University of Alberta

INTERLINKING HYDROLOGICAL BEHAVIOR AND INORGANIC NITROGEN CYCLING IN A FORESTED BOREAL WETLAND

by

NIMROD KALEF ©

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The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled **INTERLINKING HYDROLOGICAL BEHAVIOR AND INORGANIC N CYCLING IN A FORESTED BOREAL WETLAND** submitted by **NIMROD KALEF** in partial fulfillment of the requirements for the degree of **MASTER OF SCIENCE** in **ENVIRONMENTAL BIOLOGY AND ECOLOGY**.

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ABSTRACT

The dominant hydrologic and inorganic nitrogen dynamics were measured in a 1.5 ha mixed wood minerotrophic forested wetland located at the base of a 53 ha headwater catchment, near Lac La Biche, Alberta. During the study (1999-2000) there was lower than average rainfall. I determined that: 1) recharge by precipitation rarely exceeded upland and wetland storage, retarding surface runoff; 2) precipitation and ground water dominated inputs while evapotranspiration dominated outputs of the wetland water budget; 3) substrate stratigraphy resulted in complex ground water connections; 4) The wetland was a potential source of NH_4^+ to ground water from bottom sediments; 5) the wetland, in contrast to the uplands, was a potential source of NO_3^- from upper soils via surface runoff flushing. However, the likelihood of flushing is low due to low precipitation. Understanding wetland hydro-biogeochemistry and connectivity to other systems is crucial to assess the wetland's role in buffering catchment disturbance.

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1 INTRODUCTION

1.1 Overview

Wetlands represent an important component of Alberta's landscape and encompass a large portion of its boreal land base (NWWG 1988). In the last two decades, interest has arisen in the hydrological and ecological role of wetlands in the Boreal forest, and more recently, to how this role differs between landscape forms and scales (e.g., local scale vs. regional scale). Furthermore, understanding wetland behaviour in the boreal plains of Alberta is an important and pressing issue due to rapid development from forestry, oil extraction, and agricultural activities (Vitt et al.1996, Global Forest Watch 2002). It is well documented that disturbances such as logging or fire may significantly increase runoff and nutrient generation from forested watersheds (eg. Likens et al. 1977, Vitousek et al. 1982, Ward 1989). Many aquatic systems in the southern limits of the western mid-boreal ecozone are phosphorous rich (Shaw 1990, Vitt et al. 1995, Evans et al. 2000, Prepas et al. 2001, Devito et al. 2000b) and a possible increase in fluxes of nutrients such as nitrogen or phosphorous as a result of disturbance may alter the chemical and ecological balance of aquatic systems.

Wetlands, and especially riparian wetlands, have been promoted as environments which reduce nutrient loads by processes such as vegetation uptake, denitrification and adsorpsion (Malhi et al. 1990, Haycock et al. 1993, Piny et al. 1993, Emmett et al. 1994). However, several studies have shown that wetlands do not always function as nutrient sinks, but can also act as nutrient sources or as transformers (Devito and Dillon 1993a,b, Devito 1995, Hill 1996, Price and Waddington 2000). These studies demonstrated that the functionality of wetlands in nutrient cycling is spatially and temporally complex. Not only are wetlands spatially heterogeneous in both horizontal and vertical dimensions with respect to hydrology, sediment characteristics, and biogeochemical processes (Bohlke and Denver 1995, Hill 1996, Devito et al. 2000a), but their chemical responses may change with variations in external fluxes depending on climate, droughts and flooding (Devito and Dillon, 1993ab, Vitt et al. 1995, Devito et al. 2000a,b).

To better understand the chemical behaviour and existence of wetlands, and to implement effective management practices, research examining the interactions of hydrological and bio-chemical mechanisms is required (Hill 1996, Cirmo et al. 1997).

Recent reviews indicate that a large portion of our understanding of wetland hydro-biogeochemistry is based on information taken from headwater wetlands with shallow confining layers and ground water, and thus simple hydrology. The need for research in different kinds of environments with more complex hydro-geological settings is therefore required. The western boreal plains represent a complex environment, exhibiting a unique combination of a sub-humid climatic zone and a complex hydro-geological setting. Research on western boreal wetlands, combining hydrological and chemical functions, are few and only recently initiated (Vitt et al. 1995, Ferone 2001). Studies examining wetland biological and chemical status such as surface chemistry, vegetation and classification have been conducted in Alberta's boreal plains (e.g., Vitt and Chee 1990, Vitt and Halsey 1996, Thormann and Bayley 1997). Also, hydrological investigations looking at the behaviour of aquatic systems in similar glaciated settings have been conducted (e.g., Winter and Rosenberry 1995, La Baugh et al. 1997, Hayashi et al. 1998a,b, Shaw et al. 1990, Webster et al. 2000). These hydrological studies, however, were done within agriculturally impacted landscapes, or under climatological settings different from the boreal plains.

While either hydrology or biology/chemistry have been pursued in previous studies, little integrated hydro-biochemical work has been done in the pristine landscape of Alberta's boreal plains. To address this gap, the following study examines the linkages of hydrology and biogeochemistry in a western boreal wetland located in the mid-boreal transition zone.

1.2 HYDROLOGICAL MECHANISMS

Investigations of the hydrological mechanisms sustaining the existence of wetlands is the first step required in the understanding of the physical and ecological role of wetlands in the landscape. Wetlands existence depends on many factors ranging from climate and regional hydrogeology, to local stratigraphy, vegetation and hydrological setting (Winter et al. 2001). Studies have shown that groundwater - surface water interactions are important in determining wetland hydrological behavior, and affect factors such as water table stability, flow reversals and storage capabilities (Mulholland et al. 1990, Devito 1996, La Baugh et al. 1997, Winter 1999). In addition, the main water sources of aquatic systems may depend on their landscape position. For example, wetlands in uplands largely function as recharge zones and are highly dependent on atmospheric inputs, while lowland wetlands have a greater chance to function as flow through or discharge systems that are partially fed by regional or intermediate flow paths (Tóth 1963, La Baugh and Winter 1999, Webster et al. 2000).

The nature of water inputs will determine the hydrological and chemical characteristics of an aquatic system (Hayashi and Rosenberry 2001,Winter 2001), and will determine its vulnerability to various anthropogenic impacts at various time scales (Bedford 1999, Hill and Devito 1997, Devito et al. 2000b). For example, a system dependent on precipitation and local flows is more likely to be affected by local disturbances (such as logging), and by changes in precipitation chemistry and duration. In this case, disturbance usually results in an immediate effect. Aquatic systems that are hydrologically connected to regional ground water flows might be less susceptible to local disturbances but more vulnerable to ground water contamination which is usually associated with longer time scales.

Surface runoff can play an important role in wetland water and nutrient budgets. The spatial and temporal contributions of surface runoff, however, can be complex and can fluctuate greatly both in quantity and in chemical composition. During rain and snow melt events, the areas that contribute surface water and return flow expand in a non-uniform fashion within the catchment. This is usually due to the creation of a transient perched or non-perched water table reaching surface elevation and promoting quick surface runoff (Dunne and Black 1970, Freeze 1972, Govindaraju et al. 1991). The expansion, contraction, shape and size of the these contributing areas depend on site physiography, anteceded soil moisture conditions and on storm or melt intensity and duration. These dynamic contributing areas are known as Variable Source Areas (VSA) and their creation is coupled with a process know as a flushing mechanism (Govindaraju 1996). Hydrologically, flushing is the creation of a quick overland flow resulting from the rapid rise of the transient water table to the ground surface. Even though the term "flushing" relates to a specific runoff mechanism, it is often used when describing the associated physical transportation of usually near-surface substances such as nutrients, toxins or sediments (e.g., Hornberger et al. 1994, Hill et al. 1999).

Generally, models and predictions of surface runoff connectivity and transport of near surface nutrients should be coupled with the understanding of VSA behaviour (Hewlett et al. 1970, Govindaraju 1996, Creed et al. 1998). In our study site VSA characteristics may help to explain wetland hydrological and biochemical behaviour and their linkages to the watershed.

In Alberta's boreal climatic region, surface water runoff can be generated during two main periods. One is during spring melt, where long cold winters may have created impermeable frozen soils, preventing infiltration and subsequently allowing snowmelt to run according to topography (Woo and Winter 1993, Hayashi et al. 1998a). The second period is during summer rains, when most of the annual precipitation occurs. The characteristics of surface runoff during summer rains will depend on factors such as event frequencies, soil characteristics and the distribution of VSA. Currently, in the boreal plains of Alberta, the importance of spring melt vs. summer rains in water and nutrient transportation is not clear, especially in the boreal transition ecozone, where the study site is located.

1.3 LINKING HYDROLOGY TO NITROGEN CYCLING

To understand the role of wetlands in nitrogen (N) cycling and transportation, it is important to examine water table fluctuations and the hydrological fluxes that control imports and exports of N (Devito and Dillon 1993ab, Creed et al. 1996, Devito et al. 2000a). Atmospheric and hydrologic seasonal fluctuations control soil environments by controlling factors such as temperature and oxygen levels which in turn affect biological activity and N transformations. Ground water can be an important mechanism influencing water table stability and has the potential to either increase or dilute nutrients in aquatic systems (Bohlke and Denver 1995, Devito 1995, Komor and Magher 1996, Pinay et al. 1998). Studies in the boreal plains and the prairies have shown that although ground water may represent a small percentage of the hydrological budget of an aquatic system, it can represent a much larger component of the chemical budget (Shaw et al. 1990, Hayashi et al. 1998a,b, Ferone 2001). In pristine areas, such as our study site, groundwater N concentrations may be expected to be low due to biological uptake.

However, in agricultural areas high fertilizer loading may result in very high ground water N concentrations (Pinay et al. 1993, Pinay et al. 1998, Hill et al. 2000).

The type of groundwater interaction with wetlands is also important in influencing nutrient dynamics. Regional and intermediate groundwater fluxes, representing longer flow path and large residence time, will be characterized by stable chemical signatures (Buttle et al. 2001). In contrast, local subsurface fluxes and chemistry are more seasonally variable, usually affected by the local catchment vegetation, soil type, shallow stratigraphy and hydrogeology. These local groundwater movements, affected by water table elevations, are often responsible for flow reversal dynamics and the chemical fluxes associated with these phenomena (Devito et al. 1997, Ferone 2001). Finally, nutrients in surface water are spatially and seasonally variable, often to a larger extent than the local subsurface flow.

To assess the potential for nitrogen movement, it is important to identify both the source areas of different runoff waters and their spatial overlap with areas containing mobilized nitrogen. The upper organic soils are usually the areas where most of the flushable nutrients are found (Persson and Wiren 1995, Creed 1998, Carmosini 2000). In our study area, clay rich soils may promote the removal of nutrients from the upper organic soils by flushing processes. Following rain storms or snowmelt, areas with low water storage, such as clay rich soils, are susceptible to flushing by promoting a quick water table rise and large expansions of the VSA. Identifying the VSA and its potential to contribute runoff is therefore extremely important when assessing nutrient cycling and transports by wetlands.

Aside from ground and surface fluxes, N loads delivered by dry deposition and precipitation can be an important source, especially for ombrotrophic systems. Atmospheric N concentrations have increased due to anthropogenic activity (IPCC, 2002) and current research suggests that long-term effects may include alteration in N retention capabilities of forested systems, increasing N exports (Aber et al. 1989, Creed 1998, Brooks et al. 1999). Understanding the role and effects of atmospheric inputs on wetland N cycling and budget, may help us understand the potential role of wetlands in handling possible elevated N depositions/exports.

1.4 SOIL N AVAILABILITY AND CYCLING

It is important to understand the dynamics of soil N availability and soil N transformation, to further understand the potential of soil N to be leached or flushed out of the soils by various hydrologic flow paths. Furthermore, it is crucial to investigate soil N cycling and mobilization in various and discrete landscape units (e.g., wetlands and hill slopes), understand their potential role as N sources, and couple it with related hydrological processes. Work by Creed et al. (1996) and Creed and Band (1998) for example, developed topographically related N flushing models, using hill slope soils as the major contributing N sources and surface water flushing as the explanatory major transport mechanism.

Coupling nutrient sources and hydrological dynamics, however, can be complex, and models may not necessarily address this complexity. Nutrient export models that use topography as the primary variable tend to either unify soils and nutrient location, or ignore landscape features that are not necessarily a direct derivative of topography (e.g., Creed et al. 1998, Butlle et al. 2001). Wetlands may be such an "ignored" or "blended" feature despite their unique soil structure and potential to significantly affect nutrient creation and consumption. Unifying soil N sources over the whole catchment may be successful only in hydrologically simple landscapes, but may fail in landscapes with complex hydrogeology and landscape features

It is commonly stated that forest soils of the western Boreal plain have low N availability, and therefore are a relatively minor N source, because microbial cycling is inhibited by the cool temperatures and short growing seasons (Van Cleve et al. 1981, Pomeroy et al. 1999, Carmosini 2001). These statements, however, are mostly based on research investigating uplands with relatively dry and shallow organic soils. Whether deeper and wetter organic soils, such as found in wetlands, show the same low N availability is largely unknown.

Soil N cycling can be affected by episodic occurrences as well as site characteristics. For example, disturbance by clear cutting has been shown to increase extractable soil nitrate of boreal mixwood stands in Saskatchewan (Walley et al. 1996), but little influence due to harvesting has been observed in the uplands of our study area (Carmosini 2001). Regardless of disturbance effects, these studies and others (e.g., Vitousek et al. 1982, Hendrickson et al. 1985, Tamm 1991, Janssen 1996) demonstrate that N cycling such as mineralization and nitrification are not only affected by external occurrences, but also by local/internal variables such as soil characteristics.

On a temporal scale, recent literature suggests that winter microbial activity may contribute significantly to soil N cycling (Jones 1999, Brooks 1999a,b, Devito et al. 1999, Carmosini 2000). As we investigate the importance of spring melt in nutrient transport at our study site, it is also important to evaluate the fate of extractable inorganic N during the winter-spring period. Also, if winter N cycling is a more active process than thought, it may have implications with respect to winter disturbances such as winter logging, a common practice in Alberta.

1.5 OBJECTIVES AND HYPOTHESES

In the following chapter, the hydro-biogeochemical linkages and behaviour of a forested wetland, located in the transition zone of Alberta's mid-boreal plains, will be described.

The main objectives are:

I) To determine how the major hydrological inflows and outflows affect the spatial and temporal dynamics of the saturated areas within the wetland.

Specific questions asked are:

1) What are the factors and mechanisms responsible for the existence of the wetland?
    a) How do the location and stratigraphy support the existence of the wetland?
    b) What are the dominant hydrological flows that maintain the wetland?
    c) How do these flows change spatially and temporally?

II) To examine how nitrate and ammonia are being transported to, transformed within and exported from the wetland system, and to identify the role of the various hydrological fluxes in these processes. These will be tied to the net function of the wetland in N retention and transformation.

Specific questions asked are:

1. How does the chemistry of incoming water, such as precipitation, surface and ground water, transform in the wetland?
2. Which hydrologic flow paths dominate nitrate and ammonium movements?
3. Does flow to and from the wetland show a significant hydrological and chemical flushing behaviour?
   a. How is flushing, especially of nitrate, the mobile form of nitrogen, facilitated by the expansion and contraction of the variable source areas during the summer?
   b. Is flushing facilitated by the frost table and snowmelt during spring?

III) To examine the possible role of soils within the wetland and at the adjacent hill slopes as nitrate and ammonia sources, and assess how water table fluctuations and oxidation affect soil N dynamics.

Specific questions asked are:

1. What are the temporal changes in, and related mechanisms affecting, extractable and leachable soil N?
   a. How does extractable inorganic soil N change spatially and temporally in the study area? This will be done by comparing the two major landscape units: wetland and hill slopes
   b. What are the possible mechanisms explaining seasonal dynamics?
   c. By looking at the effects of water table fluctuations on oxidation levels at the upper soil layer; is there a relationship between water table fluctuation/saturated areas and upper-soil pore water N concentrations?

My hypotheses are:

A) i) The wetland is fed by both local shallow and regional ground water flows as it is located in a topographic low of the catchment and in a lowland position on a regional scale. The wetland, therefore, will exhibit a relatively stable water table (WT) and will be quick to respond to precipitation events. **ii)** Surface runoff will be an important component in water contributions during spring melt, facilitated by impermeable frozen soils and snowmelt, and will also be a major component during summer when precipitation is highest. However, summer precipitation will facilitate higher surface runoff than snowmelt due to its higher mean input (about four times larger).

B) i) Ground water in this pristine area would be characterized by low N concentrations. Based on previous studies in the region (e.g., Vitt at al. 1995), wetlands should also show low and relatively constant nitrate concentrations over depth, while ammonia concentration will increase with depth. **ii)** Due to spatial differences in soil characteristics, storage, and water table elevation between the wetland and its surrounding, the wetland will be the main site to show well defined VSA expansion that may include nearby connecting saturated areas. This behaviour will facilitated flushing events associated with a quick rise of the WT, especially during summer rains. Nitrate flushing thereby might be observed either following summer rains, depending on preceding WT fluctuations and soil oxidation in the wetland, or during spring, if snowmelt run, mostly over impermeable ground, will leach nitrate from the organic layer of the upland/hill slopes soils.

C) Soil at the wetland should show low nitrate concentration and high ammonia levels if WT levels are seasonally stable. However WT draw down during early summer may introduce oxygen to the upper soil layers thereby reducing ammonia concentrations and elevating extractable soil nitrate, which in turn will be flushed at a later date during summer rains. Extractable inorganic N of hill slopes is expected to be generally low and further decrease during the growing period. Nitrate levels are expected to be higher than the wetland soils, if wetland WT is seasonally stable.

2 METHODS AND MATERIALS

2.1 STUDY AREA

2.1.1 Site description

The study site (55.1 °N, 113.8 °W) is located in the sub-humid low eco-climatic region of north-central Alberta (Ecoregions Working Group 2002), about 70 km north of Lac La Biche (Fig 2.1A). The boreal region is dominated by aspen mixed-wood. The 1.5 ha mixed wood minerotrophic forested wetland is located in a topographic depression near the outflow of a 53 ha aspen dominated headwater catchment. The catchment discharges into Moose lake (Fraser et al. 2002) (Fig 2.1B). Using 1:40,000 air photos and a topographic survey of the area (Fraser et al. 2002), the catchment was divided into four sub catchments. Three sub-catchments A (14.6 ha), B (5.5 ha) and C (18.3 ha), contribute surface water through three tributaries to the wetland, as well as ground water to sub-catchment D that hosts the wetland.

Approximately 50% of the upper part of the catchment was logged in 1996 (Fig 2.2A) initially, as part of the "Terrestrial and Riparian Organisms Lakes and Streams" (TROLS) project to investigate the role of buffer strips in the Boreal Mixedwood zone and longer term project examining the Hydro-chemical function of headwater catchment (Devito unpublished data).

The region is characterized by warm summers and very cold winters. Long term (1970-99) average precipitation and potential evaporation are 468 mm and 530 mm yr^{-1} respectively, with approximately 80-100 mm falling as snow (Alberta Environment 2002). Long term mean annual temperature is 1.5 °C, and mean January and July temperatures are –16.7 °C and 16.3 °C, respectively.

The region is underlined with soft sedimentary bedrock of upper Cretaceous shale, siltstones, and sandstones. The glacial drift ranges in thickness from 20 to 200 m and contains complex inter-beds of clay, sand, and gravel (Tokarsky and Epp 1987). Mineral soils in the region range from well to poorly–drained, where Gray Luvisols and Eutric Brunisols dominate uplands and hillslopes, and Gleysosls and organic soils dominate low-lying areas.

Trembling aspen (*Populus tremuloides*), white spruce (*Picea glauca*) and paper birch (*Betula papyrifera*) dominate the south facing slopes of the wetland. River alder (*Alnus rugosa*), paper birch and balsam poplar (*Populus balsamifera*) dominate the wetland area. White Spruce was common in the southern part of the wetland and on the north-facing slope.

2.1.2 Stratigraphy

The stratigraphy of the wetland was largely determined by two main transects (north-south, east-west) where cores ranging from 1.5 to 5 m in depth were assessed. Sixteen cores from upstream of the wetland down to the lake (~ east - west) and 10 cores from a northern slope to the wetland (~ north - south) were taken (Fig 2.2).

2.2 HYDRO-METEOROLOGICAL ASSESSMENTS

2.2.1 Atmospheric fluxes

A meteorological station was established in the upper cut block section of the catchment (Fig 2.1B). For the summer of 1999, precipitation amounts were determined by a tipping bucket and air temperature was recorded. From November 1999 to March 2000 total daily precipitation was collected by a "Belfort Universal Precipitation Gauge" and daily min and max air (1.5 m) and soil temperatures (10 cm) were recorded every 30 minutes. Regional rainfall data were obtained from Alberta Environment records of nearby fire towers. Literature values for canopy storage capacity were used with our own data to estimate interception amounts. Canopy storage capacity of 1 mm (Elliott et al. 1998) was subtracted from each rain event if more than two days had passed since the preceding rain event. Results were than compared to total interception values from similar sites (Hogg 1999, Elliott et al. 1998).

Snow distribution, losses, and water equivalents (SWE) were assessed periodically during March-April 2000 using seven 50 m transects, with 5 sampling points each. Sampling locations were roughly near the soil core locations (Fig 2.3).

Evapotranspiration (ET) from the wetland was not measured directly in the site and was estimated using the Thornthwaite equation (Thornthwaite et al. 1957) calculating potential evaporation (PET):

$$E = 16C\left(\frac{10T_m}{I}\right)^a \qquad (2.1)$$

where E = the potential evapotranspiration (mm); C = the daylight coefficient; T_m = average monthly temperature (°C),

$$I = \sum\left(\frac{T_m}{5}\right)^{1.51} \qquad (2.2)$$

and a = an exponent derived from the heat index (I)
$a = (67.5\text{x}10^{-8}I^3) - (77.1\text{x}10^{-6}I^2) + (.0179I) + (.492)$

2.2.2 Surface water

Surface water fluxes were assessed by using two input weirs at the tributaries of sub-catchments C and B leading to the main channel that enters the wetland. One weir located at the western output channel of the wetland was used for outgoing runoff measurements (Fig 2.3 and 2.2E). The weirs were made of plywood and metal sheets embedded 0.3 m into the ground and were designed to block surface runoff but not ground water. During runoff periods, weirs were manually measured for water discharge from 1 to 4 times a day in addition to continuous measurements by automated water level recorders. Rating curves were constructed from measurements of stage and discharge at the weirs during discharge events between 1998-2000. During spring 1999, the continuous data for C weir were not recorded successfully due to technical problems.

2.2.3 Ground water

Ground water fluxes and water table elevation were estimated by using wells and piezometers in two main transects: 1) "stream" (approx east-west) with 5 nests, and 2) "hillslope-wetland" (approx north-south) with 5 nests (Fig 2.3). Wells were fully perforated and were wrapped in 'well sock' prior to installation. Piezometers were constructed from 0.0191m ID PVC pipe and coupled to 0.2 m slotted and shielded PVC heads. Wells were inserted into pre-bored holes extending into the underlying confining material and back filled with aggregate. Piezometers were inserted into pre-bored 0.5 and up to 5 m depth holes.

Subsurface stratigraphy was mapped using information from soil cores logged at and between piezometers nests, and hydraulic conductivities were determined from bail tests of piezometers (Hvorslev 1951) and wells (Papadopoulos et al. 1973) using Starpoint software "Super Slug©" version 3.02. As hydraulic conductivities for a given soil type are subject to large uncertainties, each soil type received an hydraulic conductivity range of two orders of magnitude in groundwater calculations to ensure groundwater calculations were conservative estimates. During 1999 the wells at "wetland edge", "Aspen hill slope" and "output A" (Fig 2.3) had pressure transducers while "wetland east", and "wetland main" had float potentiometers. In 2000 only pressure transducers were used, and in the wetland area were installed in wells "wetland edge", and "wetland main".

2.2.3.1 Shallow groundwater: Shallow ground water flow was calculated according to Dupuit-Forchheimer model (Freeze and Cherry 1979) and was based on data from three locations: 1) a northern slope representing local hill slope of D catchment, 2) east slope, representing the input channel and inputs from the upper sub-catchments (A, B and C) and 3) wetland outlet, representing the wetland's west runoff output.

2.2.3.2 Deep ground water: Yearly losses to recharge were estimated using Darcy's law and data from wells and piezometers from the "wetland south" location.

Deep ground water seepage was also estimated using winter water table draw down as rain and evapotranspiration fluxes during winter are considered negligible. Average recharge was estimated by using the specific yield and water table elevations at three well locations: "wetland main", "wetland east" and "wetland south" between 22 Oct 1999 and 24 March 2000. The winter recharge values were then extrapolated to a yearly value. Frost table advance, however, may affect water table elevations during winter and may result in overestimation of seepage rates (Price 1983). However, prior to winter the water table at the wetland dropped 1 m below the ground surface, most likely below the range of the frost table.

Specific yield (S_y) was estimated using the differences in the volumetric soil moisture between upper organic layer cores (10 cm) and deeper mineral soils taken on 04

July 1999, when water table was at or near surface (100% saturation), to samples taken under low water table conditions on 26 August 1999 (drained conditions). Soils comprised of a mix of mineral and organics received an average value between mineral and organics. Mineral S_y values were found to be comparable to literature values (Fetter 1994).

2.2.3.3 Saturated areas: Saturated zones in the wetland site and in the entire catchment were mapped several times each year. Areas were considered saturated if moderate pressure applied by a boot forced water from the soil, or if standing water was observed. Saturated areas were paced off between stakes, and at right angles to stakes in a saturated area network (Fig 2.3). In addition rough drawings of the visible standing water were made. Field notes were then digitized for estimation of the saturated area at each given date.

2.2.4 Infiltration and Frost table measurements

Frost table depths during spring melt at the wetland and adjacent hill slopes were estimated by pounding a metal rod into the ground to a point were the solid frozen soil blocked further movement (Young et al. 1997). Seven transects along the catchment (Fig 2.3) were measured every 5 days. Each transect contained 5 sampling points 10 m from each other and at each sampling point two measurements were taken.

Soil infiltration capabilities were assessed by insertion of metal rings 15.5 cm in diameter and 17 cm in height. The rings were inserted 10 cm into the ground in the fall before soil frost. The rings were then filled with water and allowed to drain freely. When the water disappeared, the tins were refilled and only then, the rate of decreasing hydrologic head was measured. Rates were calculated based on the total decrease in head vs. overall time. Three transects, each containing seven rings, were established, one at an aspen hill slope, one at a conifer hill slope and one in the wetland area (Fig 2.3).

2.2.5 Water Budget calculations

The water budget for the wetland was calculated for the duration of one hydrological year starting on 01 June 1999 and ending on 31 May 2000. This period exclude the effects of spring 1999 that was not fully investigated, and allows the better integration of more detailed data taken from June 1999 to the end of spring 2000.

The wetland water budget was calculated as

$$\frac{\Delta V}{\Delta t} = P_n + S_i + G_i - ET - S_o - G_o \qquad (2.3)$$

Where: $\frac{\Delta V}{\Delta t}$ = change in volume of water storage in wetlands per unit time, t; P_n = net precipitation; S_i = surface inflows; G_i = groundwater inputs; ET = evapotranspiration; S_o = surface outflows and G_o = groundwater outputs including the wetland bottom.

A negative result indicates water losses over time t, while a positive result indicates water gain, both results are usually confirmed by changes in water table position, where magnitude is dependent on the specific yield of the wetland soils.

2.3 CHEMICAL ANALYSIS

2.3.1 Water sampling procedures

Surface water samples were collected periodically at or near wells and piezometer nests and weirs (Fig 2.4). Samples were coarse filtered with 80 μm mesh, and stored in either acid washed Nalgene bottles or 140 ml plastic cups. Precipitation near the met station was collected by a bucket lined with a clear plastic bag. Wells and piezometers were usually pumped out dry a day before sampling to replace stagnant water with fresh water. However, due to large variation in recharge rates, water samples were taken within several hours to two weeks after pump-out. Samples from shallow seepers were extracted using a syringe to minimize oxygen introduction.

All water samples (precipitation, ground and surface) were fine filtered within 24 hrs using Millipore 0.45 μm filter paper. Parts of the samples were frozen for further N and P analysis while the rest was kept refrigerated at 4 °C for cation, anion and DOC analysis.

2.3.2 Chemical analysis of water

At the University of Alberta limnology lab, water samples were examined for pH and conductivity (μS/cm) before filtration. Dissolve Organic Carbon (DOC) was assessed using a TOC-5000A Total Organic Carbon Analyzer, applying methods recommended by Greenberg et al. (1992). Anions were analyzed using a Dionex 2000i/SP Ion Chromatograph, applying methods recommended by Pfaff (1993). Cations were assessed using a Perkin Elmer Atomic Absorption Spectrometer 3300, applying methods recommended by Stainton et al. (1977). Nitrate and ammonium were analyzed using a Technicon Autoanalyzer II system, applying recommended manufacturer techniques (Technicon 1973a,b).

2.3.3 Oxidation zone

Oxygen presence in the upper soil layers was estimated by evaluating rust marks on metal rods (Silins 1997). Next to each key well two to three metal rods were inserted to a maximum depth of 0.6 meters. Every two weeks the rods were pulled out and obvious rust marks were noted and depth to them measured. The rods were then polished using sandpaper to remove any rust signs before re-insertion. In the open air and under humid conditions in the wetland area, the metal rods showed rust marks within 4-5 days. Data was collected for the 1999 season starting after ground thaw in June. However, during spring melt 2000 data were not collected due to ground frost, not allowing for rod insertion or pullout.

2.3.4 Inorganic N budget

The wetland's inorganic N budget was calculated as

$$\frac{\Delta N}{\Delta t} = (C_p * P_n) + (C_{Si} * S_i) + (C_{Gi} * G_i) - (C_{So} * S_o) - (C_{Go} * G_o) \qquad (2.4)$$

where: $\frac{\Delta N}{\Delta t}$ = change in N species (NO_3^--N or NH_4^+-N) storage in wetlands per unit time, t; C_x= N species concentration of input or output mechanisms. P_n = net precipitation; S_i = surface inflows; G_i = groundwater inputs; S_o = surface outflows and G_o = groundwater outputs including the wetland's bottom.

A negative result for NO_3^--N or NH_4^+-N indicates loss over time t, suggesting the wetland is a source, while a positive result indicates consumption or transformation of N, suggesting the wetland is a sink. Calculation intervals for weighing the chemical mass of inputs and outputs were defined by the date midway between sampling (La Baugh et al. 1997). This method was used for shallow and deep groundwater seepage, surface runoff and for rain. Sampling dates and frequencies are reported in Figure 2.4. Shallow groundwater NO_3^--N and NH_4^+-N mass inputs and outputs were calculated by multiplying mean concentrations of all shallow piezometers by the volume of seepage for each corresponding input and output time segment. Inorganic N mass flux in seepage from the wetland was determined using chemistry from deep piezometers at "wetland main" and "wetland south", an area where most of seepage is estimated to occur. Bulk wet deposition mass was determined from the concentration multiplied by the precipitation volume for the weighted interval on each rain event. Snow contribution was based on mean pre-melt snow chemistry multiplied by snow water equivalent volume. In both the NH_4^+-N and NO_3^--N budgets, percent retention was calculated as the residual divided by the total inputs, multiplied by 100.

2.4 Budget Error estimations

The variance of water and chemical budgets was calculated from the sum of squares error (Winter, 1981, Devito 1989):

$$S_T^2 = S_p^2 + S_{Gi}^2 + S_{Si}^2 + S + _{ET}^2 + S_{Go}^2 + S_{So}^2 \qquad 2.5$$

where S_T is the standard deviation of a total time period for which a water budget was calculated, P is precipitation, G_i is ground water inputs, G_o is ground water outputs, ET is evapotranspiration, S_i and S_o are surface water inputs and outputs respectively. To obtain S^2_T, total water volumes based on time intervals between chemical sampling, were multiplied by their associated factional error (CV) and then squared and summed, (Devito et al. 1989). Measurement errors in water chemistry and hydrology are assumed independent; thus covariance terms are not included. Variance of the nutrient loading term (a product) can be approximated by (Mood et al. 1974):

$$VAR(X,Y) \cong \bar{X}^2 * S_Y^2 + \bar{Y}^2 * S_x^2 + S_Y^2 * S_x^2 \qquad 2.6$$

This is equivalent to adding the square of the percent errors for both hydrological and chemical measurements. The variance associated with nutrient mass was determined for each sampling time interval and summed to produce an annual value.

Interception values and associated error were taken from the literature (Hogg 1999, Elliott et al. 1998). Precipitation errors were based on both in-field equipment measurements and on literature values for uncertainties due to gauge height and area coverage (Winter 1981, Wentz et al. 1995) and variance was calculated on a monthly basis. Evapotranspiration error was based on literature values where in-site potential evaporation calculated by Thornthwaite equation was considered the upper limit. Snowpack error was based on in field variation in snowpack SWE just before snowmelt. Variation in groundwater flows was based on two methods. The first method was using in field measurements of K_{sat} in various wells and piezometers. Each classified soil layer (e.g., sand, sand-silt) had several K_{sat} values from identical or similar soil layers within the catchment, and the range in K_{sat} of these layers was used as a basis for calculating mean and deviation. As K_{sat} deviations can reach values of over an order of magnitude, uncertainties were also calculated by using ground water as a residual of inputs and outputs. Calculating maximum and minimum residuals of ground water incorporating the deviations of each mass balance component, allowed for a relatively reasonable range in ground water uncertainties that conforms to the overall accuracy of the other components of the mass balance, which in turn usually carry much lower uncertainties. As ground water chemical sampling was infrequent, the variance associated with nutrient mass was determined from CV in chemistry from several piezometers located in similar strata and depth. Errors in surface water estimates were calculated by combining uncertainties of discharge measurements within the field, with published data on error associated with automatic samplers, and with in field estimation of leakage from the weirs.

2.5 SOILS

2.5.1 Soil core collection

Extractable soil N pools were estimated by coring periodically from March to November 1999 and March and April 2000 (Fig 2.3). Initially seven transects were chosen, two hillslopes (conifer and aspen), a wetland edge, and four wetland sites. To minimize site disturbance, from July the number of transects was reduced to two hillslopes, one foothill and two- three wetland sites. In 2000 four transects were assessed, two hillslopes, one foothill and one wetland site.

For a reliable estimate of soil properties, 5 to 10 cores in each transect are recommended (Crepin and Johnson 1993). Therefore, 5 to 7 pairs of cores (two depth: 0 to 10 cm and 10 cm to 20 cm) were taken. During the study period, soil temperatures at four sites (Fig 2.3) were continuously measured every two hours at depths of 5 and 15 cm using a Onset Instruments "HoboTemp" temperature sensor and data logger.

2.5.2 Soil analysis

At the lab, soil cores were weighed and then hand-mixed within their bags until visually homogenize and subsampled by placing approximately 5 gram (dry weight) with 50 ml of 2 M KCL. The mix was then shaken for 1 hr to extract inorganic N. The extracts were gravity filtered through pre-washed Fisherbrand 0.45um filter paper and frozen until analyzed for NO_3^--N and NH_4^+-N content using a Technicon autoanalyzer (Technicon 1973a,b). Volumetric soil water content was determined for every core by either drying the whole core or a sub-sample of hand –mixed soil from the core at 65 °C for a minimum of 4 days. pH was measured using a glass electrode where organic soil to water ratio was 1:5 and 1:2 for mineral soils (Carter 1993).

2.6 STATISTICS

Most of the soil core data was not normally distributed and therefore the non parametric Kruskal-Wallis ANOVA was used to assess nutrient pools and moisture levels of all sites. For a comparison of two sites the Mann-Whitney U test was used. The data was analyzed using StatSoft STATISTICA 6.0, computer software.

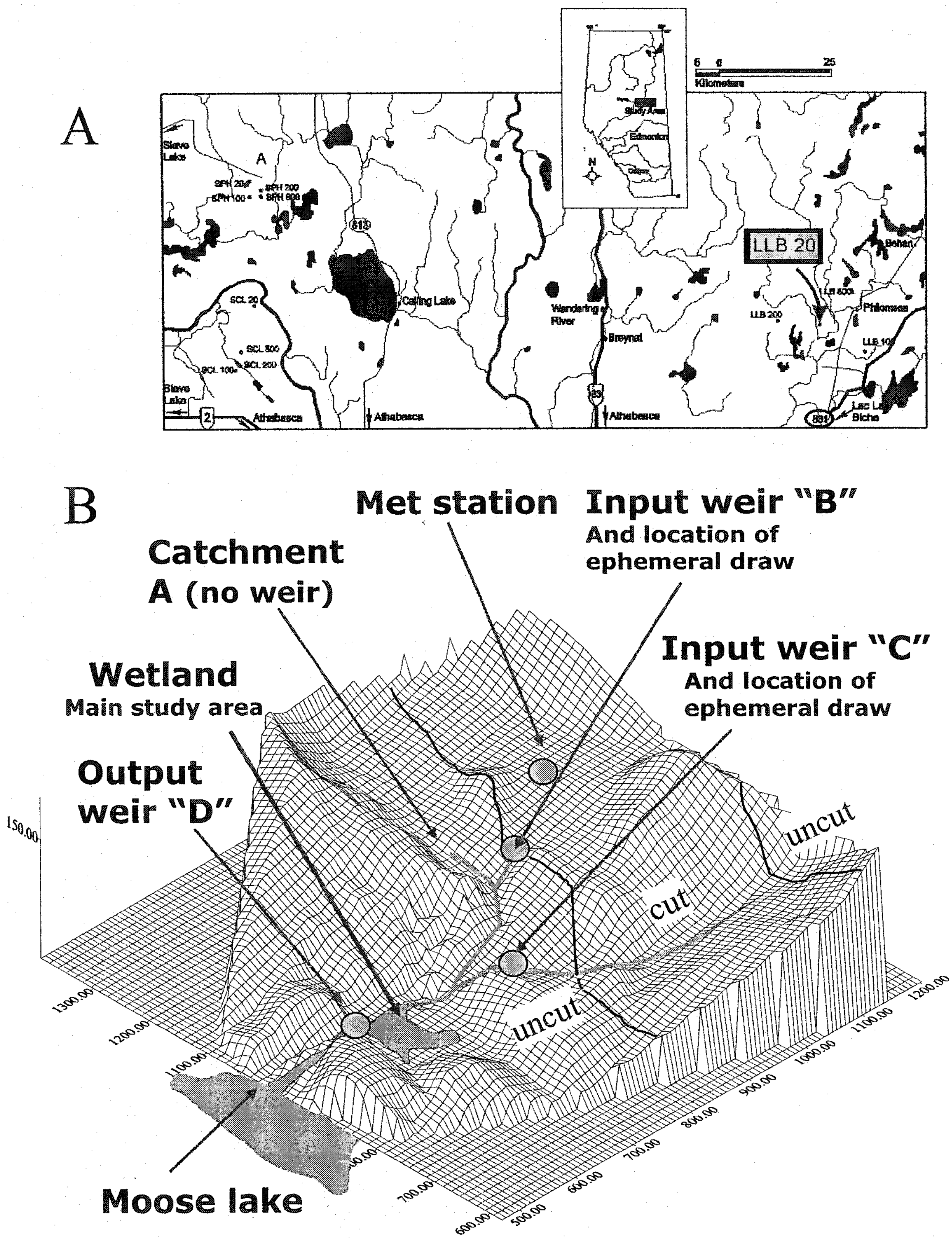


Figure 2.1 A) Location of study site, near Lac la Biche, Alberta, Canada and, B) a three dimensional schematic of the catchment illustrating the major features.

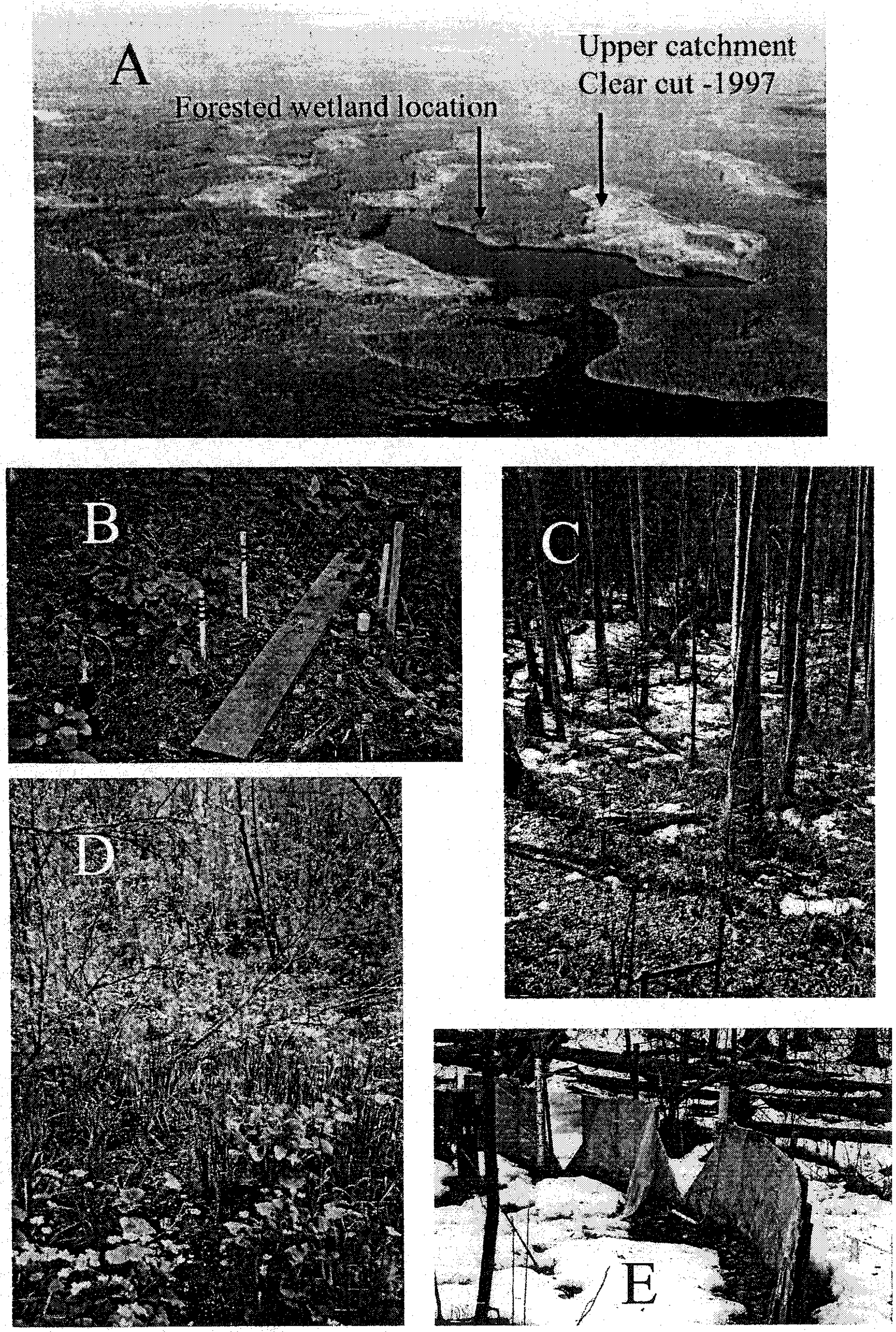


Figure 2.2 A) air photo of study area; B) well and piezometers nest (wetland east location); C) aspen hill slope; D) "Wetland main" study area, and E) wetland output weir (D weir)

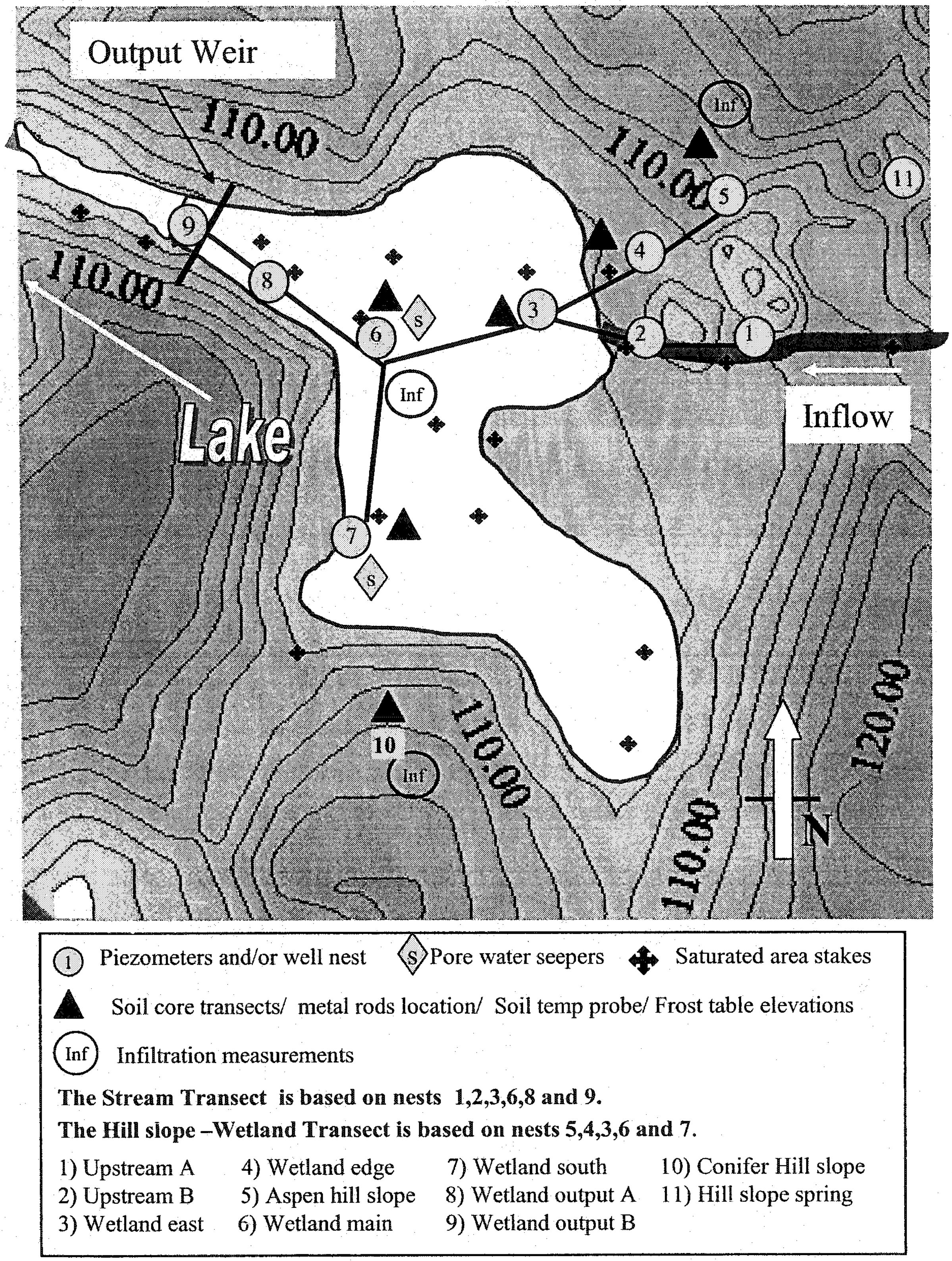


Figure 2.3 Map of study wetland, illustrating elevations, equipment locations, and sampling sites.

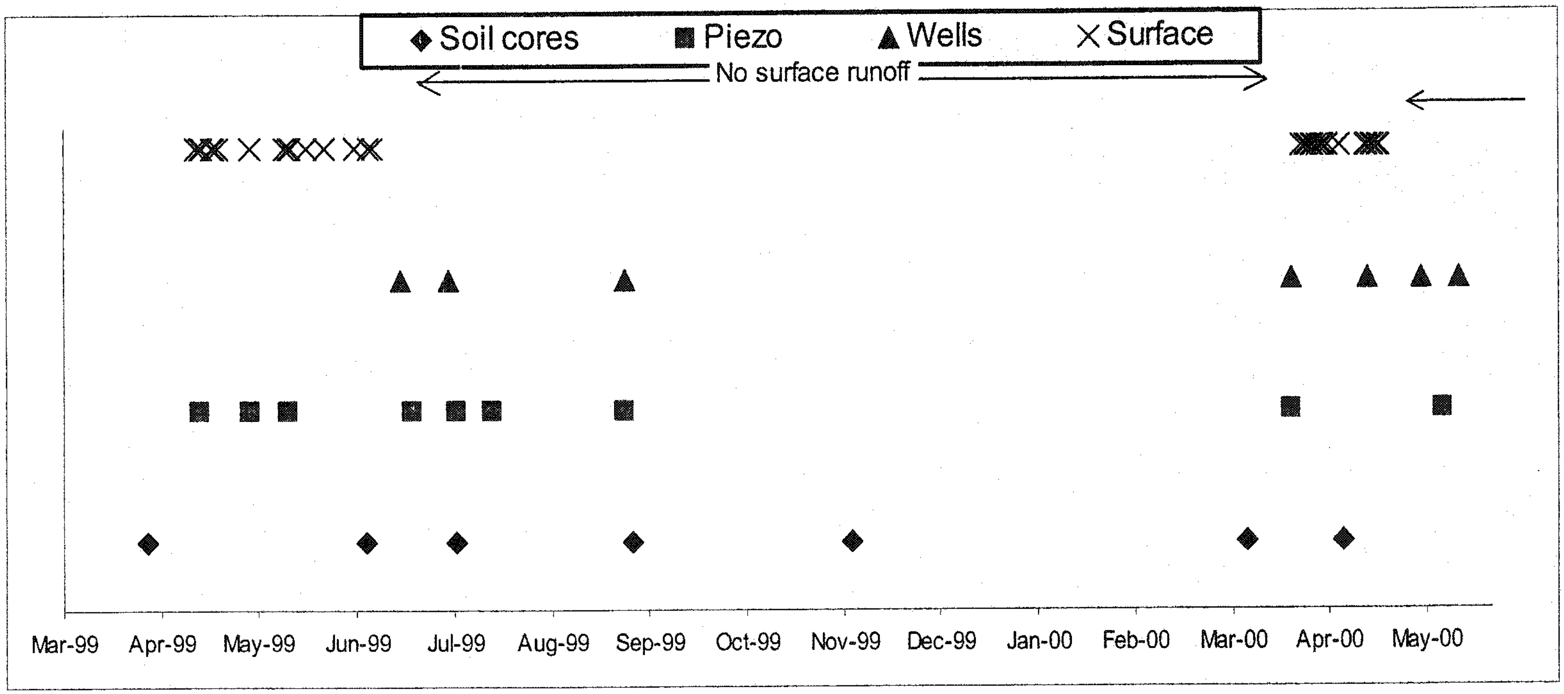


Figure 2.4 Nutrient sampling frequencies of groundwater, surface runoff and soil cores. On several occasions, complete piezometer sampling needed more than one date due to low recharge periods.

3 RESULTS

3.1 ANNUAL WATER BUDGET

A summary of hydrologic inputs and outputs is presented in Table 3.1. During the study, water outputs exceeded inputs resulting in a decrease in WT elevation at the end of the hydrological year on 31 May 2000 (Fig 3.1). Atmospheric fluxes were the dominant components of the wetland water budget. Precipitation less interception and evapotranspiration plus sublimation accounted for 60 and 88% of total inputs and outputs, respectively. Ground water inputs accounted for about 40% and were dominated by shallow ground water. The small outputs by ground water, however, were dominated by deep seepage.

The budget unexplained-residual of –232 mm was much larger than the calculated change in wetland storage of –17 mm based on water table elevations. In addition, the input budget, affected by a large percent of shallow ground water, was associated with the largest errors.

3.2 SEASONAL HYDROLOGICAL TRENDS

3.2.1 Precipitation and Evaporation

During the study, seasonal patterns followed the long-term trend (1970-99), with highest precipitation occurring during the summer months when evapotranspiration is the highest (Fig 3.2). Precipitation amounts for each month, however, were 40-80% less than the long term mean (Fig 3.2). Total annual precipitation of 259 mm and spring and summer rainfall of 190 mm were about half the long term averages of 470 mm and 350-400 mm, respectively. In addition, large rain events were absent and events greater than 10 mm were rare (Fig 3.3). Interception was estimated to be 17%, similar to other mix wood stands (Elliott et al. 1998, Hogg 1999), reducing direct precipitation by about 30 mm. Of total precipitation, 69 mm fell as snow in 1999-2000, about 25 mm less than the long term average. However, 51 mm of SWE was measured in March 2000 indicating the effect of sublimation. Snow pack density and distribution were found to be relatively even along most of the catchment, with similar accumulations in the wetland itself vs. catchment A and B (Table 3.2).

During the study, temperatures were similar to the annual means with the exception of spring 2000 when temperatures were slightly lower (Fig 3.3). Thornthwaite estimation of PET for the study period was 558 mm/yr, which is similar to the long term PET mean of 530 mm/yr. Seasonally, the highest monthly PET values co-occur with the largest rain fall periods during the summer (Fig 3.2). Annually, mean PET values are larger than long term mean precipitation suggesting water deficit in the site on a long term basis.

3.2.2 Spatial and Temporal variations in Water Table fluctuations

For most of the year, water fluxes to the wetland were insufficient to maintain the water table (WT) above ground level.

The depth of the WT from ground level varied substantially among sites within the wetland due to surface topography. For example, the distance between the water table and the surface of wetland wells could vary from 0.2 to 0.8 m below surface, while at the wetland edge the WT could reach up to 2 m below surface (e.g., "main" vs. "edge"). Overall, at the site denoted as "main" (the hydrological focus zone of the wetland) the WT fluctuated up to 1 m while at the edge site it fluctuated by over 3 m.

In both the wetland and the surrounding hill slopes water table fluctuations followed a similar pattern: a sharp increase in WT elevations during spring followed by a sharp decrease in elevations below ground level during summer interrupted only by small increases following small rain events (Fig 3.1). The WT continued to decline during fall and winter at a moderate rate.

During the study, the WT gradient consistently followed the topographic relief along the stream transect and hence no flow reversals were observed (Fig 3.1b). However, WT fluctuations along the hill slopes-wetland transect were less synchronized resulting in a period of flow reversals. From April to May 2000 the focal zone of ground water flow changed from the northwestern section of the wetland to the southern section, thus creating a ground water flow reversal where water table gradient was from the wetland into the north hill slope. At the same time, three small and transient flow reversals were observed to occur between the northern hill slopes and the wetland (Fig 3.1a).

3.2.3 Stream runoff and distribution of saturated areas

Only small patches of surface water and stream runoff were produced in the catchment during the study period (Fig 3.4). Most of the observed surface water and stream runoff were generated during the spring season with much higher stream flow during spring 1999. During spring 1999 stream runoff from upper tributaries connected to the wetland and during March and May there was surface outflow from the wetland to the lake.

Following the dry summer and fall, stream runoff decreased substantially and there was no surface water connection among the upper tributaries and the wetland nor between the wetland and the lake. As a result, there was no wetland outflow during summer and fall. Therefore, during the study period (01 June 1999 to 31 May 2000) the wetland did not receive or release surface runoff.

The low runoff inputs and precipitation affected the distribution of the saturated areas within the wetland (Fig 3.5). From spring 1998 to 1999, the saturated surfaces in the wetland shrank more than half (Fraser et al. 2002) and with the low summer precipitation the saturated areas disappeared by mid-summer 1999. Saturated areas at the wetland's periphery were the first to dry out and retreated inward toward a focal zone at the "wetland main" site (Fig 3.5). At the same location, a perched water table over ice during spring 2000 resulted in localized stagnating surface water. Similarly, during the same period, localized surface water over frozen ground were observed at the area of the input weirs. The only site that had continuous surface water year around was a small spring at the hill slopes denoted "h.s. spring" (Fig 3.5- site 11)

3.2.4 Wetland stratigraphy and ground water Characteristics.

The geology of the wetland supports water accumulation and complex ground water movements. The wetland is positioned in a topographic low of a concave depression underlined by a confining layer of silt-clay and sand mix (Fig 3.6). One to two m of fibric-humic organic layers have developed over the semi-confining layer depression, and these were overtopped by 0.3 to 1 m of organic-sand mix. This layer in turn is covered by up to 0.5 m of well humified peat, apparently subject to greater oxidation and decomposition processes than the lower substrate. Sand layers pinch the confining matrix of the wetland bottom, creating channels of higher conductivity stratum, potentially allowing for larger groundwater movements in comparison to the surrounding matrix. At the outlet of the wetland, 1 m below the surface, a cobble layer was found and appears to continue down to the lakeshore. Surrounding hill slope structure is of shallow 0.1-0.15 m LFH and organic layer overlaying 1 to 4 m of sand loam soil, which lies on top of clay and clay-silt-sand matrix.

Supported by wetland geology, shallow and deep ground water fluxes constantly fed the wetland. The piezometer nests along the stream transect revealed a continuous ground water flow-through system receiving shallow and deep ground water discharge from the upper portion of the wetland (site "east") that had its focal point at the wetland's "main" site (Fig 3.7).

The hill slope–wetland piezometer transect revealed a second flow-through system internal to the wetland. This flow was created by deeper ground water discharge at the northern part of the wetland and a deeper ground water recharge zone at the southern area of the wetland (Fig 3.7). Discharge by longer flow path was also observed at sites above the wetland such as a spring at site 11 (Fig 3.5). Northern hill slopes also contributed shallow ground water flow, the magnitude of which was highly variable. Flow paths from the stream and hill slope transect indicate ground water flow from NE to SW. During the wetting up of the wetland in May 2000, ground water flow reversals occurred on the north hill slope as shallow and deep ground water coming down the stream created a WT mound in the wetland.

To further support the observed groundwater recharge-discharge behaviour within the wetland, chemical characteristics of ground water based on DOC levels and conductivities were examined (Fig 3.8). Ground water at the northern part of the wetland showed a chemical signature similar to ground water at the northern foothill. In both, water chemistry showed lower DOC levels than the southern part of the wetland and slightly lower conductivities, that would be explained if the northern part is influenced by ground water discharge. On the other hand, the southern part of the wetland has high DOC concentrations indicating it is a recharge zone receiving DOC rich flow from the wetland itself. Upstream shallow ground water changed its signature during the study period from high conductivity and lower DOC in late summer 1999, to lower conductivity and higher DOC at spring melt 2000, reflecting changes in WT elevations and water source. In late 1999, low flow increased the residence time and therefore might have elevated conductivity while low WT elevations reduced interaction with upper organic layers reducing DOC concentrations. Spring melt 2000, however, resulted in an increase in "new" water inputs (snowmelt), reducing conductivities but increasing inputs of fresh DOC leached from the upper wetland sediments and uplands LFH layer.

3.3 SPRING MELT

3.3.1 Environmental conditions controlling surface runoff generation.

A quick snowmelt at the hill slopes during spring 2000 (Fig 3.9) of relatively low SWE generated a very little stream runoff (Fig 3.4) and unlike spring 1999, failed to connect the upper tributaries to the wetland. To further understand controls on spring melt runoff and reasons for surface runoff discontinuity, examination of frost table (FT.) elevations and infiltration rates were performed. Deep and rapidly regressing FT. was observed at the hill slopes. Similarly, stream channels located downstream from the input weirs exhibited deep FT during springmelt (Fig 3.10 and 3.11). The above data suggests that the upland soils rapidly thawed and had high permeability at the upper soils, increasing greatly the soil moisture storage capacity. Furthermore, the hill slopes held infiltration volumes that were on average 4 times larger than the actual SWE amounts on the hill slopes prior to snowmelt, indicating high soil storage capacity relative to snowmelt volume. No surface flow was observed during spring 2000 from upland soils with high infiltration capacity and/or with low FT. Standing surface water and some surface runoff were observed only at the main wetland and at the ephemeral draws of input B and C weirs where a long lasting frost table existed (Fig 3.10 and 3.11). The wetland has also shown low infiltration capacities (Table 3.3), suggesting low water storage capacity. Frost table disappearance a few meters downstream of the input weirs (Fig 3.11) permitted infiltration of runoff and prevented surface runoff connectivity from the upper catchment to the wetland. Furthermore, the wetland's output zone at D weir had a patchy frost table that most likely allowed infiltration and hence increased storage capacity that may explain the lack of stream runoff at this weir.

3.4 Wetland inorganic N budget

The annual inorganic N budget is presented in Table 3.4, summarizing the role of hydrologic N exports. Precipitation dominated NO_3^--N inputs (>90%) while shallow and deep ground water inputs were small. Although deep seepage was the dominant non-meteoric outflow, there was little NO_3^--N export via this flow path. NO_3^--N export via shallow ground water, on the other hand, dominated outputs (>99%) and represented 50% of total inputs.

Although the large uncertainties in ground water estimates make it difficult to determine if the wetland was a sink or a source of NO_3^--N, it suggests low NO_3^--N retention. In addition, an increase in NO_3^--N exports over a short period during spring, indicts that the wetland was a large NO_3^--N source at that time.

The wetland annual budget indicated very little NH_4^+-N retention, and a potential to be a NH_4^+-N source. In contrast to the NO_3^--N budget, precipitation was not the dominant NH_4^+-N input (39%) and was exceeded by shallow and deep ground water flow (~60%). Due to low surface runoff, there was no stream input. Surface runoff, however, could be more important when upland-lake connections occur, such as seen in 1999. Deep recharge, that represents the largest non-meteoric output in the water budget, is also potentially a very large NH_4^+-N export due to high NH_4^+-N concentrations. This export potential is as large or larger than the NH_4^+-N inputs on an annual basis.

3.5 Transportation and cycling of Inorganic N by Internal and external Hydrological fluxes and processes

3.5.1 Inorganic N in Atmospheric fluxes

Snow pack and rain showed a similar range of variations in inorganic N concentrations. However, volume weighted means varied considerably among the two forms. During the study period, rainfall NO_3^--N concentrations ranged from below detectable levels to over 700 µg/L with a volume-weighted mean of 223 µg/L. NH_4^+-N concentrations ranged from below detectable levels to about 1000 µg/L with volume-weighted mean of 442 NH_4^+-N. Shortly before spring melt, snow pack NO_3^--N concentrations were similar to rain with a mean of 189 μg/L. However, snow pack NH_4^+-N concentration was much lower than rain with a mean of 88 μg/L.

3.5.2 Spatial and temporal variations in N concentrations of ground and pore water

During the study period, NO_3^--N concentrations ranged from below detection limit to about 50 μg/L NO_3^--N in shallow ground water inputs from the upper catchment and surrounding hills slopes and deep ground water discharge and recharge from the wetland (Fig 3.12). These stable nitrate levels were lower than mean precipitation, and were not related to changes in WT elevations, or recharge-discharge processes.

Wells however, showed large spatial and temporal variability in NO_3^--N concentrations (Fig 3.13). Low concentrations were observed from summer 1999 to early spring 2000. With rewetting in mid-spring 2000, a substantial one to two order increase in concentrations of above ground water was observed. NO_3^--N levels, however, did not increase uniformly among wells. The wetland sites that experienced the least WT draw down and stayed saturated the longest (e.g.,, site "main"), had lower NO_3^--N levels, similar to concentrations of wells at uplands (data not shown in graph). In contrast, wells at the wetland that experienced large WT fluctuations had much higher NO_3^--N concentrations (e.g., sites "south" and "east").

Samples from groundwater wells represent integrated water chemistry from shallow to deep depths. Therefore, the observed nitrate increase can not be traced to contributions from a specific depth or a soil layer. Analysis of WT fluctuations, oxidation zone and surface pore water at selected sites provided further understanding of the source of high NO_3^--N .

Based on metal rods, oxygen was progressively introduced to the upper soil layers of all sites following the rapid draw down of the water table during the growing season of 1999 (Fig 3.14). Oxidation of metal rods was not observed within 30 cm of the WT, indicating that the capillary fringe was at least this high and prevented deeper oxidation in the wetland soils.

A consistent capillary fringe of about 30 cm above the water table prevented deeper oxidation in wetland sites. Differences in WT elevations among sites resulted in differences in oxidation depth, dropping to only 20 cm in the "wetland main" site where well NO_3^--N levels remained relatively low. Oxidation depth exceeded 40 cm for the wetland east, south and edge sites where well NO_3^--N levels were high. The oxidation zone in hill slopes rapidly dropped to below 80 cm in depth, however NO_3^--N levels remained low in wells of these sites.

Examination of two wetland sites has shown that although temporal fluctuations in inorganic N of shallow pore water were similar, concentrations differed substantially between the two sampling sites.

Pore water NO_3^--N levels increased rapidly following WT draw down at the wetland "south" site, an area affected by rapid WT drop and relatively early and deeper oxidation zone (Fig 3.15A). At this site, shallow seepers of 10, 20 and 40 cm responded similarly, showing NO_3^--N levels as low as ground water (between detection limit and 60 μg/L) at the beginning of summer when WT was at or near surface, and rapid increase in NO_3^--N levels of up to 700 μg/L as the water table fell. Nitrate concentrations of pore water at 60 cm depth demonstrated the delayed response of the deeper sediments to WT draw down, initially declining to detection limit and rapidly increasing only after WT fell below 40 cm (Fig 3.15A). At the end of the summer, all seepers dried out. In early spring re-wetting of the surface due to perched water table over ice was associated with high NO_3^--N concentrations (> 5000 μg/L NO_3^--N) in the 10 cm seeper. Also, an increase in water table elevations during mid spring resulted in levels of about 2000 μg/L from the 60 cm seeper, indicating that high NO_3^--N concentrations found in wells at the same period of time originated from surface soils (also see the following "soils" section).

The "main" wetland site differed from the "south" site by having moderate WT fluctuations, prolonged saturation and shallower oxidation depth, leading to a comparable but much more moderate trend of pore water NO_3^--N levels (Fig 3.15B). NO_3^--N concentrations at the shallow seepers of 10 and 20 cm were similar in range and temporal behavior, displaying low NO_3^--N levels of less than 10 μg/L NO_3^--N at the beginning of summer when WT was at the surface, and a moderate increase in NO_3^--N levels to about 100 μg/L when the water table fell. Similarly, deep pore water of 40 cm and 60 cm at the south site had a delayed response to WT draw down, increasing NO_3^--N levels only after the WT levels fell below 20 cm.

In contrast to NO_3^--N levels, NH_4^+-N concentrations in shallow and deep ground water were spatially variable but temporarily relatively stable (Fig 3.16). Low levels were constantly observed at the northern wetland edge, an area receiving water from mineral hill slopes via shallow sub-surface flow and ground water discharge. High concentrations of NH_4^+-N (up to 9000 μg/L) were observed at the southern part of the wetland, an area receiving ground water recharge from deep organic sediments. The recharge area of the wetland, responsible for large NH_4^+-N losses, was assumed to be able to transport the positively charged NH_4^+-N despite the possibility of high cation exchange capacity of the wetland organic soils (Foth 1990). This assumption was based on the findings of very high NH_4^+-N levels at the mineral layers at depths of 1 and 2.5 m below the organics (Fig 3.16-wetland south), suggesting transport away from the organic layers. Furthermore, deep mineral layers not in a recharge zone did not show as high NH_4^+-N levels (Fig 3.16-wetland main).

Concentrations of NH_4^+-N in wells were also spatially variable but temporarily relatively stable (Fig 3.13). Lower concentrations were constantly observed at the northern wetland edge while higher concentrations were measured at the southern part of the wetland. However, as the water table rose to the surface in spring 2000, concentrations in most wells decreased.

Ammonium concentration in shallow seepers (10-60 cm), showed a non-linear inversed behaviour to nitrate, especially noted in the "south" site (Fig 3.17 and Fig 3.18). An initial increase in concentrations of NH_4^+-N to levels of up to 2000 μg/L was followed by a sharp decrease to levels of about 200 μg/L, with water table draw down (Fig 3.17) and an increase in oxidation depth (Fig 3.14).

3.5.3 Inorganic N characteristics of ephemeral localized surface runoff during spring

During spring melt, localize stream runoff at the upper catchment revealed some of the characteristics of N movements in surface runoff at the upper tributaries and the potential inputs to shallow subsurface flow. There was large variation in NO_3^--N concentrations of surface runoff, with the highest values occurring during the initial days of spring melt (Fig 3.19). Comparison to snow pack concentrations showed that runoff at the input B weir, had a much lower volume-weighted mean with 37 μg/L NO_3^--N, while flow at the input C weir showed a higher value than snow, with 513 μg/L NO_3^--N.

Despite a lack of surface runoff at the output zone of D-weir, values of up to 6000 μg/L NO_3^--N were observed in the shallow subsurface flow just below the weir, suggesting high nitrate output by shallow subsurface flux (Fig 3.19).

Unlike nitrate, NH_4^+-N concentrations of surface runoff were less varied than snow pack concentrations and ranged from below detectable levels to about 100 μg/L NH_4^+-N (Fig 3.19). Mean concentrations of runoff from B and C weirs were lower than snow pack concentration, with volume-weighted means of 43 μg/L and 17 μg/L respectively. NH_4^+-N levels in the ephemeral subsurface flow below D-weir were similar to the levels found in surface runoff of the input weirs.

3.6 Soils as N sources: Spatial and Temporal Characteristics

3.6.1 Soil moisture and temperature

Both wetland and surrounding hill slopes showed similar and significant ($P<0.001$) seasonal changes in soil moisture levels of the organic layers (Fig 3.20A). High moisture levels were measured at the beginning of spring 1999 followed by lower soil moisture during the growing season and an increase in soil moisture in winter and early spring. Wetland sites were significantly wetter (volumetric = 11% to 99%, mean 63%) than hill slopes (1% to 65% mean 18%)($P<0.001$). The mineral soils of the hill slopes had much lower soil moisture than the organic soils, averaging 12% (data not shown).

Over the growing season soil temperatures showed weekly to bi-weekly cycles with a magnitude of ± 4 °C (Fig 3.21). On average, soil temperatures increased slightly from May to August and dropped thereafter. During the growing season the upper 7 cm of the wetland soil was constantly and significantly ($P < 0.001$) warmer than the wetland edge, ranging from 2.5 °C warmer at the beginning of the season to 0.5 °C at the end of August. This difference may relate to the lower vegetation cover at the wetland in comparison to the wetland edges. At the end of the growing season and during the winter, the wetland and the wetland edge showed similar temperatures. During the winter both wetland and hill slope showed relatively constant near-zero temperatures, most likely due to the insolating thermal properties of snow cover (Jones 1999). Sharp temperature drops during late winter in the wetland might be related to soil ice development, usually occurring in greater proportion in organic soils (Nyberg et al. 2001), and/or related to delayed response to cold peaks. The depth of the upper frozen layer was not measured directly, but was estimated to have reached 100cm in depth based on various observations such as the behaviour of piezometers, seepers and wells.

3.6.2 Seasonal changes in soils extractable NH_4^+-N and NO_3^--N.

During the study period, the presence of extractable nitrate in the hill slopes was at or below detection limits at the mineral layer (data not shown) as well as the forest floor LFH (Fig 3.20B for organics, mineral soil data not shown). Cores extracted within the wetland ("wetland main, south and east") had significantly higher extractable NO_3^--N than those from the hill slopes ($P < 0.001$), and were seasonally variable. Wetland edge showed intermediate values between the wetland sites and the hill slopes. Extractable NO_3^--N decreased during the growing season and increased at the end of the growing season when oxidation levels were the greatest. Finally, NO_3^--N decreased over winter to near detection limits after spring melt (Fig 3.20B).

The apparent effect of vegetation uptake and WT fluctuations on NO_3^--N is further illustrated in Fig 3.22. During the non-growing months, appreciable soil extractable NO_3^--N was only observed when soil moisture levels were above 35% and below 95%. However, during the growing season no relationship was observed, as NO_3^--N levels were low regardless of soil moisture levels.

In the organic layers of the hill slope soils, soil moisture levels were usually below 40% and extractable NO_3^--N was very low (Fig 3.22), indicating that only the wetland sediments were sources of NO_3^--N to surface waters.

Seasonal trends in extractable NH_4^+-N were similar between hill slopes and wetland sites ($P>0.1$). These trends showed a general decrease during the growing season regardless of location, and did not show a significant increase at the end of the growing season, but did increase at all sites during the winter (Fig 3.20C). After spring melt 2000, only the "wetland main" site had high NH_4^+-N levels while the hill slopes showed low levels. Extractable NH_4^+-N was near or below detection during all seasons in the mineral layers of the hill slopes and wetland edge (data not shown).

During the non-growing season, extractable NH_4^+-N in the wetland was positively correlated to soil moisture (R^2=0.4) (Fig 3.23). During the growing season, however, there was no clear effect of soil moisture on soil extractable NH_4^+-N. The highest NH_4^+-N concentrations were produced during the non-growing season in soils with volumetric moisture levels above 80%. At the hill slopes, soil extractable NH_4^+-N did not show an observable trend or correlation to soil moisture, regardless of season (Fig 3.23).

Table 3.1 Water balance of the wetland from 01 June 1999 to 31 May 2000. Fluxes are measured in mm +/- error.
* Interception was assessed based on literature values.

INPUTS		mm/yr	error	% of inputs
	Stream	0	+/- 0	0
	Rain	190	+/- 17	47
	Snow	69	+/- 11	17
	Shallow ground water	128	+/- 355	32
	Seepage	15	+/- 29	4
	TOTAL	402	+/- 357	
OUTPUTS				% of outputs
	Stream	0	+/- 0	0
	Shallow ground water	4	+/- 9	1
	Deep recharge	40	+/- 90	6
	PE T	558	+/- 56	86
	Interception *	32	+/- 5	5
	Sublimation	18	+/- 3	3
	TOTAL	652	+/- 106	
	ABSOLUTE (in-out)	-249	+/- 372	
	Measured change in storage by WT	-17	+/- 7	
	Unexplained residual	-232	+/- 372	

Table 3.2 Snow water equivalent (SWE) measurements at the sub catchments and at the wetland. Based on two survey dates.

Year	Sampling Date	Watershed	SWE (mm)	Std error.(mm)
2000	11-Mar-00	A	53	2.5
	11-Mar-00	B	51	3.7
	11-Mar-00	C	35	5.7
	11-Mar-00	D	47	2.7
	28-Mar-00	D-wetland	51	6.3

Table 3.3 Rates of moving hydrologic heads (initial infiltration) in three locations on 31 March 00 : hill slopes (aspen and conifer) and the wetland. The data is based on an infiltration experiment done prior to snowmelt. Five data points from each transect show the high but variable rates of moving heads. Variability may indicate the role of macropores on quick infiltration rates.

Rates of moving heads (cm/min)		
Hill slopes Aspen	**Hill slopes Conifer**	**Wetland**
4.5	38.2	0
72	33.6	0
0.5	6.3	0
2.8	84.0	0
11.1	24.0	0

Table 3.4 Annual NO_3^--N and NH_4^+-N budgets for the wetland from: 01 June 1999 to May 29 2000. Wetland area equals 1.5 ha

INPUT (kg/yr/ha)	kg/yr		error	% of inputs
Stream	0.00	+/-	0.00	0
Rain	0.32	+/-	0.04	66
Snow	0.13	+/-	0.08	27
Shallow ground water	0.02	+/-	0.03	5
Seepage	0.01	+/-	0.03	3
TOTAL	0.48	+/-	0.09	
OUTPUT(kg/yr/wetland)				**% of outputs**
Stream	0.00	+/-	0.00	0
Shallow ground water	0.25	+/-	0.48	99
Deep recharge	0.00	+/-	0.01	1
TOTAL	0.25	+/-	0.48	
ABSOLUTE (in-out)	0.23	+/-	0.49	
RELATIVE	48%			

INPUT (kg/yr/ha)	Kg/yr		error	% of inputs
Stream	0.00	+/-	0.00	0
Rain	0.42	+/-	0.05	39
Snow	0.03	+/-	0.04	3
Shallow ground water	0.54	+/-	0.77	50
Seepage	0.10	+/-	0.19	9
TOTAL	1.09	+/-	0.77	
OUTPUT (kg/yr/wetland)				**% of outputs**
Stream	0.00	+/-	0.00	0
Shallow ground water	0.39	+/-	0.57	16
Deep recharge	1.98	+/-	4.47	84
TOTAL	2.37	+/-	4.51	
ABSOLUTE (in-out)	-1.29	+/-	4.57	
RELATIVE	-118%			

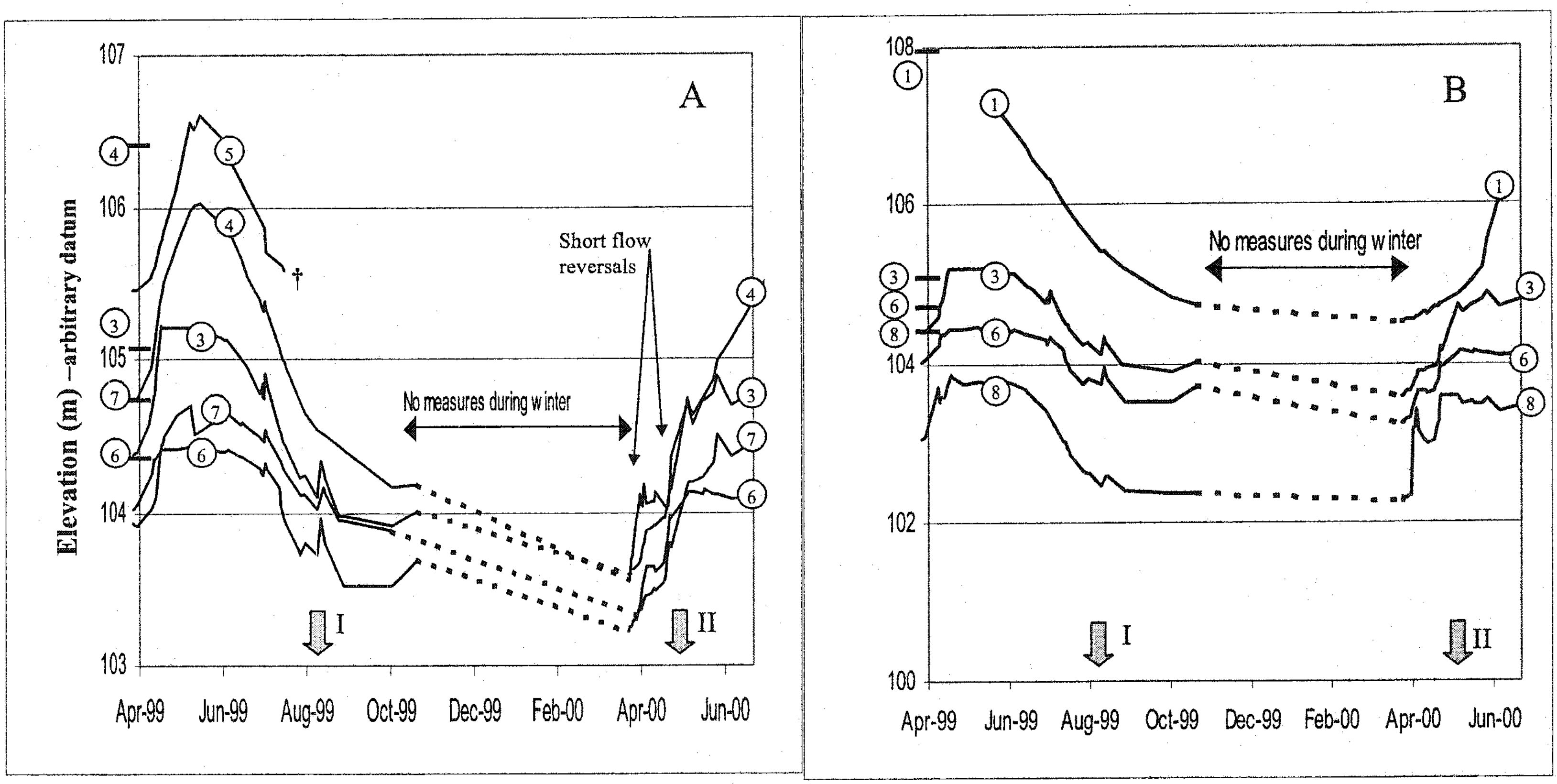


Figure 3.1 Seasonal changes in water table (WT) elevations from the (**A**) north-south transect (aspen hill slope to wetland) and (**B**) the east-west transect (upstream-downstream). Wells and piezometers location: upstream A (1), wetland east (3), wetland edge (4), aspen hill slope, (5) wetland main (6), wetland south (7), and wetland outlet A (8). Dates for flow diagram in Fig 3.5 are indicated by arrows ( ). **I**-low WT. **II**- high WT. Ground elevation for each well is marked on the left side of y axis. Ground level for well 5 = 108.55; † well dried out

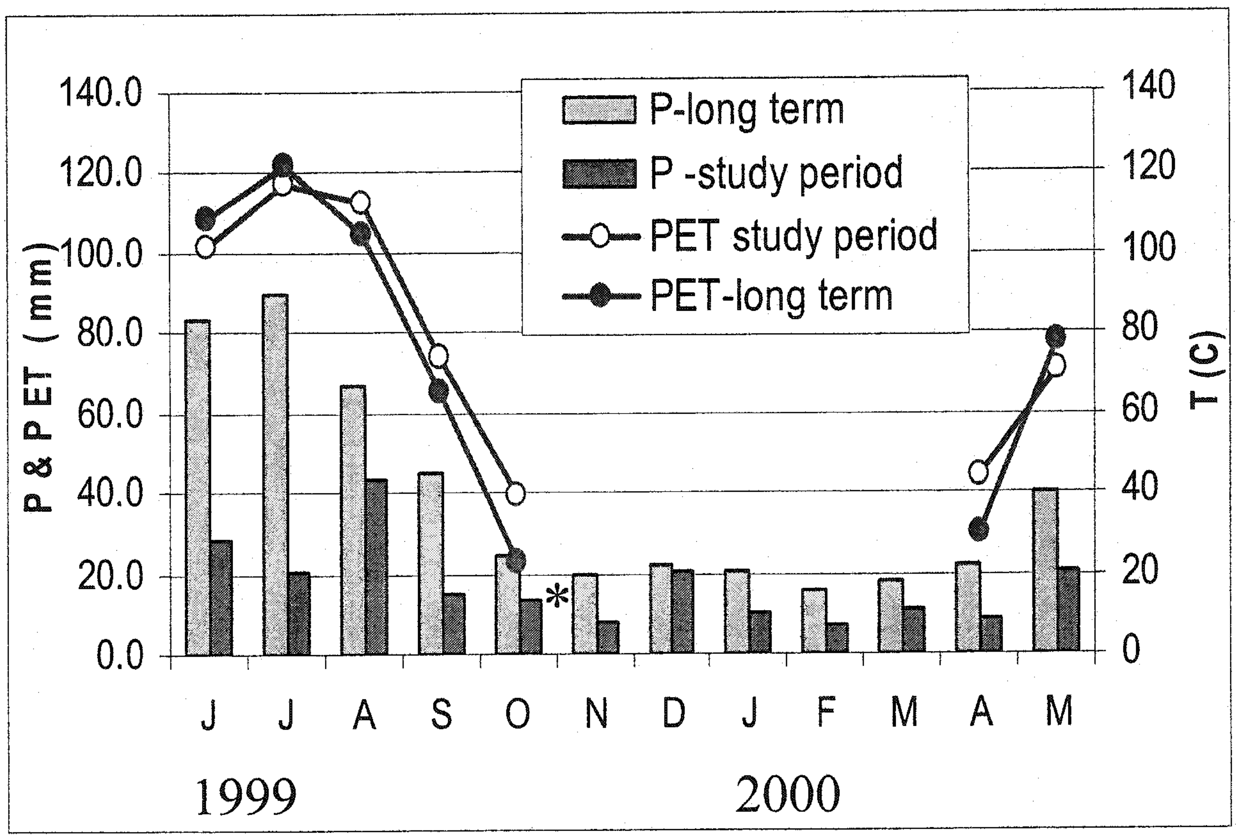


Figure 3.2 Atmospheric fluxes: study period vs. 30 years mean. Monthly values for precipitation (P), potential evaporation (PET) based on Thornthwaite model, and temperatures (T). * Data missing due to equipment failure were estimated by nearby Lac la Biche meteorological station..

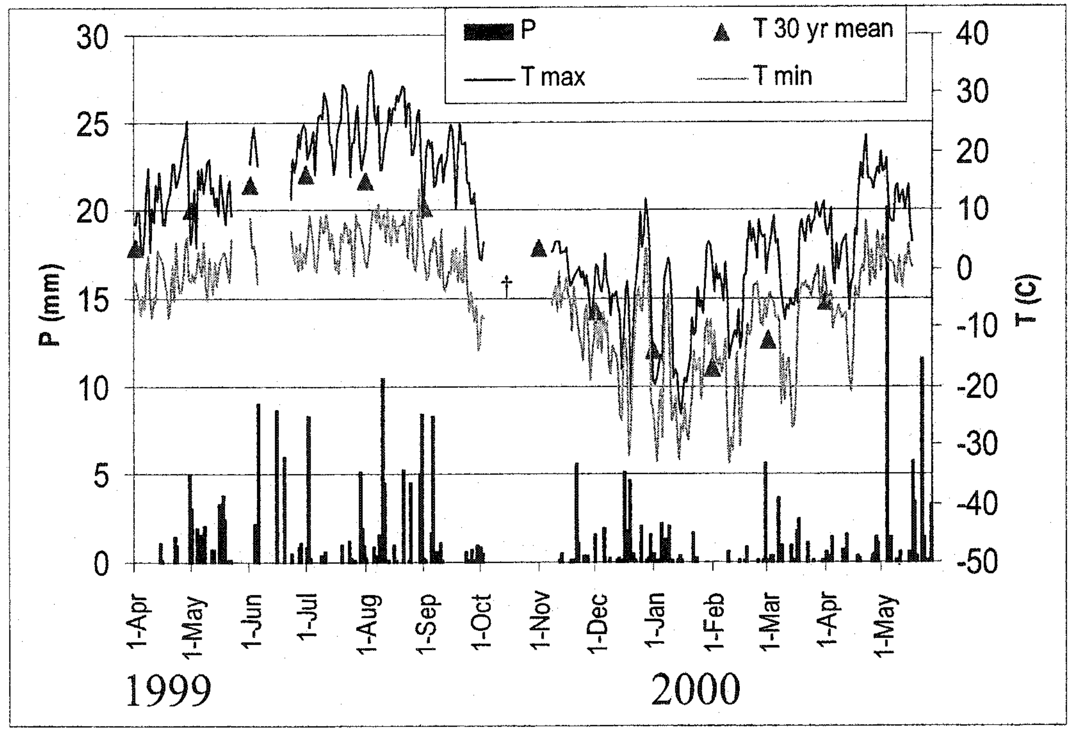


Figure 3.3 Daily inputs of precipitation (rain and snow) and minimum and maximum temperatures (T °C) during the study period. Thirty year mean monthly T is given for comparison. † Data missing due to equipment failure.

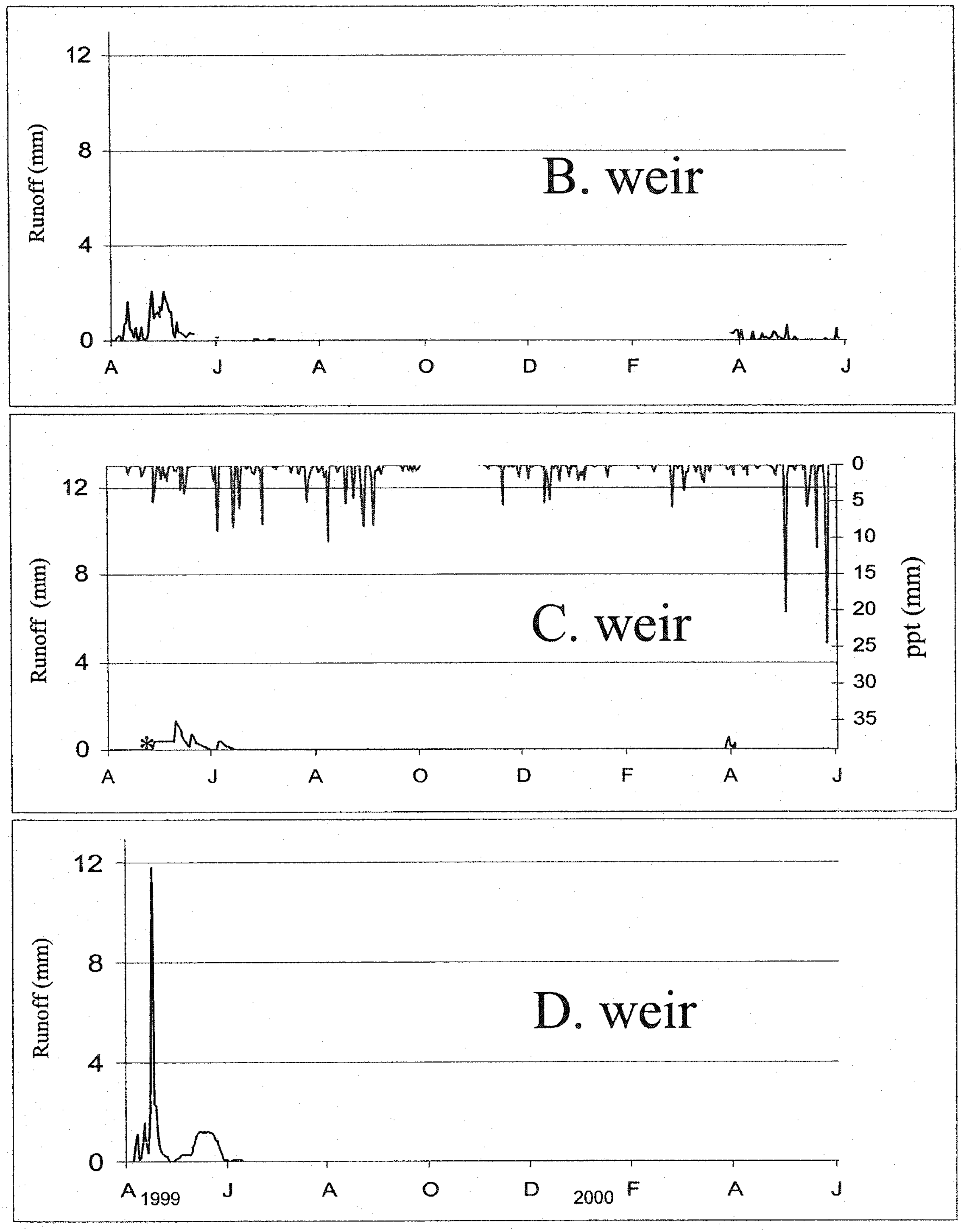


Figure 3.4 Surface water runoff (mm d^{-1}, based on wetland area) at the weirs and daily precipitation values. Study period - 01 June 1999 to 31 May 2000. Weirs total water runoff in mm relative to wetland area: **B weir**: Spring 1999 (01 April to 20 May) Q= 15 mm; Spring 2000 (29 Mar to 30 May) Q= 6.2 mm; **C weir**: Spring 1999 (29 April 31 May) Q= 29 mm, Summer 1999 (06 June to 14 Jun) Q= 2 mm, Spring 2000 (31 Mar to 05 Apr) Q= 1 mm; **D weir**: Spring 1999 Q= 49 mm, Spring 2000 (29 Mar to 30 May) Q= 0 mm.
* Indicates problems with measuring equipment.

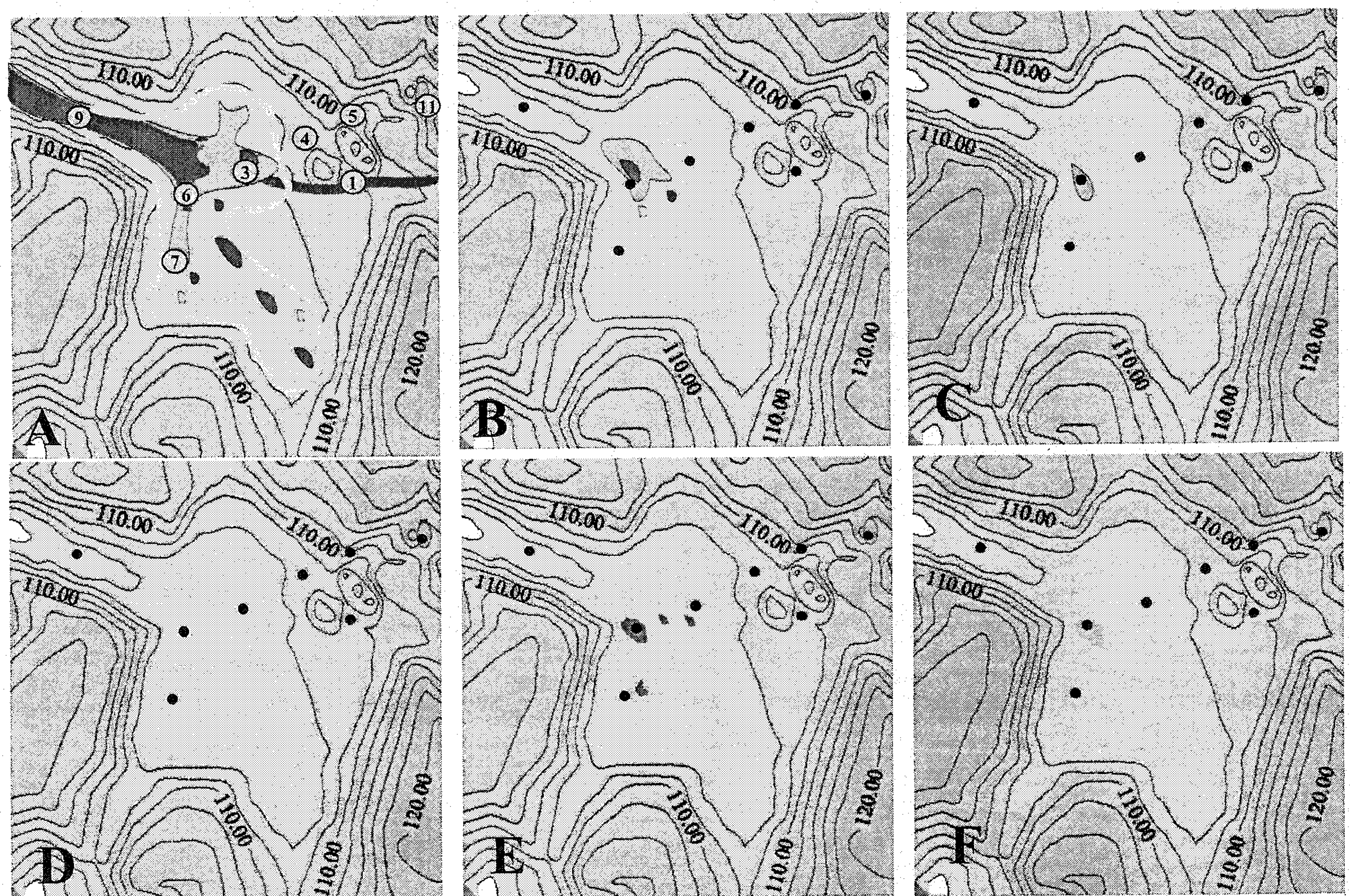


Figure 3.5 Spatial and temporal changes in surface saturation of the study wetland. Dark blue represent surface water while light blue represents saturated area. Yellow perimeter indicates surface saturation in 1998. 15 April 2000 shows surface water under perched WT (frozen soils). A) 17 May 99, B) 16 June 99, C) 02 Jul 99, D) 25 Aug 99, E) 15 April 00, and F) 29 May 2000. labeled sites names: (3) East, (4) edge, (5) h.s. aspen, (6) main, (7) south, (9) output B, and 11) h.s. spring

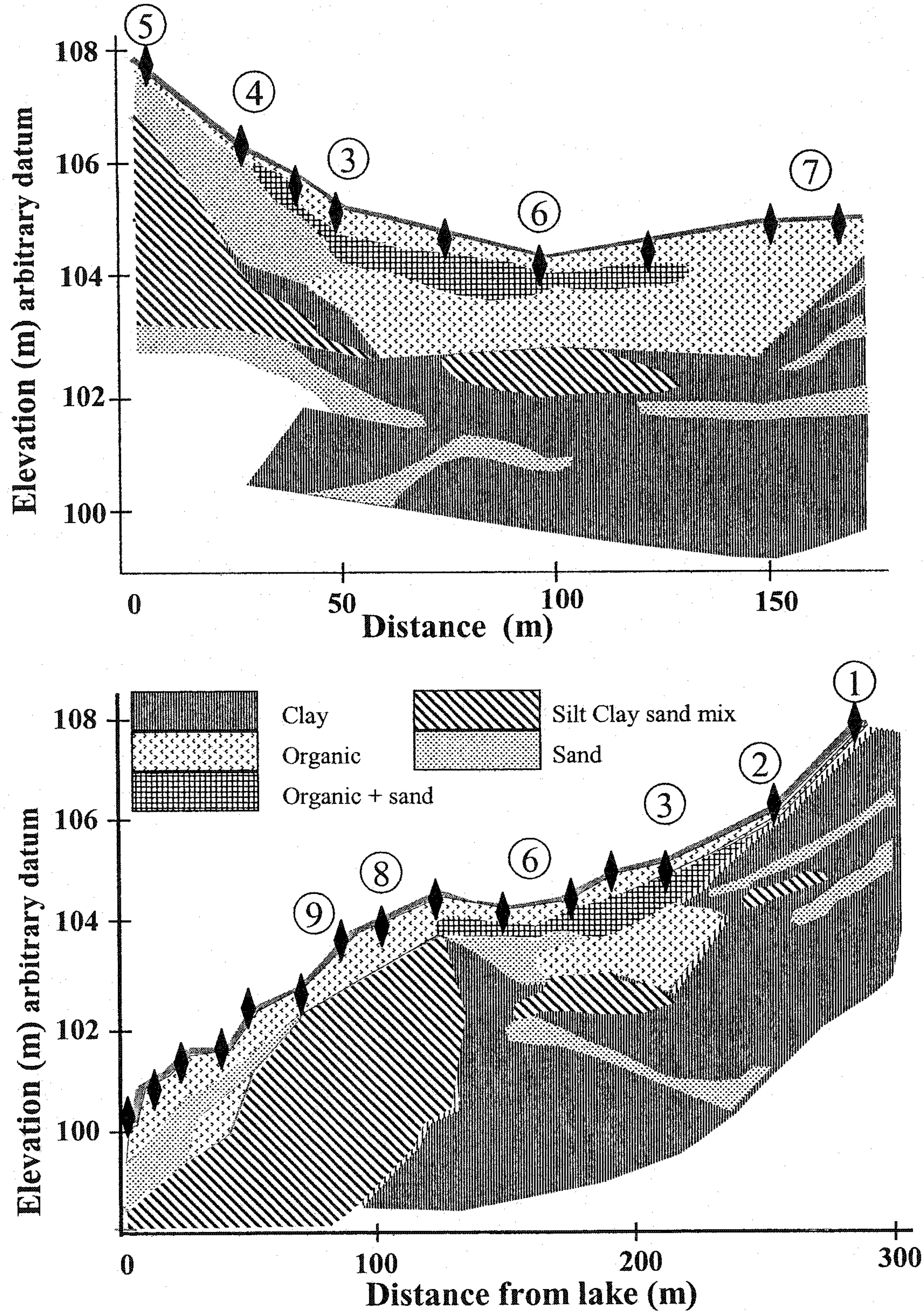


Figure 3.6 Wetland stratigraphy. Diamonds (♦) represent deep coring locations (1.5 to 5 m). 1) upstream A, 2) upstream B, 3) wetland east, 4) wetland edge, 5) aspen hill slope, 6) wetland main, 7) wetland south, 8) wetland output A, 9) wetland output B.

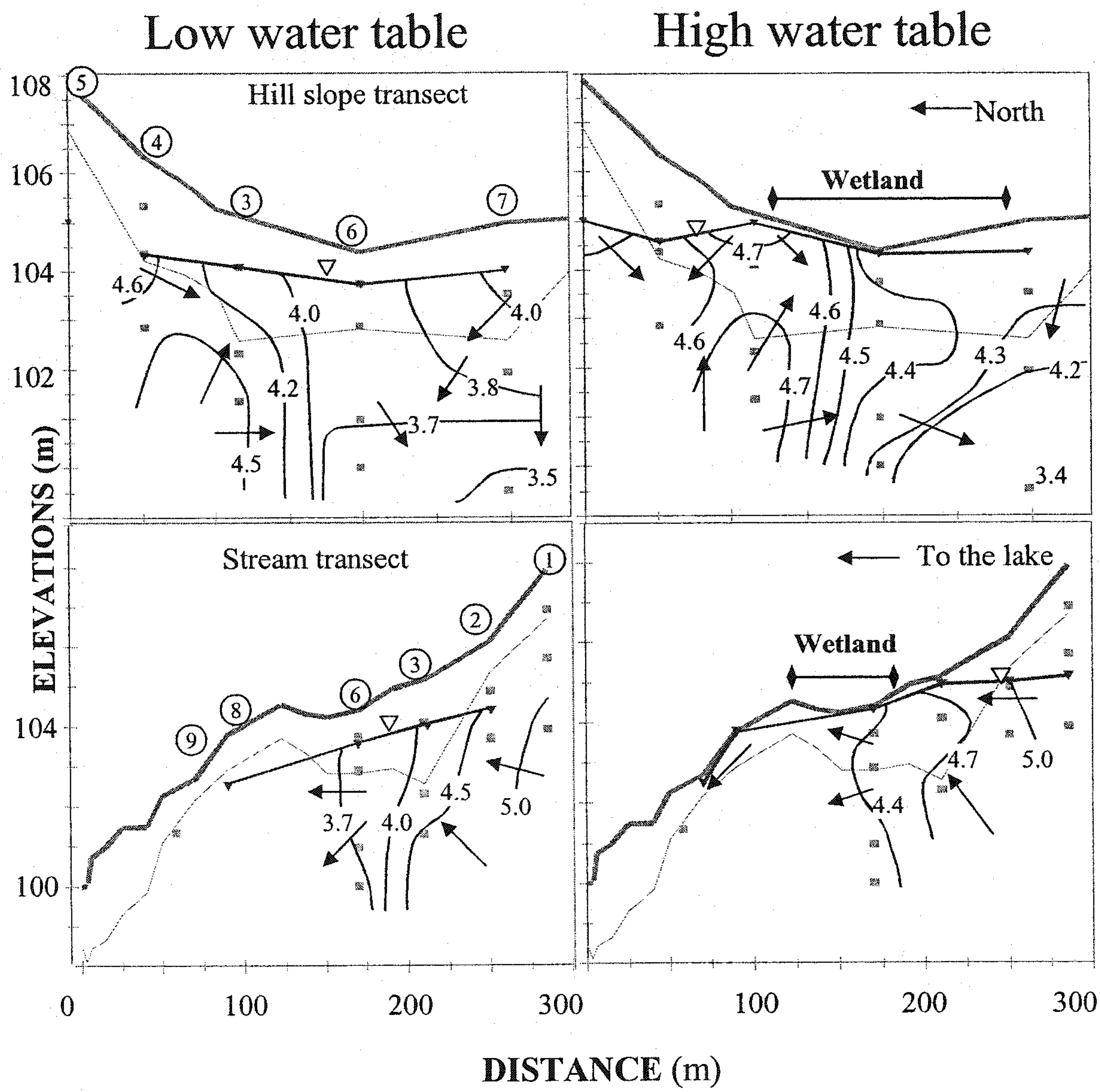


Figure 3.7 Flow diagrams during low WT elevations (10 Aug 99) and during high WT elevation (06 May 00) at both transect. The hill slope transect indicates discharge recharge conditions, while the stream transect (to the lake) indicates a flow through system. See Fig 3.1 to relate the position of the WT on the above dates to the seasonal trend in WT elevations. Numbers in Circles (9) represent wells and piezometers nests (see Fig 2.2 and 3.4)

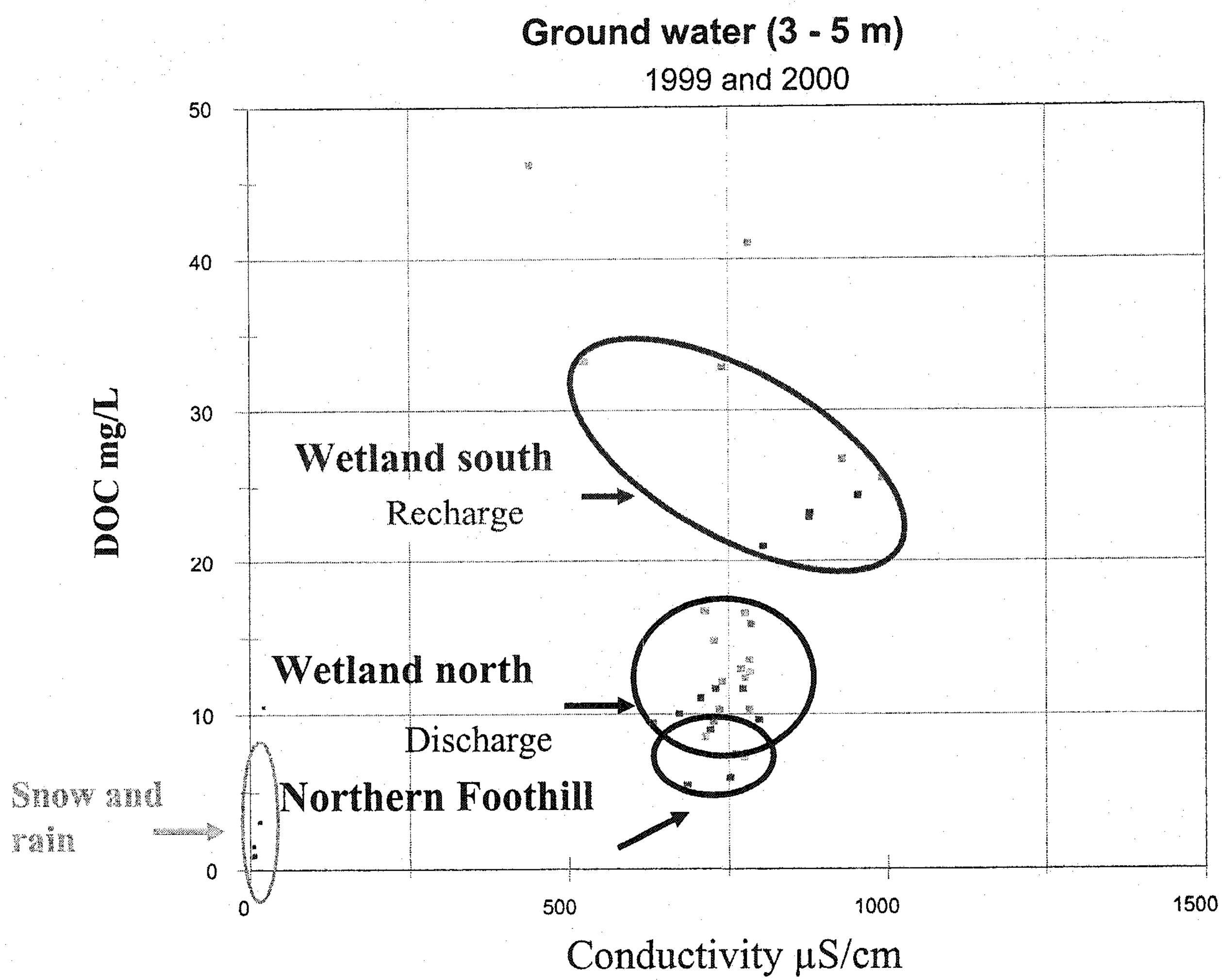


Figure 3.8 Wetland geochemical signatures of precipitation, and deep ground water (3-5m). The northern part of the wetland and foothill show lower DOC levels than the southern part.

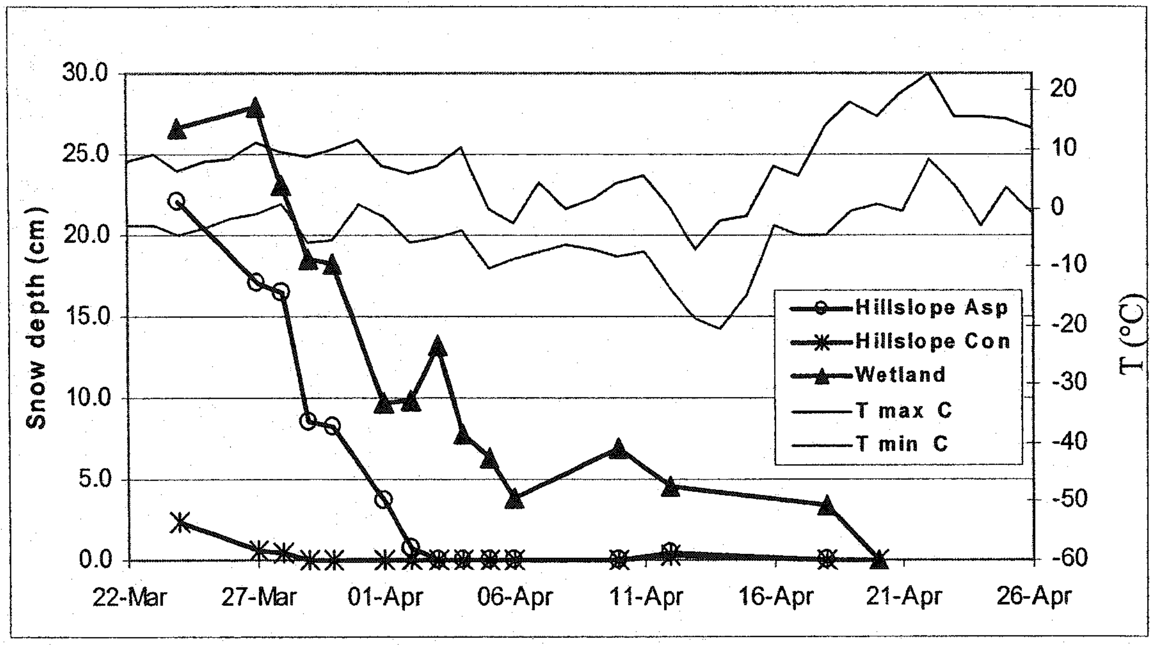


Figure 3.9 Snow depth and minimum and maximum temperatures (T) during spring melt 2000. Data were collected from two hill slope (aspen and conifer) transects and a wetland transect, each 70 m in length. Snow density on 28 March : hill slope aspen = 0.26, hill slope conifer = 0.26, wetland = 0.26 g/cm³.

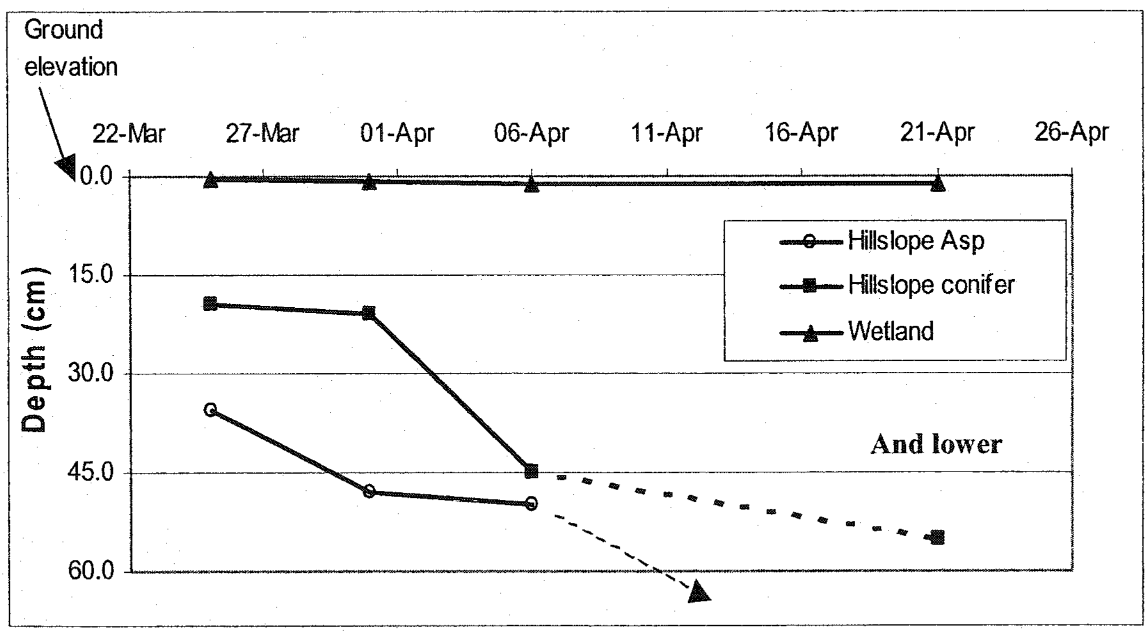


Figure 3.10 Frost table (FT) elevations during spring period. Data were collected from two hill slope (aspen and conifer) transects and a wetland transect (n=14 for each data point). Dotted line and arrows indicate that some measurements showed FT elevations greater than 50 cm depth.

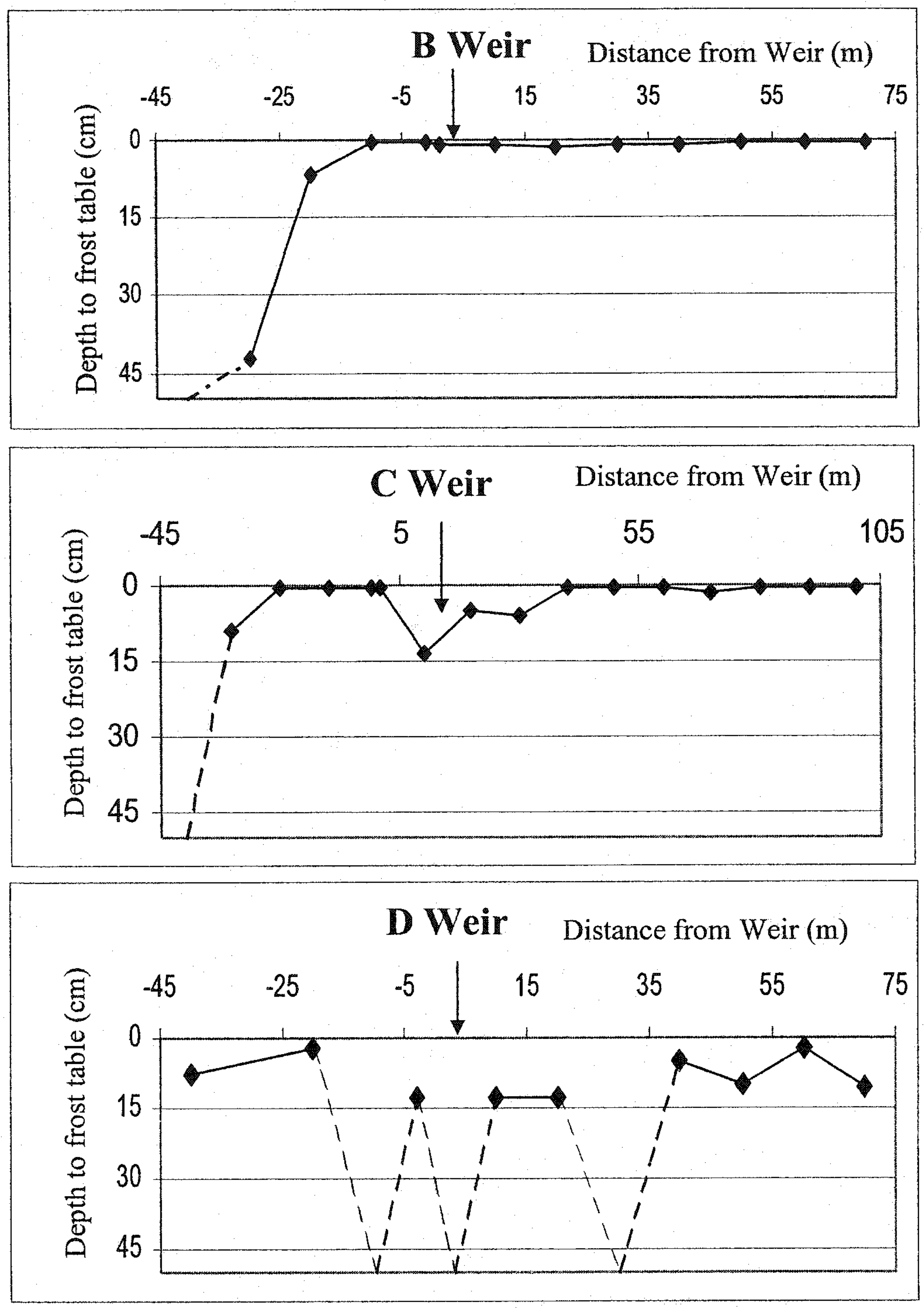


Figure 3.11 Frost table elevations upstream and downstream from the weirs on 04 April 02 (C and B weirs), and on 05 April 02 (D weir). Solid line represents frozen areas while dotted line represents areas where frost table is discontinues or greater than 50 cm depth.

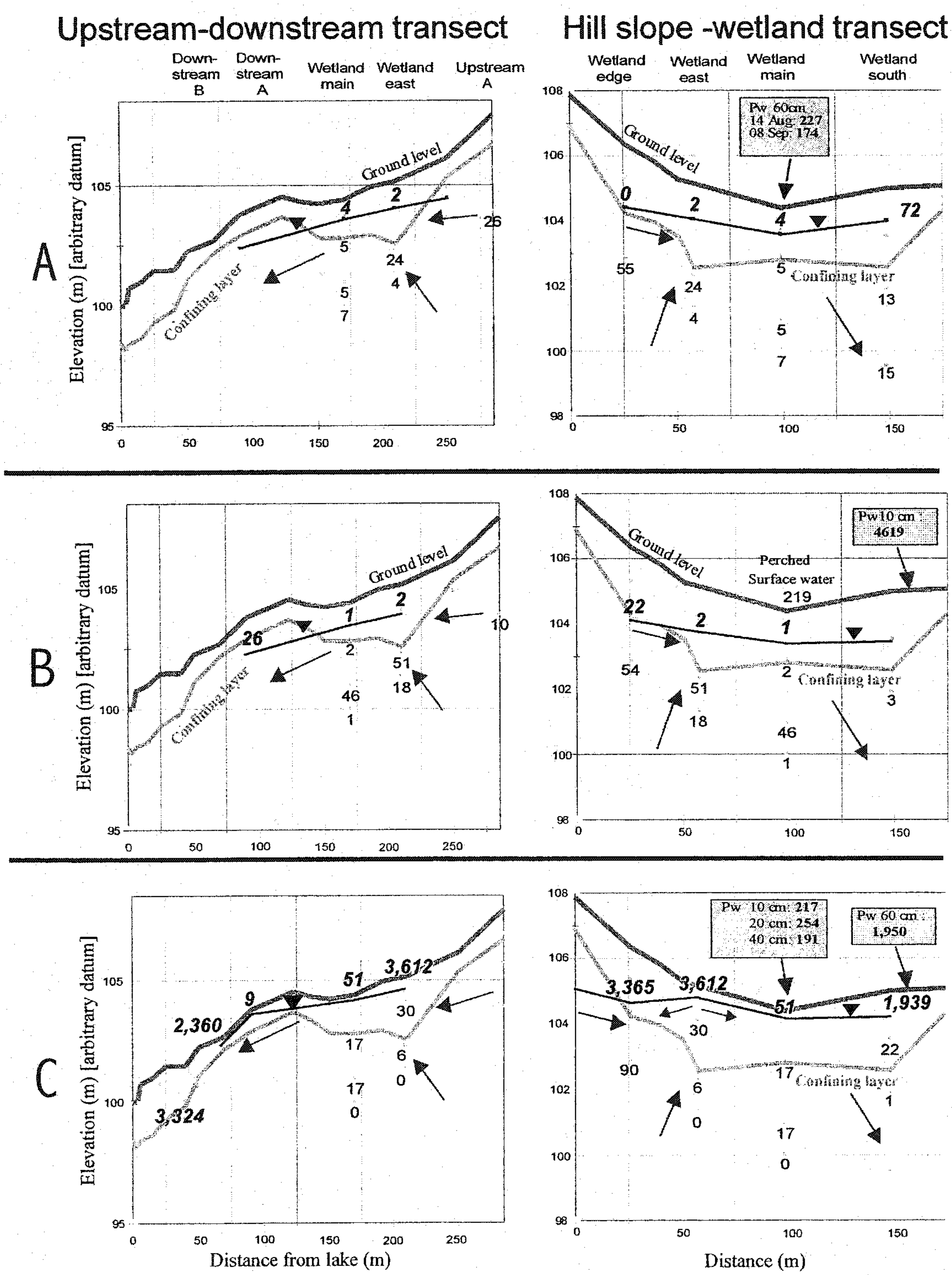
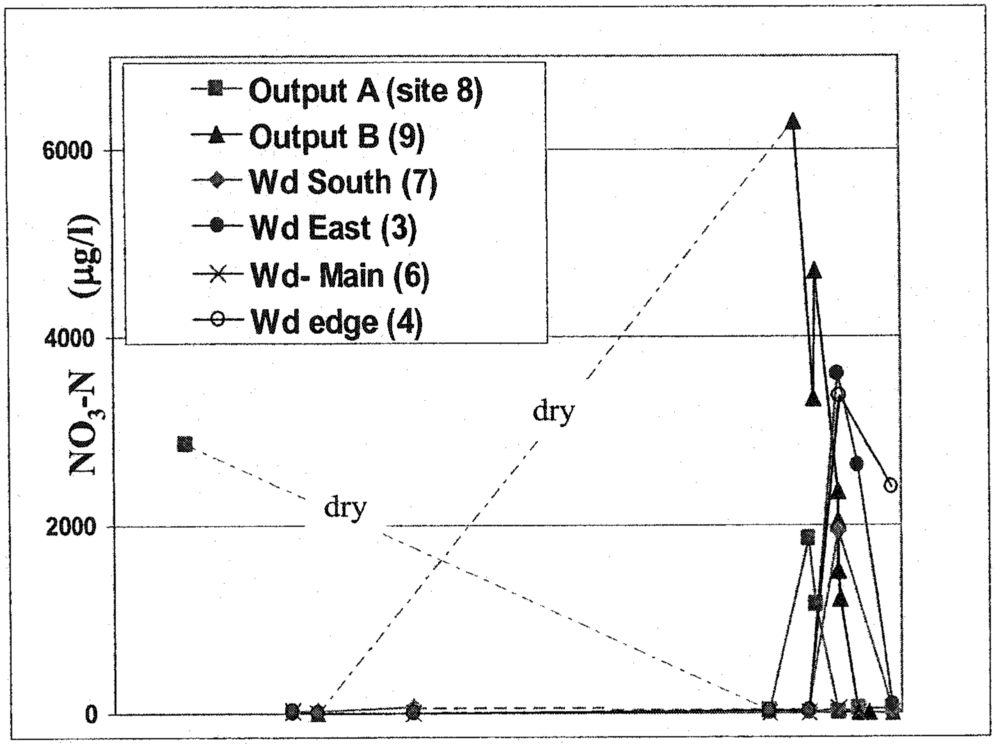


Figure 3.12 Vertical cross-sections of the wetland area showing nitrate levels in ground water on 3 representative dates: **A.** late summer (24 Aug 1999), **B.** end of winter (26 Mar 2000), and **C.** post snow melt, with high WT elevations (06 to 13 May 2000). Confining layer, piezometer slot zone (gray dots) and estimated ground water flow (arrows) are included. Numbers are nitrate concentrations in piezometers while ***Italicize bold*** numbers represent nitrate concentrations of wells. Gray boxes present information from shallow Pore Water (**PW**) seepers (10, 20,40 and 60 cm below ground level).



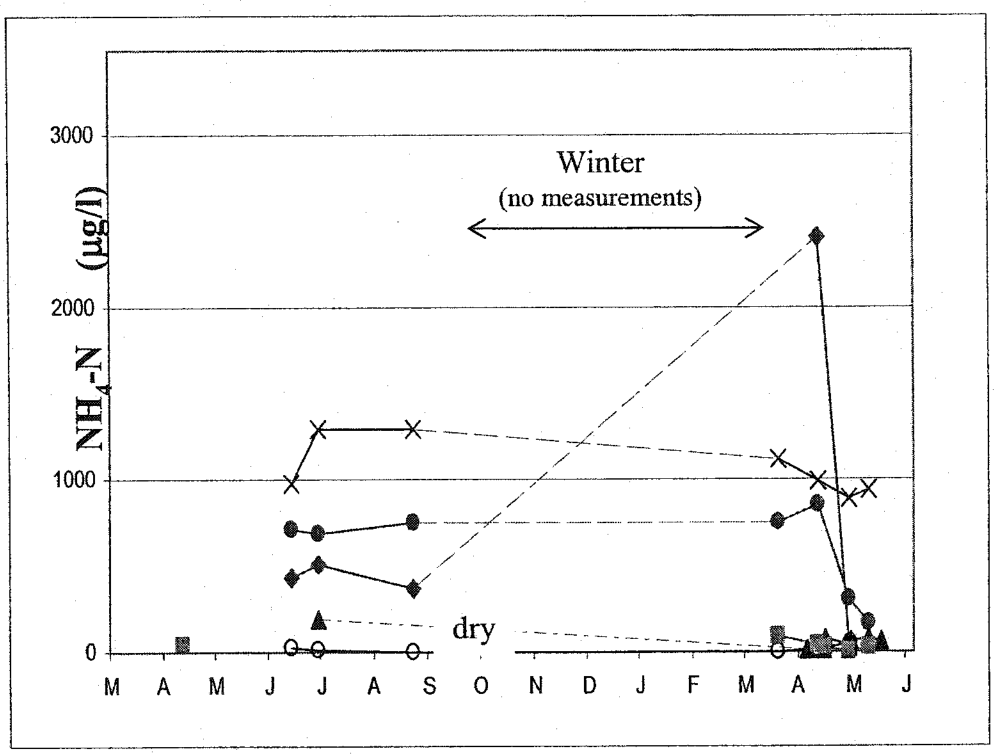


Figure 3.13 Nitrate (top) and ammonium (bottom) levels in wells over the study period.

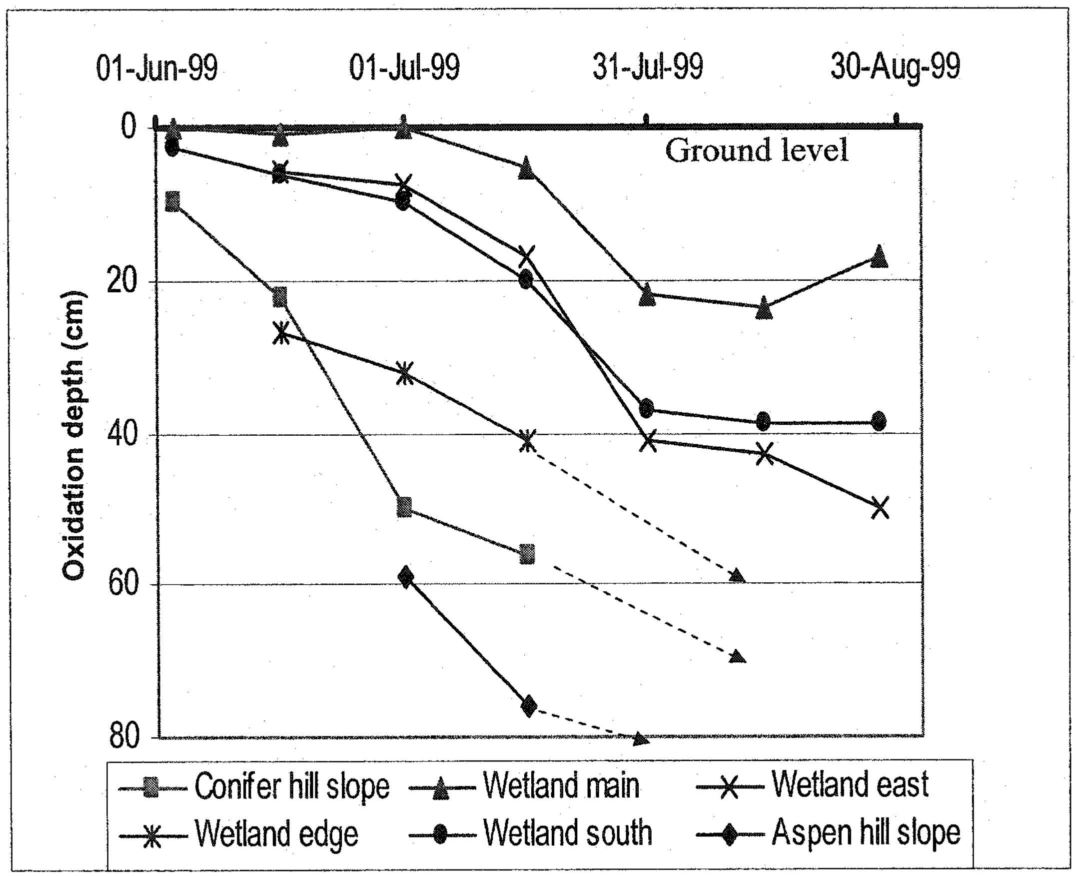


Figure 3.14 Depth of rust on iron rods indicating oxidation depth of upper soil layers in various locations in the study area. (dashed arrows indicate oxidation greater than 75 cm in subsequent measurements).

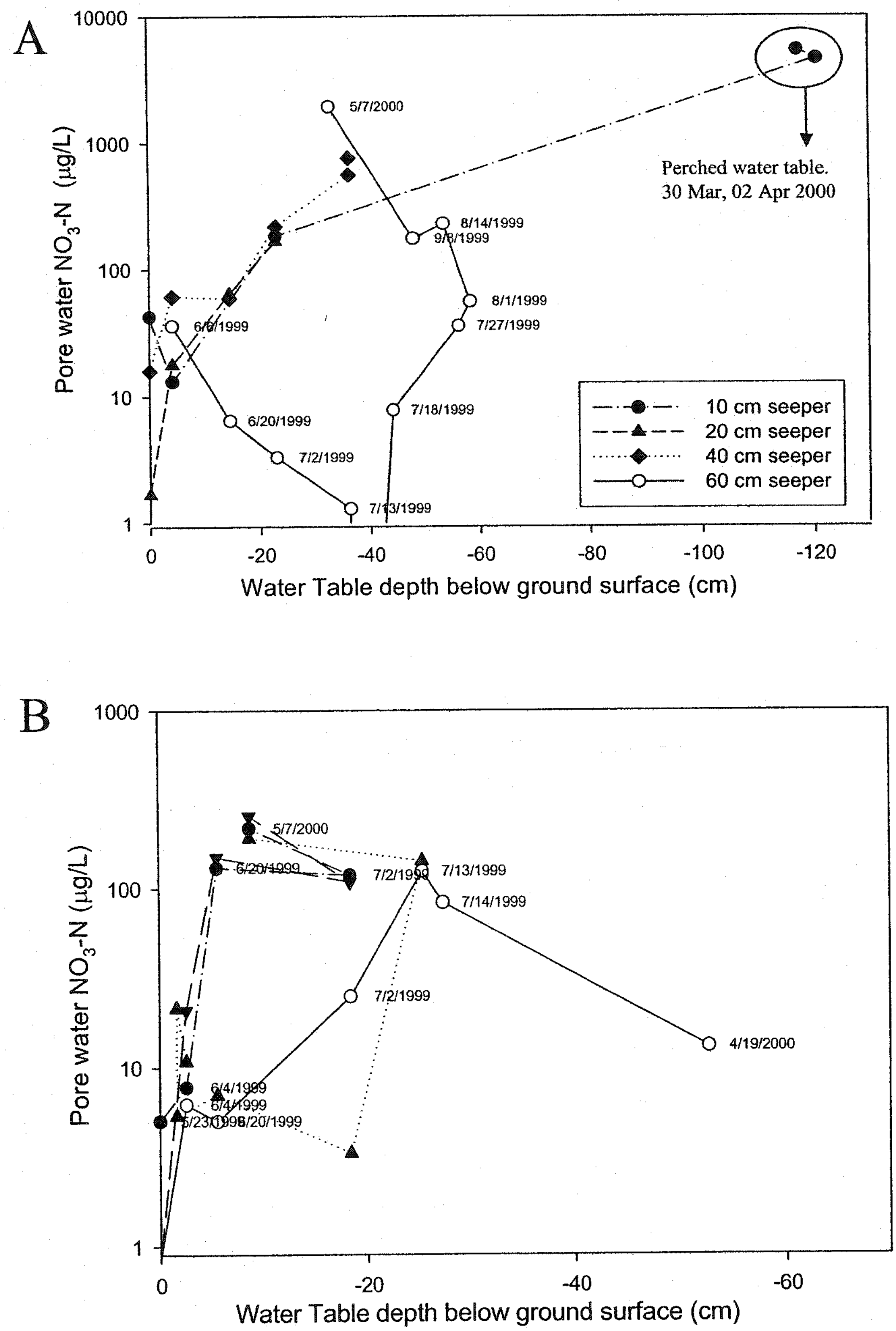


Figure 3.15 NO_3^--N concentrations in pore water as a function of water table elevations. The drier "wetland south" site (A) vs. the largely saturated "wetland main" site (B). Data collected from shallow seepers (10 to 60 cm depth).

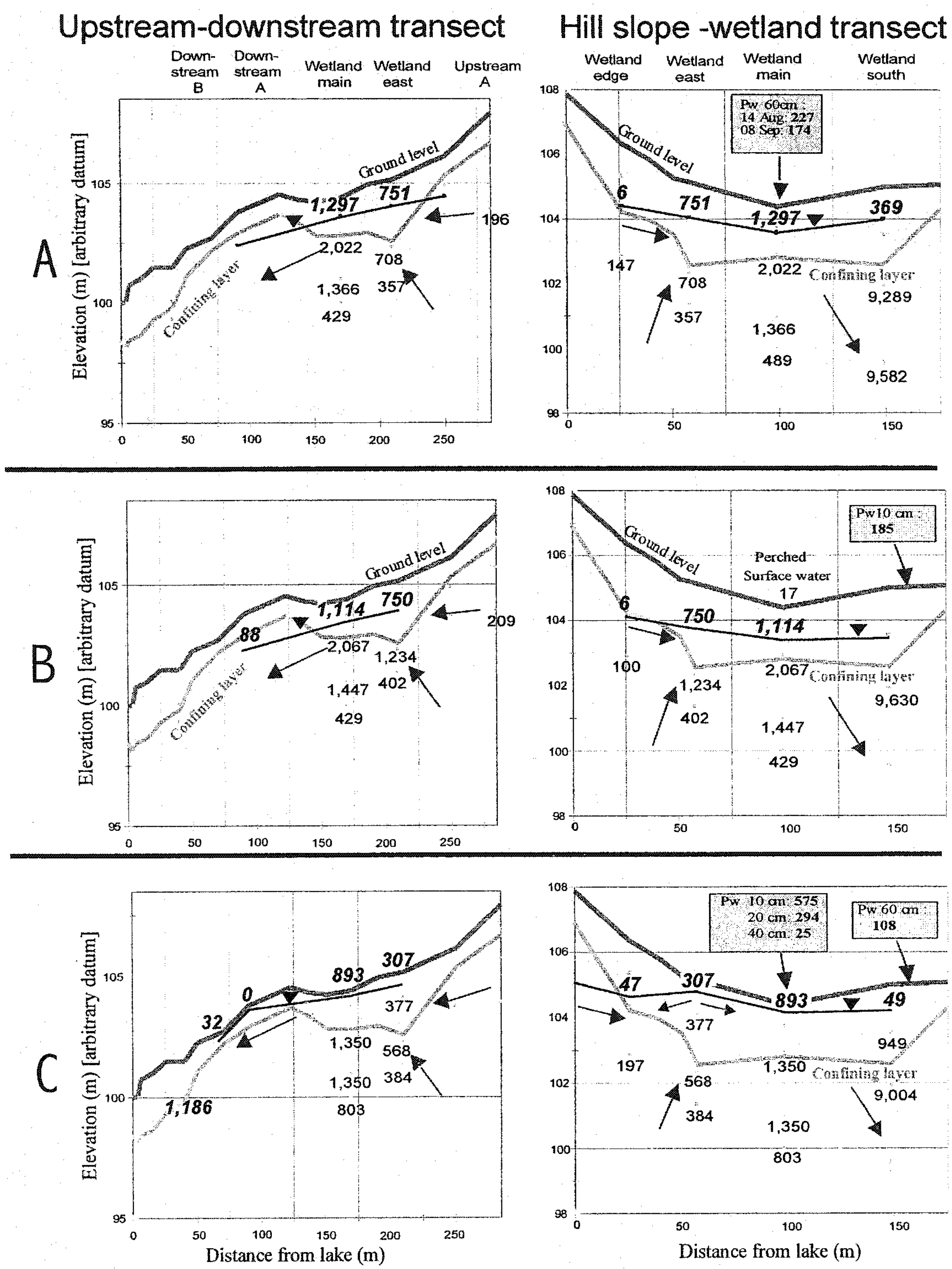


Figure 3.16 Vertical cross-sections of the wetland area showing ammonium levels in ground water on 3 representative dates: **A.** late summer (24 Aug 1999), **B.** end of winter (26 Mar 2000), and **C.** post snow melt, with high WT elevations (06 to 13 May 2000). Confining layer (gray line), piezometer slot zone (gray dots) and estimated ground water flow (arrows) are included. Numbers are nitrate concentrations in piezometers while ***Italicize bold*** numbers represent ammonium concentrations of wells. Gray boxes present information from shallow Pore Water (PW) seepers (10, 20,40 and 60 cm below ground level).

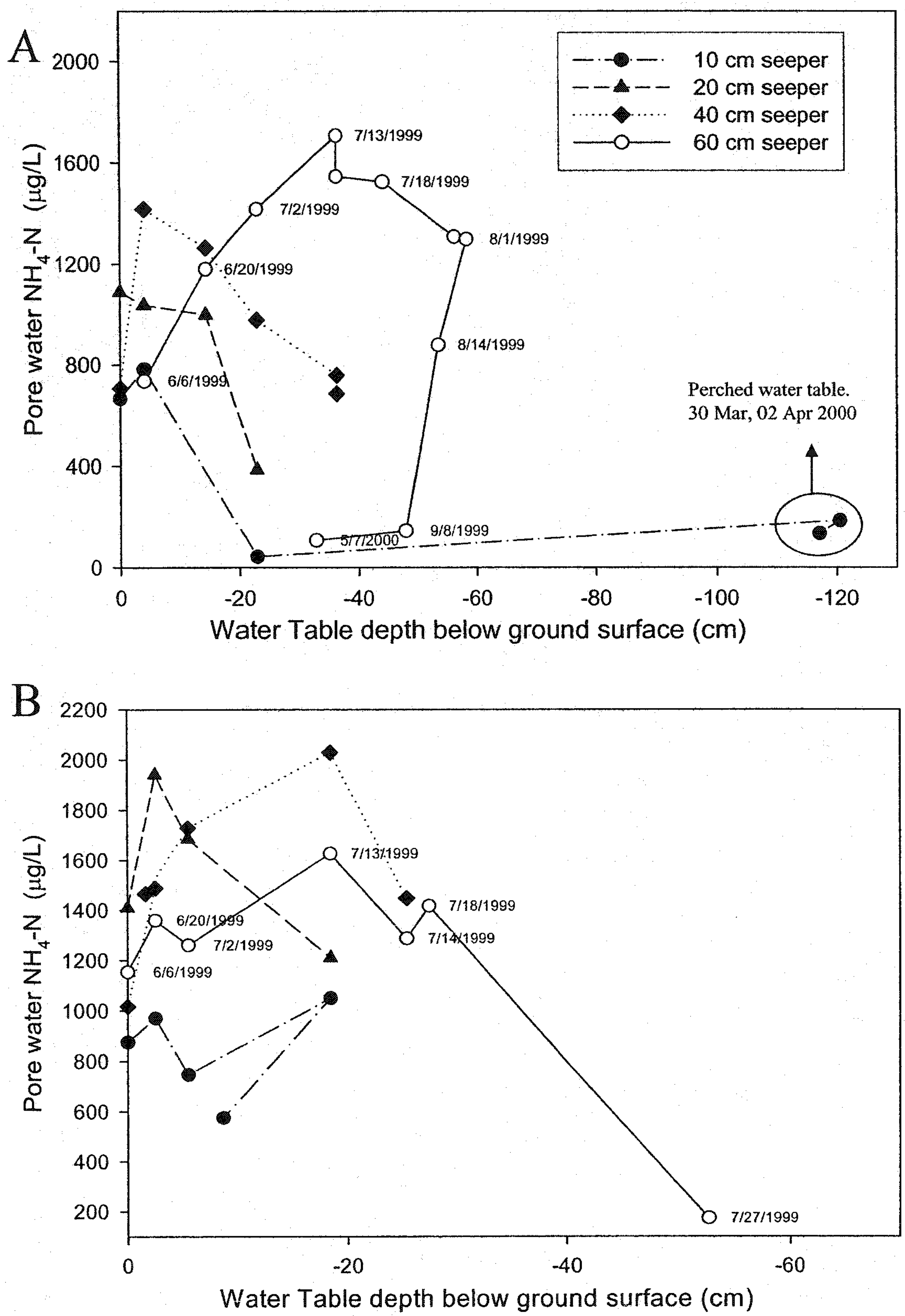


Figure 3.17 NH_4^+-N concentrations in pore water as a function of water table elevations. The drier "wetland south" site (A) vs. the largely saturated "wetland main" site (B). Data collected from shallow seepers (10 to 60 cm depth).

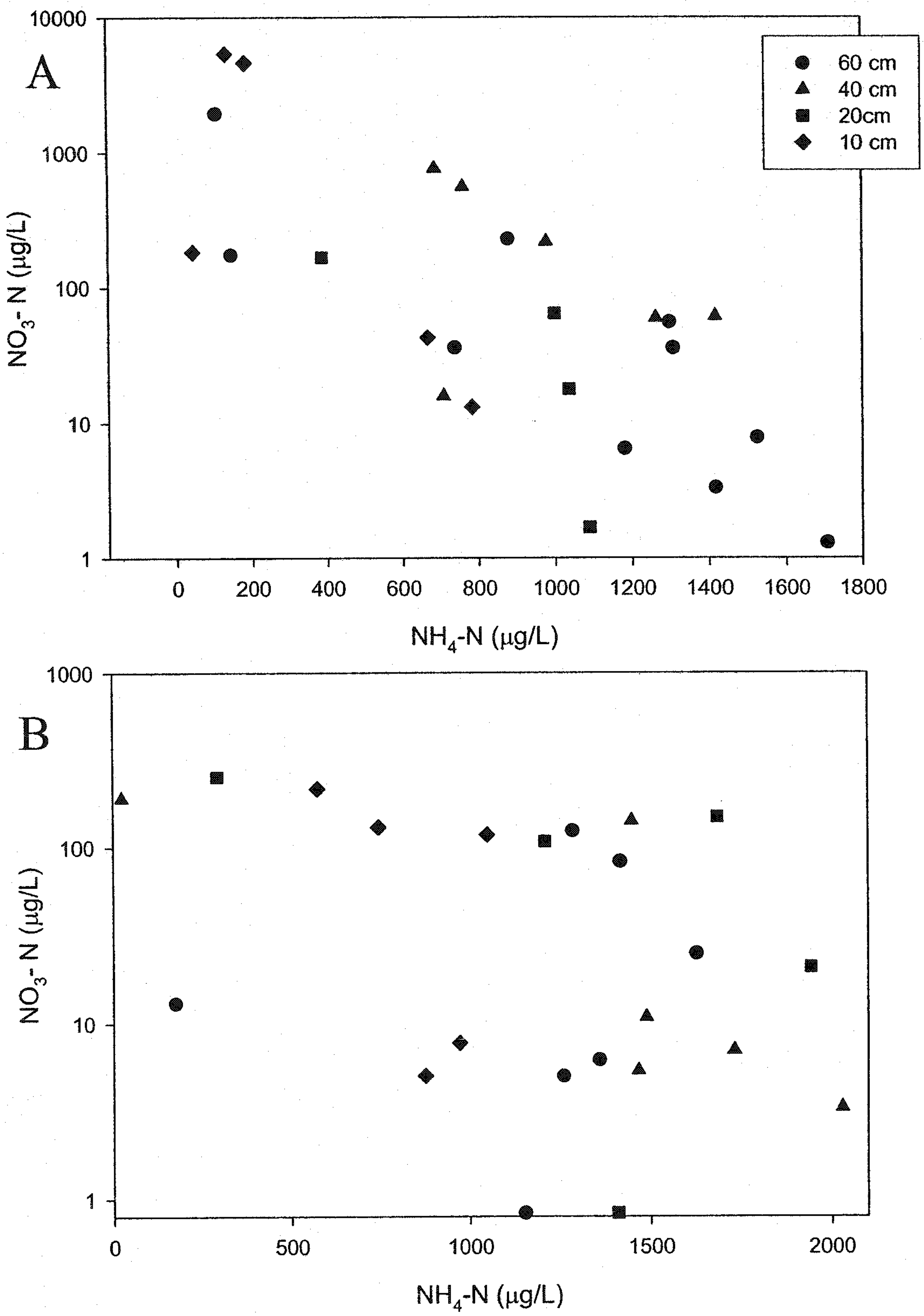


Figure 3.18 Pore water NO_3^--N vs. NH_4^+-N concentrations at the "wetland south" site (A) and at the "wetland main" site (B). Data collected from shallow seepers (10 to 60 cm depth).

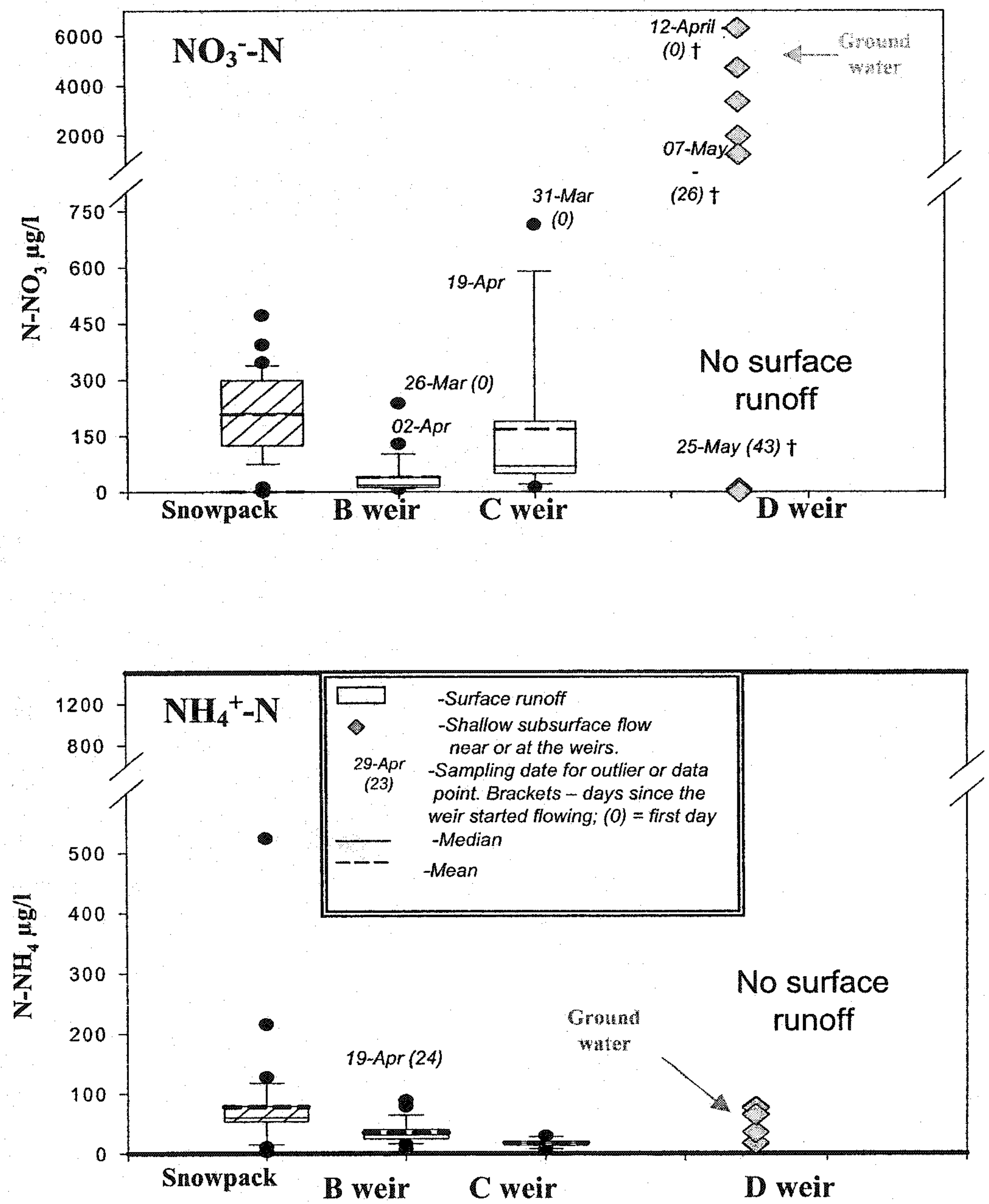


Figure 3.19. Nitrate and ammonium concentrations of surface runoff at the weirs, of snow, and of shallow subsurface flow near the outflow weir during spring 2000. Most of the data are taken from spring melt. Snow pack data are based on one survey before spring melt.
† Days since the shallow subsurface flow appeared in shallow wells (weir did not flow)
Chemistry data range for each weir: ***B weir:*** 26 March to 30 May 00. ***C weir:*** March 31 to 05 April 00. ***D weir:*** Shallow subsurface flow, 12 April to 25. May ***Snow pack:*** 25 March 2000, just before spring melt.

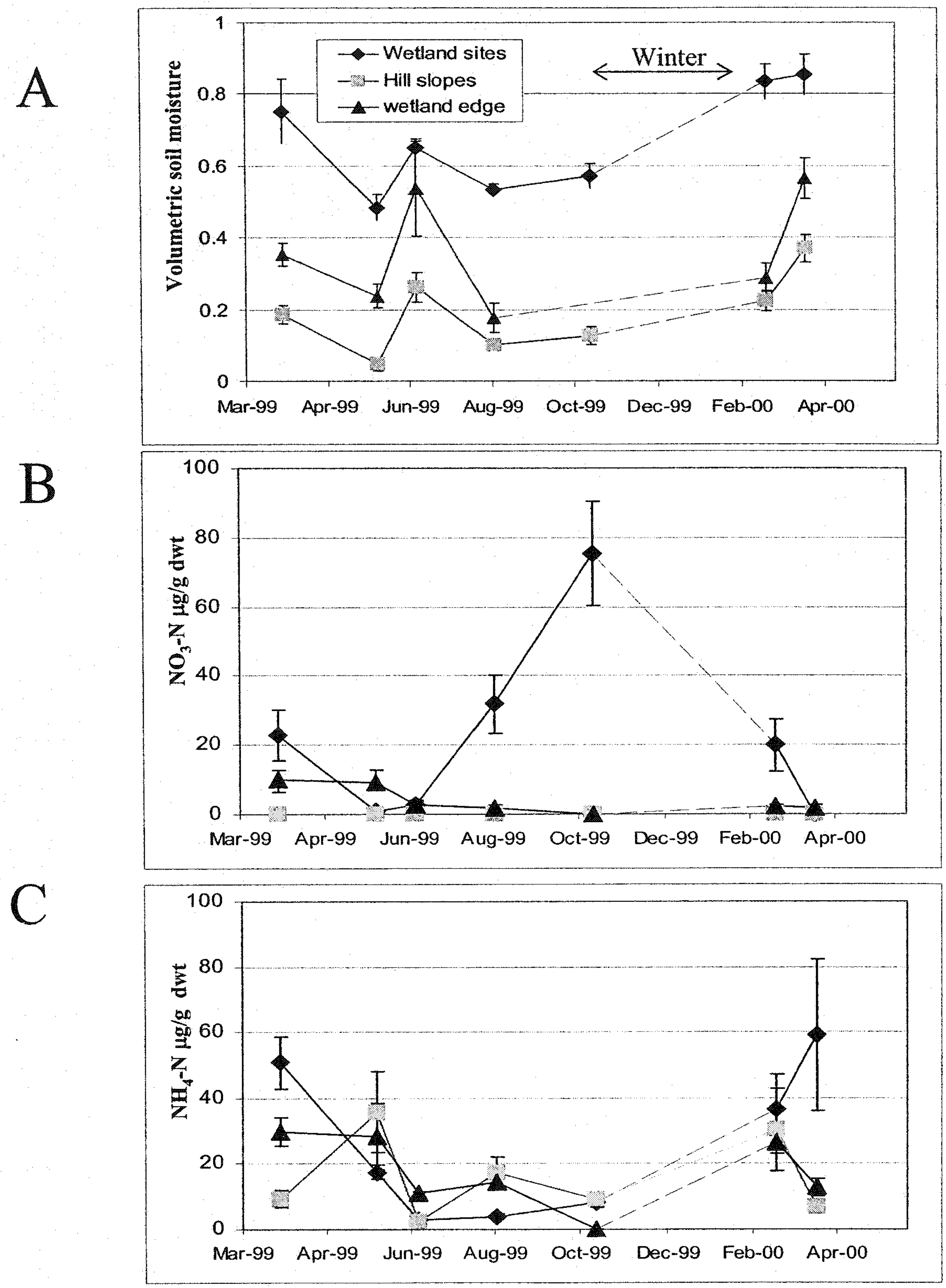


Figure 3.20 Mean seasonal changes in gravimetric soil water content ± se (A), extractable NO_3^- ± se (B), and extractable NH_4^+ ± se (C), in the top organic layers (5-12 cm) of three landscape features. Wetland – mean of 3 locations (n = between 7 to 21); hill slopes – mean of two locations (n= between 8 to 14); wetland edge mean of 1 location (n= between 4 to 7).

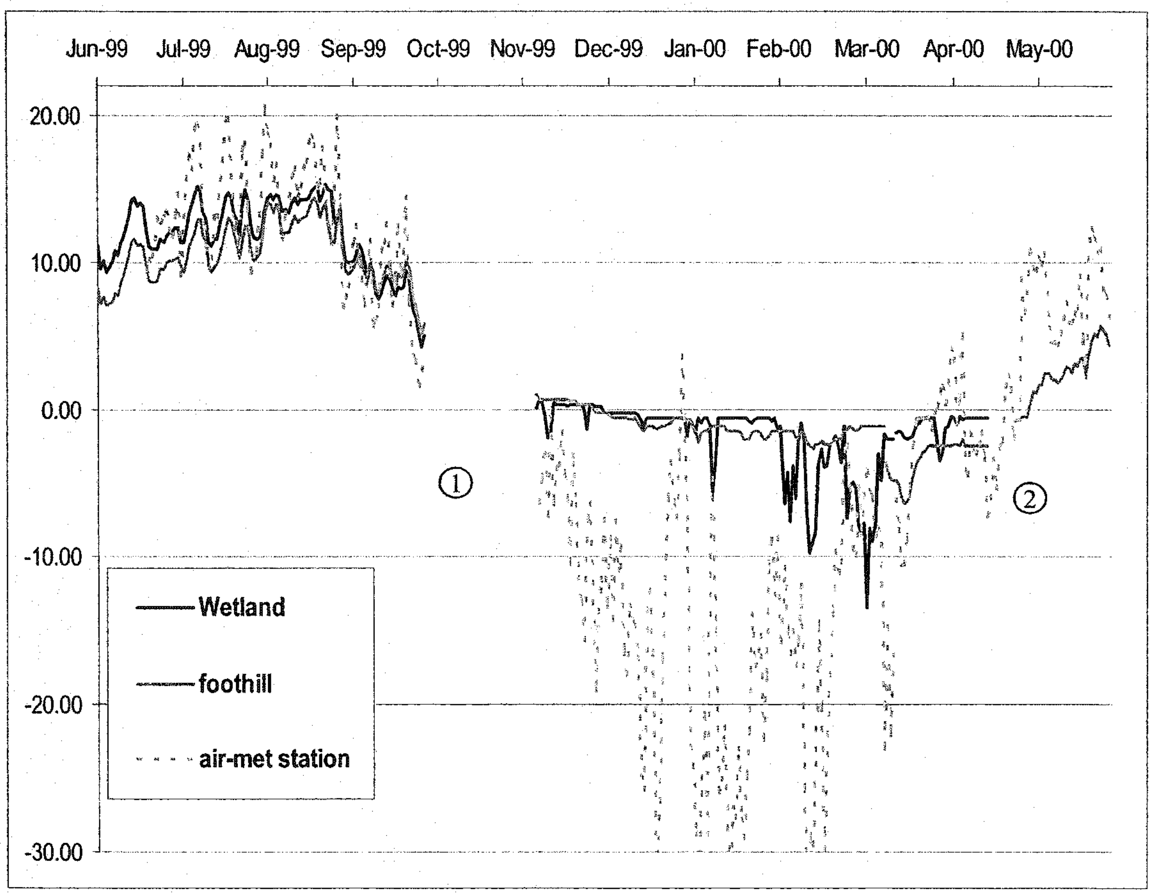
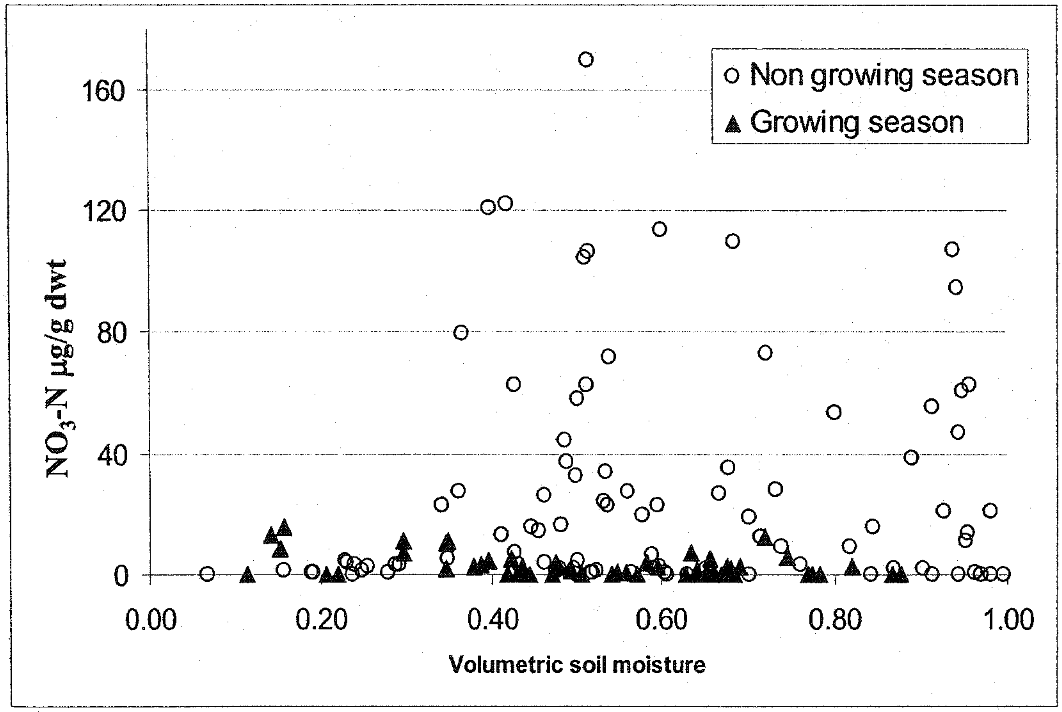


Figure 3.21 Daily mean temperatures of air and soil (7 cm depth) during the study period. (1) data loggers modified and re-installed on November 1999. (2) Wetland temp probe – malfunction from 13 Apr 00

Wetland sites



Hill slopes sites

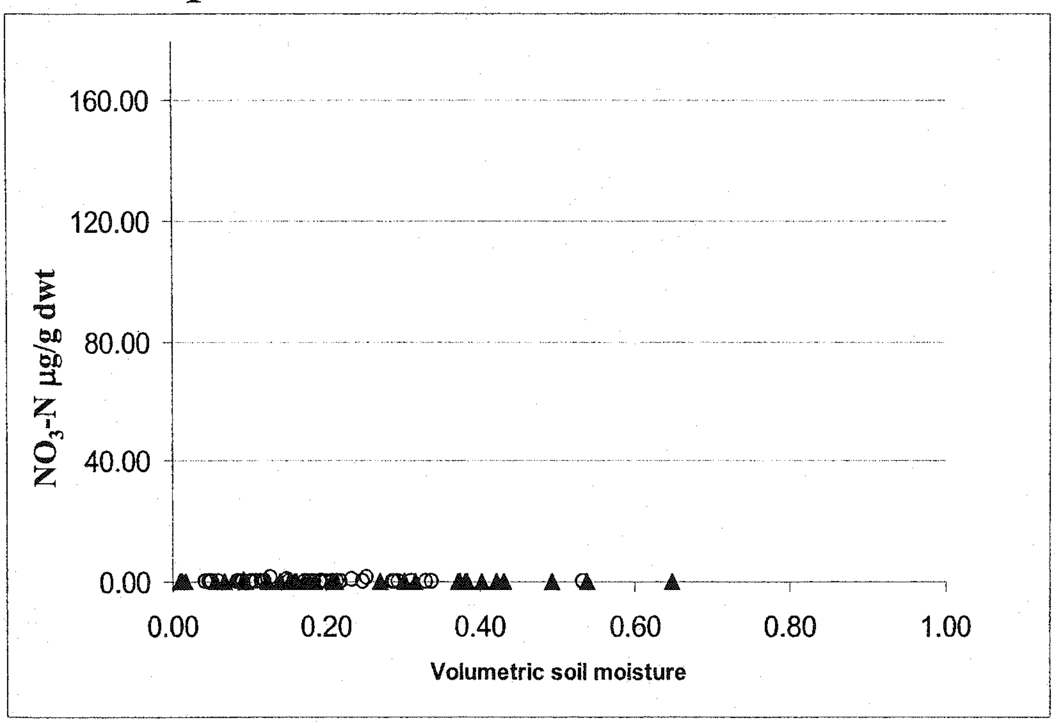
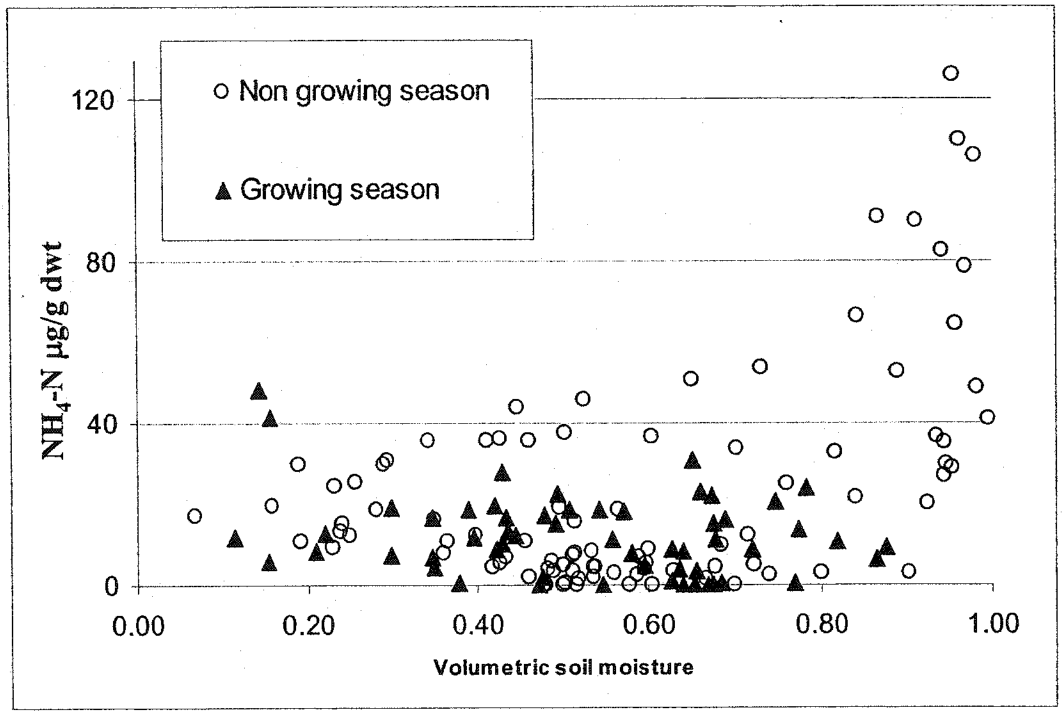


Figure 3.22 Extractable soil NO_3^--N vs. volumetric soil moisture in both the non-growing and the growing seasons, in the wetland (top) and hill slopes (bottom).

Wetland sites



Hill slope sites

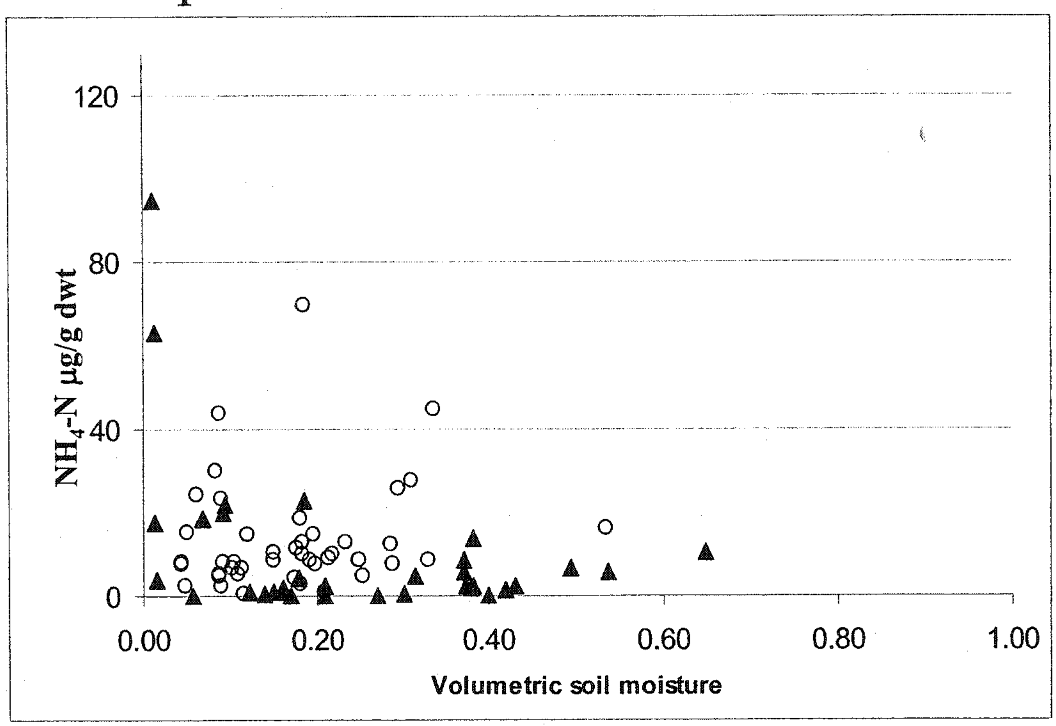


Figure 3.23 Extractable soil NH_4^+-N vs. volumetric soil moisture in both the non-growing and the growing seasons in the wetland (top) and hill slopes (bottom).

4 DISCUSSION

4.1 THE HYDROLOGICAL AND GEOLOGICAL FACTORS MAINTAINING THE WETLAND

4.1.1 Water balance and net function

The wetland was found to function as a small sink controlled by evaporative processes. Overall, however, the calculated water fluxes did not closely balance, where the budget unexplained residual of –230 mm was 65% of inputs. Several factors may account for the calculated imbalance: First, PET values are probably an over-estimation of actual in-site ET. While the Thornthwaite model of potential ET is a good indicator for evaporation losses from shallow open water (Ferone 2001), actual ET in the sub–humid area and from a temporally open water system is probably lower. Instrumented estimation of ET in similar aspen stands showed values of between 320 to 480 mm/yr (Black et al. 1996, Elliott et al. 1998, Barr et al. 2000) while pan evaporation showed values of 395 and 420 mm/yr (Ferone 2001). Surface water at the wetland was ephemeral and rainfall was low, thus actual ET in the studied wetland was probably lower than PET estimates but possibly higher than mean forested stand ET. Overall, ET values of 400-450 mm/yr are probably more realistic for the wetland site and incorporating these lower values would substantially improve the budget balance. Nevertheless, these estimates of actual ET still indicate that ET represented the largest output flux, controlling wetland water levels.

A second problem in mass balance estimation relates to large uncertainties associated with measurements of ground water (Winter 1981, Fetter 1994, LaBaugh 1997). Incoming shallow ground water to the wetland, that were important component of inputs, could have been underestimated. In addition, the seepage area of both deep ground water recharge and discharge was difficult to determine and large errors can be expected. Nevertheless, the estimated net effect of the two deep flows of -25 mm/yr, was of the same order of magnitude as -69 mm/yr, calculated by independent measurement based on wetland specific yield and over-winter water table decline. This outcome increased our confidence in the calculated net effect of deep seepage.

4.1.2 The role of atmospheric fluxes

Despite the unusually low amounts of precipitation resulting in rapid WT decline, precipitation was the major water source to the wetland. The large effect of atmospheric fluxes on aquatic systems has been reported for prairie wetland systems (Winter and Woo 1990, Hayashi and van der Kamp 1998, Ferone 2001), however, the actual amounts and seasonal distribution of atmospheric fluxes differ between prairies and the study region. Rates of ET are much lower in our region relative to wetlands of the southern bordering prairies (NWWG 1988), implying lower water losses and a potential for higher wetland sustainability. However, surface or shallow subsurface flows can be limited due to infiltration capacities of the forested soils and high ET during summer. Similarly, snowmelt, that is usually an important hydrological event in many parts of Canada such as the prairies and eastern boreal regions (Jones 1999), did not produce high surface runoff despite low ET during spring. This was related to low SWE before melt and high infiltration capacities of the hill slope soils (see below). In addition, redistribution and accumulation of snow in depressions, which is a large factor in maintaining wetlands in agricultural landscapes of the southern prairies (Hayashi et al. 1998a), was not observed in this study. Tree and shrub cover in our study site retards such redistribution and prevents this additional water input during spring melt.

4.1.3 The effects of stratigraphy and ground water fluxes on wetland existence.

The study was conducted during one of the driest years, and thus provides an opportunity to determine the importance of ground water in maintaining wetland existence. Although many wetland systems in the prairies are known to function as recharge systems, many wetlands can be affected by ground water discharge, especially systems found in topographic lows (Winter and Woo 1990, Winter et al. 1995, van der Kamp 1998, but see Ferone 2001). In our site, groundwater was an important component of the water budget consisting about 40% of water inputs.

Both shallow subsurface flow, affected by snowmelt, and deep ground water inputs combined with the wetland's low-lying topographic position and underlying semi-impermeable soils to create discharge in the wetland and moderate the effect of ET. Although there was a large drop in water table during this study, on a long term basis the deep ground water system is probably preventing complete drying of the swamp regardless of low precipitation. Local subsurface flow may be more important in maintaining hydric soils and vegetation in the site during wetter years.

Fluctuations in ground water dynamics may result in flow reversal that in turn can be very important for wetland-hill slopes chemical interactions (Hayashi and van der Kamp 1998, Ferone 2001). During spring, transient flow reversals occurred within the wetland and near the northern foothill. Many factors can account for flow reversals in various wetlands and lakes, such as surface evaporation, ET by peripheral vegetation, droughts and spring melt, or interactions between shallow and deep ground water flows (Meyboom 1966, Devito et al. 1997, Hayashi and van der Kamp 1998, Hayashi et al 1998a, Winter 1999). In the study site, during spring 2000, flow reversals were affected by low WT elevation at the hill slopes and uneven inputs to the wetland, with higher discharge to the northern part of the wetland.

4.1.4 Surface runoff and spring melt

During spring 1999, extensive VSA was observed connecting the upper tributaries of the catchment to the lake. However, during the hydrological year of the study, surface runoff at the catchment was low, and restricted to the upper tributaries. Uplands adjacent to pristine ponds in the Utikama region of Alberta, northwest of the study site, showed a similar lack of surface or storm runoff, even during spring melt (Ferone 2001). The very low precipitation, high ET, and high infiltration rates of the forested upland soils, limited contributions to the valley wetland and surface runoff from the catchment. Furthermore, relatively even snow distribution may have reduced soil frost and soil impermeability, and prevented surface runoff and snow accumulation in wetland depressions that is commonly observed in more southern prairie wetlands (Winter and Woo 1990, Hayashi and van der Kamp 1998).

Areas showing surface runoff were mostly active during spring melt and had the characteristics of frozen soil, impermeability, and connection to nearby snow pack. These areas were part of the previous year active (wet) variable source area. Any variable source areas that were not active (dry) during the study such as the stream channels between the upper catchment and the wetland did not show these characteristics and hence surface runoff. The above may suggest a link between the size of the active VSA in a given year to the extent of surface runoff and its spatial connectivity at a subsequent spring melt.

4.2 INORGANIC N CYCLING: NET FUNCTION AND HYDROLOGICAL SOURCES AND CONTROLS

4.2.1 The net function of the wetland in N retention

The wetland annual budget indicated very little inorganic N retention. Due to high uncertainties associated with ground water measurements the wetland may range between a small sink to a small source of NO_3^--N and a sink to a large source of NH_4^+-N. In addition, due to internal NO_3^--N cycling, the wetland has the potential to become a large nitrate source depending on spatial and temporal processes related to surface runoff (see below).

Wetlands and riparian wetlands have been known to act as nitrogen retaining or transforming systems (e.g., Pinay et al. 1993, Emmett et al. 1994, a summery by Hill and Devito 1997). However, the potential of a pristine wetland to become a large ammonium source via ground water or a nitrate source (also observed by Devito and Dillon 1993a), is not commonly stated in the literature.

4.2.2 Importance of precipitation contributions to the annual inorganic N budget

Direct precipitation in the form of rain and snow was very important in the wetland's nutrient budget, accounting for 95% of NO_3^--N and 45% of NH_4^+-N inputs. However, discrepancies were found between the two forms of N in precipitation. While nitrate levels in rainfall were similar to snow pack, ammonia levels in snow were much lower than rain concentrations. The lower levels in snow might be related to the evaporative properties of ammonia (Goslink 2000, cf Pomeoy et al. 1999), the effects of microbial activity in the snow pack (Jones 1999, but see Brooks et al. 1999a,b) or related to lower atmospheric pollution during winter due to lower fire frequency or lower wind blown fertilizers. Considering the unusually low precipitation during the study, the wet deposition of 1.3 kg ha^{-1} yr^{-1} inorganic N is comparable or slightly lower than neighboring Saskatchewan with 2.1 kg ha^{-1} yr^{-1} (Huang et al. 1997), north-western

Ontario with about 4 kg ha^{-1} yr^{-1} (Linsey et al. 1987) and northern Michigan (3 kg $ha^{-1}yr^{-1}$, Stottlemyer et al. 1999- study period mean – not annual mean).

4.2.3 The role of groundwater fluxes on inorganic N movements.

Studies have shown that ground water fluxes can play a large role in the nutrient budget of aquatic systems even if their relative share of the water budget is small (e.g., Shaw et al. 1990, Hayashi et al.1998a,b, Ferone 2001). Similarly, during the course of our study, ground water at the wetland appeared to play a dominant role in the inorganic N budget, despite being a smaller portion of the water budget, especially of outputs.

For most of the year, ground water did not play a large role in nitrate movements, as concentrations were low and corresponded to levels found in other western boreal wetlands (Vitt et al. 1995). However, following a high NO_3^--N spike in shallow sediments during spring the importance of shallow subsurface flow increased substantially, exporting NO_3^--N mass equal to half of the yearly total NO_3^--N inputs to the wetland. At the same time, ground water flow reversals seemed to result in flow of high nitrates from the wetland toward the wetland edge and the hill slopes. This however, was not accompanied with N accumulation at the edge beyond the duration of the nitrate spike.

The above behaviour stresses the inaccuracies that may be associated with sampling wetlands on an infrequent or pre-determined schedule without considering the *in situ* hydro-biochemical processes of the site.

NH_4^+-N concentrations in ground water increased more than twenty fold within the recharge zone of the wetland, emphasizing the effects of the wetland's internal flow system on ammonium export. Ammonia accumulation in wetlands is common (Williams 1974, Vitt et al. 1995) due to their continuous anaerobic conditions, organic sediments and high residence time. Discharge-recharge dynamics, that represent important mechanisms for nutrient movements in wetlands (Devito et al. 2000a), facilitated NH_4^+-N exports, suggesting wetlands in similar hydro-geologic settings are potential NH_4^+-N sources to ground water and neighboring aquatic systems.

However, enumerating mass losses to deep seepage in the southern part of the wetland was with considerable error, and spatial concentrations of NH_4^+-N at the bottom of the wetland varied more than ten fold. Also, large cation exchange capacity of the organic soils may retard NH_4^+-N losses (Foth 1990, Caravaca et al. 1999). Nevertheless, due to relatively high seepage and very high ammonium concentrations, it is possible that the usually large cation exchange capacity has reached saturation and allowed NH_4^+-N export from the wetland as measured at the mineral layers below the wetland organic soils.

4.2.4 NO_3^--N transformation and export

Water table fluctuations proved to exert large controls on internal N cycling. WT draw down during the summer of 1999 was shown to result in oxidation of the upper organic layers of the wetland and encouraged nitrate production and ammonium depletion (also see soils section below). The measured soil extractable NO_3^--N following WT draw down and the spike in NO_3^--N concentration of pore water as the water table raised to the surface further supported the importance of the upper organic layers as a major nitrate source to shallow ground water. NO_3^--N increase following WT draw down and during dry out of surface soils have been reported in agricultural areas (e.g., Hill and Shackleton 1989, Humphery et al. 1996), and in pristine wetlands (Devito and Dillon 1993a).

Another possible explanation or co-factor for the spike in NO_3^--N levels during spring is winter ground freezing. The advancing frost front can affect lyzing of plants and microbial cells and possibly creates root damage, leading to leaching of N to the upper water table during soil thaw (Boutin and Robitalille 1994, Mitchell et al. 1996, Brooks et al. 1997).

Regardless of the processes increasing surface soil NO_3^--N, quantifying the amount is useful to interpret potential flushing from the wetland. To conservatively estimate the NO_3^--N mass produced internally by the wetland during the spike, minimum values were used. These involved the pulse minimum NO_3^--N levels (approx 1500 µg/L NO_3^--N), a minimum affected water depth of 0.2 m (porosity of 0.9), and 2/3 of the wetland area, as water table was below ground elevation resulting in a smaller surface area connecting the upper soils. The results suggests that the nitrate spike was 3 times larger than the annual imports to the wetland with a value of at least 1.7kg NO_3^--N ha^{-1} yr^{-1} . To further support this finding, an independent estimate using 12 March 00 soil extraction, suggested that the wetland had about 2.4 kg extractable NO_3^--N, just before snowmelt. The assessment used the same area and depth as above, average bulk density of 0.17 g cm^3 , and conservatively used a median of 10.9 µg/g dwt NO_3^--N (the mean was 20 µg /g dwt, n=7). Denitrification and biological uptake are assumed to eventually consume the excess nitrates (Gosselink et al. 2000, Hill et al. 2000).

The above clearly illustrates the potential of the wetland to act as a NO_3^--N source to downstream aquatic systems. To further pursue the likelihood of large NO_3^--N exports, examination of flushing mechanisms and the probability of their occurrence is needed.

During spring, frozen sediments reduce permeability greatly and increases the potential for surface flow, when soil nitrate levels are still high in the wetland. One could speculate that a large rain event during spring could have flushed the ephemerally NO_3^--N –rich upper sediments of the wetland, possibly creating a major nitrate movement and converting the wetland to a nitrate source. During the study, however, the lack of surface runoff resulted in little flushing from the wetland. Examination of nutrient concentrations at the upper weirs under the limited scope of localized flow, indicated some flushing of NO_3^--N but was not sufficient to indicate whether it was the outcome of soil flushing or of snow nutrient flushing - "solute removal" (Brimblecombe et al. 1987, Brooks et al 1999b). Nevertheless, surface runoff connectivity was observed during early spring 1999 and large surface fluxes were observed in 1997 during an unusually wet year (Fraser et al. 2002). Episodic runoff events are important in nutrient transport (Devito and Dillon 1993a,b Devito 1995, Vitt et al. 1995) and their general importance should be assessed based on occurrence probabilities and timing. In our study site, the fate of a "successful" flushing event relies on a relatively narrow temporal range. Furthermore, the timing and effectiveness of spring flushing will vary from year to year, depending on preceding events such as water table fluctuations, winter temperatures, snow cover, and antecedent soil moisture.

To understand the probability of spring flushing in our area, runoff data from the nearby Logan and Owl rivers were examined (Alberta Environment 2002). Between 1984 and 1999 during mid spring (Julian days 100-140), substantial runoff occurred only twice, while mid to low runoffs were observed three times. All other years showed low runoff rates similar to the ones observed in 1999.

Precipitation and nearby river runoff suggest that summer rains have the potential to generate flushing, especially during successive wet years or abnormally wet summers (Alberta Environment 2002, Environment Canada 1994, Devito, unpublished data). Although the hydrologic fluxes can be large during the growing season, soil nitrate is kept low regardless of landscape feature (wetland or hill slopes). Therefore, lacking substantial NO_3^--N sources, major nitrate flushing during the summer is unlikely to occur. Furthermore, while WT draw down increases soil NO_3^--N concentrations, it also reduces the chance of surface runoff generation due to increase storage. For the above reasons and the low probability of flushing during spring, large nitrate movements are unlikely to occur during most years.

4.3 SOILS AS N SOURCES: THE ROLE OF THE WETLAND VS. HILL SLOPES

4.3.1 Instantaneous extractable soil nitrates

The examination of soil NO_3^--N in the wetland and adjacent hill slopes revealed that the wetland was the only site producing excess amounts of NO_3^--N while the surrounding hill slopes were found to be very low in extractable NO_3^--N. This data together with the N dynamics observed in shallow pore water and wells emphasize the importance of the wetland upper organic sediments as a nitrate source.

Several factors may account for the observed differences in NO_3^--N availability between the upper organics of the wetland vs. the upper organics of the hill slopes. In 1999, the dry conditions lowered the wetland's soil moisture to a threshold state in which oxygen was introduced to the upper soil layers, as indicated by the metal rod experiment. Following water table decline, oxygen introduction to the upper organic layers of the wetland may have promoted rapid nitrification, (Williams 1974, Devito and Dillon 1993a, Gosslink 2000), as also seen in pore water of the wetland. Although the WT dropped below the surface, moisture levels observed in the organic soils of the wetland were great enough to facilitate microbial nitrifying activity (Williams 1974, cf. Humphrey et al. 1996). Soil moisture levels less than 20%, as observed in the hill slope soils, has been shown to retard microbial activity resulting in low nitrate production (Klinka et al. 1994).

Differences in extractable NO_3^--N between wetland and hill slopes might also be co-related to other factors such as C:N ratios (Janssen 1996, Stottlemyer et al. 1999), variation in microbial activity (Carmosini 2000), or the effect of physical factors such as temperatures. However, in our study no relationship was observed between temperature and nitrogen concentration. While in lab studies, a correlation is often found (e.g., Huang et all 1997), in the field due to the effects of co-factors, investigations rarely find strong correlation between temperatures and NO_3^--N concentrations (e.g., Hill and Sackelton 1989, Pinay et al. 1993,Westbrook 2000, Carmosini 2000).

Large seasonal variations in extractable soil NO_3^--N, especially in the wetland, can have a large influence on the timing and magnitude of NO_3^--N exports. Wetland soils showed low NO_3^--N levels during the growing season regardless of moisture levels, possibly affected by plant uptake (Hill and Sackelton 1989, Haycock et al. 1993, Pinay et al. 1998, Stottlemyer et al. 1999, Devito et al. 1999; but see Jonasson and Shaver 1999). An increase in NO_3^--N levels was measured during the fall, when biotic demand decreased, oxidation status was high, and large leaf foliage resulted in excess nutrients (Haycock et al. 1993, Hung et al. 1997, Jonasson and Shaver 1999). A successive over winter reduction in soil NO_3^--N levels, may be related to microbial activity either by direct consumption-assimilation (Stottlemyer et al. 1999, Jones 1999) or by nitrate conversion (transformation) to ammonia under low aerobic conditions beneath snow and ice (Brooks et al. 1997). The latter possibility is supported by the observed increase in soil ammonia concentrations over winter. Direct microbial research may clarify some of the above speculations.

4.3.2 Instantaneous extractable soil ammonia

While significant or detectable NO_3^--N levels were almost exclusive to wetland soils, extractable NH_4^+-N at the upper 5-10 cm organic layers did not differ between hill slopes and wetland upper soils. However, the mineral soils had much lower NH_4^+-N than the organics and the deeper organic layer of the wetland relative to the uplands could result in a greater NH_4^+-N mass if deeper portions were considered.

Strong seasonal variations in extractable soil NH_4^+-N were observed in both the hill slopes and wetland sites. NH_4^+-N was low during the growing season, most likely due to plant uptake and increased microbial immobilization (Carmosini 2000). The increase in NH_4^+-N over the winter has been observed in other studies and can be expected as ammonifying microorganisms may still be active under harsh winter conditions, taking refuge at favorable, protective micro sites (Clein and Shimel 1995, Brooks et al. 1996, Hobbie and Chapin 1996, Devito et al. 1999b, Brooks et al. 1999a). Possibly, the production or maintenance of extractable soil NH_4^+-N is encouraged by recent fall foliage, less aerobic conditions under snow and ice, and the substantial decrease in plant and microbial uptake (Jonasson and Shaver 1999).

In all sites, extractable soil ammonia was generally 20-50% lower than found two years earlier by Carmonisi (2000) in the uplands of the same research area and could be explained by the drier conditions and resultant water stress on microbial and plant communities during this study. Also, similarities were found between the seasonal patterns observed in my study and the patterns observed by Carmosini (2000). In both studies higher NH_4^+-N levels were usually observed in late winter and spring followed by a decline over the growing season.

4.4 THE EFFECT OF LOGGING ON THE WETLAND HYDRO-BIOCHEMISTRY

Currently it is not clear whether or not logging at the upper part of the catchment two years prior to this study (Carmosini 2000) created substantial changes in the wetland's hydro-biochemistry. The wetland was not studied prior to the disturbance, and my study did not look directly at links between the effects of logging and the wetland. It is commonly stated that disturbance by logging is hydrologically and chemically manifested by increases in surface runoff (e.g., Likens et al. 1977, Vitousek et al. 1982, Ward 1989). However, during the study, surface runoff connectivity and ultimately outflow to the lake did not exist, and the probability of catchment surface flow is low. Also, fast aspen recovery (Carmosini 2000) and lower precipitation to soil storage ratio (Fraser et al. 2002) may have reduced logging–runoff effects.

Overall, it appears that concerns regarding nitrate exports from the catchment should be directed toward the wetland system and not the hill slopes. Therefore, logging the nitrate deficient uplands may affect nitrate transports only if in turn it will trigger nitrate release from the wetland. This scenario is not expected during dry years due to no-flow conditions. During wet years, logging may increase WT elevations and runoff, and may induce more frequent flushing from the wetland. However, wetter conditions due to logging may also result in a decrease in nitrate flushing due to a decrease in soil oxidation, possibly leading to sink behaviour.

5 CONCLUSIONS

The study was conducted under dry conditions and reflects short term trends, however, certain patterns of wetland hydrologic and biogeochemical function are evident and might be enhanced under wetter or drier conditions.

As hypothesized, the wetland was hydrologically controlled by both atmospheric fluxes and ground water. Patterns of recharge and discharge that were hydrologically and chemically important illustrated the significance of complex stratigraphy on wetland hydro-biochemistry. In contrast to the initial hypothesis, summer rains and spring melt did not result in significant surface runoff. High infiltration rates during spring and high ET and infiltration during summer minimized runoff. However, due to lower water storage and ground freezing, the wetland and upper tributaries were the only landscape features with a potential to create surface runoff.

The wetland annual inorganic N budget indicated very little retention and a potential function as a NO_3^- source via surface runoff and a NH_4^+ source via deep seepage. Despite their lower share in the hydrologic budget, ground water outputs dominated inorganic N exports. However, the largest NO_3^- outputs are assumed to be controlled by vegetation and possibly denitrification as observed during the growing season and late spring.

As hypothesized, hill slope soils had low inorganic N year-around and therefore were not a potential large source to water fluxes. In contrast, wetland soils affected by water table draw down and soil oxidation, showed elevated NO_3^- concentrations during fall and winter and combined with higher potential to generate surface runoff, illustrated the overall potential of the wetland to act as a nitrate source during spring. Evidently, during spring, shallow ground water exports increased substantially when the WT approached the surface and nitrate was removed from the upper soils. Shallow subsurface transportation, though important, is relatively slow in comparison to the transportation potential of surface runoff. However, unlike initial assumptions, flushing of soil nutrients via surface runoff was not confirmed, due to low disconnected fluxes. Based on long term hydrologic data from nearby streams, the probabilities of flushing during spring when NO_3^- levels are still high are small due to low SWE, high infiltration rates in the uplandsand the nature of precipitation patterns. NH_4^+ exports, on the other hand, are assumed to be more stable as these are linked to stable deep recharge discharge processes and are not depended on fluctuant conditions such as imposed by WT behaviour and surface runoff generation.

The study indicates that wetlands are not necessarily buffers for nutrient exports. Furthermore, concerns regarding nitrate exports due to disturbance such as logging should be directed toward the wetland system and less toward the forested uplands. In addition, the study illustrates that catchment scale models that assess N exports, incorporating strong links to hydrology and flushing probabilities, are necessary to simulate and predict nutrient movements, disturbance effects, and sustainability of the Western Boreal ecozone.

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