Swedish University of
Agricultural Sciences

Element balances and retention for wetlands in the forest environment – case study Bohyttan fen

Gina Lucci

Supervisors: Lars Lundin & Torbjörn Nilsson

MSc thesis at the Department of Forest Soils, SLU

Uppsala, 2007 Nr. 17

Element balances and retention for wetlands in the forest environment – case study Bohyttan fen

Ämnesbalanser och retention för våtmarker i skogslandskapet – exempel Bohyttekärret

Gina Lucci

PREFACE

This MSc thesis was carried out at the Department of Forest Soils at the Swedish University of Agricultural Sciences and corresponds to 20 credits at D level and fulfils the requirements for a Master of Science thesis in Soil Science within the Soil and Environment program.

In a society facing increasing demands on forest biomass, the nutrient and other element cycling in the catchments are of crucial importance. Forest is of interest for many purposes and harvesting is high on the agenda. However, impacts from forestry measures on soils and water attracts high interest and the element leaching to water courses and lakes is under debate. In the commune of Nora, the drinking water supply from a lake experienced deterioration and restrictions were established to protect the water quality. Within this catchment, forestry anyhow was carried out and this was studied in fairly long-term monitoring in a sub-catchment with discharge measurements and water sampling for chemical analysis. The current report includes trends in water chemistry, effects from forestry measures and the retention of elements in an ordinary sedge fen before inflow to the water supply lake. Such investigations are of great significance for understanding the impacts of enhanced forestry on important water bodies for the society. This is a good contribution in the ongoing environmental work taking consideration of environmental services.

Supervisor professor Lars Lundin and deputy supervisor Dr. Torbjörn Nilsson

Uppsala
May 17, 2007
Lars Lundin

TABLE OF CONTENTS

SUMMARY. The importance of water quality, protection of watersheds and restoration of aquatic ecosystems has steadily been gaining attention. Wetlands perform many functions that are beneficial to water quality, and also play an important role in forested landscapes. Peatlands in forested catchments act as both sinks and sources for different nutrients. The effects of leached nutrients from a forested catchment on a fen and the retention properties of the fen were investigated using data collected from the Bohyttan fen in the commune of Nora, Sweden. In addition, the effects of forestry management on nutrient loading and how the fen affects the flow of nutrients into the nearby lake were also investigated.

Runoff measurements into and out of the fen were calculated using v-notch weirs and water level recorders. Water samples were collected monthly and used to calculate nutrient balances and retention of different elements. The effects of different forest management practices on nutrient leaching were investigated using the calibration period and control area technique.

The peatland retention was obvious for nitrate (34%), ammonium (6%) phosphate (5%), potassium (8%) and sulfate (7%). Nitrate retention is beneficial for water quality and helps to mitigate the effects of nutrient leaching, however the peatland exported organic nitrogen (-6.5%) and iron (-9%). Results show that the effects of forestry practices like clear-cutting increased the transport of nitrate and potassium even six years after it was carried out. Ditching of a cleared forest stand lowered the groundwater and also caused a rise in the pH of nearby stream water.

Leaching from forest land showed a correlation between high water color, DOC and iron concentrations. These conditions furnish additional organic matter and lead to a deterioration in water quality.

Key words: Retention, element balances, wetlands, peatland, water quality, forestry

SAMMANFATTNING. Betydelsen av vattenkvalitet, skydd av vattentäkter och restaurering av akvatiska ekosystem har fått ökad uppmärksamhet. Våtmarker bidrar till många viktiga funktioner som är välgörande för vattenkvalitet och de spelar en viktig roll i skogslandskapet. Torvmarker i beskogade avrinningsområden fungerar både som näringsfällor och näringskällor. Effekterna av näringsläckage från skog på ett kärr och retentionsegenskaper av kärret undersöks med hjälp av data samlat från Bohyttekärret i Nora kommun. Dessutom har skogsbrukets effekter på näringsläckage samt hur kärret påverkar utflödet av näring till en intilliggande sjö undersökts.

Vattenflödet till och från kärret beräknades med vattenföringsstationer. Vattenprover togs varje månad och användes för att beräkna närningsbalanser och retention av olika ämnen. Effekterna av skogsbruket på närningsutlakning undersöktes med kalibrering och kontrollmetoden.

Torvmarkens retention var uppenbar för nitrat (34%), ammonium (6%), fosfat (5%), kalium (8%) och sulfat (7%). Nitrat retention förbättrar vattenkvalitet och bidrar till att minska näringsbelastningen, däremot exporterade våtmarken organiskt kväve (-6.5%) och järn (-9%). Resultat visade också att effekterna av avverkning ökade transporten av nitrat och kalium upp till sex år efter åtgärd. Dikning av en avverkad skogsmark sänkte grundvattennivån och orsakade en höjning av pH i närliggande bäckvatten.

För utlakning från skogsmark noterades samband mellan hög vattenfärg, DOC och järnkoncentrationer. Detta visar på ökad mängd organiskt material och leder till försämrad vattenkvalitet.

Nyckelord: Retention, näringsbalanser, våtmark, torvmark, vatten kvalitet, skogsbruk

INTRODUCTION

Deterioration of water quality is a problem in many parts of the world. Agriculture, forestry, and the burning of fossil fuels are just a few of many pressures on the environment that have lead to an increase in nutrients in aquatic ecosystems. Agriculture, forestry management practices and atmospheric deposition increase nutrient loading and negatively affect surface water chemistry. Attention has increasingly turned to the use of wetland systems to mitigate the effects of water-bound pollutants.

Wetlands may make up only a small fraction of the landscape, but because they are found in depressions, much of the runoff and some groundwater water must pass through them. Therefore wetlands have a great impact on the chemical transformations, biodiversity and water quality of ecosystems downstream. Through natural adsorption and biochemical transformations like sedimentation and denitrification, wetlands are able to remove nutrients from through-flowing water. Understanding wetland dynamics, nutrient retention and transport is vital for the evaluation water quality issues.

Driving forces

Waterborne nutrients in surface water are vital for production, but nutrient loading can be regarded as a form of water pollution once natural concentrations are exceeded. Plant nutrients, together with environmental pollutants like suspended solids and metals can come from a variety of sources. Here the underlying environmental pressures from agriculture, forestry, peat mining and atmospheric deposition are briefly summarized.

Agriculture

The loss of nutrients from agriculture is unprofitable both for the farmer and the environment, and is a result of unsustainable nutrient cycling. Some of the traditional farming practices are to blame. It is common practice to leave the soil surface bare at some stage in crop rotation. While the soil is bare nitrate (NO_3^-) is more readily lost because there is no vegetation to capture it and also because its negative charge makes it more susceptible to leaching (Addiscott, 2005). Erosion and the generation of sediment in waterways are most prevalent when the soil is bare. Agricultural drainage systems, designed to lower the groundwater table, can also be detrimental to water quality and can provide a more direct path for fertilizers, pollutants and pesticide residues to waterways.

Animal husbandry is also to blame for anthropogenic nutrient loading. Manure and dilute farm wastes, like dairy washings, are potential sources of nutrient runoff and the spreading of such wastes is therefore often subject to regulation. Apart from the direct effects of manure handling, livestock farming may also indirectly influence water quality. Livestock trample the soil and thereby increase the bulk density and reduce infiltration rates of the soil. This leads to increased erosion and sediment in watercourses. (Harrod & Theurer, 2002).

Peat Mining

Peat is an important resource utilized in NW Europe, Ireland and Canada. In order to harvest peat for energy or as growing medium, the peatland must first be drained. The drainage water from peatlands affects the quantity and quality of the water in nearby streams and watercourses. The increase in nutrient loading to recipient watercourses is a result of increased runoff and mineralization rates (Lundin, 1996; Rydin & Jeglum, 2006) in the peatland. A compilation of investigations of Swedish peat harvesting (Stenbeck, 1996) showed that the amount of nitrate in peatland drainage water was an average of 6.4 times higher than the natural conditions in watercourses at elevations less than 250 meters above sea level. The increase in inorganic nitrogen concentration is mainly due to the higher mineralization rates in the drained peat and the lack of vegetation able to assimilate the mobilized nutrients (Heikkinen, 1990). Drainage water from peat mining activities also decreases pH and increases suspended solids in recipient water courses. The amount of suspended material increases drastically throughout the first half year after drainage, because of the disturbance of the peat surface. Peatlands such as bogs are naturally acidic and therefore drainage water from peat mining has a lower pH than the recipient water (Stenbeck, 1996; Heikkinen, 1990; Rydin & Jeglum, 2006).

Atmospheric Deposition

Acidic deposition of sulfate and nitrate can lead to the acidification of surface waters and is regarded as one of the major environmental dilemmas in Europe and North America. Many counties have developed environmental control programs aimed at the reduction of acidic deposition and ground level ozone (for example the Swedish Environmental Objectives: *Clean air* and *Natural acidification only,* Swedish EPA, 2007). As a result of these programs, sulfate concentrations in stream waters have decreased in Europe and in North America (Kleemola & Forsius, 2006; Skjelkvåle et al., 2005). Atmospheric deposition of nitrogenous compounds can exacerbate eutrophication and lead to alterations in species composition and biodiversity. The amount of atmospheric deposition is highly variable and depends on the amount and intensity of precipitation, and the proximity to the source of emissions.

Forestry

Although nutrient leaching from forests is relatively small, forests can be an important source of nutrients to rivers and lakes. This is because of their extensive area (more than 50% of Sweden is covered by forest) and forest management practices. Modern forestry practices like clear-cutting and scarification can directly and indirectly affect the hydrology and water chemistry of adjacent water courses and drastically change the forest environment. Clear-cutting removes vegetation and consequently alters nutrient uptake and hydrology. After clear-cutting the vegetation transpiration ceases which results in an increase in runoff and leaching of nutrients (Rosén, 1984). It also exposes the soil surface which results in more extreme temperature differences. The effects of clear-cutting depend on the method used, soils, topography and climate (Chang, 2003).

The construction of forest roads is another contributor to the deterioration of water quality. Although essential, forest roads and connected drains are sources of erosion and soil compaction. Forest roads also interfere in natural drainage patterns, and create convergences which can turn into gullies (Chang, 2003). Scarification and mechanical site preparation includes plowing, hummocking and ditching. From the Nurmes-study in Eastern Finland, Ahtiainen (1992) reported that clear-cutting and scarification increased the concentration of nitrogen, phosphorous and iron in forest streams. Three years after the treatments, many of the water quality variables had not returned to their pre-treatment values. The study also revealed a sharp increase in suspended solids that was 200 times higher than before the clear-cutting and scarification took place (Ahtiainen, 1992).

The practice of prescribed burning after clear-cutting and logging is used to prepare sites for revegetation and reduce wildfire risk. However, prescribed fires have both positive and negative effects on water quality. Fire causes an increase in mineralization and leaching of some nutrients but it is beneficial for species composition and forest health. Controlled surface fires tend to have little effect on stream hydrology because the majority of the forest stand remains (Chang, 2003).

Pressures

Chang (2003) divides water pollutant pressures into the following categories: sediments, plant nutrients, oxygen-demanding wastes, disease-causing agents, heat, inorganic chemicals and minerals, synthetic organic chemicals and radioactive substances. The principal pollutants involved in this study belong to the second category, but some of the others are also worth mentioning.

Nitrogen

Nitrogen is an essential element that is an ingredient in proteins, nucleic acids and chlorophyll. Nitrogen is taken up by plants in the form of nitrate (NO_3^-) and ammonium ions (NH_4^+), but organic nitrogen can also be used directly by some plants (Näsholm et al. 1998). Nitrate-N is more susceptible to leaching because it is not adsorbed by most soil colloids on account of its negative charge. Nitrogen can also be lost to the atmosphere in the gaseous form (N_2 and N_2O) through denitrification. The nitrification/denitrification mechanism is the principle route through which nitrogen is removed from wetlands (Vymazal et al, 1998). Nitrate is formed through nitrification in the upper oxic zone near the surface of the wetland. It is later denitrified to gaseous N after it diffuses down to the waterlogged anoxic zone. In flooded soils, where pH is high enough, nitrogen in the form of ammonium ions can be lost through volatilization as ammonia gas (NH_3).

Phosphorus

Phosphorous is another essential plant nutrient that is responsible for eutrophication and resulting hypoxia (low oxygen) of surface waters. Phosphorus can be either taken up directly by plants and bacteria or it can be complexed or adsorbed to minerals like aluminum or iron hydroxides and humic substances (Leonardson, 2002). The exchange capacity of the soil, together with microbial uptake determines the amount of P available to plants (Richardson & Marshall, 1986). Generally, the uptake of P through biological accumulation is of minor significance (Väänänen et al, 2006). Inorganic P may also precipitate as insoluble iron or aluminum phosphates. These particles become part of the sediment and the retention or release from complexation depends on pH and the availability of oxygen (Vymazal et al. 1998).

Metals

Metals occur in both soluble and particulate forms, but soluble ionic forms are the most available to plants. Metallic ions may be adsorbed to the surfaces of soil colloids or plants. The redox potential (Eh) of the soil, together with pH, will significantly affect the precipitation and solubility of metals. As long as pH remains relatively high, there is little risk for the leaching of metals and they will instead accumulate in the soil.

Suspended Solids

Sediments and suspended solids in water can cause problems by restricting light and by clogging streams. Nutrients, metals, organic chemicals and fertilizers can be adsorbed to soil particles. Consequently, erosion of sediments can lead to chemical, and not only physical, water quality problems. (Chang, 2003; Harrod & Theurer, 2002).

Organic Compounds

Wetlands typically contain waters with dissolved organic compounds. BOD and COD (Biological- and Chemical Oxygen Demand, respectively) are means to measure the amount of biodegradable organic matter in waste water. In natural wetlands the total organic carbon (TOC) is more often used to indicate the amount of carbon in the water. Because organic compounds are quickly oxidized by microorganisms, high organic content can indicate the potential risk for hypoxia and dead zones in aquatic environments.

Forestry Nutrient Sources

Nutrients leached from clear-cut forest soils can originate from logging residues like branches and tree tops, or the remaining roots of felled trees and the organic humus layer that covers the soil surface. Palviainen et al. (2004) calculated the concentration of elements in different fractions of logging residues and found there was great variation both between different tree species (pine, spruce and birch) and between residue types (foliage, roots and branches). They found that the initial concentrations of K and Ca were highest in the foliage, while the concentrations of Fe and Al were highest in the roots. The individual elements were also lost at unequal rates through decomposition. Potassium concentrations in foliage, for example, were reduced by almost 90% after the first year, while iron concentrations in branches and foliage tended to increase in subsequent years. The changes in nutrient concentration were highly variable depending on the residue fraction.

Impact

Increase in the nutrient input to aquatic ecosystems (eutrophication) has a wide array of effects. The most visible and often associated is the explosion of phytoplankton growth that creates mats of biomass called algal blooms. Algal blooms floating on the water surface reduce the amount of light available deeper down in the water column profile and thereby the species composition. The amount of dead organic material in the water increases significantly after phytoplankton death. The accelerated decomposition of the organic material soon exhausts the dissolved oxygen in the water, resulting in the suffocation of bottom-dwelling organisms (Skei et al., 2000).

Erosion and sediment can create problems by clogging channels and may lead to flooding. Sediment also coats the bottom of stream beds and may obstruct the spawning of fish by deteriorating the interstitial water (Harrod & Theurer, 2002).

Mitigation

Wetlands

The term wetland is used to describe a wide variety of areas and ecosystems. The most consistent feature of a wetland, as Kadlec and Knight (1996) point out, is the presence of water such that the plant species present are adapted for growth in flooded conditions for at least part of the year. Municipal wastewater treatment facilities are modelled after natural wetland systems and purify water by many of the same processes. However, space for waste water treatment is limited and the compressed treatment area requires additional labor and energy inputs to optimize the transformation of pollutants by microorganisms (Hammar, 2002). In order to meet modern environmental goals, interest in wetlands as more cost effective, natural water purifiers has increased.

Mechanisms of filtration

Wetlands reduce the amount of pollutants by physical, chemical and biological processes. Sedimentation is a physical process and is the action of gravity on suspended particles. Chemical processes include the precipitation of solids and the sorption of substances to the surface of soil particles. In some instances, purification is the result of a combination of processes. Positively charged elements are attracted to the negative surfaces of soil particles and these particles, together with adhered elements, will sink and be removed from the water by sedimentation. Microorganisms also have surface charges which cause them to be adsorbed to the soil matrix and removed by sedimentation. Biological processes include the uptake of nutrients by vegetation and biological transformations by microorganisms. One of the most important biological transformations is denitrification. Through denitrification nitrate is converted into nitrogen oxides and nitrogen gas and lost from the system. The microbial population's biological transformation processes are considerable compared with plant uptake. According to Hammar (1992), less than 5% of nutrients are taken up by plants in wetlands. Instead, the biological transformations of nutrients are primarily carried out by fungi, bacteria, algae, and protozoa.

Wetland dynamics

Hydrology is the central characteristic that shapes both the biotic and abiotic processes in wetlands. The duration and depth of inundation mediate the variety of plants and animals present in a wetland. This is because there are few plants adapted to survive in flooded conditions. Consequently, a change in the hydrology will result in a change in the species composition. Nutrient cycling is also related to the flow regime of rivers and wetlands. Nutrients are alternately mobilized or immobilized according to the availability of oxygen. Decomposition rates of organic matter in anoxic conditions are generally 10% of that in oxic environments (Hammar, 1992). This is the reason for peat accumulation in waterlogged areas like bogs and fens.

Retention time and preferential flow

Retention time is the length of time water stays in a wetland. Retention time can range from hours to years and is an important factor used to judge wetland efficiency. The more time the effluent is in contact with the organic and inorganic components of the wetland, the more time for chemical transformations and soil adsorption processes to occur. The theoretical retention time (RT) of a wetland is calculated by dividing the system volume (V) by the flow rate (Q) (Hammar, 1992).

$$RT = V/Q$$

Theoretically calculated retention times have shortcomings because they do not take into consideration preferential flow and stagnant regions. Using tracer studies, Smith et al. (2005) found that mean retention times were approximately 50% of the theoretical retention times for a constructed wetland. Over time it is common that preferential flow channels form that allow the water to flow through wetlands more quickly and bypass the filtering properties of the wetland. For wetlands to act as effective filters, the formation of preferential flow channels should be minimized (Smith et al. 2005).

Depth

Wetland depth affects the settling time of particles and therefore the amount of nutrients, such as phosphorous, that will become incorporated in the sediment. In a comparison of constructed wetlands (CWs, depth <0.5m) and ponds (depth > 0,5m) Uusi-Kämppä et al. (2000) found that the average total phosphorous retention for CWs was 47% while ponds retained only 17%. Several studies (Braskerud, 2002; Tonderski et al., 2002; Uusi-Kämppä et al., 2000) have found that shallow constructed wetlands are superior because they allow macrophyte growth and promote rapid particle settling.

Vegetation

Plant species in wetlands can range from small, simple, nonvascular species like algae, to large and familiar cattails and reeds like *Phragmites australis*. Common to all wetland plants is their ability to flourish in periodically flooded environments. Vegetation increases the sedimentation of particles by reducing the water velocity. The presence of vegetation in wetlands also hinders erosion and the re-suspension of settled particles. Vegetation increases infiltration rates and it can also serve to insulate against frosts in the winter and keep the soil cooler in summer months (Vymazal et al. 1998). Plants are able to take up and remove nutrients, but some of the nutrients are returned again via litter fall. With increased nutrient loading, the removal rate by plants declines sharply (Leonardson, 2002; Hammar, 1992; Brix, 1997). However as Gottschall et al. (2007) points out, studies aimed at determining plant nutrient uptake usually sample only above-ground plant biomass and underestimate nutrient storage in the roots and rhizomes below the surface.

The most important function of plants in a wetland is to create environments for microorganisms. The biofilms that coat the roots, stems and leaves of macrophytes are responsible for most of the microbial process that occur in wetlands (Vymazal et al. 1998). One example is the oxic zone that surrounds the roots of wetland plants in an otherwise oxygen deprived environment. These small oxic zones are important for the biological transformations of nitrogen and other elements (Hammar, 2002).

Wetland age

Another important aspect in the evaluation of wetland efficiency is how the wetland changes with time. Most studies of constructed wetlands are carried out a few years after the wetland construction is completed, and do not reflect the possible decline in retention capacity as wetlands age (Gottschall et al, 2007). Over time, wetlands subject to high loading rates become saturated with nutrients and their effectiveness as nutrient sinks decreases. In some cases it is necessary to dredge the wetland and recreate the soil filter and plant matrix. Natural wetland systems are seldom exposed to such high loading rates and nutrient saturation is rarely an issue except in disturbed ecosystems (Vitousek et al., 1979).

Wetland Types

Because of global and local environmental programs, like the Swedish Environmental Objectives (*Thriving Wetlands* and *Flourishing Lakes and Streams*), and the Ramsar Convention on Wetlands, there has been increased attention upon the creation and restoration of wetlands. Some wetlands are created as compensation for past government programs that encouraged draining to create more arable land (Arheimer & Bergström, 2002). Other wetlands are constructed for the improvement of water quality and act as green filters for effluents that originate from sources such as waste treatment and agriculture (Söderberg et al., 2000). The different types of wetlands used for water treatment can be grouped into three basic categories: constructed surface flow (SF) wetlands, constructed subsurface-flow (SSF) wetlands and natural wetlands (figure 1). Buffer zones can belong to either natural or constructed wetlands. They are an intermediate type of protective riparian zone that are preserved in their natural state or recreated where such areas have been destroyed.

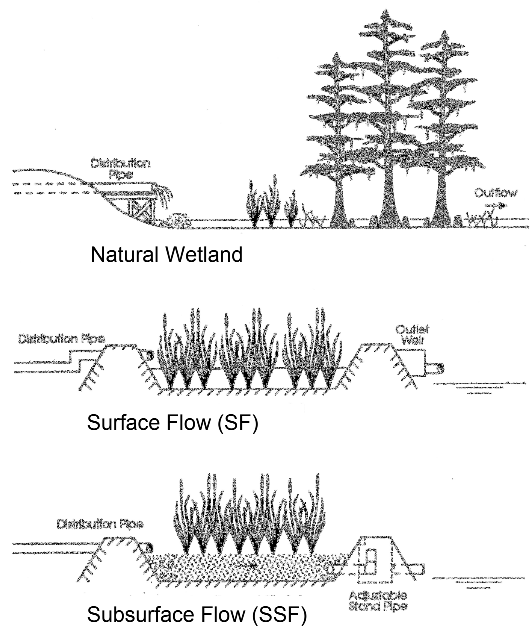


Figure 1. Treatment wetland types. From Kadelec and Knight (1996).

Constructed Wetlands

In Europe and the US constructed wetlands are mainly used for polishing the effluent of mechanically and chemically treated waste water from treatment plants. Wetlands can be designed in two ways related to how the water, or effluent, flows through the system. Surface flow (SF) wetlands are similar to natural vegetated wetland systems and may have open water areas. These wetlands are often planted with macrophyte vegetation and have specially designed inlet points and outlet points to promote even distribution of the effluent (Kadlec & Knight, 1996).

In sub-surface flow (SSF) wetlands the effluent flows below the soil surface through a bed of low permeability soil or gravel. The beds are planted with vegetation and the waste water comes in close contact with roots and the soil matrix with alternating aerobic and anaerobic sites. The water is cleaned through physical filtration and by biochemical processes (Kadlec and Knight, 1996).

Buffer zones

In many countries it has become common practice to leave zones of vegetation or "buffer zones" along the edges of waterways to take up nitrogen and other nutrients before they are transported into recipient waterways (Broadmeadow & Nisbet, 2004). Buffer zones are implemented in both agricultural and forest environments.

A buffer zone can either be an intact strip of vegetation left bordering a stream or watercourse, or an area created for filtering purposes. As with natural and constructed wetlands, the effectiveness of buffer zones depends on the area, soil type, vegetation and loading rates. Channeling can be a problem which reduces the uptake of nutrients and metals by reducing residence times and limiting the distribution of runoff (Väänänen et al. 2006). An important function of buffer zones is to reduce suspended solids and sediment. In steep areas, buffer zones should be made wider than the distance sediment can travel down slope. Forested buffer zones are susceptible to wind damage, and if the area is prone to strong winds the area should be increased (Chang, 2003).

In Finland, forest buffer zones were found to decrease the concentration of suspended solids by 70% (Nieminen et al., 2005) and NO_3-N by 50% (Vasander et al., 2003). While buffer zones reduce suspended solids and nitrogen concentrations, buffer zones on restored peatlands may cause increased P loading if the area has been previously fertilized with phosphorus (Vasander et al., 2003). Buffer zones also help to regulate temperature and shading in sensitive riparian ecosystems.

Peatlands

This study is mainly focused on a natural wetland in a forested ecosystem; a peatland. Peatlands may be classified in a number of ways related to their vegetation, chemistry or hydrology. The most straight forward method is hydrology based and defines bogs as peatlands where water input comes exclusively from precipitation (ombrotrophic). The result is that bogs are nutrient poor and have a low pH because of their isolation from mineral soil and ground water. Because of their special hydrology, natural bogs will only be affected by deposited nutrients. Fens, however, receive input from precipitation, ground water and runoff from adjacent mineral soils (minerotrophic). Fens have therefore higher pH and nutrient levels than bogs. Fens can be further sub-divided according to species richness and pH, encompassing poor (pH 4 – 5.5) to extremely rich fens (pH 7 – 8) (Mitsch & Gosselink, 2000; Rydin & Jeglum, 2006).

In natural bogs and mires nutrients are more carefully conserved than in other ecosystems. Hemond (1983) found in his thorough investigation of Thoreau's bog that even though N fluxes are small, almost 80% of the annual N input is retained. Another characteristic of mires is peat accumulation that results from low decomposition rates. Decomposition in bogs is lower than other ecosystems because of the cool, waterlogged, and acidic conditions prevalent in most mires. Peat accumulation can act as a long-term sink for nutrients like phosphorous (Richardson & Marshall, 1986). Organic compounds found in peat have a high amount of negatively charged sites and therefore a high cation exchange capacity (CEC) and ability to hold positively charged ions. The result is that there are relatively few cations available in the soil solution.

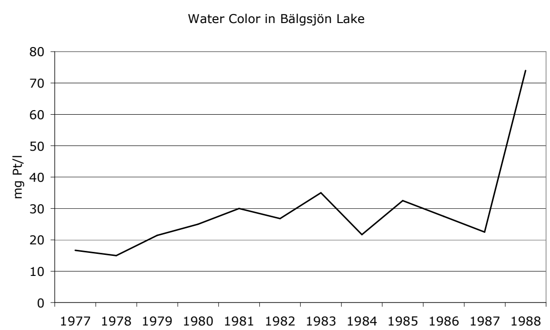


Figure 2. Average water color concentration in Bälgsjön Lake 1977 – 1988. Source: Länsstyrelsen Örebro.

Case Study: Bohyttan Fen

In 1987 the county administrative board in Nora received hundreds of complaints concerning the color of the drinking water originating from Bälgsjön Lake. The color of the raw water before treatment was on average 20 – 30 mg Pt l^{-1} (figure 2), but in 1987 and 1988 individual measurements of water color reached 85 and 100 mg Pt l^{-1} (Länsstyrelsen Örebro). Water color varies naturally with season and with the precipitation intensity and it is linked with iron concentration (Lundin, 1991). However, it was suspected that the unusually high water color was a result of the intense forestry activities around the lake. As mentioned previously, clear-cut areas and protective ditching alter hydrology and soil dynamics and can have a direct effect on water quality by increasing the amount of nutrients, metals and organic materials. Organic compounds can cause problems with water treatment. Chlorine used in water purification can react with the organic material in the water to form halogenated trihalomethanes

(THMs) which have been linked to cancer (Gopal et al., 2006).

The water color has not been the only issue of concern for Bälgsjön Lake. The Lake has also been limed to increase the pH of the acidified waters (background pH = 5.4) in an attempt to maintain the fishing population of Roach (*Rutilus rutilus*) (Länsstyrelsen Örebro, 2003). The Lake was limed from the 1970's to 2000, but direct liming of the lake has since ceased because it is now a protected watershed. Now only the smaller lakes upstream of Bälgsjön are limed in order to keep the pH of the lake above 6.

The water quality issues associated with Bälgsjön Lake made it an interesting site to study. More specifically, it was relevant to investigate the importance of a fen in the forested sub-catchment for the retention or release of organic material, metals, nitrogen and phosphorous to the lake.

Aim of Study

The purpose of this study is to investigate the importance of a fen in the flow of nutrients in a forest landscape and to put the results in context with other landscapes and wetland types.

This study is divided into two parts. The first part, you already read, explores water quality issues and the effects of different wetland parameters on water quality. The second part is a case study of a forested catchment in Sweden that investigates the effects of forestry management on the nutrient loading in streams and how the fen affects the flow of nutrients into the nearby lake.

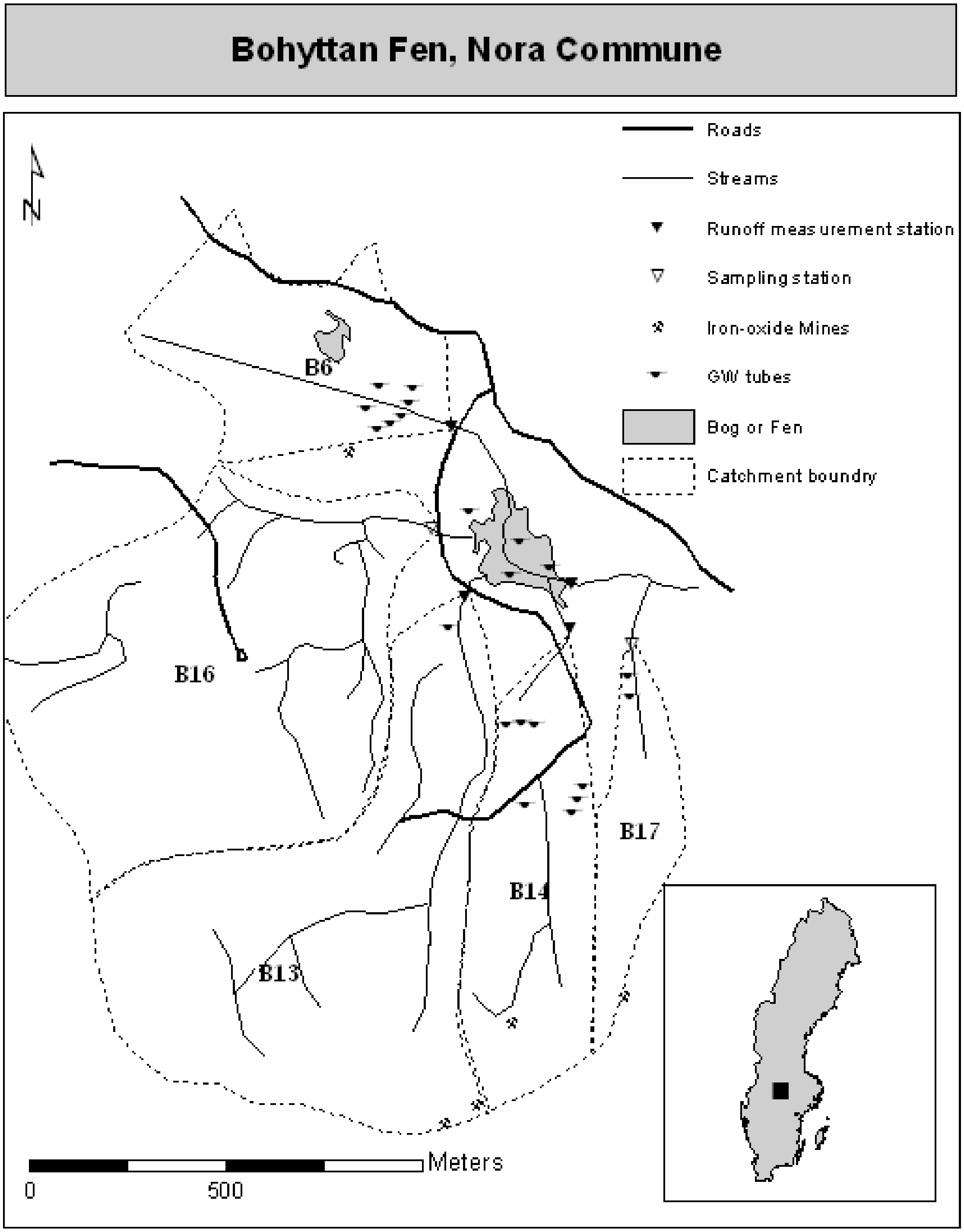


Figure 3. Bohyttan fen catchment. The total catchment is 243ha.

METHODS

Site description

General

The site of this investigation is the 243 ha Bohyttan catchment, located in Nora commune in central Sweden (N 59°34'; E 14°54'). The soil in this forested catchment is predominantly sandy loam and there are a number of peatlands scattered throughout the catchment. The catchment drains to a poor fen with an outlet to Balgsjön Lake (2.82 km^2), from which Nora derives its communal drinking water. The geology of the area is mainly acid sodium-rich metavulcanits (Lundin, 1998). The area around Nora is historically known for its mining industry. According to the Geological Survey of Sweden (2007) the Nora region is rich in iron-oxides. In sub-catchments B6, 13 and 14 are the remains of small iron-oxide quarries (figure 3). The climate is cold and moist (table 1) with a mean annual temperature of +5°C.

Sub-catchment description

The catchment area is divided into four sub-catchments ranging from 29-81 ha plus one control area outside the catchment (table 2). 40% of the forest in the catchment is less than 25 years old (table 3) while the rest of the area is made up of mature forest and mires. The tree layer is primarily *Pinus silvestris* and *Picea abies* with additional deciduous species. Of the four sub-catchments, B14 had the most intensive forestry management with 28% (8ha) of the area clear-cut, scarified or drained since 1995 (table 3, figure 5). In the northernmost sub-catchment, B6, 23% of the area was clear felled since the start of the observation period. This sub-catchment is also the site of the highest water color in the catchment area (table 2). Since 1992 sub-catchment B16 had 23% of its area harvested. B13 is over one third young forests and there were no forest activities since the observation period started. Another monitoring station

Table 1. Climate and hydrology at Bohyttan Fen (Raab and Vedin, 1995)

Annual mean temperature (°C)	+5	Precipitation (mm)	850
Length of vegetation period (days)	190	Runoff (mm)	400
Snow-covered period (days)	125	Evapotranspiration (mm)	450

Table 2. Sub-catchment areas, pH, water color and other characteristics

Catchment	Area (ha)	Characteristics	Average pH	Average water color (mg Pt/l)
B6	31	23% clear cut, fen and forested peat soils	4.42	294
B13	73	Mature & young forest, no recent clear-cutting	4.94	132
B16	81	Mostly young forest with some recent clear felling, 23%	4.71	148
B14	29	28% recently cut forest	5.31	93
Fen + surroundings	7 + 22	Wet soils and peatland	4.70	158
B7	243	Includes B6, B13, B16 & B14	4.70	158
B17	13	Mature forest, control	5.08	95

was set up in a nearby stream in an old coniferous forest where there have been no forestry activities (B17).

Table 3. Forestry activities for each sub-catchment. Bold text indicates forestry activities completed during measurement period 1992-2005.

Sub-Catch ment	Management Practice	Area (ha)	% of sub-catch-ment	Date
B6	**Clear-cut**	**7**	**23**	**16/10/2000**
	Scarification	**7**	**23**	**27/9/2001**
B13	Clear-cut	8	11	1/1/1982
	Clear-cut	19	26	1/10/1986
B16	Clear-cut	22	27	1/1/1982
	Clear-cut	**10**	**12**	**1/10/1998**
	Scarification	**10**	**12**	**1/10/1999**
	Clear-cut	**9**	**11**	**5/10/2003**
	Scarification	**9**	**11**	**1/12/2004**
B14	Clear-cut	7	24	1/1/1973
	Clear-cut	6	21	1/10/1986
	Clear-cut	**5**	**17**	**1/6/1995**
	Ditching	**3**	**10**	**1/8/1995**
	Scarification	**2**	**7**	**1/9/1996**
	Clear-cut	**3**	**10**	**15/9/2003**
	Scarification	**3**	**10**	**15/11/2004**
B7	**Clear-cut**	**4**	**14**	**1/6/1995**
Total catchment cut 1992-2005:		**38**	**16**	

Fen description

Bohyttan fen is an open poor fen with a layer of sedge peat that varies from 0.5 – 2.0 m (Lundin, 1998). The ground water table in the fen is on average 10 cm below the surface, but varies from 10cm over the fen surface to -40 cm below (figure 4).

There is widespread coverage of *Sphagnum* species (*S. palustre*, *S. fallax*) along with sedges like *Carex lasiocarpa* and cottongrass *(Eriophorum vaginatum)*. Other vascular plants found on the site include buckbean (*Menyanthes trifoliata*), heather (*Calluna vulgaris*), cranberry (*Vaccinium oxycoccos*), bog rosemary (*Andromeda polifolia*) and the dwarf marsh violet (*Viola epipsila*). The nitrogen-fixing shrub sweetgale (*Myrica gale)* along with a few small birches (*Betula*) and pines (*Pinus sylvestris*) also grow in the fen.

Sampling procedures and calculations

Water sampling

The flow of water into and out of the fen was measured by V-notch weirs and water level recorders. The water flow measurement stations were located on three of the four sub-catchments (B6, B13 and B14) that enter the fen and on the stream outlet (B7) leaving the fen. Water flow measurements were continuously recorded except during periods of ice. When the watercourses were frozen, the measurements were taken weekly and then daily values were calculated. The discharge was determined using the Thompson equation (Anon, 1979). The discharge was not measured in sub-catchment B16, and was estimated using a relationship to the adjacent sub-catchment B13.

Water samples were taken about once a month from February 1992 onward. In the sub-catchment used as a control, B17, the samples were generally taken every second month. The water samples were taken in plastic bottles that were rinsed 2-3 times first with the sample stream water. Care was taken to not disturb the bottom sediment and thereby contaminate the water samples. The water samples were stored no more than two days before they were analyzed. pH, HCO_3, DOC, Si, Na, K, Ca, Mg, Fe, Al, Mn, Cl, SO_4, PO_4-P, Tot-P, NO_3-N, NH_4-N, Tot-N and water color were determined by chemical analysis according to Swedish standards (SIS, 1986) (table 4).

There are 21 tubes installed in the catchment for measuring the ground water level: 7 in B6, 7 in B14, 4 in the fen, one in B13 and two in B17 (figure 3). Ground water level measurements were generally carried out twice a month (figure 4).

Peat Sampling

Peat core samples were taken at four depths and at three different sites in the fen. Peat core samples were taken at the following depths: 0-30cm, 30-50 cm, 50-100 cm and 100-150cm. For the depths 0-30 cm and 30-50 cm 5 core samples were taken at each site. For the depths 50-100 and 100-150 2 core samples were taken. The samples taken at the same site and at the same depth were mixed together and kept in plastic bags. Peat core samples were taken at three different sites in the fen: site 1 was at the edge of the fen, 1-2 m from the mineral soil uplands; site 3 was 1 m from the edge of the stream flowing through the fen; and site 2 was in-

Table 4. Sample analysis methods

Sample	Analysis	Reference
DOC	Carlo Erba analyzer	Pella and Colombo (1973)
Base cations & Si	Inductively coupled plasma- emission spectrometry (ICP)	
Cl, SO_4	Ion Chromatography (IC)	
NH_4^+, NO_3^-	Flow Injection Analysis (FIA)	Ruzicka and Hansen (1975)
P	Spectrophotometric method (Mo-blue)	Murphy and Riley (1962)

Table 5. Peat chemistry of samples taken at four different depths at three different sites. Site 1 was close to the edge of the fen, while site 3 was approximately one meter from the stream flowing through the middle of the fen.

Sample site & depth (cm)	pH (H_2O)	SO_4 (mg/100g)	NH_4^+ (mg/100g)	NO_3^- (mg/100g)	%C	%N
1: 0-30	4.54	63.0	0.62	0.18	48.3	2.13
1: 30-50	4.66	20.2	0.58	0	52.6	1.92
1: 50-100	4.68	14.3	0.87	0	50.7	1.85
1: 100-130	4.62	21.1	0.73	0	46	1.71
2: 0-30	4.45	87.6	-	0	47.5	2.04
2: 30-50	4.54	30.6	0.67	0	52.9	1.87
2: 50-100	4.58	21.7	0.57	0	51.2	1.93
2: 100-150	4.57	22.8	0.51	0	35.1	1.49
3: 0-30	4.54	101.6	-	0	38.4	1.94
3: 30-50	4.68	38.6	0.62	0	35.4	1.74
3: 50-100	4.81	35.1	0.57	0	33.2	1.55
3: 100-150	4.90	29.3	0.41	0	23	1.07

between sites 1 and 3 at a groundwater measuring station. After collection the samples were kept in cold storage (<5° C) until analysis. Part of the fresh sample was used for the determination of NH_4^+-N and NO_3^--N and the rest was dried and sieved. The fresh sample was refrigerated until the extraction was made 10 days later. The rest of the sample was dried two days after collection.

NH_4^+-N and NO_3^--N was extracted with 2 Molar KCl and the filtrate was analyzed, following standard methods (NH4-N method nr. GB-352-87 Rev.1; NO3-N method nr. GB112-94 Rev.1) on a Bran-Luebbe TrAAcs 800 autoanalyzer. An H_2O extraction of the dried peat was used to determine SO_4, Cl^-, and PO_4 using Ion Chromatography (ICS-90). The pH of the peat samples was also measured using the H_2O extraction. The dried peat was used to determine the percent nitrogen and carbon by incineration of the samples at 1250° C with a LECO CNS-1000 elemental analyzer.

Water and nutrient balances

The inputs to the fen are measured as stream inflows (R_{in}) from B6, 16, 13 and 14, and the output as the stream outflow (R_{out}) from B7. Only the waterborne nutrients were measured in this study, gas fluxes were not included. It was assumed that groundwater inputs and outputs were negligible and that evapotranspiration was minor for a fen of this size.

The nutrient budget for the fen and retention (RT) was calculated simply as the sum of the nutrient loads in R_{in} minus the load in R_{out}. The nutrient loading was estimated by combining the continuous stream flow measurements with the sampled water chemistry results. Where the sampling data was missing (see: Missing data), interpolated values were used. The nutrient load of each individual stream was multiplied by the fraction of the total catchment area. The loading of all input streams were summed to arrive at the total inputs (mass area^{-1} time^{-1}). For the fen and surrounding area (7 + 22 ha), loading values from the stream

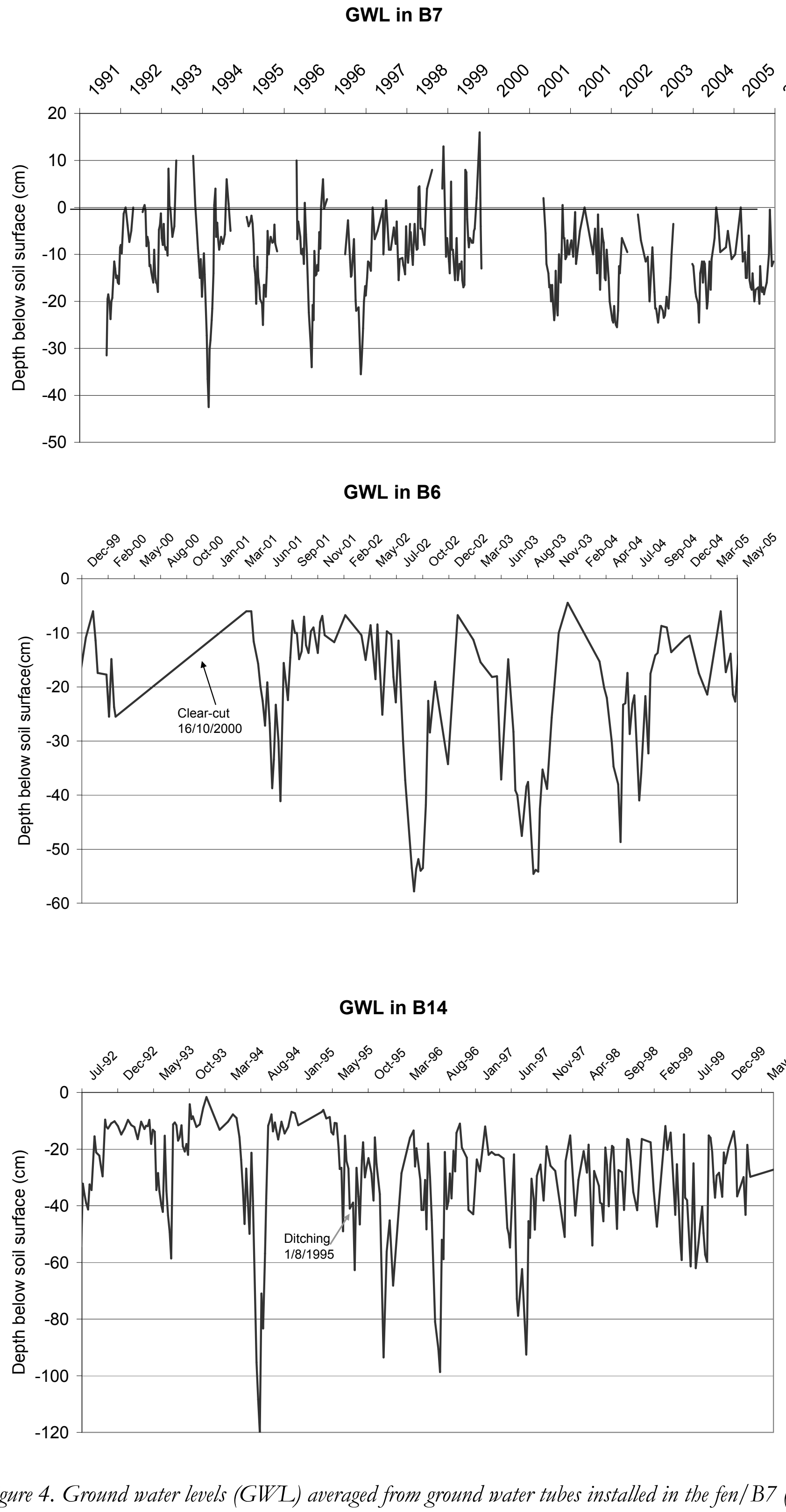


Figure 4. Ground water levels (GWL) averaged from ground water tubes installed in the fen/B7 (n = 4), B6 (n = 7), and B14 (n = 7). Measurements given in cm below soil surface. Note the different scales on y-axis and the different time frames on the x-axis. Ground water measurements were not taken in B6 from February 2000 to February 200.1

output, B7, were used. The retention was calculated:

$$\%RT = [(\Sigma R_{in} - R_{out}) / \Sigma R_{in}] \times 100$$

Except for the first two years of monitoring, the sum of the stream inflow (ΣR_{in}) was not equal to R_{out}. The possible reasons for this discrepancy are discussed later on. In order to correct for this inconsistency the outflow from B7 (R_{out}) was increased. For each year:

$$\Sigma R_{in} / R_{out} = X$$

X was a specific value for each year. The transport values for B7 were multiplied by X in order to even out the flow differences. The differences were not completely eliminated (Δ = 1.3%, table 10) because the outflow, B7, is also used for part of the inflow calculations.

Nutrient concentration and transport

Because of the natural variation in concentration and transport, it was not always apparent what fluctuations could be attributed to forest activities. Using catchment B17 as a control, the calibration period and control area technique was used. Except for pH, most regressions yielded poor correlations and so another method was tried to asses the effects of forestry on nutrient leaching and water quality.

Instead, the mean concentration over a three year period before forest treatment was calculated. Then a comparison was made to the averages of the years following treatment. In order to take into account the natural variation of substances the concentrations from B6 and B14 were normalized with the values from B13 in the following manner:

During the same time period, in both subcatchments (B6 and B13), the mean concentration after management was subtracted from the mean concentration before management resulting in Δ_{B6} and Δ_{B13}. Then their difference was calculated:

$$\Delta_{B6} - \Delta_{B13} = \Delta_{C} \text{ (corrected difference)}$$

Then Δ_C was divided with the mean concentration in B6 before management to arrive at the average difference.

Missing data

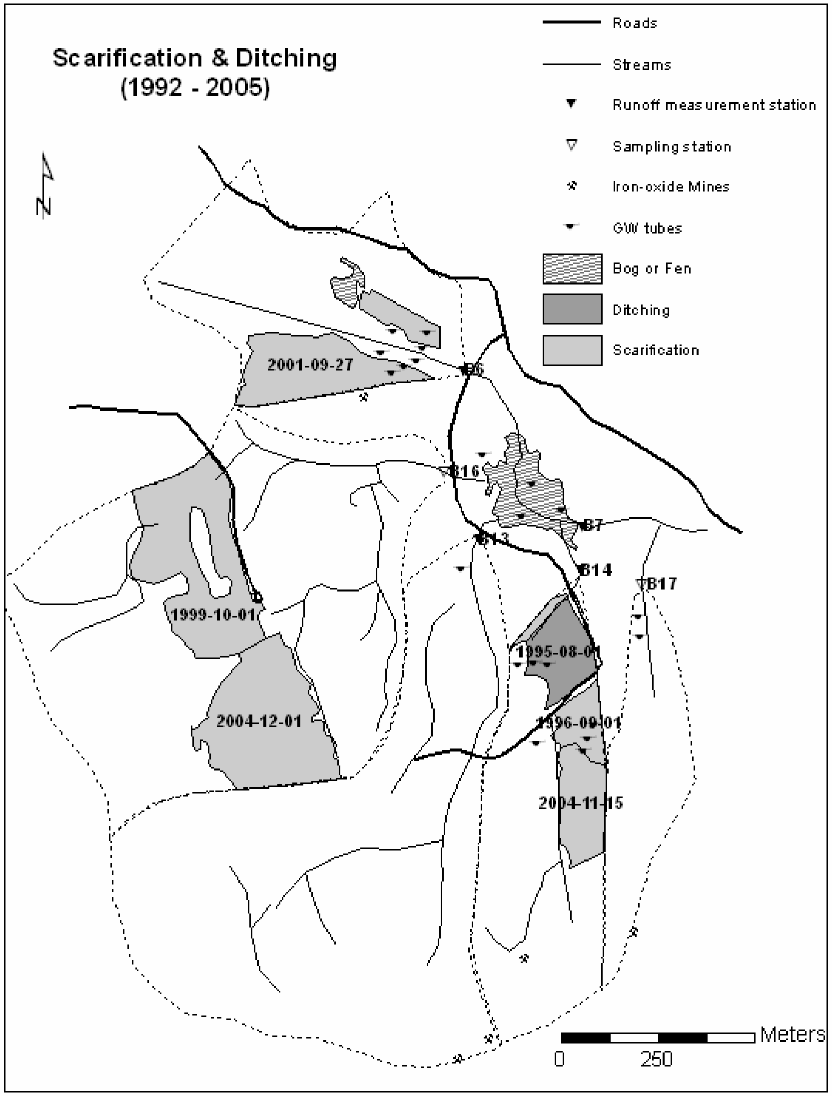
During a period of reorganization at SLU, some of the data from water samples were lost for the following dates:

Phosphorous (Tot-P, PO_4):

- 9/12/1995 – 6/7/1996
- 19/1/1998 – 13/12/1998

Nitrogen (all species):

- 16/6/1996 – 13/12/1997
  (with 2 exceptions)
- NH_4^+: 1/1/1998 – 30/12/1998



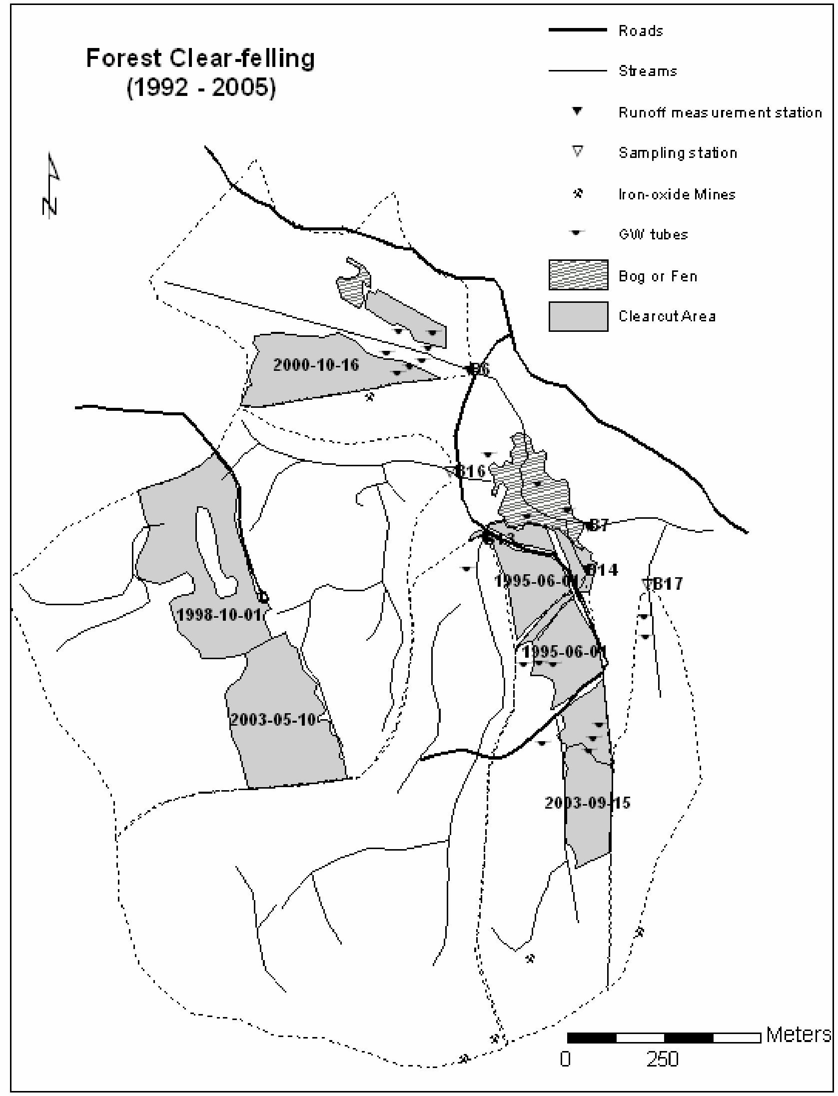


Figure 5. Scarification (left) and clear-cut (right) in Bobyttan catchment with dates of treatment

Results

Trends

General

The annual runoff in the Bohyttan catchment ranges from less than 15 mm per month to over 60 (figure 6). The highest periods of runoff in the catchment occur in April and from November to January. Notable is the fairly high runoff in winter. The summer months; June, July and August, are the driest. After 1998 the input into Bohyttan fen was always greater than the output (figure 7). The largest deficits (input > output) most often occur in the early spring months of February and March (figure 8).

The average monthly nitrate concentration is highest from January to March; just before the uptake by growing vegetation begins (figure 9). However the concentration of total nitrogen in stream water is at its maximum in the late summer. In addition, total phosphorous, DOC and water color all have their highest concentrations in the late summer (figure 10). This coincides with the lowest levels of runoff.

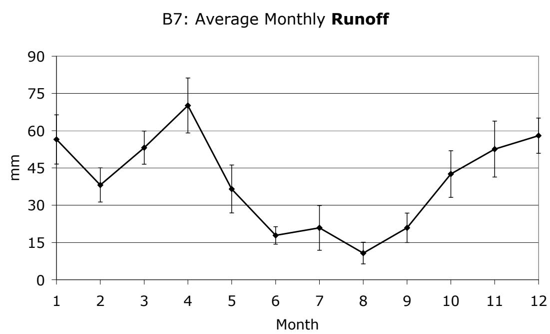


Figure 6. Monthly mean runoff from Bohyttan fen calculated from the years 1992 – 2005 (± 1 SE)

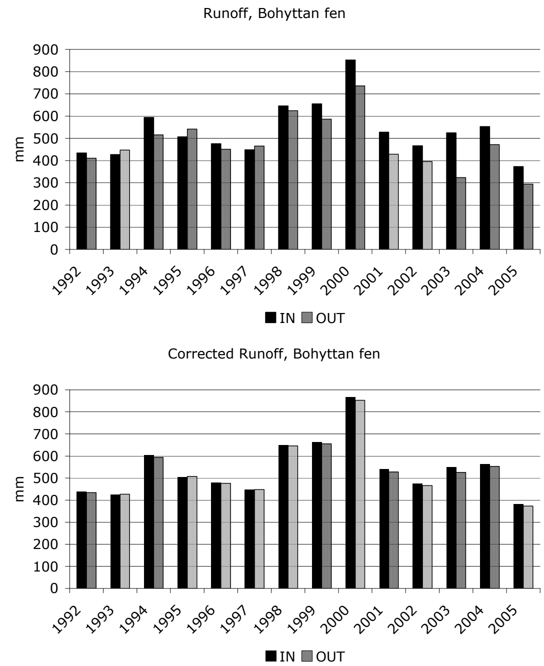


Figure 7. Above: Observed inflow to and outflow from Bohyttan fen. Below: Corrected runoff to and outflow from Bohyttan fen.

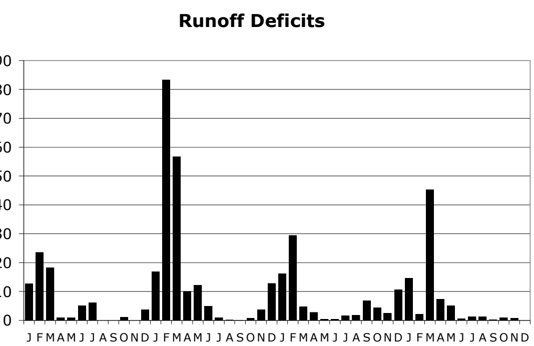


Figure 8. Runoff Deficits; INFLOW – OUTFLOW for the period 2002-2005.

The forest in Bohyttan catchment is actively managed and 40% of the catchment has been harvested since 1973. Forest revegetation was expected to lead to a decline in nitrate concentration. However no connection with the forest growth and declining nitrogen concentration levels was detected.

Sulfur, Iron and Water color

The concentration of SO_4 in the forested sub-catchment streams declined during the 14 year observation period (figure 11). There has also been an increase in iron concentrations in the stream water during the same period. This increase in iron coincides with the decrease in sulfate concentrations in the same sub-catchments. A strong correlation was found between iron and sulfate in the forested sub-catchments B13 and B16 (R^2 = 0.72 and 0.67; figure 12). Additionally, there was a strong correlation (R^2 = 0.86) in the concentration of iron in stream water and water color in sub-catchment B6 (figure 13). Sub-catchment B6 has by far the highest average water color of all the sub-catchments (table 2) and it also has the highest iron concentration. In this part of Sweden it is not unusual to find iron deposits and there are a number of small former iron-oxide mines found in the catchment (figure 3).

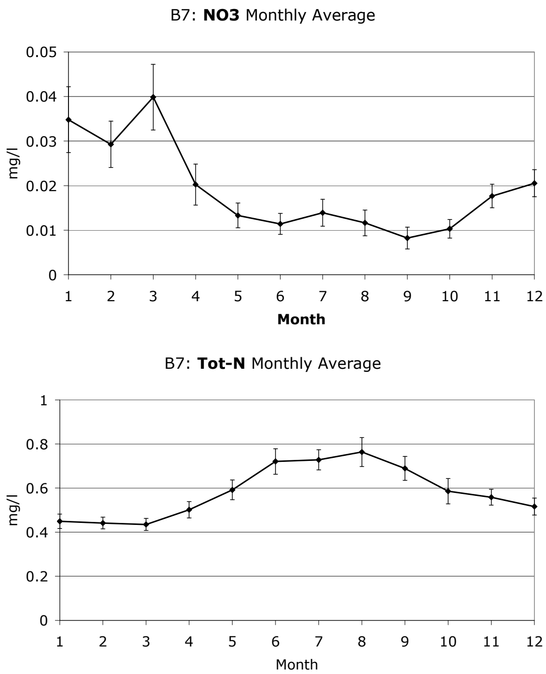


Figure 9. The monthly mean concentration of nitrate and total nitrogen in B7, calculated for the years 1992-2005 (including ± 1 SE).

In B6 there was also a strong correlation between water color and DOC (figure 14, $R^2 = 0.90$).

Forestry Practices

All of the sub-catchments, besides B13 and the control (B17), have experienced some type of forest management during the measurement period from 1992 to 2005. This study focuses on the forestry activities in sub-catchments B6 and B14.

In B6, 7 ha of forest was cut in October of 2000 and one year later the soil was scarified on the same site (figure 5). The harvested site is 23% of the sub catchment area (table 3).

In sub-catchment B14 there were two events of clear-cutting and soil scarification. The first clear-cutting occurred in June 1995 and in August of the same year 3 ha was ditched. One year later the remaining cleared land was scarified. The harvested site was 17 % of the sub-catchment area (table 3) and located near the stream outlet to the fen. The second forest cutting event in the same sub-catchment was in September 2003, comprising 10% of the sub-catchment. One year later the harvested site was scarified (figure 5).

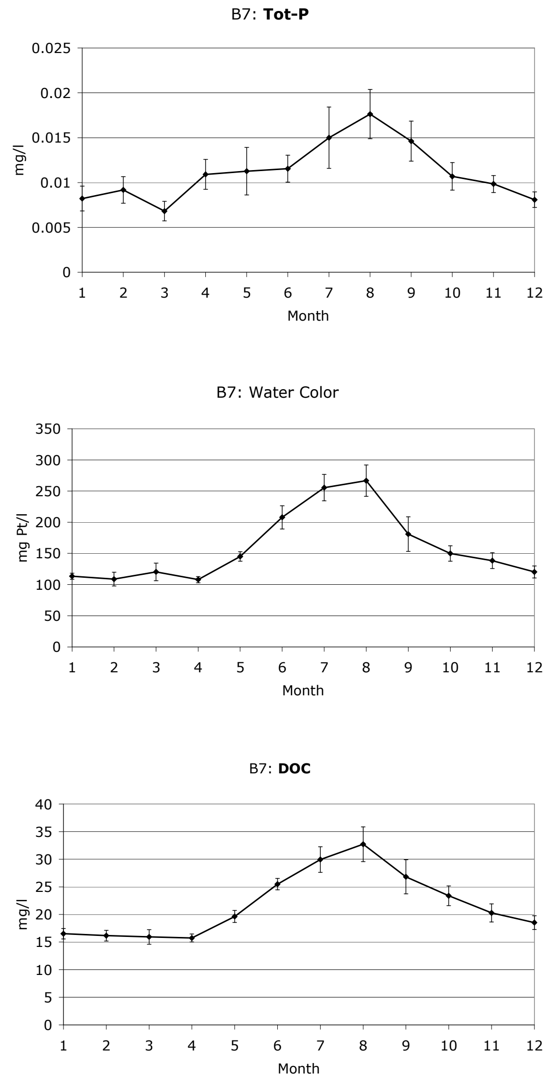


Figure 10. The monthly mean concentration of total P, water color and DOC in B7, calculated for the years 1992-2005 (including ± 1 SE)

pH

The calibration period and control area technique was used to discover if any relationship could be found between the harvested forest plots and the control catchment. When sub-catchment B6 was plotted against the control B17, no relationship was satisfactory to predict the development of nutrient loading. However, a strong correlation ($R^2 = 0.66$) in pH was found between B17 and B14, and was used to predict what the pH in B14 would have been without treatment (figure 15).

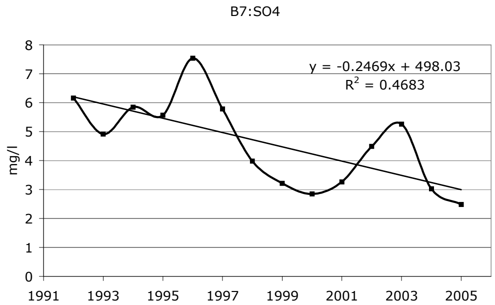


Figure 11. Decreasing SO_4 concentration in Catchment B7

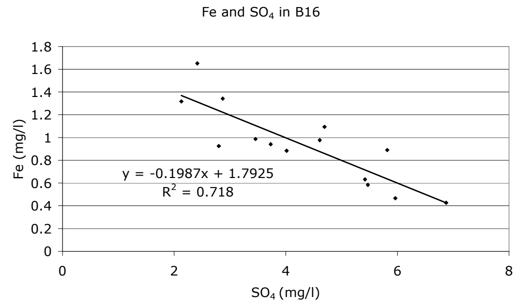


Figure 12. Relationship between iron and sulfate concentrations in sub-catchment B16.(R2 values for B13 and B17 are 0.67 and 0.61 respectively.)

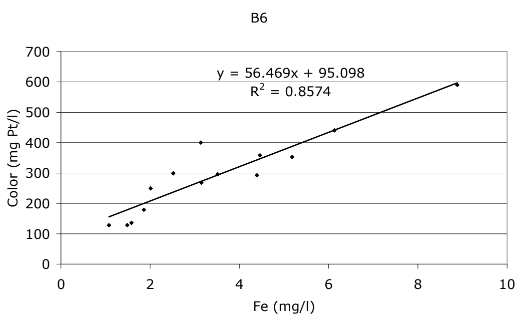


Figure 13. Relationship between water color and iron concentrations in B6 (1992-2005).

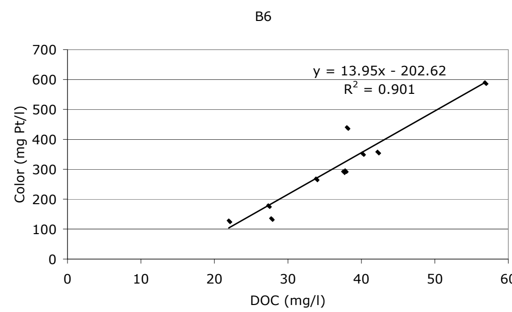


Figure 14. Relationship between water color and DOC in B6 (1992-2005).

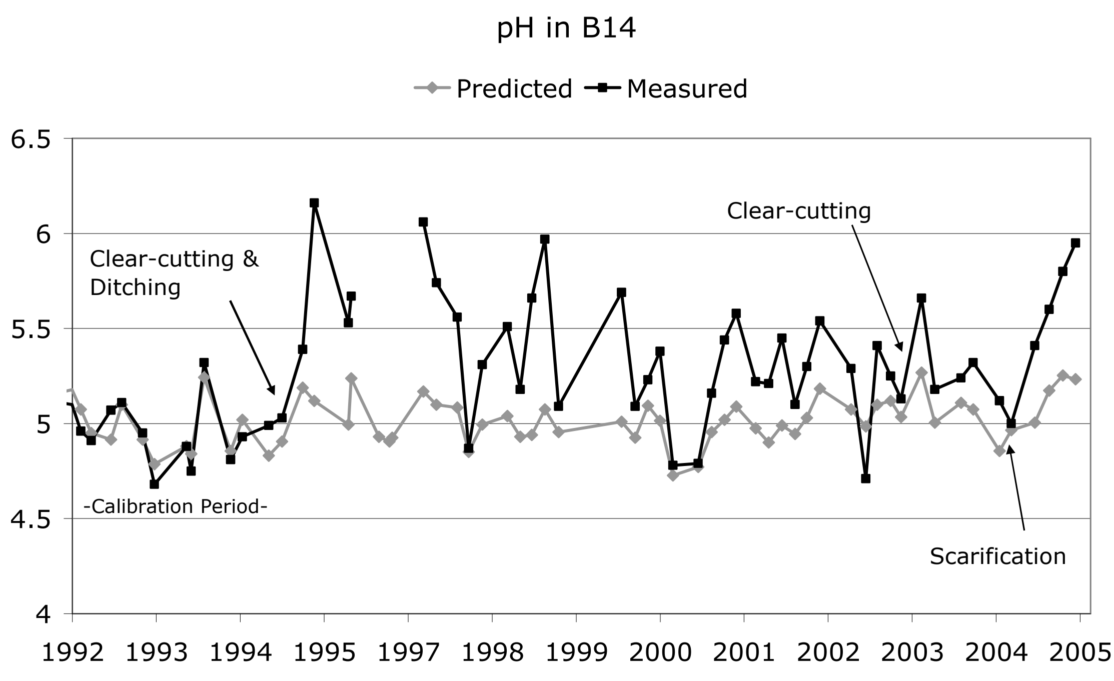


Figure 15. Measured and predicted pH values in B14. (Linear regression with B17: $y = 0.4962x + 2.4991$, $R^2 = 0.66$)

The pH increased from 5.1 to 6.2 after forest cutting and drainage in sub-catchment B14 and has since remained higher than the predicted values. The cutting in this area was followed by ditching to lower the groundwater table. It is likely that the lowering of the groundwater table caused the rise in pH.

Concentration

The mean concentration before and after forestry activities was calculated and normalized with concentration values from sub-catchment B13. One year after clear-cutting in B6, the concentration of many elements was lower than the average before harvest (table 6). Ca, Mg and Si returned to their pre-treatment values after about three years. In the first year, the water color, DOC, Tot-C, Al and Fe concentrations were all lower than the average concentration before harvest, but in the second year the concentrations increased to above average. In the third year the concentration values were again lower, followed by higher concentrations in the fourth year after clear cutting.

Nitrate concentrations after clear-cutting were on average over 80% higher than the three year average concentration before clear-cutting. How-

Table 6.The three year average concentration of elements in sub-catchment B6 before forest cutting compared with the average concentration for the first, second, third and fourth year after the operation. Concentration values were compared with the values from B13 as a control. Positive values indicate an increase in concentration and negative values indicate decrease in concentration. Clear-cut date 2000/10/16.

	Average Concentration (mg/l)	**Average Change**			
		2000-2001	**2001-2002**	**2002-2003**	**2003-2004**
pH (H_2O)	4.35	1%	-1%	-1%	0%
Conductivity	37.4	-9%	1%	4%	6%
Color (mg Pt/l)	305	-2%	25%	-51%	80%
DOC	38.5	-4%	16%	-41%	38%
Tot-C	40.1	-7%	16%	-43%	50%
Ca	2.49	-10%	-4%	-11%	8%
Mg	0.722	-12%	-8%	-21%	6%
K	0.296	62%	83%	68%	73%
Na	2.53	-4%	-2%	-2%	-1%
Fe	3.98	-20%	35%	-64%	105%
Al	0.811	-7%	13%	-35%	54%
Si	4.56	-1%	-8%	5%	7%
Cl	2.91	4%	13%	11%	3%
SO4	3.09	-19%	-10%	52%	22%
NO3-N	0.028	55%	37%	151%	109%
NH4-N	0.093	-18%	45%	-45%	-50%
Org-N	0.597	-8%	20%	-35%	-1%
Tot-N	0.718	-7%	24%	-29%	-3%
PO4-P	0.004	7%	6%	-29%	61%
Tot-P	0.025	-6%	15%	-36%	25%

Table 7. The three year average concentration of elements before forestry operations in sub-catchment B14 (Clear-cutting: 1995/6/1 and ditching: 1995/8/1) compared with the average concentration for the first, second, third and fourth year after the operations. Positive values indicate an increase in concentration and negative values indicate decrease in concentration. Concentration values were compared with the values from B13 as a control. Values for nitrogen concentrations are missing or few during the years 1996 & 1997.

	Average Concentration (mg/l)	**Average Difference**			
		(1995-96)	**(1996-97)**	**(1997-98)**	**(1998-99)**
pH	5.05	10%	13%	13%	11%
Conductivity	42.5	17%	10%	4%	2%
Color (mg Pt/l)	104	-18%	-8%	-19%	-19%
DOC	13.71	-51%	-14%	-34%	-21%
Tot-C	14.17	-41%	-6%	-21%	-15%
Ca-tot	2.818	21%	23%	23%	6%
Mg-tot	0.719	44%	31%	36%	11%
K-tot	0.574	61%	64%	44%	38%
Na-tot	2.705	14%	19%	10%	1%
Fe-tot	0.599	-41%	-29%	-44%	-57%
Al-tot	0.484	-5%	-6%	9%	17%
Si	4.229	13%	3%	4%	-5%
Cl	3.510	1%	1%	-7%	-7%
SO4	9.302	31%	22%	16%	-4%
NO3-N	0.016	491%	†	445%	369%
NH4-N	0.052	101%	†	-	69%
Org-N	0.509	38%	†	-2%	1%
Tot-N	0.577	56%	†	15%	17%
PO4-P	0.003	-48%*	30%	6%	20%
Tot-P	0.009	341%*	36%	0%	-39%

*only 3 measurements

† only 2 measurements during this period

ever nitrate makes up only a small fraction of the total nitrogen species in the catchment (figure 16) and the other nitrogen species concentrations showed different trends.

NH_4-N concentration was 50% lower than the pre-treatment average four years after clear-cutting. The concentration of Org-N and Tot-N was slightly lower the first year and the concentration during the second year after clear cutting was 20% higher than pre-treatment average. In the third year after clear-cutting the concentration of Org-N and Tot-N was 30% lower than pre-treatment. The average concentration the fourth year after clear-cutting, was about the same as pretreatment values.

Potassium (K) concentration was higher in the years following forest cutting. After four years the concentration of K was 73% higher than the period proceeding cutting. This increase in K loading to streams occurred in both B6 and B14.

The results in table 7 are a combination of the effects of both clear-cutting and ditching in B14. Here, in contrast with B6, the concentrations of elements were most often higher after the treatments than before. Water color, DOC, Tot-C and Fe were the exceptions with lower than average concentrations after management. The concentration of these elements had not returned to the pretreatment average after four years.

Table 8. Annual average loads in sub-catchment B6 calculated from the three years prior to clear-cutting (1997 – 2000) and the change in transport for the three years following clear-cutting (2000 – 2003).

B6	**Average transport (kg/ha)**	**Δ 2000 - 2003**
Ca	14.12	**3%**
Mg	4.48	**-1%**
K	2.57	**75%**
SO4	31.75	**7%**
DOC	138.75	**0%**
NH4-N	0.29	**1%**
NO3-N	0.19	**88%**
Org-N	2.28	**8%**
N-tot	2.76	**10%**
P-tot	0.06	**-1%**
Q (mm/yr)	637	8%

The nitrate concentration increased drastically following clear-cutting and ditching. It was unfortunate that there was data missing for nitrogen concentrations in the years following ditching and it is difficult to draw conclusions from the remaining data. Organic nitrogen concentrations were higher the first year, but then returned to pre-treatment levels in subsequent years. The concentration of total nitrogen was highest the first year after harvest and remained higher than pre-treatment average the in following years.

Total phosphorous concentration in the first two years following treatment was higher than pre-treatment levels, but in subsequent years the concentration fell below pretreatment levels.

Transport

The average annual loading was calculated for the three years before and for the three years after forest management. These values were compared with sub-catchment B13 and the percent change calculated for sub-catchments B6 and B14 (table 8 & 9) relative to B13. In sub-catchment B6 the most significant changes in nutrient transport after clear-cutting were for potassium and nitrate (table 8). Potassium loading increased 75% and nitrate loading was almost 90% higher than the average after three years.

In B14 the changes in nutrient loading after clear-cutting and ditching were much more pronounced (table 9). The clear-cutting and ditching were performed in 1995 and two three year averages (1995 – 1998 & 1998 – 2001) were calculated for the period after ditching. Both the total nitrogen and total phosphorous loading were higher than average during the first three years after forest management. However, during the following three years (1998 – 2001) the average N-tot returned to pre-treatment levels, while the P-tot loading was much lower than before clear-cutting and ditching. As in B6 the annual loading of nitrate increased dramatically in B14. In contrast to B6, the levels of DOC in B14 decreased in both time periods. The concentration during the last three years was 40% lower than the average before treatment.

Table 9. Annual average loads in sub-catchment B14 calculated from the three years prior to clear-cutting (1992 – 1995) and the change in transport three years following clear-cutting (1995 – 1998) and the following three years (1998 – 2001)

B14	**Average transport (kg/ha)**	**Δ 1995 - 1998**	**Δ 1998 - 2001**
Ca	15.23	**-3%**	**-5%**
Mg	3.29	**14%**	**7%**
K	3.27	**32%**	**11%**
SO4	47.77	**3%**	**-8%**
DOC	73.09	**-21%**	**-40%**
NH4-N	0.28	**33%**	**-8%**
NO3-N	0.05	**924%**	**848%**
Org-N	2.75	**14%**	**-10%**
N-tot	3.07	**31%**	**4%**
P-tot	0.05	**22%**	**-45%**
Q (mm/yr)	554	-6%	-22%

Nutrient Balances

Element balances were calculated for the Bohyttan fen. As mentioned previously, the runoff inflow was not often equal to the outflow from the fen. The average yearly difference of the inflow and outflow during the period from 1992 to 2005 was 11% (table 11). Most years the inflow was greater than the outflow. The difference seemed to increase after 2000 with 2003 showing the greatest discrepancy (figure 7). The outflow from B7 was increased (or decreased) in order to reduce this margin of error. With the corrected results the yearly difference between inflow and outflow was reduced to 1% (table 10). Hereafter nutrient balance results referred to will be the corrected version unless otherwise stated.

The calculated nutrient balance (table 10) shows that the fen acted both as a sink and a source for different nutrients. Some elements, like aluminum, calcium, DOC, total phosphorous (Tot-P) and total nitrogen (Tot-N), appeared to be unaffected by the wetland. The fen acted as a sink for nitrate and ammonium, retaining almost 34 % in the case of nitrate and 6% in the case of ammonium. However, the fen exported enough organic nitrogen (6.5%) to make up the difference in the total nitrogen budget. Orthophosphate (PO_4^{3-}) and potassium were also retained in the fen, 5% and 8% respectively. The fen acted as a source for the metals iron and manganese, and as a sink for copper.

Annual budget trends

The average annual transport to the fen was dominated by the org-N fraction (84%) with NH_4-N (9%) and NO_3-N (7%) making up a much

Table 10. ***Corrected*** *annual average nutrient balances in Bohyttan fen, including runoff (q), from 1992-2005. Unless otherwise noted, the amounts are given in kg · ha⁻¹. Positive differences indicate retention, while negative values indicate export from the fen.*

	IN	OUT	Δ
q (mm)	542	535	1.3%
K	2.58	2.37	8.2%
Ca	11.2	10.8	3.5%
Mg	3.52	3.64	-3.2%
Na	12.9	12.4	4.2%
Fe	5.82	6.33	-8.8%
Al	2.52	2.53	-0.7%
Mn	0.05	0.09	-75.1%
Si	21.3	20.7	2.8%
Cu	0.42	0.40	5.2%
SO4	26.6	24.8	6.9%
Cl	13.9	14.4	-3.6%
DOC	106	108	-2.4%
NH_4-N	0.24	0.22	6.1%
NO_3-N	0.20	0.13	33.9%
Org-N	2.39	2.54	-6.5%
Tot-N	2.83	2.90	-2.6%
PO_4-P	0.02	0.01	5.0%
Tot-P	0.05	0.05	1.2%

Table 11. ***Uncorrected*** *annual average nutrient balances and runoff from Bohyttan fen (1992-2005) including runoff (q). Unless otherwise noted, the amounts are given in kg · ha⁻¹. Positive differences indicate retention, while negative values indicate export from the fen.*

	IN	OUT	Δ
q (mm)	535	478	10.6%
K	2.55	2.10	17.5%
Ca	11.1	9.7	12.7%
Mg	3.48	3.29	5.6%
Na	12.8	11.0	14.2%
Fe	5.74	5.63	1.9%
Al	2.49	2.28	8.1%
Mn	0.05	0.09	-84.3%
Si	21.0	18.3	12.8%
Cu	0.42	0.38	8.6%
SO4	26.3	22.3	15.3%
Cl	13.7	12.9	5.8%
DOC	104	97	7.5%
NH_4-N	0.25	0.23	9.0%
NO_3-N	0.20	0.12	40.8%
Org-N	2.36	2.29	2.9%
Tot-N	2.79	2.60	6.7%
PO_4-P	0.02	0.01	14.3%
Tot-P	0.05	0.05	11.0%

smaller part of the total nitrogen input (2.8 kg ha^{-1} yr^{-1}) (figure 16). The average annual N output from the fen was 2.9 kg ha^{-1} yr^{-1} and was primarily in the form of org-N (88%) followed by NH_4-N (8%) and NO_3-N (4%).

Transport of NO_3-N and NH_4-N had the most variation between years (figure 17) while the transport of org-N remained generally constant throughout the observation period. The annual minimum and maximum transport values for NO_3-N were 0.09 and 0.5 (kg ha^{-1} yr^{-1}), and for NH_4-N the transport values ranged from 0.15 to 0.5 (kg ha^{-1} yr^{-1}).

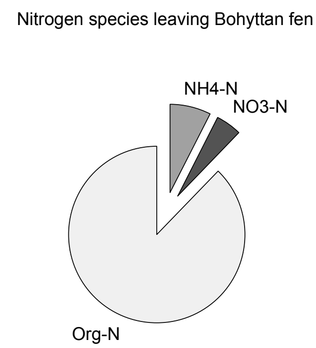
Without exception, the annual input of NO_3-N was higher than the output, meaning that the fen was a reliable sink for nitrate. For ammonium the pattern is more variable; some years the fen acted as a sink and other years a source. Except for one year, the output of org-N was higher than the input and so the fen was a source for org-N.

In the first eight years of monitoring (1992-1999), the fen was both a sink and source for PO_4-P (figure 18). In the last five years, the input of PO_4-P was always greater than the output from the fen. There were no clear trends for the tot-P balances.

With regard to metals, in 9 out of 14 years there was a net export of Fe from the fen. For Cu, there was net retention in the fen during 12 of the 14 years studied (figure 18).

The annual average transport of SO_4 to the fen was larger than the export. Thus, the fen was a sink for sulfate (7%). Net DOC export from the fen was slightly larger than the input (-2%) but there were a few years where the input exceeded the output (figure 18).

The nutrient budget for Bohyttan fen was compared to three other peatlands in the US and Canada (table 13). The peatlands all vary in size and nutrient loading, in addition to climate, soil and other factors. One thing common to all catchments is the retention of nitrate, ammonium and phosphate. Tot-N and Tot-P retention was almost non-existent in Bohyttan fen, while in other wetlands it was over 60%.



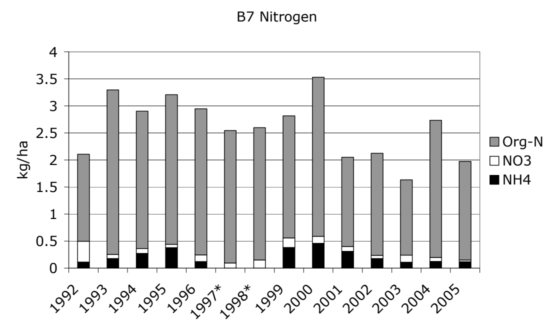


*Figure 16. Average uncorrected nitrogen transport (kg ha^{-1}) leaving Bohyttan fen. *Measurements of ammonium ion concentration for the years 1997 & 1998 are missing, not zero.*

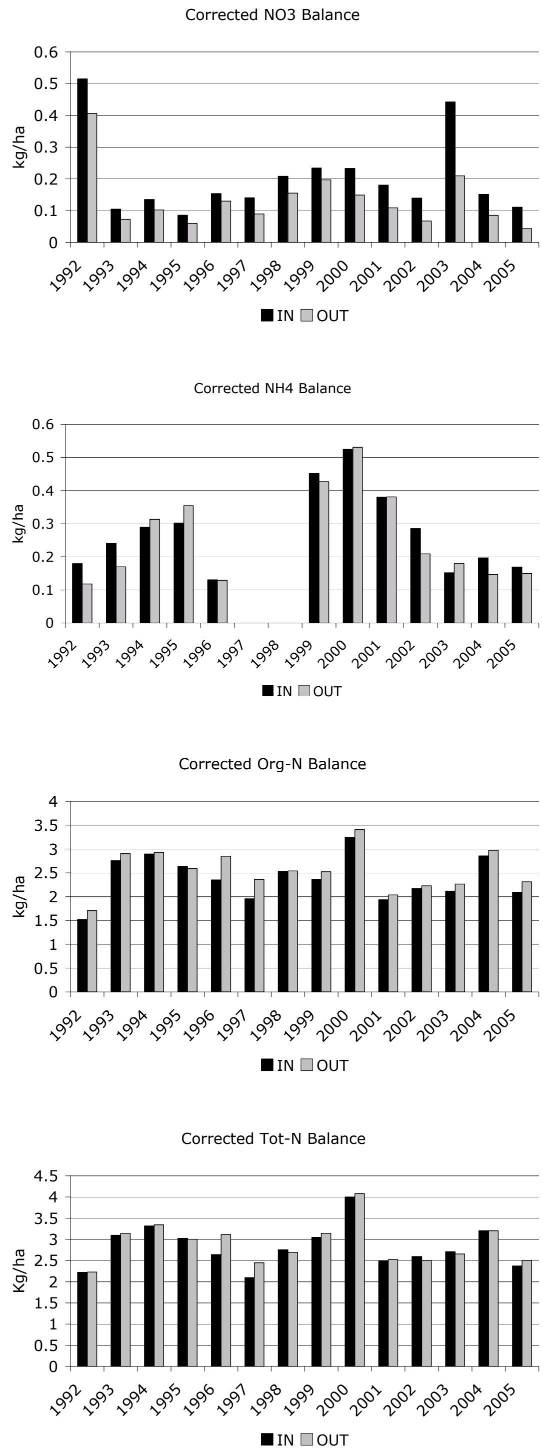


Figure 17. Nutrient balances for the different species of nitrogen entering and leaving the fen. Observe the different scales on the y-axis and that Data is missing for NH_4 in 1997 & 1998, not 0.

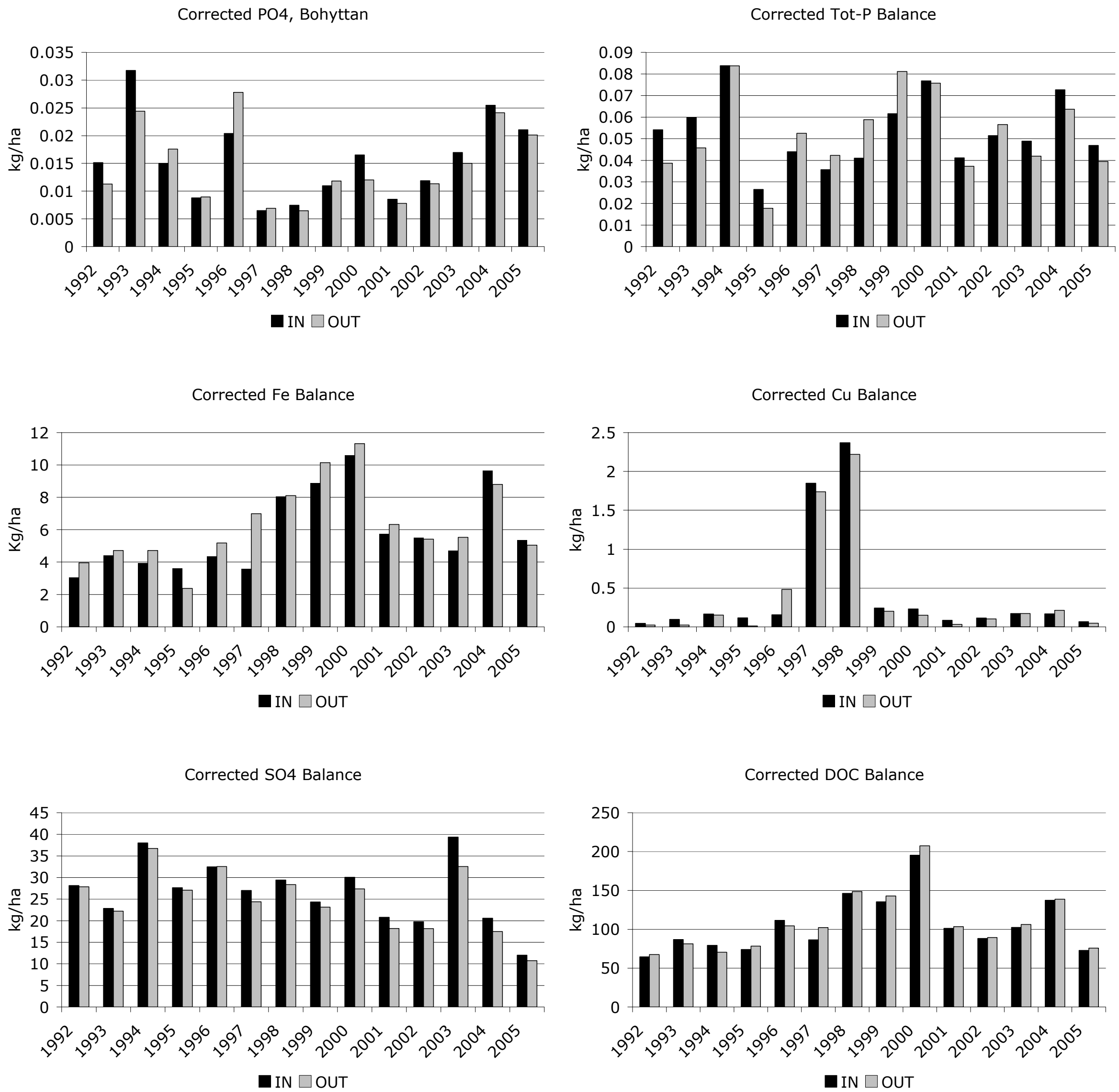


Figure 18. Annual nutrient balances for PO_4, Fe, Cu, SO_4 and DOC (dissolved organic carbon) in Bohyttan 1992-2005.

Table 13. Nutrient budget for Bohyttan fen, Marcell Experimental Forest, Plastic Lake and Paint Lake

Reference	Nutrient (kg ha^{-1} yr^{-1})					
	Total N	Total P	NO_3	NH_4	PO_4	K
This paper	Bohyttan fen, Nora Sweden (7 ha)					
Input*	2.83	0.053	0.20	0.24	0.016	2.58
Output*	2.90	0.053	0.13	0.22	0.015	2.37
Retention	-3 %	1 %	34 %	6 %	5 %	8 %
Verry & Timmons 1982	Marcell Experimental Forest, Minnesota USA (Black Spruce peatland, 3.24 ha)					
Input	12.696	1.167	2.041	2.249	0.383	11.059
Output	6.374	0.460	0.278	0.706	0.154	6.120
Retention	50 %	61 %	86 %	69 %	60 %	45 %
Devito & Dillon 1993	Plastic Lake, Ontario Canada (*Sphagnum*-conifer swamp, 2.12 ha)					
Input	16.4	0.292	6.4	3.7	0.086	---
Output	14.7	0.281	3.1	0.4	0.04	---
Retention	10 %	4 %	51 %	89 %	53 %	---
Devito et al. 1989	Paint Lake catchment #1, Ontario Canada (sedge fen, 0.1 ha)					
Input	105	2.97	35.7	3.93	---	---
Output	99.4	2.47	27.0	2.60	---	---
Retention	5 %	17 %	24 %	34 %	---	---

* 1992-2005 Average (corrected)

DISCUSSION

Discharge

Runoff balance discrepancies

The inflow and the outflow of water from the fen was calculated with v-notch weirs and water level recorders, but there may have been some ungauged runoff. A dam was built on the eastern side of the fen to form a water reservoir for a mill. This dam is probably not water-tight and some of the inflow could be expected to leave the fen via leaks in the dam. Along the eastern part of sub-catchment B14 (figure 3) a ditch was dug to install the weir and channelize the stream water. However, this ditch is known to occasionally overflow its banks and some water from the sub-catchment would then be lost.

As mentioned earlier, the greatest discrepancies occur during the early spring concurrently with freezing periods and snow melt (figure 8). It is possible that runoff generated during these periods while the ground is frozen is more likely to overflow stream banks, adding to the ungauged runoff. The deviation can also be due to error in the calculation of runoff during frozen stream conditions. A small overestimate in the calculation of each of the streams would compound the error for the total inflow to the fen. As these are cold months, it is unlikely that evapotranspiration is a significant contributor to the measured deficit.

Trends

Runoff

An increase in temperature due to climate change seems to have influenced runoff patterns. In the past, there was little runoff in the late winter months and then a sharp runoff peak in April coinciding with snowmelt. Because of rising average temperatures, more runoff occurs in the late winter months of Feburary and March, and in the fall. The result is a less dramatic runoff peak in the spring. The Bohytttan catchment has only been monitored since 1992 but measurements at Buskbäcken, located 40 km to the north east, date back to the 1970's. A comparison of the monthly runoff averages there during the period 1972 – 1987 and 1988 – 2000 shows a decease in runoff in April and a higher early spring average in the more recent period compared with the previous (figure 19). This effect of climate change was modeled by Forsius (2004) using data from ICP IM. His climate change scenario was calculated for 2050, but is already relevant today.

Sulfate

The decreased concentration of sulfate in the forest streams of Bohyttan catchment is probably a result of decreased atmospheric deposition of S. Environmental control programs restricting emissions are the most likely cause of reduced deposition. The international UNECE monitoring programs ICP IM and ICP Waters have both detected a decline in the sulfate concentration of streams in Europe and North America (Kleemola & Forsius, 2006; Skjelkvåle et al., 2005), and have attributed the downward trend in sulfate concentration to the decrease in atmospheric deposition.

The soil organic carbon status may also effect the leaching of SO_4^{2-}. Gustafsson and Jacks (1993) suggested that SO_4^{2-} adsorption could be predicted from the concentrations of extractible Fe/Al, TOC, SO_4^{2-}, and pH. The degree of SO_4^{2-} saturation in the soil is also an important factor affecting the extent of SO_4^{2-} sorption, but no measurements of extractable SO_4^{2-} in the forest soil were made. The chemical analysis of peat taken from the fen (table 5) shows that sulfate in top 30 cm was almost three times as high as further down in the profile. However, because the fen acted as a sink for sulfate (6.9%) it is unlikely that the fen surface was completely saturated with sulfate.

An increase in iron concentration in the surface water outflow was observed concurrently with the decrease in SO_4^{2-}. Sulfur in the soil and soil water can bind with iron and form, for example, iron sulfide (FeS). A reduction in the amount of sulfur deposited may have caused a shift in chemical equilibrium, and iron released from sulfur complexes into the soil water.

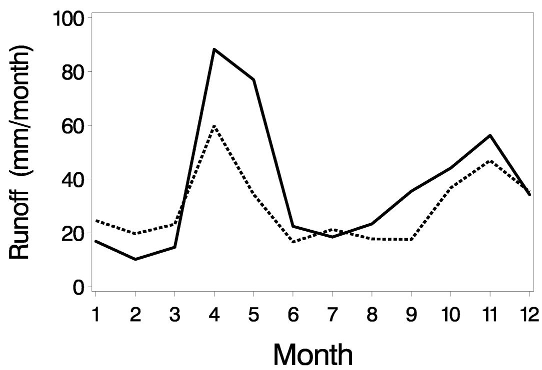


Figure 19. Monthly average runoff from. Buskbäckens catchment during the period 1972-1987 (black line), compared with the period 1988-2000 (dashed line).

Water color and Fe & DOC

Water color is often used as an indicator of the amount of organic content in water. In the soil, the water color is highest in the illuvial horizon where organic material and metals are precipitated or deposited. Metals like iron are also strongly sorbed to organic matter and high water color levels increase concurrently with DOC and Fe concentration (Lundin 1991). The results from B6 show this relationship very well with a strong correlation between water color and iron, and water color and DOC (R^2= 0.86 and 0.90 respectively).

Forestry Practices

The magnitude of nutrient leaching from forest soils depends on hydrology, climate and the mineralogy of the site (Rosén et al., 1996). Clear-cutting tends to cause a decrease in pH, an increase in water runoff and the leaching of metals, org-N and DOC. Subsequent scarification incorporates organic matter in the soil which stimulates microbial growth and mineralization (Norrström, 2002). However, drainage operations can reverse some of these effects, for example lowering DOC and raising pH (Ramberg, 1981). Only 16% of the total forest catchment was cut during the measurement period. This study focuses mainly on two management events: clear-felling in B6 where 23% of the sub-catchment was harvested, and clear-felling and ditching in B14 where 17% of the sub-catchment was harvested. The results in water chemistry differ for the two catchments and while site characteristics play a part, the main reason is probably the difference in management (i.e. ditching).

pH

The pH of stream water after clear-cutting was expected to decrease but in B6 it remained generally unchanged after clear-cutting and no change in the groundwater level was observed (figure 4). However, there was an increase in pH after clear-cutting and subsequent ditching in sub-catchment B14 (figure 15). When the groundwater is lowered by ditching or trenching, the infiltrating water percolates through more of the mineral soil. The longer contact time with mineral soil raises the pH of the drainage water. In saturated conditions after clear-felling, the water instead runs over the acidic surface of the humus layer causing a drop in pH. The elevated pH in the stream water of B14 will continue as long as the ditches in the area are functioning. In comparison, after the clear-cutting in B6, there was no change in the pH of the stream water. Decomposition of forest residues after logging increases the amount of base cations and thereby causes a rise in pH. However high water levels may mitigate this effect.

An increase in pH can lead to changes in ion concentrations. At higher pH there are less positively charges sites (anion exchange capacity) in the soil and this might partially contribute to the increase in SO_4^{2-} and NO_3^- concentration after ditching.

Concentration and Transport: Common trends

The only common trends in concentration after management in sub-catchments B6 and B14 were for K and NO_3-N. In both sub-catchments the concentration and transport of NO_3-N and K was clearly higher than before management. For the other elements the results are often conflicting, probably as a result of the differences in ground water levels. Part of sub-catchment B14 was ditched and had lower ground water levels than B6. The concentrations of Ca and Mg, for example were increasing in B14 and decreasing in B6.

Nitrogen

Even though the concentration of total nitrogen was variable, it was clearly higher than before clear-cutting. Rosén et al. (1996) also found increased levels of nitrate after clear-cutting, along with tot-N and K.

The clear-cutting of vegetation decreases the competition for nutrients and can lead to excess nitrate, as observed in the years after clear-cutting. The warmer soil temperatures on a clear-cut site stimulate decomposition resulting in mineralization and accumulation of ammonium, while possibly immobilizing nitrate. Biological processes operating in the soil, like competition for limiting nutrients may prevent, or at least delay nitrifying bacteria from being established (Vitousek & Melillo, 1979; Vitousek et al. 1979; Tamm et al., 1974).

In both sub-catchments the concentration of nitrate was higher than the background levels four years after management. The increase in nitrate concentration and transport was much lower in B6 than in B14. This can be due to differences in site characteristics. On more fertile sites, there is less available NO_3^- because pioneer plants begin using the excess nitrogen in the soil and competition for nutrients increases (Vitousek, 1981). B6 could have had lower mineralization rates than B14 because of greater amounts of logging residue. The high C:N ratio from woody debris increases the immobilization of nitrogen.

Wiklander et al. (1991) reported elevated values of NO_3-N in the spring water of a clear-cut area four years felling. However, the forested catchment in south Sweden studied by Wiklander et al. (1991) had a much higher percentage of NO_3-N (83% of tot-N) leaching, in contrast to Bohyttan fen (7% of tot-N). The area studied by Wiklander receives elevated levels of N through atmospheric deposition and this is likely the reason for the high levels of nitrate leaching.

A trend for NH_4-N was hard to establish, in part due to missing data that affected the results in B14. The concentration of NH_4-N in B6 was generally lower than pre-treatment, but no net change in the transport average three years after treatment was seen. In B6 the concentration first decreased, then increased, and the third year decreased again. This can be due to changes in the rates of mineralization and immobilization. Once the nitrifiers have become established, the concentration of ammonium should decrease as it is used by the nitrifiers. Such a case could be responsible for the decrease in ammonium which is seen in years 3 and 4 following the forest harvest in B6.

Potassium

Potassium concentrations and transport was higher after clear-cutting in both sub-catchments. This is consistent with the findings of Martin et al. (1984) where K concentrations were higher in catchments where at least 20% of the area was clear-cut. The increase in K may originate from logging residues, especially decomposition of needles. Palviainen et al. (2004) found that 90% of the initial amount of K was lost in three years from logging residue.

DOC, Al and Fe

It was interesting to observe that although the trends were not the same, in both B6 and B14 the water color, DOC, tot-C and Fe all varied in the same way.

In a comparative investigation by Lundin (1991), increased water color, DOC and metal concentrations were observed after clear-cutting. The increase in DOC is attributed to the higher ground water levels and increased runoff which increases the contact time with the organic material in the upper layers of the soil. In B6 the concentrations of DOC, Tot-C, Fe and Al were higher than pre-treatment values in the second and fourth year after clear-cutting. The fact that these values were not consistently higher than pre-treatment values may be attributed to changes in water levels and redox conditions. Another explanation may be found in the high average water color. In B6 the average water color is almost twice as high as in the other catchment streams (table 2). It is possible that the conditions that promote the export of organic material (i.e. high ground water, thick humus layer) already exist in the catchment and that DOC was only slightly affected by clear-cutting. Concentrations of humic substances have a large natural variation that fluctuates with season, temperature and amount of precipitation (Löfgen & Lundin, 2003) and it may be difficult to distinguish between natural variation and human influence.

After drainage in B14 there was a marked decrease in the concentration and transport of DOC, Tot-C and Fe after clear-cutting and ditching. This is also in agreement with the findings of Lundin (1991). Drainage lowers the groundwater and water percolates through more of the mineral soil. Lowering the ground water level stimulates decomposition and more carbon is used by microorganisms.

Nutrient Balances

Nutrient leaching from the forested catchment is highest in the spring due to the high water flow and absence of established vegetation to take up nutrients. In contrast, the sphagnum in the fen is actively growing and able to take up nutrients whenever conditions are favorable.

Nitrogen

Many studies have found that peatlands are sinks for nitrate and sources for organic nitrogen (Jacks et al., 1997; Devito et al., 1989…) and this study of Bohyttan fen was no exception. The fen exported 6.5% more Org-N than entered the fen, and retained 6% of the NH_4-N entering the fen. The retention of nitrate (34%) was higher than that of ammonia, showing that denitrification is an important process in the fen. Hemond (1983) found in his study of Thoreau's bog that the most important nitrate sinks were denitrification (≤ 75%) and reduction to ammonium and incorporation into the exchange complex (≈ 25%).

The climate directly affects the concentration and transport of nitrate. Burt et al. (1988) proposed two different climatic conditions to explain the variation in NO_3-N. "Transport-limited" conditions are warm and dry while "Supply-limited" conditions are cool and wet. The warm, dry conditions result in elevated levels of nitrate due to increased mineralization of NO_3, and low leaching. This can lead to high levels of NO_3 being washed out with sub-surface runoff in the winter. The opposite conditions lead to supply-limitation. During cool, wet years N leaching is high and the remaining NO_3 is exhausted. The conclusions of Burt et al. (1998) are made in relation to soil, but may also apply to certain similar conditions in peatlands.

Phosphorus

There was no Tot-P retention over the 14 years studied, but net retention for the individual years was alternatingly positive and negative (figure 18). Temperature and degree of water saturation probably caused the variation between years. Richardson and Marshall (1986) found in microcosm studies that plant uptake is not a major sink for P in fens and that most P added was removed by microorganisms and fine sediments.

P export also is dependent on the redox conditions in the soil or sediment. In reducing conditions Al- and Fe-oxides dissolve and P is released. Reddy et al. (1998) found a 35% decrease in P sorption capacity in anoxic conditions compared with oxic.

Terrestrial ecosystems are able to retain and store higher levels of P than wetlands because of the larger pools of Al and Fe sesquioxides (Richardson, 1985). Therefore the removal of phosphorus is more likely to occur before water reaches the fen.

Preferential flow

The stream from B6 forms the main channel through Bohyttan fen (figure 3). The inflows from the other streams join this channel and therefore the water input is not evenly spread out over the entire area of the fen. However, due to several inflow locations, the waters were partially distributed over a larger part of the fen. The hydraulic conductivity in peat decreases with depth (Rydin and Jeglum, 2006, Hammar, 1992) which suggests that the greater part of the water flow is at the surface of the peatland. Using steel rods as redox indicators, Jacks et al. (1992) found that the upper 5 cm of a peatland were oxidizing; the next 5 cm were sulfate reducing and below 10 cm were unaltered. Therefore water that is not channelized can undergo both oxidation and reduction processes while passing through the fen. Water that passes through the fen in channels misses these redox processes and remains largely unchanged. Unfortunately, it is unknown how much of the runoff inflow is filtered through the fen and how much preferentially flows through the channels. Also, it would have been interesting to determine the average residence time of water passing through the fen.

CONCLUSIONS

- Forestry activities had an effect on the water quality of the catchment, but it was minor and on the same magnitude of natural variation. However, only parts of the catchment studied were affected by management.
- The fen was an important sink for inorganic nitrogen species along with phosphate, potassium and sulfate.
- The fen was a source for organic N and iron.
- The input/output balances of fens are very complex and further investigations, specially targeted, need to be carried out and this would contribute considerably to the understanding of fen ecosystem hydro chemical functions

ACKNOWLEDGEMENTS

I would like to thank my supervisor Lars Lundin and deputy supervisor Torbjörn Nilsson for all their time, feedback and patience. I am grateful to Kenth Andersson and Kjell Larsson who helped me with chemical analysis on my samples. I also want to thank Fredrik Gunnarsson at Sveaskog who helped me with the forest management data and Max Ahlstedt who was responsible for the essential field sampling.

References

Addiscott, T. 2005. *Nitrate, Agriculture and the Environment*. CABI Publishing. Oxfordshire, UK. pp. 279.

Ahtiainen, M. 1992. The effects of forest clear-cutting and scarification on the water quality of small brooks. *Hydrobiologia* 243/244: 465 – 473.

Anon. 1979. *Vattenföringsbestämning vid vatten undersökningar*. SMHI/Naturvårdsverket. Liber förlag, Jönköping. pp 110.

Arheimer, B. and Bergström, S. 2002. Våtmarkers påverkan på vattenbalans och storskaliga flöden. In: *Våtmarksboken*. VASTRA rapport 3. AB C O Ekblad & Co., Västervik. pp. 26-30.

Braskerud, B. C. 2002. Factors affecting phosphourous retention in small constructed wetlands treating agricultural non-point source pollution. *Ecological Engineering* 19:41-61

Brix, H., 1997. Do macrophytes play a role in constructed treatment wetlands*? Water Sci. Technol.* 35(5), 11–17.

Broadmeadow, S. and Nisbet, T.R. 2004. The effects of riparian forest management on the freshwater environment: a literature review of best management practice. *Hydrology and Earth System Sciences*. 8(3) 286–305

Burt, T.P, Arkell, B.P., Trudgill, S.T., and Walling, D.E. 1988. Stream nitrate levels in a small catchment in southwest England over a period of 15 years. *Hydrological processes* 2:267 – 234

Chang, M. 2003. *Forest Hydrology: an introduction to water and forests*. CRC Press LLC, Boca Raton, FL. pp. 373.

Devito, K. J. and Dillon, P. J. 1993. The influence of hydrologic conditions and peat oxia on the phosphorus and nitrogen dynamics of a conifer swamp. *Water Resources Research* Vol. 29, No. 8: 2675–2685

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

Forsius, M. 2004, Use of ICP IM data for the assesment of climate/global change effects. In: Kleemola, S. Forsius, M. (eds). 13th Annual Report 2004. UNECE ICP Integrated Monitoring. *The Finnish Environment* 710:33–36.

Geological survey of Sweden (SGU), 2007. Mineral and bedrock resource database. Geological survey of Sweden, Uppsala. (http://maps.sgu.se/sguinternetmaps/oredep/default.htm :Visited 2007-05-09)

Gopal, K., Tripathy, S., Bersillon, J. and Dubey, S. 2006. Chlorination byproducts, their toxicodynamics and removal from drinking water. *Journal of Hazardous Materials* 140 (2007) 1–6.

Gottschall, N., Boutin, C., Crolla, A., Kinsley, C., and Champagne, P. 2007. The role of plants in the removal of nutrients at a constructed wetland treating agricultural (dairy) wastewater, Ontario, Canada. *Ecological Engineering* 29:154–163.

Gustafsson, J.P. and Jacks, G. 1993. Sulphur status in some Swedish podzols as influenced by acidic deposition and extractible organic carbon. *Environmental Pollution* 81: 185–191.

Hammar, D.A., 1992. *Creating freshwater wetlands*. Lewis Publishers, Chelsea, MI. pp. 298.

Harrod, T.R. and Theurer, F.D. 2002. Sediment. In: *Agriculture, Hydrology and Water Quality*. CABI Publishing, Wallingford, UK. pp. 155-170.

Heikkinen, K. 1990. Transport of organic and inorganic matter in river, brook and peat mining water in the river basin of the river Kiiminkijoki. *Aqua Fenn*. 20:143 – 155.

Jacks, G., Joelsson, A., Norrström A.C., Johansson, U. 1997. Management of a forest wetland after clearcutting. In: *Groundwater and surface water ecotones - Biological and hydrological interactions and managements options*. Cambridge University Press-UNESCO International Hydrology Series. pp. 204-207.

Kadlec, R.H. and R. Knight. 1996. *Treatment Wetlands*, Lewis Publishers, Boca Raton, FL. pp. 893.

Kleemola, S. and Forsius, M. 2006. Trend assessment of bulk deposition, throughfall and runoff water/soil water chemistry at ICP IM sites. In: Kleemola, S., Forsius, M. (eds), 15th Annual report 2006. ICP Integrated Monitoring. *The Finnish Environment* 30/2006 22-48.

Leonardson, L. 2002. Hur avskiljer våtmarker kväve och fosfor? In: *Våtmarksboken*. VASTRA rapport 3. AB C O Ekblad & Co., Västervik, Sweden. pp. 41-66

Lundin, L. 1991. Influence of siviculture on content of organic matter and metals in water. *Finnish Humus News* 3:21-26.

Lundin, L. 1996. Effects of peat-winning on the water environment at a sedge fen ecosystem. *Proceedings from the 10th International Peat Congress*, 27 May - 2 June, Bremen, Germany. Vol. 2, 426-432

Lundin, L. 1998. Hydrochemical retention in mirelands. *Proceedings of the British Hydrological Society International conference*, Exeter, July 1998. Vol 1, 445-454.

Länsstyrelsen Örebro (Örebro County Administrative board). 2003. Åtgärdsplan för kalkningsverksamheten i sjöar och vattendrag 2003-2007, Arbogaån (122) och Kolbäcksån (125) Bilaga 5. Publ nr 2003:14

Löfgren, S. and Lundin, L. 2003. Mer humus i svenska vatten – bidrar även skogsbruket? *Fakta Skog* Nr 15, 2003

Martin, C.W., Noel, D.S. and Federer, C.A. 1984. Effects of forest clearcutting in New England on stream chemistry. *J. Environ. Qual.*, Vol 13, no. 2: 204–210.

Mitsch, W. and Gosselink, J. 2000. *Wetlands.* John Wiley and Sons, New York, New York, USA. pp. 920

Murphy, J. & Riley, J.P. 1962. A modified single solution method for the determination of phosphate in natural waters. *Analytica Chimica Acta*, 27, 31-36.

Nieminen, M., Ahti, E., Nousiainen, H., Joensuu, S. and Vuollekoski, M. 2005. Capacity of riparian buffer zones to reduce sediment concentrations in discharge from peatlands drained for forestry. *Silva Fennica* 39(3): 331 – 339.

Norrsström, A-C. 2002. Även skogen läcker näring. In: *Våtmarksboken*. VASTRA rapport 1. AB C O Ekblad & Co. Västervik, Sweden. pp. 105-122

Näsholm, T., Ekblad, A., Nordin, A., Giesler, R., Högberg, M., and Högberg, P. 1998. Boreal forest plants take up organic nitrogen. *Nature* Vol. 329 914-916

Palviainen, M., Finér, L., Kurka, A-M., Mannerkoski, H., Piirainen, S., and Starr, M. 2004. Release of calcium, iron and aluminium from Norway spruce, Scots pine and silver birch logging residues. *Plant and Soil* 259: 123 – 136.

Pella, E. and Colombo, B. 1973. Study of carbon, hydrogen and nitrogen determination by combustion-gas chromatography. *Mikrochim. Acta* 5: 697-7 19

Raab B., Vedin H. (1995): Klimat, sjöar och vattendrag. *Sveriges National Atlas*

Ramberg, L. 1981. Increase in stream pH after a forest drainage. *Ambio* Vol.10 1:34-35

Reddy, K. R., OConnor, G. A. and Gale, P.M. 1998. Phosphorus sorption capabilities of wetlands soils and stream sediments impacted by dairy effluent. *J. Environ. Qual.* Vol. 27:438 – 447

Richardson, C. and Marshall, P. 1986. Processes controlling movement, storage, and export of phosphorus in a fen peatland. *Ecological Monographs*, 56(4) 279–302

Richardson, C.J. 1985. Mechanisims controlling phosphorus capacity in freshwater wetlands. *Science*, Vol. 228: 1424 – 1426.

Rosén, K. 1984. Effect of clear-felling on runoff in two small watersheds in Central Sweden. *For. Ecol. Manage.* 9: 267 – 281.

Rosén, K., Aronson, J-A. and Eriksson, H.M. 1996. Effects of clear-cutting on streamwater quality in forest catchments in central Sweden. *Forest Ecology and Management* 83: 237 – 244.

Ruzicka J, and Hansen H. 1975. Flow injection analysis. Part I. A New concept of fast continuous flow analysis. *Anal Chim Acta* 78:145

Rydin, H. and Jeglum, J. 2006. *The Biology of Peatlands.* Oxford University Press, Oxford. pp. 343.

SIS, 1986. *Swedish standards for water analysis.* Standardisation Committee of Sweden.

Skei, J., Larsson, P., Rosenberg, R., Johnsson, P., Olsson, M. and Broman, D. 2000. Eutrophication and Contaminants in Aquatic Ecosystems. *Ambio* Vol. 29 No. 4-5 184-194

Skjelkvåle, B.L., Stoddard, J.L., Jefferies, D.S., Tørseth, K., Høgåsen, T., Bowman, J. Mannio, J., Monteith, D.T., Mosello, R., Rogora, M., Rzychon, D., Vesely, J., Wieting, J., Wilander, A. and Worsztynowicz, A. 2005. Regional scale evidence for improvements in surface water chemistry 1990–2001. *Environmental Pollution* 137: 165–176

Smith, E., Gordon, R., Madani, A. and Stratton, G. 2005. Cold climate hydrological flow characteristics of constructed wetlands. *Canadian Biosystems Engineering/ Le génie des biosystèmes au Canada* 47: 1.1-1.7

Stenbeck, G. 1996. Torvbruk – Miljö. Effekter och åtgärder (Peat harvesting - environmental impact and restoration alternatives). *SNV Rapport 4596.* Naturvårdsverket förlag, Stockholm. pp. 94

Swedish EPA, 2007. Swedish Environmental Objectives Portal Environmental Objectives Secretariat. Swedish Environmental Protection Agency, Stockholm. (*http://miljomal.nu/index.php* :visited 2007-04-10)

Söderberg, H., Tonderski, K. & Johansson, E. 2000. Hållbar lösning eller grönt filter? Användning av markväxtsystem för behandling av spillvatten i Sveriges kommuner.*Vatten* 56:103-110.

Tamm, C.O., Holmen, H., Popovik, B. and Wiklander, G. 1974. Leaching of plant nutrients from soils as a consequence of forestry operations. *Ambio* Vol. 3 No. 6 211-221.

Tonderski, K.S., Weisner, S., Landin, J. & Oscarsson, H. (eds.). 2002. *Våtmarksboken*: Skapande och nyttjande av värdefulla våtmarker. VASTRA rapport 3. AB C O Ekblad & Co., Västervik

Uusi-Kämppä, J., Braskerud, B., Jansson, H., Syversen, N. and Uusitalo, R. 2000. Buffer Zones and Constructed Wetlands as Filters for Agricultural Phosphourous. *J. Environ. Qual.*, Vol. 29:151-158

Vasander, H., Tuittila, E.-S., Lode, E., Lundin, L., Ilomets, M. Sallantaus, T., Heikkilä, R., Pitkänen, M.-L. and Laine, J. 2003. Status and restoration of peatlands in northern Europe. *Wetlands Ecology and Management* 11: 51 – 63.

Vitousek, P.M. 1981. Clear-cutting and the nitrogen cycle. *Ecol. Bull.* 33: 631 – 642.

Vitousek, P.M. and Melillo, J.M. 1979. Nitrate losses from disturbed forests: patterns and mechanisms. *Forest Science* 25: 605 – 619.

Vitousek, P., Gosz, J., Greir, C., Melillo, J., Reiners, W. and Todd, R. 1979. Nitrate Losses from disturbed Ecosystems. *Science* Vol. 204: 469 – 474.

Vymazal, J., Brix, H., Cooper, P., Haberl, R., Perfler, R. and Laber, J. 1998. Removal mechanisms and types of constructed wetlands. In: *Constructed wetlands for wastewater treatment in Europe*. Backhuys Publishers, Leiden NL. pp. 366.

Väänänen, R., Nieminen, M., Vuollekoski, M. and Ilvesniemi, H. 2006. Retention of phosphorus in soil and vegetation of a buffer zone area during snowmelt peak flow in southern Finland. *Water, Air, and Soil Pollution* 177:103–118.

Wiklander, G., Norlander, G. and Andersson, R. 1991. Leaching of nitrogen from a forest catchment at Söderåsen in Southern Sweden. *Water, Air, and Soil Pollution* 55: 263-282.

MSc THESES FROM THE DEPARTMENT OF FOREST SOILS, SLU SINCE 2001.

1. Gustafsson, Maria. 2001, Carbon loss after forest drainage of three peatlands in southern Sweden.
2. Isberg, Susanna. 2002. Elementkoncentrationer i gran utmed en markfuktighetsgradient.
3. Munter, Fredrik. 2002. Kloridhalter i gran utmed en depositions-gradient för havssalter.
4. Poggio, Laura. 2002. Epiphytic algae on Norway spruce needles in Sweden – geographical distribution, time-trends and influence of site factors.
5. Zander, Niclas. 2002. Beskogad åkermark – Förändringar av mark-pH efter plantering.
6. Bergkvist, Åsa. 2002. Små skogliga vattendrag i Värmland – Generell beskrivning, förekomst av traktorspår samt spårens inverkan på bottenfauna.
7. Gille, Emma. 2002. Den bäcknära zonen vid små skogliga vattendrag i Värmland – Generell beskrivning, förekomst av traktorspår samt kvicksilverhalter i körpåverkat ytvatten.
8. Herbertsson, Sofia. 2003. Sjunkande pH i Västerbottens humuslager – en kvantitativ analys.
9. Hedstrand, Ylva. 2003. Effects of ammonium oxalate treatment on interlayer materials in 2:1 layer silicates from a podzol
10. Hansson, Karna. 2004. Bok- och grankonkurrens i Sydsverige – markegenskaper och naturlig föryngring.
11. Granlöf, Jonatan. 2005. Stormfällning och dess riskfaktorer i skyddszoner längs skogliga vattendrag i Västra Götaland.
12. Chaminade, Guillermo. 2005. Topography, soil carbon-nitrogen ratio and vegetation in boreal coniferous forests at the landscape level.
13. Torgnyson, Beatrice. 2005. Student attitudes toward incentives to reduce automobile use.
14. Berg, Kristin. 2006. Naturlig föryngring på torra sedimentmarker. Ståndortsegenskaper och jordartskartan som beslutsunderlag för att undvika schablonmässig markberedning.
15. Hultnäs, Mikael. 2006. Skötselåtgärder vid nyetablering av skyddszoner vid bäckar i södra Värmland.
16. Stråhle, Erik. 2006. Sumpskogens motståndskraft vid stormen Gudrun år 2005.
17. Lucci, Gina. 2007. Retention and element balances for wetlands in the forest environment -case study Bohyttan fen.

This series includes MSc theses published at the Department of Forest Soils, SLU. Printed copies can be ordered from the department by phone; 018-672212 if still available. The MSc theses in PDF versions can be downloaded from the department's website: www.sml.slu.se.

ISSN 1650-7223
ISBN 978-91-576-7233-9

Department of Forest Soils
SLU
Box 7001
750 07 Uppsala