Hydrol. Process. 12:57-72

Bridgham, S. D., K. Updegraff and J. Pastor. 1998. Carbon, nitrogen and phosphorus mineralization in northern wetlands. Ecology 79:1545-1561.

Brooks, K. H., P. F. Ffolliott, H. M. Gregersen and L. F. DeBano. 2003 *Hydrology and the Management of Watersheds* (3rd Ed). Iowa State University Press. 574 pp.

Canadian Council on Ecological Areas. 2006. Ecozones of Canada. Available from http://www.ccea.org/ecozones/ [accessed on 24 January 2006].

Chambers, P. A., J. M. Culp, N. E. Glozier, K. J. Cash, F. J. Wrona and L. Noton. 2006. Northern Rivers Ecosystem Initiative: nutrients and dissolved oxygen – issues and impacts. Environ. Monit. Assess. 113:117-141

Chapin, III, F. S., W. C. Oechel, K. Van Cleve and W. Lawrence. 1987. The role of mosses in the phosphorus cycling of an Alaskan black spruce forest. Oecol. 74:310-315.

Clymo, R. S. 1987. Interactions of *Sphagnum* with water and air. In: *Effects of Acidic Deposition and Air Pollutants on Forests, Wetlands and Agricultural Ecosystems* (T. C. Hutchinson and K. M. Meema, eds) Berlin: Springer-Verlag. pp. 513-529.

Clymo, R. S. 1992. Models of peat growth. Suo 43:127–36.

Cooke, S. E. and E. E. Prepas. 1998. Stream phosphorus and nitrogen export from agricultural and forested watersheds on the Boreal Plain. Can. J. Fish. Aquat. Sci. 55: 2292-2299.

Conference on Canadian Wetlands Stewardship. 2003. Report No. 03-2. North American Wetlands Conservation Council (Canada). 145 p.

Dalva, M. and T. R. Moore. 1991. Sources and sinks of dissolved organic carbon in a forested swamp catchment. Biogeochemistry 15:1-19.

Devito, K. J., P. J. Dillon and B. D. Lazerte. 1989. Phosphorus and nitrogen retention in five Precambrian shield wetlands. Biogeochemistry 8:185-204.

Devito, K. J., A. R. Hill and N. Roulet. 1996. Groundwater-surface water interactions in headwater forested wetlands of the Canadian Shield. J. Hydrol. 181:127-147.

Dillon, P. J. and L. A. Molot. 1997. Effect of landscape form on export of dissolved organic carbon, iron, and phosphorus from forested stream catchments. Water Resour. Res. 33:2591-2600.

Ecological Stratification Working Group. 1996. A National ecological framework for Canada. Centre for Land and Biological Resources Research, Research Branch, Agriculture and Agri-food Canada, Ottawa, ON. 125 p.

Edmondson, W. T., G. C. Anderson and D. R. Peterson. 1956. Artificial eutrophication of Lake Washington. Limnol.Oceanogr. 1:47-53.

Emsley, J. and D. Hall. 1976. *The chemistry of phosphorus*. Halsted Press: New York. 562 pp.

Environment Canada. 2004. Threats to water availability in Canada. National Water Research Institute, Burlington, ON. NWRI Scientific Assessment Report Series No. 3 and ACSD Science Assessment Series No. 1. 128 p.

Environment Canada. 2006. Digital archive of the Canadian climatological data (surface). Atmospheric Environment Service. Canadian Climate Centre, Data Management Division, Downsview, ON.

Franklin, S. E., M. B. Lavigne, M. A. Wulder and G. B. Stenhouse. 2002. Change detection and landscape structure mapping using remote sensing. For. Chron. 78:618-625.

Gibson, J. J., E. E. Prepas and P. McEachern. 2002. Quantitative comparison of lake throughflow, residency, and catchment runoff using stable isotopes: modeling and results from a regional survey of boreal lakes. J. Hydrol. 262:128-144.

Glenn, M. S. and M. K. Woo. 1997. Spring and summer hydrology of a valley-bottom wetland, Ellesmere Island, Northwest Territories, Canada. Wetlands 17: 321-329.

Gomolka, R. E. 1975. An investigation of atmospheric phosphorus as a source of lake nutrient. M.Sc. Thesis. University of Toronto. 166 p.

Gordon, A. M, C. Chourmouzis and A. G. Gordon. 2000. Nutrient inputs in litterfall and rainwater fluxes in 27-year old red, black and white spruce plantations in Central Ontario, Canada. Forest Ecol. Manag. 138:65-78.

Gorham, E. 1991. Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. Ecol. Appl. 1:182-195.

Grant, D. M. and B. D. Dawson. 1997. *Isco open channel flow measurement handbook.* Isco Inc, Lincoln, NE. 501 p.

Halsey, L. A. and D. H. Vitt. 1997. Alberta wetland inventory standards. *In* Alberta Vegetation Inventory Standards. Version 2.2. Alberta Environmental Protection, Resource Data Division, Edmonton, AB.

Hansen, J., K. Reitzel, H. Jensen and F. Andersen. 2003. Effects of aluminum, iron, oxygen and nitrate additions on phosphorus release from the sediment of a Danish softwater lake. Limnol. Oceanogr. 492:139-149.

Hewlett, J. D. and A. R. Hibbert. 1967. Factors affecting the response of small watersheds to precipitation in humid areas. In: W. E. Sopper and H. W. Lull (eds) Proceedings of the International Symposium on Forest Hydrology, Pergamon Press, Oxford, pp 275-290.

Hobbie, J. E. and G. E. Likens. 1973. Output of phosphorus, dissolved organic carbon, and fine particulate carbon from Hubbard Brook watersheds. Limnol. Ocreanogr.18:734-742.

Hopkinson, C., L. E. Chasmer, G. Sass, I. F. Creed, M. Sitar, W. Kalbfleisch and P. Treitz. 2005. Vegetation class dependent errors in lidar ground elevation and canopy height estimates in a boreal wetland environment. Can. J. Remote Sens. 31:191-206.

Hornbeck, J. W., S. W. Bailey, D. C. Buso and J. B. Shanley. 1997. Streamwater chemistry and nutrient budgets for forested watersheds in New England: variability and management implications. Forest Ecol. Manag. 93:73-89.

Huang, W. Z and J. J. Schoenau. 1997. Seasonal and spatial variations in soil nitrogen and phosphorus supply rates in a boreal aspen forest. Can. J. of Soil Sci. 77:597-612.

Jenssen, P. D. and T. Krogstad. 2003. Design of constructed wetlands using phosphorus sorbing lightweight aggregate (LWA). Adv. Ecol. Sci. 11:259-272.

Johnson, D., L. Kershaw, A. MacKinnon and J. Polar. 1995. Plants of the western boreal forest and aspen parkland. Lone Pine Publishing, Edmonton, AB. 392 p.

Kellogg, L. E. and S. D. Bridgham. 2003. Phosphorus retention and movement across an ombotrophic-minerotrophic peatland gradient. Biogeochemistry 63:299-315.

Koelliker, Y., L. A. Totten, C. L. Gigliotti, J. H. Offenberg, J. R. Reinfelder, Y. Zhuang and S. J. Eisenreich. 2004. Atmospheric wet deposition of total phosphorus in New Jersey. Water Air Soil Pollut. 154:139-150.

Lafleur, P.M., R. A. Hember, S. W. Admiral and N. T. Roulet. 2005. Annual and seasonal variability in evapotranspiration and water table at a shrub-covered bog in southern Ontario, Canada. Hydrol. Process. 19:3533-3550.

Li, J. and W. Chen. 2005. A rule-based method for mapping Canada's wetlands using optical, radar and DEM data. Int. J. Remote Sens. 26:5051-5069.

Likens, G. E., F. H. Bormann, K. S. Pierce, J. S. Eaton and N. M. Johnson. 1977. *Biogeochemistry of a Forested Ecosystem*. Springer-Verlag, New York, 146 pp.

Linsey, G. A., D. W. Schindler and M. P. Stainton. 1987. Atmospheric deposition of nutrients and major ions at the Experimental Lakes Area in northwestern Ontario, 1970 to 1982. Can. J. Fish. Aquat. Sci. 44 (Suppl. 1): 206-214.

Locky, D. A., S. E. Bayley and D. H. Vitt. 2005. The vegetational ecology of black spruce swamps, fens, and bogs in southern boreal Manitoba, Canada. Wetlands 25:564-582.

Marin, L. E., T. K. Kratz and C. J. Bowser. 1990. Spatial and temporal patterns in the hydrogeochemistry of a poor fen in northern Wisconsin. Biogeochemistry 11:63-76.

McEachern, P., E. E. Prepas, and D. S. Chanasyk. 2006. Landscape control of water chemistry in northern boreal streams of Alberta. J. Hydrol. 323: 303-324.

McKeown, R. A., G. Putz and J. Arnold. 2005. Modifications of the Soil and Water Assessment Tool (SWAT-C) for streamflow modelling in a small, forested watershed on the Canadian Boreal Plain. Proc. 3rd Internat. SWAT 2000 Conf., Zurich, Switzerland. K. Abbaspour and R. Srinivasan (eds). Texas Water Resources Institute, College Station, TX.

McKillop, R., N. Kouwen and E. D. Soulis. 1999. Modelling the flood modification effects of a mid-sized unregulated wetland. In *New applications in modeling urban water systems*, Monograph 7, James W (ed.). Computational Hydraulics International (CHI): Guelph, ON.

McNabb, D. H., A. D. Startsev and H. Nguyen. 2001. Soil wetness and traffic level effects on bulk density and air-filled porosity of compacted Boreal Forest soils. Soil Sci. Am. J. 65:1238-1247.

Menzel, D. W. and N. Corwin. 1965. The measurement of total phosphorus in seawater based on the liberation of organically bound fractions by persulfate oxidation. Limnol. Oceanogr. 10:280-282

Mitchell, P. and Prepas, E. 1990. Atlas of Alberta Lakes. University of Alberta Press, Edmonton, AB.

Mitsch, W. J. and J. G. Gosselink. 2000. *Wetlands. 3rd Ed.* John Wiley & Sons Inc., New York. 920 p.

Morris, D. M., A. G. Gordon and A. M Gordon. 2002. Patterns of canopy interception and throughfall along a topographic sequence for black spruce dominated forest ecosystems in northwestern Ontario. Can. J. For. Res. 33:1046-1060.

Moustafa, M. Z. 1999. Nutrient retention dynamics of the Everglades Nutrient Removal Project. Wetlands 19:689-704.

Munn, N. and E. Prepas. 1986. Seasonal dynamics of phosphorus partitioning and export in two streams in Alberta, Canada. Can. J. Fish. Aquat. Sci. 43:2464-2471.

National Wetlands Working Group. 1986. Wetland regions of Canada. In Canada's wetlands. Map folio. Energy, Mines and Resources Canada and Environment Canada. Ottawa, ON.

National Wetlands Working Group. 1988. Wetlands of Canada. Ecological Land Classification Series, No. 24. Sustainable Development Branch, Environment Canada, Ottawa, ON, and Polyscience Publications Inc., Montreal, QC. 452 p.

National Wetlands Working Group. 1997. The Canadian Wetland Classification System B .G. Warner and C. D. A. Rubec (eds). National Wetland Working Group, Wetlands Research Centre, University of Waterloo. 68 p.

Nesby, R. 1997. Alberta vegetation inventory standards manual, version 2.2 ed. Alberta Environmental Protection, Edmonton, AB.

Newman, E. I. 1995. Phosphorus inputs to terrestrial ecosystems. J. Ecol. 83:713-726.

Paterson, A. M., P. J. Dillon, N. J. Hutchinson, M. N. Futter, B. J. Clark, R. B. Mills, R. A. Reid and W. A. Scheider. 2006. A review of the components, coefficients and technical assumptions of Ontario's Lakeshore Capacity Model. Lake Reserv. Manage. 22:7-18.

Pollman, C. D., W. M. Landing, J. J. Perry Jr. and T. Fitzpatrick. 2002. Wet deposition of phosphorus in Florida. Atmos. Environ. 36:2309-2318.

Pomeroy, J. W., J. Parviainen, N. Hedstrom and D. M. Gray. 1998. Coupled modeling of forest snow interception and sublimation. Hydrol. Process. 12:2317-2337.

Prepas, E. E. and F. H. Rigler. 1982. Improvements in quantifying the phosphorus concentration in lake water. Can. J. Fish. Aquat. Sci. 39:822-829.

Prepas, E. E. and D. O. Trew. 1983. Evaluation of the phosphorus-chlorophyll relationship for lakes off the Precambrian Shield in western Canada. Can. J. Fish Aquat. Sci. 40:27-35.

Prepas, E. E., B. Pinel-Alloul, D. Planas, G. Méthot, S. Paquet and S. Reedyk. 2001a. Forest harvest impacts on water quality and aquatic biota on the Boreal Plain: introduction to the TROLS lake program. Can. J. Fish. Aquat. Sci. 58:421-436.

Prepas, E. E., D. Planas, J. J. Gibson, D. H. Vitt, T. D. Prowse, W. P. Dinsmore, L. A. Halsey, P. M. McEachern, S. Paquet, G. J. Scrimgeous, W. M. Tonn, C. A. Paszkowski and K. Wolfstein. 2001b. Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: a comparison of wetland- and upland-dominated catchments. Can. J. Fish. Aquat. Sci. 58:1286-1299.

Prepas, E. E., J. M. Burke, D. S. Chanasyk, D. W. Smith, G. Putz, S. Gabos, W. Chen, D. Millions and M. Serediak. 2003. Impact of wildfire on discharge and phosphorus export from the Sakwatamau watershed in the Swan Hills, Alberta, during the first two years. J. Environ. Eng. Sci. 2:S63-S72.

Prepas, E. E., J. M. Burke, I. R. Whitson, G. Putz and D. W. Smith. 2006. Associations between watershed characteristics, runoff, and stream water quality: Hypothesis development for watershed disturbance experiments and modelling in the Forest Watershed and Riparian Disturbance (FORWARD) project. J. Environ. Eng. Sci. 5 (Suppl. 1):27-37.

Price, J. S., B. A. Branfireun, J. M. Waddington and K. J. Devito. 2005. Advances in Canadian wetland hydrology, 1999–2003. Hydrol. Process. 19:201-214.

Putz, G., J. M. Burke, D. W. Smith, D. S. Chanasyk, E. E. Prepas and E. Mapfumo. 2003. Modelling the effects of boreal forest landscape management upon streamflow and water quality: basic concepts and considerations. J. Environ. Eng. Sci. Suppl. 2:87-101.

Quinton W. L. and N. T. Roulet. 1998. Spring and summer hydrology of a subarctic patterned wetland. Arctic Alpine Res. 30: 285-294.

Reddy, K. R. and W. F. DeBrusk. 1987. Nutrient storage capabilities of aquatic and wetland plants. In: *Aquatic Plants for Water Treatment and Resource Recovery*, K. R. Reddy and W. H. Smith, Eds. Magnolia Publ., pp. 337–357.

Reddy, K. R., R. H. Kadlec, E. Flaig and P. M. Gale. 1999. Phosphorus retention in streams and wetlands: a review. Crit. Rev. Environ. Sci. Technol. 29:83-146.

Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228:1424-1426.

Richardson, C. J. and P. E. Marshall. 1986. Processes controlling movement, storage, and export of phosphorus in a fen peatland. Ecol. Monograph. 56:279-302.

Sader, S. A., D. Ahl and W. Liou. 1995. Accuracy of Landsat-TM and GIS rule-based methods for forest wetland classification in Maine. Remote Sens. Environ. 53:133-144.

Schiff, S., R. Aravena, E. Mewhinney, R. Elgood, B. Warner, P. Dillon and S. Trumbore. 1998. Precambrian shield wetlands: hydrologic control of the sources and export of dissolved organic matter. Clim. Chang. 40:167-188.

Schindler, D. W. 1974. Eutrophication and recovery in experimental lakes: implications for lake management. Science 184:897-899.

Schindler, D. W., R. W. Newbury, K. G. Beaty and P. Campbell. 1976. Natural water and chemical budgets for a small Precambrian lake basin in central Canada. J. Fish. Res. Board. Can. 33:2526-2543.

Shaw, R. D., A. M. Trimbee, A. Minty, H. Fricker and E. E. Prepas. 1989. Atmospheric deposition of phosphorus and nitrogen in central Alberta with emphasis on Narrow Lake. Water Air Soil Poll. 43:119-134.

Sjörs, H. and U. Gunnarsson. 2002. Calcium and pH in north and central Swedish mire waters. J. Ecol. 90:650-657.

Smith, D. W., E. E. Prepas, G. Putz, J. M. Burke, W. L. Meyer and I. R. Whitson. 2003. The Forest Watershed and Riparian Disturbance study: a multi-discipline initiative to evaluate and manage watershed disturbance on the Boreal Plain of Canada. J. Environ. Eng. Sci. Suppl. 2:1-13.

Sophocleous, M. 2002. Interactions between groundwater and surface water: the state of the science. Hydrogeol. J. 10:52–67.

Stainton, M.P., M.J. Capel and F.A.J. Armstrong. 1977. The chemical analysis of freshwater. 2nd ed. Can. Fish. Mar. Serv. Misc. Spec. Publ. No. 25. 180 p.

Tarnocai, C. 2006. The effect of climate change on peatlands in the Canadian boreal and subarctic. Proceedings from the International Conference on Hydrology and Management of Forested Wetlands (April 8-12, 2006): New Bern, NC. 597 p.

Taylor, C. H. 1997. Runoff processes in temperate headwater wetlands. In *Ecology of Wetlands and Associated Systems,* Majumdar SK, Miller EW, Brenner FJ (eds). The Pennsylvania Academy of Science: Pittsburgh, PA. 168-181 p.

Thormann, M. N., S. E. Bayley and A. R. Szumigalski. 1998. Effects of hydrologic changes on aboveground production and surface water chemistry in two boreal peatlands in Alberta: Implications for global warming. Hydrobiologia 362:171-183.

Thormann, M. N., S. E. Bayley and R. S. Currah. 2001. Comparison of decomposition of belowground and aboveground plant litters in peatlands of boreal Alberta, Canada. Can. J. Bot. 79:9-22.

Tsukuda, S., M. Sugiyama, Y. Harita and K. Nishimura. 2004. A methodological re-examination of atmospheric phosphorus input estimates based on spatial microheterogeneity. Water Air Soil Poll. 152: 333-347.

Verhoeven, J. T. A., E. Maltby and M. B. Schmitz. 1990. Nitrogen and phosphorus mineralization in fens and bogs. J. Ecol. 78:713-726

Verry, E. S. and D. R. Timmons. 1977. Precipitation nutrients in the open and under two forests in Minnesota. Can. J. For. Res. 7:112-119.

Verry, E. S. and D. R. Timmons. 1982. Waterborne nutrient flow through an upland-peatland watershed in Minnesota. Ecology 63:1456-1467.

Vitt, D. H. and W.-L. Chee. 1990. The relationships of vegetation to surface water chemistry and peat chemistry in fens of Alberta, Canada. Plant Ecol. 89:87-106.

Vitt, D. H., L. A. Halsey, I. E. Bauer and C. Campbell. 2000. Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. Can. J. Earth Sci. 37:683-693.

Walbridge, M. R. and J. P. Struthers. 1993. Phosphorus retention in non-tidal palustrine forested wetlands of the Mid-Atlantic Region. Wetlands 13:84-94.

Wang, H., J. W. Jawitz, J. R. White and M. D. Sees. 2005. Rejuvenating the largest municipal treatment wetland in Florida. Conference Proceedings from the 9th International Symposium on Biogeochemistry of Wetlands (March 20 – 23, 2005): Baton Rouge, LA. 122 p.

Welch, H. E. and J. A. Legault. 1986. Precipitation chemistry and chemical limnology of fertilized and natural lakes at Saqvaqjuac, N.W.T. Can. J. Fish. Aquat. Sci 43:1104-1134.

Wetzel, R. G. 2001. Limnology, 3rd Ed. Academic Press: San Diego, CA. 1006 p.

White, J. R., K. R. Reddy and M. Z. Moustafa. 2004. Influence of hydrologic regime and vegetation on phosphorus retention in Everglades stormwater treatment area wetlands. Hydrol. Process. 18:343-355.

White, J. R., K. R. Reddy and J. Majer-Newman. 2005. Performance of stormwater treatment wetlands receiving low phosphorus agricultural drainage water: implications for design and management. Conference Proceedings from the 9th

International Symposium on Biogeochemistry of Wetlands (March 20 – 23, 2005): Baton Rouge, LA. 122 p.

Whitson, I. R., D. S. Chanasyk and E. E. Prepas. 2003. Hydraulic properties of Orthic Gray Luvisolic soils and impact of winter logging. J. Environ. Eng. Sci. 2:S41-S49.

Wilson, S., M. Griffiths and M. Anielski. 2001. The Alberta GPI Accounts: Wetlands and Peatlands. Pembina Institute for Appropriate Development; Report #23. 29 p.

Wulder, M. A, J. A. Dechka, M. A. Gillis, J. E. Luther, R. J. Hall, A. Beaudoin and S. E. Franklin. 2003. Operational mapping of the land cover of the forested area of Canada with Landsat data: EOSD land cover program. For. Chron. 79:1075-1083.

Zar, J. H. 1998. Biostatistical analysis. 4th ed. Prentice Hall, Englewood Cliffs, NJ.929 p.

Zoltai, S.C. and Pollett, F.C. 1983. Wetlands in Canada: their classification, distribution, and use. *In* Mires: swamp, bog, fen, and moor. B. Regional studies. *Edited by*: A.J.P. Gore. Elsevier, Amsterdam. 245 – 268 p.

APPENDICES

APPENDIX A

CHI SQUARE COMPARISONS

TABLE A.1. WIIT and AWI abilities to identify soil types within watersheds

	Observed		Expected*		Obs-Exp		(Obs-Exp)^2		(Obs-Exp)^2/Exp	
	WIIT	**AWI**	**WIIT**	**AWI**	**WIIT**	**AWI**	**WIIT**	**AWI**	**WIIT**	**AWI**
Organic	35	13	39	9	-4	4	14	14	0.36	1.52
Gleysolic	27	3	24	6	3	-3	8	8	0.31	1.34
Other	5	0	4	1	1	-1	1	1	0.23	0.96

4.73

TABLE A.2. WIIT and AWI abilities to identify soil types in Millar Western FMA

	Observed		Expected*		Obs-Exp		(Obs-Exp)^2		(Obs-Exp)^2/Exp	
	WIIT	**AWI**	**WIIT**	**AWI**	**WIIT**	**AWI**	**WIIT**	**AWI**	**WIIT**	**AWI**
Organic	36	18	38	16	-2	2	3	3	0.07	0.16
Gleysolic	4	1	4	2	0.5	-0.5	0.3	0.3	0.08	0.18
Other	6	1	5	2	1	-1	1	1	0.26	0.59

1.33

TABLE A.3. WIIT and AWI abilities to identify polygon areas in Millar Western FMA

	Observed		Expected*		Obs-Exp		$(Obs-Exp)^2$		$(Obs-Exp)^2/Exp$	
	WIIT	AWI	WIIT	AWI	WIIT	AWI	WIIT	AWI	WIIT	AWI
< 1 ha	436	99	386	149	50	-50	2482	2482	6.43	16.68
1-30 ha	3047	1205	3069	1183	-22	22	494	494	0.16	0.42
31-60 ha	75	49	90	35	-15	15	211	211	2.35	6.10
61-90 ha	21	19	29	11	-8	8	62	62	2.15	5.57
91 ha +	15	13	20	8	-5	5	27	27	1.34	3.49

44.7

TABLE A.4. WIIT and AWI abilities to identify tree heights in Millar Western FMA

	Observed		Expected*		Obs-Exp		$(Obs-Exp)^2$		$(Obs-Exp)^2/Exp$	
	WIIT	AWI	WIIT	AWI	WIIT	AWI	WIIT	AWI	WIIT	AWI
No trees	377	187	407	157	-30	30	907	907	2.23	5.78
1-5 m	1012	498	1090	420	-78	78	6078	6078	5.58	14.47
6-10 m	912	371	926	357	-14	14	199	199	0.21	0.56
11 m +	1293	329	1171	451	122	-122	14930	14930	12.75	33.09

74.6

APPENDIX B

REGRESSION ANALYSIS

TABLE B.1. Deriving precipitation values for wetland using Environment Canada Data (Equation 2)

Regression Statistics	
R Square	0.655772
Standard Error	6.982061
Observations	23

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	1950.265	1950.265	40.0061	0.000003
Residual	21	1023.733	48.74918		
Total	22	2973.997			

TABLE B.2. Deriving wetland outflow using Willow outflow prior to log transformation (Equation 4).

Regression Statistics	
R Square	0.584581
Standard Error	0.836712
Observations	11

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	8.866519	8.866519	12.66487	0.006131
Residual	9	6.300789	0.700088		
Total	10	15.16731			

TABLE B.3. Deriving TP concentration leaving wetland from wetland outflow rate (Equation 5).

Regression Statistics	
R Square	0.37
Standard Error	0.307999
Observations	15

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	0.733861	0.733861	7.736056	0.016
Residual	13	1.233212	0.094862		
Total	14	1.967074			

TABLE B.4. Deriving TP concentration leaving wetland from wetland outflow rate (Equation 6).

Regression Statistics	
R Square	0.42
Standard Error	0.269016
Observations	15

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	0.689398	0.689398	9.526090	0.009
Residual	13	0.940803	0.072369		
Total	14	1.630201			

APPENDIX C

ANOVA ANALYSIS and TUKEY HSD

TABLE C.1. Volume of water reaching the ground under open, coniferous and deciduous canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	74.34
Coniferous	5	28.73
Deciduous	2	57.58

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	4743.43	2	2371.72	17.864	0.0011
Within Groups	1062.12	8	132.765		
Total	5805.55	10			

TUKEY

Groups	*Difference*	*Standard Error*	*Probability*
Deciduous – Coniferous	28.8537	9.64	0.01725
Open – Coniferous	45.6104	7.729	0.00036
Open – Deciduous	16.7566	9.979	0.13162

TABLE C.2. Wet TP concentrations deposited beneath deciduous, coniferous and open canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	165.3
Coniferous	5	668.4
Deciduous	2	265.5

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	615065	2	307532	8.5215	0.0104
Within Groups	288710	8	36088.8		
Total	903775	10			

TUKEY

Groups	*Difference*	*Standard Error*	*Probability*
Deciduous – Coniferous	-402.915	158.9	0.03498
Open – Coniferous	-503.092	127.4	0.00424
Open – Deciduous	-100.177	164.5	0.55948

TABLE C.3. Wet TDP concentrations deposited beneath deciduous, coniferous and open canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	65.65
Coniferous	5	378.6
Deciduous	2	165.8

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	226506	2	113253	5.1892	0.0359
Within Groups	174599	8	21824.9		
Total	401105	10			

TUKEY

Groups	*Difference*	*Standard Error*	*Probability*
Deciduous – Coniferous	-212.824	123.6	0.12339
Open – Coniferous	-312.927	99.1	0.01344
Open – Deciduous	-100.102	127.9	0.45650

TABLE C.4. Total amount of wet TP deposited beneath deciduous, coniferous and open canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	13.94
Coniferous	5	15.58
Deciduous	2	11.85

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	20.6505	2	10.3253	0.42714	0.6664
Within Groups	193.386	8	24.1732		
Total	214.036	10			

TABLE C.5. Total amount of wet TDP deposited beneath deciduous, coniferous and open canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	8.514
Coniferous	5	8.149
Deciduous	2	6.907

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	3.52412	2	1.76206	0.23245	0.7978
Within Groups	60.6419	8	7.58204		
Total	64.1661	10			

TABLE C.6. Dry TP concentrations deposited beneath deciduous, coniferous and open canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	37.5
Coniferous	5	41.72
Deciduous	2	39.12

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	40.3596	2	20.1798	0.37545	0.6985
Within Groups	429.989	8	53.7487		
Total	470.349	10			

TABLE C.7. Dry TDP concentrations deposited beneath deciduous, coniferous and open canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	22.11
Coniferous	5	26.25
Deciduous	2	28.05

Source	*SS*	*Df*	*MS*	*F*	*P-value*
Between Groups	59.6313	2	29.8157	1.1241	0.3713
Within Groups	212.2	8	26.525		
Total	271.831	10			

TABLE C.8. Total amount of dry TP deposited beneath deciduous, coniferous and open canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	11.25
Coniferous	5	12.51
Deciduous	2	11.74

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	3.63236	2	1.81618	0.37545	0.6985
Within Groups	38.699	8	4.83738		
Total	42.3314	10			

TABLE C.9. Total amount of dry TDP deposited beneath deciduous, coniferous and open canopies.

ANOVA

Groups	*Count*	*Average*
Open	4	6.634
Coniferous	5	7.721
Deciduous	2	8.415

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	4.88862	2	2.44431	0.98533	0.4144
Within Groups	19.8456	8	2.4807		
Total	24.7343	10			

TABLE C.10. Difference in concentration of TP in wet and dry deposition in open collectors.

ANOVA

Groups	*Count*	*Average*
Wet	4	13.93
Dry	4	11.25

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	14.4522	2	14.4521	1.75949	0.2329
Within Groups	49.2831	8	8.21386		
Total	63.7353	10			

TABLE C.11. Difference in concentration of TDP in wet and dry deposition in open collectors.

ANOVA

Groups	*Count*	*Average*
Wet	4	8.514
Dry	4	6.635

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	7.06516	2	7.06516	1.15978	0.3229
Within Groups	36.5508	8	6.09180		
Total		10			

TABLE C.12. TP concentrations in peatland, peatland margin and upland wells.

ANOVA

Groups	*Count*	*Average*
Peatland	5	179.3
Peatland margin	5	389.1
Upland	2	41.53

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	515563	2	237782	82.713	<0.0001
Within Groups	71680.9	23	3116.56		
Total	587244	25			

TUKEY

Groups	*Difference*	*Standard Error*	*Probability*
Peatland margin – Peatland	209.844	27.13	<<0.0001
Upland – Peatland	-137.722	26.32	<<0.0001
Upland – Peatland Margin	-347.567	27.13	<<0.0001

TABLE C.13. TDP concentrations in peatland, peatland margin and upland wells.

ANOVA

Groups	*Count*	*Average*
Peatland	5	90.22
Peatland margin	5	70.18
Upland	2	15.83

Source	*SS*	*df*	*MS*	*F*	*P-value*
Between Groups	26530.1	2	88382.8	189.01	<0.0001
Within Groups	10755.2	23	13265.1		
Total	37285.3	25			

TUKEY

Groups	*Difference*	*Standard Error*	*Probability*
Peatland margin – Peatland	-20.0472	10.51	0.068982
Upland – Peatland	-74.3889	10.19	<<0.0001
Upland – Peatland Margin	-54.3417	10.51	<<0.0001