FORESTS AND GREENHOUSE GASES

FLUXES OF CO₂, CH₄ AND N₂O FROM DRAINED FORESTS ON ORGANIC SOILS

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Distributed by:
Department of Water and Environmental Studies
Linköping University
SE-581 83 Linköping
Sweden

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Forests and Greenhouse gases
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Cover illustration and layout by Tomas Eklund

ISSN: 0282-9800

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Department of Water and Environmental Studies

UniTryck, Linköping, 2004
LIST OF PAPERS


# TABLE OF CONTENTS

## INTRODUCTION

AIMS OF THIS THESIS

BACKGROUND

Forest drainage

Pre-drainage conditions

Impact of drainage

Drainage intensity

Problems associated with scaling fluxes

Complexities in production and consumption

Temporal and spatial variation in fluxes

Temporal variation

Spatial variation

Within-site spatial variation

Among-site spatial variation

Net GHG fluxes between the atmosphere and poorly drained forests

Management of peatlands

Up-scaling

STRATEGIES FOR DEVELOPING EMISSION FACTORS FOR POORLY DRAINED FORESTS

Study sites

Measurements of GHG

Biotic and abiotic variables measured

RESULTS AND DISCUSSION

Do groundwater level and air temperature explain temporal variation in GHG fluxes?

Does distance to trees affect the emissions of GHG?

Do the soil emissions of GHG differ among sites differing in fertility and tree species?

Are poorly drained forests sources or sinks for GHG?

Net emissions of GHG at poorly drained forest sites

Sensitivity analysis

Reallocation of carbon

Are poorly drained forest sites larger net sinks than well-drained and virgin sites?

Comparison between poorly drained and virgin sites

Comparison between poorly drained and well-drained sites

Do drained sites contribute significantly to the Swedish GHG budget?

MAIN CONCLUSIONS AND FUTURE RESEARCH

REFERENCES

ACKNOWLEDGEMENTS
INTRODUCTION

In the atmosphere there are gases, referred to as greenhouse gases (GHG), which restrict the outward flow of infrared radiation. These gases cause a net warming of the Earth’s surface called the greenhouse effect. If no GHG were present in the atmosphere, the global temperature would be 33°C lower, i.e. the mean global temperature would be −18°C instead of the current 15°C (IPCC, 1990). Thus, the greenhouse effect is essential for most of the life forms that have developed on Earth. At present, however, the concentrations of GHG in the atmosphere are increasing, promoting further global warming. The exact effects these changes will have are not known, but according to the IPCC (2001a) it is possible that both ecological and socio-economic systems may be irreparably damaged. In addition, the probability of extreme weather events, such as periods with very high or very low temperatures, extreme floods, droughts, tropical cyclones, and storms will increase, as will the probability of large-scale singular events, such as the collapse of the West Antarctic ice sheet or shutdown of the Gulf Stream. In Sweden, modelling suggests that by the year 2050 the temperature will be on average 2.5-4.5°C higher and precipitation 8-23% greater than today (Räisänen et al., 2003). Furthermore, although the potential productivity of forests and agricultural crops is likely to be higher in a warmer and rainier climate, any such gains could be undermined by conditions becoming more favourable for harmful insects, diseases and changes in soil moisture (Mattsson and Rummukainen, 1998).

Carbon dioxide (CO\textsubscript{2}), methane (CH\textsubscript{4}) and nitrous oxide (N\textsubscript{2}O) are regarded as the most important greenhouse gases, accounting for an estimated 80% of the total global warming (IPCC, 2001b). The global warming potential (GWP), which describes the cumulative warming over time caused by the emission of a gas, differs among the gases. The two basic factors governing each gas’s GWP value are its radiative forcing, i.e. the infrared absorption of an incremental amount of the gas in the atmosphere, and its rate of decay in the atmosphere. The GWP of CO\textsubscript{2} is set to 1, and the corresponding figures for CH\textsubscript{4} and N\textsubscript{2}O are 23 and 296, respectively (IPCC, 2001b), i.e. it takes 23 or 296 CO\textsubscript{2} molecules to cause the same warming as one molecule of CH\textsubscript{4} or N\textsubscript{2}O, respectively. There are two ways of reducing the concentrations of GHG in the atmosphere: to increase the strength of the sinks or to decrease the strength of the sources. Forestry can help reduce national emissions in either of two ways. Firstly, forests can accumulate carbon in their biomass or soil and, secondly, the produced biomass can be used as substitutes for other products, most notably fossil fuels, but also materials that are produced by energy-consuming processes, e.g. cement and plastics.
The major part of Sweden’s land surface (52% or 23.5 Mha) is covered with productive forestland (SCB, 2004). Forestry activities are, therefore, of major importance when assessing Sweden’s national GHG budget. However, not all forests are sinks for CO₂. For example, Lindroth et al. (1998) and Lohila et al. (2004) found that drained forests on organic soil could act as sources for atmospheric CO₂. In addition, these soils may be major sources of N₂O (Martikainen et al., 1993; Maljanen et al., 2003a).

AIMS OF THIS THESIS
The general aim of the work underlying this thesis was to elucidate how forests on drained organic soils function in the context of GHG exchange.

Specific aims were to:
- determine the most important factors regulating the emissions of CO₂, CH₄ and N₂O in drained forests on organic soil both temporally (Papers I and II) and spatially (Papers I, II, III and IV)
- determine the net fluxes of greenhouse gases from poorly drained forests on organic soil (Papers I and II)
- establish management strategies for poorly drained soils in order to minimize their GHG source strength or maximize their GHG sink strength (Papers I and II)
- estimate total net GHG exchange between the area of drained forest on organic soil in Sweden and the atmosphere (Paper V)

BACKGROUND
In this section the state-of-the-art knowledge about the fields addressed in this thesis are presented. At the end of each subsection a hypothesis is formulated, which relates to the specific aims.

Forest drainage
Land used for forestry has commonly been drained in various areas of the world, especially Fenno-Scandia and the former USSR (Paavilainen and Päivänen, 1995). To date, about 15 Mha of peatlands and wetlands have been drained for forestry in boreal and temperate regions (Paavilainen and Päivänen, 1995).

Pre-drainage conditions
The sites that have been drained for forestry had high groundwater levels before drainage. Decomposition in anaerobic environments occurs through
the cooperative action of several microbial populations and results in the
production of CH$_4$ (Guijer and Zehnder, 1983; Conrad, 1989). Anaerobic
decomposition is less effective than aerobic decomposition, resulting in
incomplete degradation of litter from the vegetation and an accumulation of
organic matter in the soil (Swift et al., 1979; Clymo, 1984). In Sweden, there
are about 10 Mha of peat-covered land, of which about 15% has been drained
(Hånell, 1990). Sixty percent of the peat-covered land area is classified as
peatland (Hånell, 1990), i.e. has a peat layer thicker than 30 cm. The remaining
40% of the peat-covered land area has a shallow peat layer (Hånell, 1990), i.e.
thinner than 30 cm, and is thus not classified as peat soil. Nevertheless, these
soils may have a high organic content and be classified as organic (which
applies to any soil with a proportion of organic matter exceeding 20% according to the FAO, 1998). Of the total area drained for forestry in Sweden, approximately 50% is situated on peat soil, while the rest has a peat layer thinner than 30 cm, according to data from the Swedish National Forest Inventory (S-NFI).

Impact of drainage

Drainage for forestry generally results in sites with high forest productivity
(Holmen, 1978), and consequently the CO$_2$ accumulation in tree biomass may
be high on drained sites. However, the accumulated organic matter in the soil
becomes available for aerobic decomposition after drainage, which promotes
high soil CO$_2$ release rates, as shown, for example, by Silvola et al. (1996a).
Furthermore, the nitrogen contained in the organic matter becomes available
for N$_2$O-producing microbes after drainage. Consequently, drained organic
forest soils have been found to be significant sources of both CO$_2$ and N$_2$O
(Martikainen et al., 1993; Laine et al., 1996; Silvola et al., 1996a; Regina et al.,
1998; Widén, 2001; Maljanen et al., 2003a; Weslien et al., XXXX). In addition,
CH$_4$ is exchanged between the atmosphere and drained organic forests (see,
for instance, Nykänen et al., 1998; Maljanen et al., 2003b; Weslien et al.,
XXXX). It has also been shown that the size of all the fluxes depends on the
type of land that is drained (e.g. Minkkinen et al., 2002).

Drainage intensity

In Finland GHG fluxes at drained forestland have been measured extensively
(see Table 2 in Paper V). Swedish forestland differs from Finnish, as the
forests are more productive due to the warmer climate. Thus, emissions
derived from measurements at Finnish drained forests cannot be uncritically
used for Swedish areas. In Sweden, only two drained sites have been studied,
one dominated by spruce and pine (Lindroth et al., 1998; Widén, 2001) and
one dominated by birch (Weslien et al., XXXX). The mean annual position of
the groundwater tables were between 40 and 100 cm (Lundblad and Lindroth,
The optimal groundwater table position after drainage is >35 cm below the soil surface for weakly decomposed nutrient poor peat and >55 cm for well decomposed nutrient rich peat (Paavilainen and Päivänäinen, 1995). When a tree stand has developed, the transpiration of the trees further lowers the groundwater table (Paavilainen and Päivänäinen, 1995). However, subsidence, through physical compaction of the soil and decomposition of the soil organic matter, result in a successive rising of the groundwater table (Eggelsmann, 1986). A large survey of drained peatlands in Finland showed that the average subsidence was 22 cm, approximately 60 years after drainage (Minkkinen and Laine, 1998). At logging, the transpiration by the trees is decreased, and thus the water table is likely to rise further (Roy et al., 1996). Consequently, complementary or remedial drainage is needed in order to maintain high productivity in drained forests.

In Sweden, the most extensive drainage period was between 1920 and World War II (Hånell, 1990) and the most intensive remedial drainage period was during the late 1970s and 1980s (Hånell, 1990). Thus, the peak in remedial drainage activity followed approximately 50 to 60 years after the peak in activities associated with dewatering land for forestry. Recently, the average area annually subjected to remedial drainage has been small, equivalent, on average, to less than 0.3% of the drained forestland or 2600 hectares per year in 1992 to 2002 (National Board of Forestry, 2003). Assuming that there is a 50 to 60 year period before drained land needs remedial drainage, the areas drained during World War II need remedial drainage now, but drainage activity during the World Wars was low (Hånell, 1990). If this is the reason for the currently low level of remedial drainage, a new peak in remedial drainage is likely to occur in the fairly near future, since the area subjected to drainage increased again after the end of World War II. However, the low remedial drainage activity may also reflect the present recognition of swamp forests and wetlands as valuable biotopes that are worth protecting (see, for instance, Rubec, 1997) and should not, therefore, be remediably drained. This view is also reflected in the fact that since 1986, drainage of wetlands has been prohibited, for environmental reasons, without a special permit. Nevertheless, it is very likely that the area of moist drained forestland in Sweden will increase in the near future. Hence, there is an urgent need to enhance our understanding of these systems in order to develop sustainable management strategies. Therefore, this thesis focuses on GHG exchange between the atmosphere and Swedish moist drained forests. Moist drained forests will be referred to as poorly drained in the following text due to their drainage depth being below the optimal.
Problems associated with scaling fluxes

Complexities in production and consumption

When calculating the net GHG balance of a drained forest, several fluxes must be included (Fig. 1). The net CO₂ exchange of a system is the sum of CO₂ uptake via plant photosynthesis, CO₂ respired from below- and above-ground parts of plants and CO₂ released from decomposition of soil organic matter, i.e. both recently added litter and organic matter accumulated before drainage (Fig. 1). If the amount of CO₂ incorporated into plant biomass exceeds the CO₂ released via the decomposition of soil organic matter the site is a net sink for CO₂ and if the CO₂ released from decomposition exceeds CO₂ incorporation into biomass the site is a net source for CO₂. CH₄ is produced in the anaerobic fraction of the soil and consumed in the aerobic fraction (Fig. 1) (Sundh et al., 1994; 1995). Consequently, soil fluxes of CH₄ are a result of the balance between CH₄ production and consumption, and drained forests can be either sources or sinks for CH₄ (Nykänen et al., 1998; Maljanen et al., 2003b; Weslien et al., XXXX). Nitrification and denitrification are the two most important processes involved in soil N₂O-production (Firestone and Davidson, 1989) and the N₂O flux is a result of the N₂O released from both nitrification and denitrification. These two processes are tightly coupled to each other and to mineralization, since nitrifiers use NH₄⁺ derived from mineralization and denitrifiers use NO₃⁻ produced by nitrification (Fig.1). Some denitrifiers can also gain energy by using atmospheric N₂O as a substrate and, therefore, water-saturated soils can be sinks for atmospheric N₂O (Blackmer and Bremner, 1976; Regina et al., 1996; Johansson et al., 2003).

Figure 1. Schematic diagram of GHG production, consumption and fluxes in terrestrial systems.
The temporal and spatial variation of these fluxes is complex since they are the net results of diverse processes (Fig. 1), all of which are regulated by multiple biotic and abiotic factors (e.g. Swift et al., 1979; Conrad, 1989; Robertson, 1989; Paavilainen and Päivänen, 1995).

Temporal and spatial variation in fluxes
A number of authors have published formulas for calculating the carbon accumulation in tree biomass (e.g. Marklund, 1988; Fridman, 1995; Peterson, 1999). These formulas have input parameters that are easily measured, e.g. diameter at breast height, age of trees and altitude. Thus, the carbon accumulation in tree biomass in forested areas can be determined quite accurately over large temporal and spatial scales based on variables that can be measured at one visit to a site. On the other hand, the spatial and temporal variation in soil fluxes of GHG is high (e.g. Matson et al., 1989). In this context the soil emissions of CO₂ represent the sum of CO₂ released from roots and decomposition. One of the aims of this thesis was to identify easily measurable variables that could explain a major part of the temporal and spatial variation in soil GHG fluxes. If possible, it would be very useful to base the up-scaling of emissions on parameters available in national databases, such as the S-NFI. Therefore, the scope for coupling the spatial variation in GHG fluxes to some of the S-NFI variables was studied.

Temporal variation
As shown by various authors (e.g. Swift et al., 1979; Conrad, 1989; Robertson, 1989) the activity of microbes producing and consuming GHG in terrestrial systems is heavily influenced by soil moisture and temperature. Total soil CO₂ release has been found to correlate positively with both soil temperature and groundwater table depth in drained, as well as undrained, peat soils (Silvola et al., 1996a; Wickland et al., 2001). Positive correlations between soil temperature and CH₄ emission rates (Frolking and Crill, 1994; Nykänen et al., 1998; Wickland et al., 2001) and negative temporal relationships between CH₄ emissions and the groundwater table position have been found at both drained and undrained peatlands (e.g. Nykänen et al., 1998). Similarly, temporal variations in N₂O fluxes have been found to be positively correlated with air temperature and groundwater table in a drained peat forest soil (Maljanen et al., 2003a).

Hypothesis I: Thus, I hypothesized that the variation in groundwater table and air temperature could explain the temporal variation in GHG soil fluxes in poorly drained forests.
Spatial variation

In this thesis the spatial variations in soil GHG emissions are considered at two different scales: within sites and among sites.

Within-site spatial variation

Forest soils are, naturally, heavily affected by trees, but the impact of the trees is not evenly distributed within the forested area. For example, higher concentrations of organic matter have been found in areas close to stems (Liski, 1995), and due to the uneven distribution of fine roots (Olsthoorn, et al., 1999), root exudates, competition for nutrients, oxygen demand and content, pH and soil moisture are all very likely to differ spatially within a forest stand. Furthermore, the amount of throughfall increases with distance from stems, and the concentrations and loads of NO₃⁻ and NH₄⁺ decrease with distance from stems (Hansen 1996; Whelan et al., 1998). All of these factors are known to affect the production and consumption of GHG (see, for instance, Swift et al., 1979; Conrad, 1989; Robertson, 1989).

The impact of distance to stems on soil fluxes has been investigated in studies of forests on mineral soil. Scott-Denton et al. (2003) found that rates of soil CO₂ release decreased with distance from trees. Similarly, Brumme (1995) found that soil CO₂ release rates are lower in gaps than under canopies in forests, which was attributed to the likelihood that the amount of CO₂ released via root activity will be higher in areas closer to stems. Butterbach-Bahl et al. (2002) have reported that the net consumption of CH₄ is lower and net emissions of N₂O higher in the soil within a 1 m radius of Norway spruce stems compared to soil more distant from tree stems. They suggested that this was an effect of the significantly higher soil nitrogen content found close to stems.

Hypothesis II: Given the above considerations, I hypothesized that the distance to stems and soil fluxes of GHG are also related in organic, poorly drained forests. If so, the number of stems per hectare, a parameter that is recorded in the S-NFI database, could be used for extrapolation purposes.

Among-site spatial variation

Drainage intensity, tree species and soil fertility could be the most important factors regulating the soil emissions of GHG in drained forests since they affect many GHG-regulating factors, for example soil oxygen content, litter quantity (Bray and Gorham, 1964) and quality (Gosz, 1981; Staaf and Berg, 1981; Johansson, 1995, Wedderburn and Carter, 1999), and thus soil nutrient conditions (Menyailo et al., 2002, Smolander and Kitunen, 2002) and pH (Menyailo et al., 2002; Smolander and Kitunen, 2002).
Silvola et al. (1996a) found that the soil CO₂ release rates increased with water table depth at a number of drained and undrained peat sites. CH₄ fluxes in drained areas have been found to decrease with increasing groundwater table depth (Nykänen et al., 1998), and although N₂O emissions are affected by many factors other than groundwater table there is, at least, a tendency for emissions to be higher at drained areas with relatively low groundwater tables compared to wetter drained areas (Regina et al., 1996). Silvola et al. (1996a) compared CO₂ fluxes from drained peat areas differing in fertility and found that differences in fluxes among sites could be partly explained by differences in fertility. Soil fertility has also been reported to affect CH₄ emissions in undrained peatlands (see, for instance, Martikainen et al., 1995; Nilsson et al., 2001) and in drained peatlands higher N₂O emissions have been found from nutrient rich than from nutrient poor, drained peat soils (Martikainen et al., 1993; Regina et al., 1996). To my knowledge no studies have compared GHG fluxes from drained organic soils dominated by different tree species. However, studies on mineral soils have shown that soil CO₂ release is higher in areas dominated by deciduous species than in areas dominated by coniferous species (Hudgens and Yavitt, 1997; Janssen et al., 1999; Longdoz et al., 2000). Hudgens and Yavitt (1997) reported that mineral soils dominated by deciduous tree species had higher net CH₄ consumption rates than mineral soils dominated by coniferous species. Borken and Brumme (1997) found similar results and attributed them to the coniferous litter having lower diffusivity for CH₄. Soil collected in stands of different tree species grown on the same mineral soil showed that N₂O emissions correlated with litter CN ratios, and increased in the order larch < pine < spruce < cedar < aspen < birch (Menyailo and Huwe, 1999).

**Hypothesis III:** Thus, I hypothesized that the GHG fluxes in poorly drained forest soils would differ significantly from the fluxes in well-drained forest soils and that soil fertility and dominating tree species are important regulators of system GHG exchange at poorly drained forests.

**Net GHG fluxes between the atmosphere and poorly drained forests**

Forests can decrease the national emissions of GHG by accumulating CO₂ in biomass or soil, and by producing biomass, which can be substituted for fossil fuels. Consequently, a drained forest could be regarded as a net sink for GHG if the CO₂ uptake by the vegetation can compensate for the decomposition of organic matter in the soil and soil emissions of CH₄ and N₂O. If the tree biomass on drained forestland was used to substitute for fossil fuels then the impact would be more complex, at least from a political perspective, because drainage causes a shift in the allocation of carbon from peat to tree biomass. In the same way that the GHG emissions from inputs of external energy in forest operations have to be included when estimating the environmental
impacts of using forest biomass for energy production (as done, for example, by Berg and Karjalainen, 2003), the decomposition of peat has to be included in estimates of the impact of using wooden material on drained forestlands for energy production. The climatic impact of using peat as an energy source has been discussed extensively in recent years, and a number of studies have shown that the impact of peat utilization is comparable to that of fossil fuels (e.g. Savolainen et al., 1994; Rodhe and Svensson, 1995), while other reports have claimed that the utilization of peat should be compared to use of forest residuals (Åstrand et al., 1997). Tree stands, which represent a renewable fuel, are ready for harvest after approximately 100 years while fossil fuels, such as coal and oil, have been embedded in the Earth’s crust for maybe 100 million years. As it takes thousands of years for peat deposits to be harvestable, peat can neither be classified as a renewable nor a fossil fuel. It has been suggested that peat should be treated separately and classified as a slowly renewable fuel (Crill et al. 2000; SOU, 2002). Thus, to use peat as a substitute for fossil fuel is more controversial than use of biofuel. Taking the allocation of carbon into consideration, some of the carbon accumulated in tree biomass in drained forests could be regarded as peat carbon. Burning tree biomass on drained forestland for energy production could therefore be viewed to some degree as burning peat.

Hypothesis IV: I hypothesized that tree accumulation of CO$_2$ more than compensates for the soil emissions of GHG at poorly drained organic forest areas, making the areas net sinks, but the major part of the carbon in the trees should be considered as peat carbon.

Management of peatlands

Assuming that poorly drained soils will increase in abundance it is of interest to know how these areas should be managed in order to keep them as large sinks (or as small sources) as possible. There are three easy options: (i) to rewet the area by closing ditch systems or merely neglecting them, and thus allow a return to a paludified state through gradual subsidence; (ii) to further lower the water table, i.e. to use complementary or remedial drainage; or (iii) to keep the areas poorly drained but prevent them returning to a paludified state.

The impact of drainage and drainage intensity has been discussed above. Complementary or remedial drainage is very likely to improve the forest growth conditions (Paavilainen and Päivänen, 1995) and decrease CH$_4$ emissions (Nykänen et al., 1998), but increase soil emissions of both CO$_2$ (Silvola et al., 1996a) and N$_2$O (Regina et al., 1996). A rewetting of the site would result in decreased carbon uptake by tree vegetation (Paavilainen and Päivänen, 1995) and, in addition, CH$_4$ emissions are likely to be increased
(Nykänen et al., 1998). On the other hand, decomposition rates of soil organic
matter (Silvola et al., 1996a) and emissions of N₂O would decrease (Regina et
al., 1996).

Hypothesis V: I hypothesized that the increased CO₂ uptake by trees and the
decreased soil emissions of CH₄ could not compensate for the increased rates
of soil CO₂ and N₂O release resulting from complementary or remedial
drainage. Similarly, I hypothesized that the decreased soil CO₂ and N₂O
releases could not compensate for the decreased CO₂ uptake by trees and the
increased soil emissions of CH₄ resulting from a rewetting. Consequently, I
hypothesized that both well-drained and rewetted organic soils are larger
sources (or smaller sinks) of GHG than poorly drained soils.

Up-scaling
I hypothesised that poorly drained areas are net sinks of GHG. Observations
have shown that well-drained areas, on the other hand, may be sources of
GHG, as net releases of CO₂ have been found from them (Lindroth et al,
1998; Lohila et al., 2004). As only 20% of the drained area is classified as wet
or moist at present in Sweden (according to S-NFI data), the major part of the
land is well-drained and may be a source of GHG. In order to determine the
impact of Swedish drained forest ecosystems on the national GHG budget,
accurate up-scaling and evaluation of the net emissions from drained forests
on organic soils is essential, and requires high quality data on fluxes from
different types of drained soils. There is, however, a paucity of flux data,
making attempts to scale up the fluxes highly uncertain. The IPCC has
developed guidelines to be applied when countries calculate and report their
national emissions and removals of GHG. In the Good Practice Guidance for
Land Use, Land-Use Change and Forestry (GPG-LULUCF) default emission
factors for drained forests on organic soils are available which could be used
for estimating the net fluxes from drained organic forest soils. However, these
data are rough and it is suggested that country-specific data on fluxes from
drained organic soils should be used if available (Penman et al., 2003).

Swedish data on GHG emissions from drained organic forest soils are scarce.
Therefore, more flux measurements are needed in order to scale up the
emissions. However, the aim of an up-scaling may not only be to provide an
exact value of the GHG exchange between the Swedish area of drained forest
on organic soil and the atmosphere, but also to obtain an estimate based on
present knowledge. Such estimates are needed by the decision makers.
Furthermore, they are useful as they may highlight sectors where more
research is needed.
Hypothesis VI: I hypothesized that the GHG contribution from drained forests on organic soils would have a significant impact on the national GHG budget.

STRATEGIES FOR DEVELOPING EMISSION FACTORS FOR POORLY DRAINED FORESTS

This section contains a presentation of the study sites as well as a discussion about the methods used for measuring GHG emission in poorly drained forests.

Study sites
To test the hypothesis stated above five drained sites differing in fertility and tree species were studied. All five sites were classified as poorly drained, with mean annual groundwater tables in the upper 30 cm of the soil (Table 1). Four of the sites were classified as having peat soils, while the fourth had a peat layer thinner than 30 cm (Table 1). The organic content in the soils was over 20% at all sites (Table 1), and thus they were classified as organic according to FAO criteria (FAO, 1998). One of the poorly drained sites was dominated by Scots pine (Pinus sylvestris (L.)), one by downy birch (Betula pubescens Ehrh.), two by Norway spruce (Picea abies (L.) Karst.) and one by black alder (Alnus glutinosa (L.) Gaertn.). The soil fertilities of the sites were based on the classification by Hånell (1991). At the poorly drained pine site the forest floor vegetation was dominated by Vaccinium uliginosum, and the site was, consequently, classified as dwarf shrub type, i.e. of low fertility (Fig. 2). At the poorly drained birch site herbs were abundant. However, most of these herbs were not indicator species listed in the classification scheme. Therefore, based on the large amounts of Trientalis europea, the site was classified as bilberry-horsetail type, i.e. of medium fertility (Fig. 3). One of the spruce sites was dominated by 40-year-old trees. It was sparsely vegetated, but the present forest floor vegetation was mainly Vaccinium myrtillus and the site was, therefore, classified as billberry-horsetail type, i.e. of medium fertility (Fig. 4). The other spruce site had older trees, about 80 years old, and was also quite sparsely vegetated. Maianthemum bifolia and Oxalis acetosella were present and the site was, therefore, classified as low herb type, i.e. highly fertile (Fig. 5). Tall herbs, such as Dryopteris species and Filipendula ulmaria, dominated the poorly drained alder site (Fig. 6). For comparison two undrained sites, one in a fen and one in an alder swamp, were chosen. These systems represent two different site types which were commonly drained for forestry. The fen was classified as impediment and dominated by tall Carex species and therefore classified as tall sedge type, i.e. of low fertility (Fig. 7).
Figure 2. The photo to the left shows the vegetation at the drained pine site. The diagram to the left shows groundwater table (blue), peat depth (brown) and position of chambers (bars), while the diagram to the right shows the position of ditches (lines, the thickness of the line corresponding to the width of the ditch) and chambers (dots).

Figure 3. The drained birch site

Figure 4. The drained spruce site with young trees
Figure 5. The drained spruce site with old trees

Figure 6. The drained alder site

Figure 7. The undrained fen
The swamp, dominated by black alder, was classified as low herb type, i.e. highly fertile, based on the presence of *Filipendula ulmaria* and *Viola* species (Fig. 8).

The sites were chosen to represent different types of poorly drained areas (Fig. 9). The classification in Fig. 9 is based on potential productivity (data from Table 1, *Paper V*), which is not directly convertible to soil fertility. While the fertility classification has been developed for determining potential productivity after drainage (Hånell, 1991), and thus focuses on the nutrient content in the soil, the potential productivity is based on both soil nutrient conditions and other factors such as groundwater table. However, the productivity of the trees is strongly influenced by the soil fertility and it is therefore assumed that a comparison is justifiable.

![Figure 8. The undrained swamp](image)

![Figure 9. Relative areas of poorly drained types of productive forestland in Sweden, as represented by areas of boxes. The total area is divided into three fertility classes, based on potential forest productivity, i.e. the box to the left represents low productivity (<4 m$^3$ ha$^{-1}$ y$^{-1}$), the box in the middle medium productivity (4-8 m$^3$ ha$^{-1}$ y$^{-1}$) and the box to the right high productivity (>8 m$^3$ ha$^{-1}$ y$^{-1}$) areas. The percentage of different kinds of poorly drained forest types in Sweden is then shown by the different coloured areas within the three boxes, i.e. peat soils (white), mineral soils (gray), coniferous (unstriped white and gray areas) and deciduous (white and gray striped areas). The types of area that the sites discussed in this thesis (i.e. the sites dominated by pine (DP), birch (DB), young spruce trees (DSy), old spruce trees (DSo) and alder (DA)) represent, are indicated by arrows.](image)
The most common poorly drained forested site type consists of peat soils dominated by coniferous tree species (Fig. 9). The areas classified as being of low, medium and high productivity were represented by the pine site (DP), the spruce site with young trees (DSy) and the spruce site with old trees (DSo), respectively (Fig. 9). However, the spruce site, which was classified as highly fertile, had approximately the same carbon accumulation in tree biomass as the medium fertility spruce site (Table 2), indicating that its fertility may be medium rather than high. Nitrogen is one of the most important nutrients, and the finding that the nitrogen content of the organic matter was similar at the two spruce sites (Table 1) further strengthens the possibility that their fertility was similar. Similarly, the birch site (DB) was chosen in order to represent the medium productivity peat forests dominated by deciduous tree species, but the growth rate at the birch site was lower than expected from the soil fertility (Tables 1 and 2). Therefore, the fertility classification may not reflect actual conditions sufficiently well, and should be treated with care.

About 40% of the poorly drained forest soils in Sweden have a peat layer thinner than 30 cm (Fig. 9). About 15% of these sites are classified as highly productive (Fig. 9), and can thus be represented by the poorly drained alder site (DA).

Table 1. Soil parameters for all sites, both poorly drained, i.e. pine (DP), birch (DB), spruce, young trees (DSy), spruce, old trees (DSo) and alder (DA), and undrained, i.e. fen (UF) and alder swamp (US).

<table>
<thead>
<tr>
<th></th>
<th>DP</th>
<th>DB</th>
<th>DSy</th>
<th>DSo</th>
<th>DA</th>
<th>UF</th>
<th>US</th>
</tr>
</thead>
<tbody>
<tr>
<td>Annual mean</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>groundwater table (cm)</td>
<td>17</td>
<td>15</td>
<td>27</td>
<td>22</td>
<td>18</td>
<td>7</td>
<td>-1</td>
</tr>
<tr>
<td>Probable time since drainage (years)</td>
<td>40</td>
<td>60</td>
<td>&gt;30</td>
<td>&gt;30</td>
<td>20</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Peat depth (cm)</td>
<td>114</td>
<td>52</td>
<td>53</td>
<td>7</td>
<td>5</td>
<td>70</td>
<td>41</td>
</tr>
<tr>
<td>Organic matter (%)</td>
<td>94</td>
<td>73</td>
<td>92</td>
<td>86</td>
<td>40</td>
<td>90</td>
<td>92</td>
</tr>
<tr>
<td>Dry bulk density</td>
<td>0.10</td>
<td>0.32</td>
<td>0.17</td>
<td>0.13</td>
<td>0.63</td>
<td>0.03</td>
<td>0.10</td>
</tr>
<tr>
<td>Porosity (%)</td>
<td>93</td>
<td>84</td>
<td>86</td>
<td>87</td>
<td>72</td>
<td>93</td>
<td>91</td>
</tr>
<tr>
<td>Tot N 0-10 cm (%)</td>
<td>1.3</td>
<td>2.2</td>
<td>1.9</td>
<td>1.9</td>
<td>2.8</td>
<td>1.2</td>
<td>2.5</td>
</tr>
<tr>
<td>Tot C 0-10 cm (%)</td>
<td>56</td>
<td>52</td>
<td>54</td>
<td>54</td>
<td>47</td>
<td>49</td>
<td>54</td>
</tr>
<tr>
<td>CN ratio 0-10 cm (%)</td>
<td>44</td>
<td>25</td>
<td>29</td>
<td>29</td>
<td>16</td>
<td>48</td>
<td>22</td>
</tr>
<tr>
<td>pH 0-10 cm</td>
<td>2.7</td>
<td>3.4</td>
<td>3.2</td>
<td>3.3</td>
<td>4.5</td>
<td>3.9</td>
<td>4.2</td>
</tr>
<tr>
<td>Productivity class</td>
<td>low</td>
<td>low</td>
<td>low</td>
<td>medium</td>
<td>medium</td>
<td>low</td>
<td>low</td>
</tr>
<tr>
<td>Humification degree</td>
<td>low</td>
<td>medium</td>
<td>medium</td>
<td>high</td>
<td>high</td>
<td>medium</td>
<td>medium</td>
</tr>
</tbody>
</table>

* measured down to a maximum depth of 120 cm

*b of organic matter

classification based on von Post and Granlund (1926).

However, the sites also differed in respects other than tree species and fertility (Table 1). For example, the thickness of the peat and the soil content of organic matter differed amongst them (Table 1).
Table 2. Tree parameters for all treed sites, both poorly drained, i.e. pine (DP), birch (DB), spruce, young trees (DSy), spruce, old trees (DSo) and alder (DA), and the undrained alder swamp (US).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>DP</th>
<th>DB</th>
<th>DSy</th>
<th>DSo</th>
<th>DA</th>
<th>US</th>
</tr>
</thead>
<tbody>
<tr>
<td>Age (years)</td>
<td>70</td>
<td>60</td>
<td>50</td>
<td>90</td>
<td>40</td>
<td>80</td>
</tr>
<tr>
<td>Height (m)</td>
<td>16</td>
<td>16</td>
<td>18</td>
<td>24</td>
<td>19</td>
<td>18</td>
</tr>
<tr>
<td>Diameter, breast height (mm)</td>
<td>200</td>
<td>150</td>
<td>180</td>
<td>290</td>
<td>220</td>
<td>220</td>
</tr>
<tr>
<td>Diameter increment (mm y⁻¹)</td>
<td>1.8</td>
<td>1.9</td>
<td>2.8</td>
<td>2.2</td>
<td>3.5</td>
<td>1.8</td>
</tr>
<tr>
<td>Number of stems of the dominating tree species (ha⁻¹)</td>
<td>1100</td>
<td>850</td>
<td>1350</td>
<td>750</td>
<td>1750</td>
<td>500</td>
</tr>
<tr>
<td>Calculated biomass growth (tonnes DW ha⁻¹ y⁻¹)</td>
<td>3.6</td>
<td>3.2</td>
<td>7.6</td>
<td>7.7</td>
<td>20.5</td>
<td>3.3</td>
</tr>
</tbody>
</table>

* a all trees taller than 1.3 m were counted

**Measurements of GHG**

The net GHG exchange was determined from dark static chamber measurements of soil GHG release and CO₂ accumulation in biomass. With this method the net exchange of CH₄ and N₂O can be estimated quite well, although transport of CH₄ and N₂O through plants may be inhibited (Sebacher *et al.*, 1985; Chang *et al.*, 1998). However, the measuring technique has several limitations with respect to determination of the net ecosystem exchange of CO₂. Firstly, the carbon accumulation in certain fractions, mainly fine roots, is difficult to estimate. The formulas used for estimating the carbon accumulation in coniferous trees (*Paper I*) do not account well for the contribution of fine roots, since they were primarily designed for estimating the amount of dry weight in different fractions of the standing biomass (Peterson, 1999). In the formulas used for deciduous tree species the fine root fraction was not included (*Paper II*). Consequently, the carbon accumulation in tree biomass was underestimated using the formulas. Secondly, the soil CO₂ release as measured in chambers is the sum of both root and decomposition activity. The contribution of root activity to measured soil CO₂ release has been shown to be on average 50% in forests (e.g. Hanson *et al.*, 2000) and around 10% in a virgin bog (Silvola *et al.*, 1996b). Due to the nature of the techniques used for measuring the CO₂ originating from roots, the 50% value includes not only direct root activity, but also decomposing activity associated with root exudates and recently dead root tissues (collectively called root-derived activity). As fine roots have a life-time of about a year (Majdi and Andersson, 2004), the decomposition of fine roots was assumed to be included in the measurements of root-derived activity. Consequently, the CO₂ allocated to fine roots is not considered in the estimate of tree carbon accumulation, but is instead subtracted from the soil CO₂ release. Thus, the estimate of the net ecosystem exchange was assumed to be accurate, in spite of the somewhat back-calculation involved.

Measurement with dark chambers also leads to the exclusion of photosynthetic activity of the forest floor vegetation. Furthermore, the forest
floor vegetation was left intact, so the measured forest floor CO₂ release also included CO₂ respired by the above-ground parts of the understory. Removing plant tissues from the chambers would have solved the problems associated with subtracting the fraction of the CO₂ release originating from forest floor vegetation. Furthermore, the cut-away vegetation could have been used to estimate the carbon accumulation in forest floor biomass. On the other hand, cutting the forest floor might have resulted in overestimation of the decomposition rates as the roots would have died off and provided the microorganisms with easily decomposable organic matter. Therefore, the forest floor vegetation was disturbed as little as possible. At the spruce sites there were only small amounts of forest floor vegetation, mostly a thin layer of mosses, so it is not likely that the forest floor respiration contributed to the CO₂ release to any great extent (Figs. 4 and 5). Compared to the carbon accumulation in trees it is also likely that the CO₂ uptake by the forest floor vegetation is negligible. At the deciduous sites, both poorly drained and undrained, the forest floor vegetation was denser (Figs. 3, 6 and 8), but still the CO₂ release from forest floor vegetation and the carbon accumulation by the forest floor vegetation was assumed to be negligible. On the other hand, both the pine site and the virgin fen had thick Sphagnum layers (Figs. 2 and 7), which might have significantly contributed both to the annual forest floor CO₂ release and to the annual carbon accumulation by vegetation. At these sites estimates of the growth of the forest floor vegetation were based on literature data. The carbon use efficiency was assumed to be 0.5 (based on Choudhury, 2000, 2001). Consequently, the same amount of CO₂ that was estimated to be annually accumulated in forest floor vegetation was also assumed to be released by forest floor vegetation, and thus subtracted from the forest floor respiration in order to obtain an estimate of its contribution to soil respiration (Paper I).

The ditches were not evenly distributed within the sites, but chambers were placed so that as much as possible of the differences in groundwater level and peat depth within the sites was covered (Figs. 2-8). For more information about the measuring techniques see Papers I-III.

**Biotic and abiotic variables measured**

The air temperature and groundwater level were measured concurrently with the gas sampling. Other variables were measured occasionally for three different purposes: to check the differences among the studied sites (Papers I and II), to test within-site variability (Paper II) and to examine the effect of distance to tree stems (Paper III). At the site level the age, height, diameter and diameter increase of trees were measured and the number of stems per hectare calculated (Table 2). Furthermore, dry bulk density and porosity of the soil were determined at site level (Table 1, Papers I and II). Mean annual
groundwater levels and peat depth were used to characterize each chamber and soil samples were collected once at each chamber within the sites and used to measure degree of humification, organic matter content, total nitrogen and carbon content in the organic matter, CN ratio and pH (Table 1, \textit{Papers I} and \textit{II}). The parameters measured in the transect between two trees were peat depth, leaf area index, dry weight of different species, pH, soil content of water, organic matter, nitrogen and carbon, CN ratio and potential denitrification (\textit{Paper III}).

\section*{Results and Discussion}
The results of the studies are structured around the six hypotheses that were formulated in the introduction.

\textbf{Do groundwater level and air temperature explain temporal variation in GHG fluxes?}
I hypothesized that the variation in groundwater table and air temperature could explain the variation in soil fluxes of GHG in poorly drained forests (hypothesis I).

Between 47 and 68\% of the temporal variations in forest floor CO$_2$ release at the poorly drained sites were explained by differences in air temperature and sometimes also groundwater level (Tables 6 and 5 in \textit{Papers I} and \textit{II}, respectively). In \textit{Papers I} and \textit{II} it was suggested that the response at high temperatures might have been underestimated as the days when the groundwater level was below the depth which could be measured, were usually warm (\textit{Papers I} and \textit{II}). However, even at the site where the largest number of samples (32\%) was excluded from the regressions (\textit{Paper I}), there was no significant difference between the shapes of the curves obtained by (i) including all of the samples, and (ii) excluding samples collected on occasions

\begin{figure}[h]
\centering
\includegraphics[width=0.5\textwidth]{figure10}
\caption{Relationships between mean temperature and mean site flux obtained when including (●) and excluding (○) chambers at which the groundwater level could not be measured. The correlation found when all data are included is represented by the full line, and the correlation found when chambers with a groundwater table below the depth that could be measured was excluded is represented by the dotted line.}
\end{figure}
when the groundwater level could not be measured (Fig. 10). Thus, this possibility is unlikely to have caused a significant error in the estimates.

The temporal variation in CH\textsubscript{4} fluxes could be explained by groundwater level and air temperature to a smaller extent than the variation in CO\textsubscript{2}, i.e. 0 to 26\% (Tables 6 and 5 in Papers I and II, respectively). Air temperature was more important than groundwater level, which only significantly affected CH\textsubscript{4} emissions at the poorly drained pine site (Table 6 in Paper I). CH\textsubscript{4} fluxes were not related to groundwater level in any easily predictable non-linear way either (Fig. 11).

In two of the poorly drained sites, i.e. the pine and alder sites, virtually none of the temporal variation in N\textsubscript{2}O fluxes could be explained by differences in groundwater level and air temperature (Tables 6 and 5 in Papers I and II, respectively). For the other three sites, i.e. the spruce sites and birch site, between 19 to 27\% of the temporal variance could be explained by these two factors.

These results show that the hypothesis - that groundwater level and air temperature are the most important temporal regulating factors of GHG emissions at poorly drained organic sites - was only supported for CO\textsubscript{2}. Consequently, at poorly drained forest sites, factors other than groundwater level and air temperature, or at least a more complex function describing the relationship between these two factors, are needed to model temporal variations in the emissions of CH\textsubscript{4} and N\textsubscript{2}O. Since the years in which the measurements were performed were warmer than the 30-year mean (Fig. 1 in Papers I and II) the mean annual forest floor CO\textsubscript{2} release may have been higher than usual and due to anticipated climatic changes, which are expected to raise temperatures in Sweden (Räisänen et al., 2003) it is very likely that the future forest floor release rates will be even higher. That the temporal

![Figure 11. Relationship between temporal variations in CH\textsubscript{4} fluxes and mean site groundwater table at the poorly drained sites, i.e. pine (○), birch (+), spruce with young trees (▲), spruce with old trees (□) and alder (△).]
variation in forest floor CO₂ release was so strongly correlated with air temperature and groundwater table has important implications for attempts to scale up emissions as it implies that estimates of annual CO₂ emissions at poorly drained sites could be based on a limited number of flux measurements.

**Does distance to trees affect the emissions of GHG?**

I hypothesized that there is a relationship between distance to stems and soil fluxes of GHG in forests on poorly drained organic soils (hypothesis II).

There were no consistent patterns in the variations in CO₂ and CH₄ fluxes in transects between two trees. On two occasions, however, the emissions in transects between trees (Paper III) and the emissions in the rest of the poorly drained spruce site with old trees (Paper I) were measured at the same time. Using all data from these two occasions, both CO₂ and CH₄ were linearly correlated (p<0.05) with distance to stems (n=51 and 28, respectively): CO₂ during the second sampling occasion and CH₄ during the first (Fig. 12). However, only 8 and 13% of the spatial variation in forest floor CO₂ release and CH₄ fluxes, respectively, was explained by distance to stems. For N₂O there was no linear correlation (Fig. 12). On the other hand, N₂O fluxes showed large spatial variations within transects with peaks (attributed to root dynamics) occurring during spring and autumn (Fig. 3 in Paper III). The emissions in these peaks were much higher than the emissions measured at the rest of the site (Fig 3 in Paper I). Thus, there also seems to be a tree effect on N₂O emissions, but it is not easily predictable in time and space.

The hypothesis that there is a relation between distance to stems and soil GHG fluxes in poorly drained forest sites was partly supported for all gases. The distance to trees should, therefore, be taken into consideration when planning sampling schemes for poorly drained organic forest soils.
Consequently, the GHG fluxes from the poorly drained sites in this study may have been over- or under- estimated as distance to stems was not taken into account when the positions of the chambers were chosen. However, the results show that the number of stems per hectare is not a useful parameter for scaling up site emissions as linear correlations were weak or non-existing.

**Do the soil emissions of GHG differ among sites differing in fertility and tree species?**

I hypothesized that soil fertility and dominating tree species are important regulators of system GHG exchange in poorly drained forest sites (hypothesis III).

The differences in soil fertility among the poorly drained sites, as determined by forest floor vegetation, were partly reflected in forest productivity (the birch and the highly fertile spruce sites being exceptions) and the CN ratio of the organic matter, except for the highly fertile spruce site. Similarly, the expected effects of tree species on the soil CN ratio and pH in the upper 10 cm of the soil were found, i.e. CN ratio decreased in the order pine > spruce > birch > alder (Johansson, 1995; Wedderburn and Carter, 1999; Menyailo et al., 2002, Smolander and Kitunen, 2002) and pH decreased in the order alder > birch > spruce > pine (Menyilo et al., 2002; Smolander and Kitunen, 2002) (Table 1). Thus, there were differences among the sites caused by differences in soil fertility and tree species.

Despite the differences in soil fertility and tree species, the forest floor CO$_2$ release did not differ significantly among the poorly drained sites (Fig. 13; Papers I and II), except that forest floor CO$_2$ release rates were significantly (p<0.05) lower in the highly fertile spruce site dominated by old trees than in the deciduous sites (Paper II). The CO$_2$ release from root activity is likely to be dependent in some way on the amount of stems per hectare, as the amount of roots is likely to be dependent on the amount of stems, and this variable differed among the sites (Table 2). For example, the number of stems per hectare was lower at the spruce site with old trees compared to the pine site (Table 2). Consequently, the CO$_2$ release originating from root activity is probably higher at the pine site. As the forest floor CO$_2$ release did not differ significantly between the two sites the results indicate that the CO$_2$ release from decomposition was higher at the more fertile spruce site.
Hence, there may be an effect of soil fertility and tree species on the CO₂ release originating from decomposition, which is masked by the respiration of the roots and forest floor vegetation. Furthermore, the sites do not differ only in terms of tree species and fertility, which weakens the conclusion that neither tree species nor fertility influence forest floor CO₂ release. However, it is possible that the high groundwater tables at these sites (Table 1) limited the decomposition rates. Oxygen status is recognized as one of the most important factors affecting decomposition rates in terrestrial systems (Swift et al., 1979). In this case the other factors, known to vary amongst the sites are unlikely to affect the decomposition rates. For example, a load of 100 kg NH₄NO₃-N ha⁻¹ y⁻¹, which should have had a major effect on soil fertility, did not significantly increase the forest floor CO₂ release at a Finnish pine bog with a mean annual groundwater table at approximately 20-30 cm below the soil surface (Nykänen et al., 2002).

The CH₄ emissions decreased significantly (p<0.05) in the order pine > birch and spruce site with old trees > spruce site with young trees and alder (Fig. 13; Papers I and II). There was no correlation with soil fertility, CN ratio or tree species. Instead, the differences in CH₄ emissions among the sites were, most likely, governed by the mean annual groundwater level (Fig. 14). Although the groundwater levels differed between the wettest and driest sites by only 10 cm, the effects of these differences may have masked the effects of tree species and fertility, which have previously been found to cause differences in CH₄ fluxes between mineral forests (Borken and Brumme, 1997; Hudgens and Yavitt, 1997) and undrained peat soils (Martikainen et al., 1995; Nilsson et al., 2001).
The N$_2$O emissions at the poorly drained sites increased significantly (p<0.05) in the order pine < spruce with young trees < birch < alder, while the N$_2$O emissions at the highly fertile spruce with old trees did not differ significantly from either the pine or the other spruce site (Fig. 13; Papers I and II). Consequently, higher emissions were found for the sites dominated by deciduous sites and the emission pattern is similar to trends Menyailo and Huwe (1999) found for tree species planted on mineral soils. Furthermore, they are highly correlated with the CN ratio in the upper 10 cm of the soil (Fig. 15). Under conditions of low nitrogen availability larger fractions of the nitrogen in leaves and needles are withdrawn before the litter falls (Gosz, 1981; Staaf and Berg, 1981). Thus, both the soil fertility and tree species affect the nitrogen concentration in the litter.

As the relationship between CN ratio and N$_2$O emissions at the poorly drained sites was so strong, data from other studies were also included (Paper IV), and the relationship remained strong after their inclusion (Fig. 15). The N$_2$O emission rates at CN ratios >25 are low, i.e. in the range 0.005 to 0.08 g m$^{-2}$ y$^{-1}$. Below this level, the emissions increase with further reductions in the CN ratio. The threshold value at a CN ratio of 25 agrees well with observations that net nitrification (i.e. accumulation of nitrate) only occurs at low CN ratios (Gundersen et al., 1998a). Net nitrification has been found to increase exponentially with reductions in CN below the threshold (Ollinger et al., 2002), as found for N$_2$O emissions in this study (Fig. 15). The findings also have analogues with observations of significant N losses by nitrate leaching in forests on mineral soils, which again occur at soil organic matter CN ratios below 25 (Gundersen et al., 1998b; MacDonald et al., 2002). This indicates that nitrification may be the rate-limiting process for N$_2$O emissions in drained organic forest soils.
There is a problem associated with the curve form for CN ratios below 15-20, which is linked to the hierarchical control of the emissions (Brumme et al., 1999). Once the rate-limiting parameter loses importance, other factors (e.g. pH, soil moisture and temperature (Robertson, 1989)) start to act as moderators of the emissions. Consequently, more data are needed to improve the curve form for the emissions at CN ratios below 15-20.

The hypothesis that soil fertility and tree species would be the most important factors causing differences in GHG emissions at poorly drained sites was only supported for N₂O. The CN ratio in the top-soil seems to be a good predictor for mean annual N₂O emissions, as shown by the strong correlation between these two variables. Attempts to scale up N₂O emissions should therefore be based on CN ratios. For CO₂ and CH₄ the groundwater table seems to be of major importance. Consequently, the groundwater table needs to be considered in attempts to scale up CO₂ and CH₄ fluxes. The results show that the forest floor CO₂ release rates from a poorly drained forest with a peat layer thinner than 30 cm were not significantly lower than those from poorly drained sites with a peat layer thicker than 30 cm. This indicates that drained sites with a peat layer thinner than 30 cm should also be included in estimates of GHG emissions from drained areas. As about 50% of the drained organic forest soils in Sweden have a peat layer thinner than 30 cm (according to data from S-NFI), the inclusion of this area would have a significant impact on estimates of GHG emissions.

**Are poorly drained forests sources or sinks for GHG?**

I hypothesized that tree accumulation of CO₂ more than compensates for the soil emissions of GHG at poorly drained organic forest areas, making the...
areas net sinks, but that the major part of the carbon in the trees should be considered as peat carbon (hypothesis IV).

Net emissions of GHG at poorly drained forest sites
The contribution of CH$_4$ and N$_2$O to the soil fluxes of GHG was small at all sites (Fig. 13). Neither the forest floor CO$_2$ release nor the estimated CO$_2$ release originating from decomposing activity differed significantly among the poorly drained sites, since the fraction originating from decomposing activity was assumed to be 50% of the forest floor release at all sites except the pine site (Paper I). Consequently, the differences in net GHG fluxes among the sites were largely due to the differences in calculated CO$_2$ accumulation in the trees.

The poorly drained sites were very different in many respects (Tables 1 and 2). However, most sites were net sinks of -0.2 to -2.7 kg CO$_2$ equivalents m$^{-2}$ y$^{-1}$, showing that the forest production at poorly drained sites, in most cases, compensated for soil emissions (Fig. 16; Papers I and II). Only the poorly drained birch site was a net source of GHG (0.4 kg equivalents m$^{-2}$ y$^{-1}$; Fig. 16; Papers I and II). This was mainly due to the very low level of carbon accumulation in the trees. The calculated carbon accumulation in tree biomass of 600 g CO$_2$ m$^{-2}$ y$^{-1}$ is equivalent to between 40 and 60% of the average calculated carbon uptake by trees in deciduous moist drained forests in the area (Paper V). Even the average growth in wet areas on peat soils, although almost half the average for moist areas, is higher than the growth at the poorly drained birch site. Accordingly, the calculated net GHG flux is probably not representative for larger areas, and most of the poorly drained forest area is most probably a sink for GHG.

Figure 16. Calculated net GHG fluxes at all sites. The drained sites, i.e. pine (DP), birch (DB), spruce young trees (DSy), spruce old trees (DSo) and alder (DA), are represented by black bars and the undrained sites, i.e. fen (UF) and swamp (US), by white bars.
Sensitivity analysis

There are large uncertainties in the estimates of net GHG fluxes at the poorly drained sites. The major uncertainty is coupled to the CO₂ fluxes. For example, no biomass functions were available for estimating the below-ground carbon accumulation for birch. Such formulas were only available for spruce and pine (Peterson, 1999), so the formula derived for spruce was used to determine the birch parameter since birch, like spruce, has a flat root system, while pine, in contrast, can have a taproot system. The above-ground growth of alder was calculated using formulas derived for birch, and although alder has a taproot system, identical formulas were used for birch and alder, i.e. the formula derived for spruce roots was also used to estimate the fraction allocated below-ground for alder. The estimates resulted in a below-ground allocation of 25-27% of the total annual biomass increment for the deciduous species (Paper II). This is similar to the 24 to 26% recommended by the Good Practice Guidance for deciduous species in temperate regions (Penman et al., 2003). Thus, the growth estimates would have been similar if default values had been used. This does not, of course, guarantee their accuracy, and more studies are needed to resolve the biomass distribution, but the assumptions regarding below-ground carbon accumulation were assumed to be correct in this sensitivity analysis.

The carbon accumulation in forest floor vegetation was assumed to be negligible for all sites except the poorly drained pine site. This may not, however, be the case. Between 0 and 30% of the total carbon annually assimilated has been found to be taken up by forest floor vegetation at productive forested sites (see, for instance, Widén, 2001). Therefore, in this sensitivity analysis, a range for carbon accumulation in biomass was used, between the values presented in Table 2 and 1.3 times these values. For the poorly drained pine site, 30% of the CO₂ annually accumulated was estimated to be taken up by Sphagnum, based on literature data on the production of the species present at the site (Paper I), and in the sensitivity analysis values of 20 and 40% were used.

To get a range for the forest floor CO₂ release, the differences among years, although seldom statistically significant (Papers I and II), and standard errors associated with the forest floor CO₂ release, were used. The standard errors were multiplied by two in order to determine the theoretical 95% confidence intervals. The part of the CO₂ release originating from roots is also uncertain. Estimates of 50% of the root-derived CO₂ release for the forested areas were used for this parameter. To check the sensitivity of the net CO₂ exchange data, values of 40 and 60% were used. The assumption that there is significant carbon accumulation in the forest floor vegetation also implies that the respiration of the forest floor vegetation contributes to the forest floor CO₂
release. However, due to the large ranges used for the other fluxes this was not included in the calculations.

When calculating the net ecosystem exchange of GHG, emissions of CH₄ and N₂O were also included. As for CO₂ the differences among years and standard errors (times two) were used to determine the respective ranges (Papers I and II).

Table 3. Results of the sensitivity analysis of the CO₂ exchange (in kg CO₂ m⁻² y⁻¹) and GHG exchange (in kg CO₂ equivalents m⁻² y⁻¹) at the drained sites, i.e. pine (DP), birch (DB), spruce, young trees (DSy), spruce, old trees (DSo) and alder (DA), and undrained, i.e. fen (UF) and alder swamp (US). The ranges arise from differences in assumptions regarding carbon accumulation in the forest floor vegetation and soil fluxes of GHG.

<table>
<thead>
<tr>
<th>Site</th>
<th>Range in annual CO₂ exchange</th>
<th>Range in annual GHG exchange</th>
</tr>
</thead>
<tbody>
<tr>
<td>DP</td>
<td>-0.9 to 0.8</td>
<td>-0.9 to 0.9</td>
</tr>
<tr>
<td>DB</td>
<td>-0.5 to 1.2</td>
<td>-0.5 to 1.4</td>
</tr>
<tr>
<td>DSy</td>
<td>-1.5 to -0.2</td>
<td>-1.6 to -0.2</td>
</tr>
<tr>
<td>DSo</td>
<td>-1.7 to -0.1</td>
<td>-1.7 to 0.0</td>
</tr>
<tr>
<td>DA</td>
<td>-4.2 to -2.2</td>
<td>-4.2 to -1.5</td>
</tr>
<tr>
<td>UF</td>
<td>-0.6 to 1.1</td>
<td>-0.6 to 1.6</td>
</tr>
<tr>
<td>US</td>
<td>-0.1 to 0.5</td>
<td>0.0 to 1.0</td>
</tr>
</tbody>
</table>

These calculations showed that the two poorly drained sites with the lowest carbon accumulation in tree biomass, i.e. the pine and birch sites, might be in equilibrium rather than being sinks or sources, while the more productive poorly drained sites are almost certainly net sinks for GHG (Table 3).

Reallocation of carbon

The carbon release estimated to originate from decomposition is due not only to the decomposition of organic matter stored before drainage. Some of it is also due to the decomposition of organic matter deposited from the growing vegetation. Consequently, the CO₂ originating from the decomposition of new plant material has to be subtracted to estimate the fraction released as a result of drainage. Organic matter is mainly deposited in the form of above- and below-ground litter from trees and vegetation. As the annual litter input was not measured, these values had to be derived from the literature. Decomposition of fine roots has already been considered above, as it is included in the CO₂ from root-derived activity. Assuming that the contributions of coarse roots is relatively small they will be excluded from the estimates. The annual litter input in a part of the poorly drained spruce site with young trees was measured as part of the LUSTRA project (LUSTRA, 2004). Preliminary data indicate that above-ground litter inputs from the trees amount to 86 g C m⁻² y⁻¹ and corresponding inputs from the field and bottom layers amount to 20 g C m⁻² y⁻¹ (Berggren et al., 2002). The above-ground litter input from trees corresponds to 20% of the total calculated carbon uptake by
trees (Table 2). The proportion of the above-ground litter originating from the forest floor vegetation was about 20%.

Given the lack of better data, 20% of the annual carbon accumulation in trees at the coniferous sites was assumed to be deposited as litter and the litter input from forest floor vegetation was assumed to be equivalent to 20% of the litter input from trees at the spruce sites. For the pine site, the previously applied figure for carbon accumulation in forest floor vegetation, i.e. 30% of the carbon accumulated in tree biomass (Paper I), was used and all that is produced during the course of a year was assumed to be added to the soil as litter. The annual tree litter input at deciduous sites has been found to be about 35% of the annual tree biomass production (Mar-Möller et al., 1954; Duvigneaud and Denaeyer-De Smet, 1970). As for coniferous forests, about 20% of the litter originates from forest floor vegetation (M. Johansson pers. comm.). These values were used in the calculations. The estimated CO2 contribution from the decomposition of recently added litter was divided by the estimated soil CO2 release, which gave the proportion originating from recently added litter. The values derived from the calculations of litter input from above-ground parts of the trees were subtracted from the carbon accumulations in tree biomass, to obtain an estimate of the fraction of the CO2 annually taken up by the trees that is allocated to the standing vegetation. The fraction of the tree carbon that should be considered as peat carbon was calculated by dividing the amount of CO2 estimated to originate from peat decomposition by the annual CO2 increment in the standing vegetation.

Table 4. Estimates of different CO2 fluxes (g C m⁻² y⁻¹) at all drained sites, i.e. the pine (DP), birch (DB), spruce young trees (DSy), spruce old trees (DSo) and alder (DA) sites, are represented by black bars and the undrained sites, i.e. fen (UF) and swamp (US), by white bars.

<table>
<thead>
<tr>
<th></th>
<th>Annual carbon accumulation in trees</th>
<th>Tree litter carbon accumulation minus litter</th>
<th>Forest floor CO2 release from decomposition of recently added litter</th>
<th>Proportion of total estimated CO2 decomposition (%)</th>
<th>Proportion of tree carbon considered peat carbon (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>DP</td>
<td>180</td>
<td>40</td>
<td>140</td>
<td>50</td>
<td>50</td>
</tr>
<tr>
<td>DB</td>
<td>160</td>
<td>60</td>
<td>100</td>
<td>10</td>
<td>70</td>
</tr>
<tr>
<td>DSy</td>
<td>380</td>
<td>80</td>
<td>300</td>
<td>20</td>
<td>100</td>
</tr>
<tr>
<td>DSo</td>
<td>385</td>
<td>80</td>
<td>305</td>
<td>20</td>
<td>100</td>
</tr>
<tr>
<td>DA</td>
<td>1025</td>
<td>360</td>
<td>665</td>
<td>70</td>
<td>100</td>
</tr>
</tbody>
</table>

Based on these assumptions, between 30 and 100% of the estimated total CO2 release from decomposition originated from decomposition of new litter at the sites (Table 4). Consequently, between 0 and 100% of the carbon in the trees could be considered as peat carbon (Table 4). Of course, there are major uncertainties associated with the estimates for each site. Nevertheless, this gives an indication of an important problem that should be addressed when assessing drained forestland. The calculations show that the low productivity sites are the most crucial, as 100% of the tree carbon could be regarded as
peat carbon at the pine and birch sites. The finding that 100% of the estimated soil CO$_2$ release originates from the decomposition of new litter at the alder site implies that poorly drained forested sites on soils with a peat layer thinner than 30 cm may not release additional soil CO$_2$ from the decomposition of organic matter stored before drainage. However, in another study a forested drained mineral soil has been found to be a net emitter of CO$_2$ (Lindroth et al., 1998).

Thus, the results support the hypothesis that tree accumulation of CO$_2$ more than compensates for the soil emissions of GHG at poorly drained organic forest areas, making the areas net sinks. The exceptions are sites with low forest growth, which may be either sources or in equilibrium. Consequently, the total area of poorly drained forestland in Sweden is likely to be a net sink of GHG. The results also support the hypothesis that most of the carbon in the trees should be considered as peat carbon. This allocation also has to be considered in attempts to scale up emissions from drained forest soils.

**Are poorly drained forest sites larger net sinks than well-drained and virgin sites?**

I hypothesized that the GHG fluxes in poorly drained forest soils would differ significantly from the fluxes in well-drained forest soils (hypothesis III) and that both well-drained and rewetted organic soils are larger sources (or smaller sinks) of GHG than poorly drained soils (hypothesis V).

**Comparison between poorly drained and virgin sites**

To estimate the impact of a management strategy involving rewetting and the restoration of previously drained areas, the emissions from the poorly drained sites were compared with emissions from undrained sites. As climatic variables, e.g. groundwater table and temperature, are known to affect the emissions to a large extent (Silvola et al., 1996a; Nykänen et al., 1998; Maljanen et al., 2003a) measurements from both poorly drained and undrained sites in the same region during the same measuring period were needed for the comparison. To obtain a possible span for undrained conditions two sites were chosen, one fen and one alder swamp, which may be representative for the pre-drainage conditions of the poorly drained sites. These two peat-covered sites differ in many respects. The pH and nitrogen content is higher for alder swamps and the CN ratio lower compared to the corresponding figures for fens and bogs (Mäkinen, 1979; Urvas et al., 1979). There is also a significant limit to the peat thickness of alder swamps, of around 50 cm, because roots of alder trees cannot extend much deeper into the mineral soil (Mäkinen, 1979). Similar differences were also found between the two drained sites in this study (Table 1). The soil emissions of GHG at the sites were found to be representative for their respective wetland types (Papers I and
II). However, the fen may not be representative for all untreed peat-covered soils. The carbon accumulation in biomass is larger in bogs than in fens (see, for instance, Tolonen and Turunen, 1996) while CH$_4$ emissions are larger in fens than in bogs (Martikainen et al., 1995; Nilsson et al., 2001). Likewise, the alder swamp may not be considered to be representative for other swamps and treed mires. The growth at the alder swamp was associated with an uptake of CO$_2$ by the trees of around 600 g m$^{-2}$ h$^{-1}$ (Paper II). The average carbon uptake by the trees in coniferous drained productive wet forests is in the order of 200 to 400 g m$^{-2}$ h$^{-1}$ (based on S-NFI data, using the same method for calculating as in Paper V). Consequently, the carbon accumulation in tree biomass at coniferous undrained swamps and treed mires is most likely lower than the carbon accumulation in the alder swamp. Emissions of CH$_4$ have been found to be lower from treed compared to untreed mires (Bubier et al., 1995; Granberg et al., 1997). However, the CH$_4$ emissions did not differ significantly between the fen and swamp, indicating that the CH$_4$ emissions were high at the alder swamp compared to other swamps and treed mires. Furthermore, N$_2$O emissions of most undrained mires have been found to be low (Martikainen et al., 1993, Laine et al., 1996; Regina et al., 1996) while the emissions from the alder swamp were significant. Thus, the net fluxes of GHG may be either smaller or larger from coniferous swamps and treed mires compared to the alder swamp. Consequently, the net GHG fluxes in the poorly drained sites in a virgin condition may have differed from the net GHG fluxes at the fen and swamp in this study. Nevertheless, the fen and the alder swamp will be used for comparison.

Comparing the mean net emissions, all poorly drained sites had smaller net emissions than the fen, and all but the poorly drained birch site had smaller net emissions than the swamp (Fig. 16). However, as for the poorly drained sites, a sensitivity analysis of the net annual fluxes had to be performed. For the fen substantial uncertainty is associated with the carbon accumulation in the forest floor vegetation. The forest floor carbon accumulation was estimated to be 200 g DW m$^{-2}$ y$^{-1}$ (Paper I). The highest Sphagnum production rate reported by Lindholm and Vasander (1990) was 380 g DW m$^{-2}$ y$^{-1}$ and Moore et al. (2002) reported net primary production for the above-ground forest floor of 360 g DW m$^{-2}$ y$^{-1}$ at a bog. In the sensitivity analysis, therefore, a range of carbon accumulation of 100 to 400 g DW m$^{-2}$ y$^{-1}$ was used. Furthermore, the contribution to the soil CO$_2$ release from roots was assumed to be 10% of the total, based on Silvola et al. (1996b). In the sensitivity analysis root-derived contributions of 0 to 20% were used. As for the poorly drained sites, the contribution of forest floor respiration to forest floor CO$_2$ release was not changed in the different calculations. The sensitivity analysis for the alder swamp was based on the same assumptions as for the poorly drained sites (see the Sensitivity analysis section above).
The sensitivity analysis shows that both the fen and the swamp may have been in equilibrium, or the fen even a sink, instead of sources during the sampling period (Table 3). The poorly drained birch and spruce sites may have net emissions similar to the undrained sites. However, although the differences between the poorly drained and undrained sites were not significant (Table 3), it is still likely that the poorly drained spruce sites were larger sinks for GHG, during the measuring period, than both the fen and the swamp. Furthermore, it is almost certain that the poorly drained alder site was a larger sink for GHG than the undrained sites. This is in accordance with Minkkinen et al. (2002) who claimed that forestry drainage had decreased the radiative forcing of Finnish peatlands.

It is not certain that the emissions from a rewetted forest are similar to those in virgin conditions. After a disturbance of an agricultural soil, 10-15 years or more of constant management is required for soil organic carbon to reach a new balance (Batjes, 1998), in which it is possible that either smaller or larger amounts of carbon can be stored in the soil. Therefore, it is likely that it will take some time for rewetted sites to return to a state similar to the virgin conditions, and in some cases the rewetted conditions may never be similar to them. However, the comparison gives an indication that the net emissions of GHG fluxes from rewetted sites are likely to be larger than those from poorly drained sites, i.e. the results support the hypothesis.

Comparison between poorly drained and well-drained sites
To estimate the impact of a remedial or complementary drainage, the emissions from the poorly drained sites were compared with emissions from well-drained sites. The average annual forest floor CO2 release in a well-drained site dominated by pine, with a mean annual groundwater table at about 70 cm (M. Lundblad pers. comm.), was 2.8 kg CO2 m\(^{-2}\) (Widén, 2001). Average annual forest floor CO2 release rates of 2.4 kg m\(^{-2}\) y\(^{-1}\) have been reported for a well-drained Swedish birch site with an average annual groundwater level at 53 cm (Weslien et al., XXXX). The average annual forest floor CO2 release rates in the pine and birch sites in the present study were 1.5 and 1.9 kg m\(^{-2}\) y\(^{-1}\), respectively (Fig. 14). Consequently, the forest floor CO2 release rates were 1.3 and 0.5 kg m\(^{-2}\) y\(^{-1}\) higher at the well-drained sites compared to the poorly drained sites. This might, however, be partly due to differences in root activity as well as in decomposition rates between the investigated areas. Nevertheless, it seems very likely that the soil CO2 release increases with drainage depth, as reported by Silvola et al. (1996a).

However, the CO2 accumulation in trees is also likely to be increased by a lowering of the groundwater table. The pine site studied by Widén (2001) was
situated in a heterogeneous area with patches of both mineral and peat soils. In this area measurements were also performed with micrometeorological techniques. The site was found to be a net source of between 0.2 and 0.8 kg CO₂ m⁻² y⁻¹ (Lindroth et al., 1998). Thus, the carbon assimilation by trees could not compensate for the soil emissions. Our poorly drained pine site was, in contrast, a net sink of 0.2 kg CO₂ m⁻² y⁻¹. In the site studied by Weslien et al. (XXXX) the carbon uptake by trees was estimated to amount to 1.0 kg CO₂ m⁻² y⁻¹. Using the same assumptions for root contributions as those used for the poorly drained birch site, the site was an estimated net source of 0.2 kg CO₂ m⁻² y⁻¹, i.e. approximately equal to the poorly drained birch site (0.3 kg CO₂ m⁻² y⁻¹). Soil fluxes of CH₄ and N₂O were reported to be on average -0.1 g CH₄ m⁻² y⁻¹ and 3.2 g N₂O m⁻² y⁻¹, respectively (Weslien et al., XXXX). Summing these fluxes makes the site a net source of 1.1 kg CO₂ equivalents m⁻² y⁻¹, while the poorly drained birch site was a net source of 0.4 kg CO₂ equivalents m⁻² y⁻¹.

Consequently, it is very likely that net emissions of GHG from the well-drained forest areas and undrained areas are higher than from poorly drained areas. This supports the hypothesis that the best management strategy for poorly drained soils is to keep them moist. However, the virgin mires are only weaker estimated sources of GHG than poorly drained forests if the reallocation of carbon is not considered (see section Reallocation of carbon). Overall, the results indicate that the poorly drained sites should at least not be subjected to complementary or remedial drainage, if the GHG perspective is considered. The results also support the hypothesis that the GHG fluxes in poorly drained forest soils differ significantly from the fluxes in well-drained forest soils. As the net emissions of GHG differ among sites that differ in drainage intensity the groundwater table has to be considered in attempts to scale them up.

**Do drained sites contribute significantly to the Swedish GHG budget?**

Based on the results of the study an attempt was made to scale up the emissions from drained organic forestland in Sweden to a national level. I hypothesized that the contribution of GHG from drained forests on organic soils would have a significant impact on the national GHG budget (hypothesis VI).

For valid up-scaling high quality emission data are required for different site types. My experimental work was focused on poorly drained sites. As less than 20% of the drained organic forestland in Sweden is classified as wet or moist (i.e. poorly drained, see Paper V), the up-scaling had to be complemented by literature data. Only Finnish and Swedish sites were included. Finnish
emission data may not accurately describe the conditions in southern Sweden, but were still used due to the paucity of Swedish data.

The emissions reported in Finnish and Swedish studies were divided into site property groups based on the climatic conditions in the areas in which the respective sites were situated, mean annual groundwater table, tree species and soil fertility (Paper V). The soil CO₂ release was higher in sites with an average annual groundwater level below 40 cm compared to sites with a groundwater level above 40 cm. Mean annual groundwater table also significantly separated the CH₄ emissions: the poorly drained sites having higher emissions. CH₄ fluxes were also significantly divided by soil fertility, the less fertile sites having higher emissions. The N₂O emissions only differed significantly among sites dominated by different tree species. Sites dominated by deciduous species had higher emissions than sites dominated by coniferous species. Consequently, the data currently available support the finding that groundwater table is the most important factor affecting CO₂ and CH₄, while tree species were of importance for the N₂O emissions.

All drained forestland on organic soils, rather than merely forestland on peat soils, was included in the up-scaling. Based on the emission factors the drained forestland in Sweden was estimated to be a net source of 2.2 M tonnes CO₂ equivalents y⁻¹ (Paper V). The N₂O emissions may have been underestimated since accurate up-scaling cannot be based on tree species alone since N₂O emissions from forest sites with organic soils that have previously been used for agriculture have been found to be high, i.e. 8-33 kg N₂O ha⁻¹ y⁻¹, regardless of tree species (Maljanen et al., 2004; Weslien et al., XXXX). Between 30 and 45% of the organic drained forestland in Sweden may have been used for agriculture previously. As this area is not accounted for separately the national N₂O emissions from drained organic forestland may have been underestimated by a factor of up to four (Paper V). These sites would most probably be recognized as highly emitting if the CN ratio was considered in the up-scaling. Such data are, however, not available in S-NFI but in the Swedish Survey of Forest Soils and Vegetation. The number of sampling plots in the latter database is smaller than for S-NFI and, consequently, the coupling between these two databases would result in a larger uncertainty in the areal estimates. Therefore, a larger survey is needed before an up-scaling can be based on CN data.

In the estimate the CO₂ exchange was of most importance (Paper V). Therefore, only the CO₂ exchange will be discussed in the following sections. The uptake of CO₂ by the vegetation was estimated to amount to 8.9 and the soil CO₂ release from decomposition to 10.4 M tonnes CO₂ y⁻¹ (Paper V). As previously discussed, the decomposition of peat should be treated separately.
The estimates were based on the same assumptions as previously used (see section Reallocate of carbon), i.e. that 20 and 35% of the carbon accumulated in trees annually in coniferous and deciduous areas, respectively, is deposited as litter, that the same amount of carbon accumulated in forest floor vegetation during the course of a year becomes litter, and that the root litter has already been accounted for by subtracting the estimated root-derived activity from the soil respiration. Of the 2.2 Mtonnes y$^{-1}$ of carbon (or 8.0 Mtonnes CO$_2$ y$^{-1}$) incorporated into tree biomass annually about 70% was attributed to growth in areas dominated by coniferous species (Paper V). Consequently, the total litter input from the above-ground tree vegetation was estimated to be 0.5 Mtonnes C y$^{-1}$ (0.3 Mtonnes C y$^{-1}$ in coniferous and 0.2 Mtonnes C y$^{-1}$ in deciduous forests). The annual carbon incorporation into forest floor biomass was estimated to be 0.2 Mtonnes C y$^{-1}$ (or 0.9 Mtonnes CO$_2$ y$^{-1}$ Paper V). If all of the 0.7 Mtonnes C produced forest floor biomass is added to the soil as litter annually and all added litter is decomposed, 2.6 Mtonnes CO$_2$ y$^{-1}$ of the soil release originates from the decomposition of litter, resulting in an estimated contribution of organic matter stored before drainage of 7.8 Mtonnes CO$_2$ y$^{-1}$. The total CO$_2$ emissions from the consumption of fossil fuels are somewhere around 50 Mtonnes CO$_2$ y$^{-1}$ (SNV, 2004). Consequently, the release of CO$_2$ from soil organic matter corresponds to approximately 15% of the total amount of CO$_2$ released from the consumption of fossil fuels.

The trees growing on drained organic forests soils could compensate for most of the soil CO$_2$ release. Still the drained forested area in Sweden was a net source of CO$_2$, and therefore a major part of the carbon in tree biomass should be regarded as peat carbon. Consequently, if peat should not be used for energy production neither should the trees growing on drained organic soils. This has profound political implications for the suitability of using the tree biomass from drained forestland for fuel.

If the main goal is to reduce the radiative impact of drained forestland the results suggest that the best alternative is to let all sites become poorly drained by not subjecting them to complementary or remedial drainage. Based on the emission factor for poorly drained soils (Paper V) this should reduce the amount of CO$_2$ released from the soil via the decomposition of soil organic matter by 30%. The tree growth would decrease, but probably not to the same extent. Assessing the average potential productivities for moist drained forests (based on S-NFI data) for the total area of organic drained forestland in Sweden, the carbon uptake by the trees would only decrease by 6%. Consequently, if the total area of drained organic forestland in Sweden becomes moister in the future, the net CO$_2$ emissions are likely to decrease significantly.
The hypothesis – that the contribution of GHG from drained forests on organic soils would have a significant impact on the national GHG budget – was not supported by the results of the up-scaling. The net emissions were small and will probably become even smaller in the future if the drained forested area is not complementary or remediably drained.

Savolainen et al. (1994) argued that it was better to use peat from cultivated soils than peat from virgin peatlands or forest-drained peatlands from a GHG perspective and that it was better to use peat from cultivated peatlands than coal for energy production. Their conclusion was based on the assumption that cultivated peatlands were net sources of GHG, while forest-drained peatlands were net sinks. This is not supported by our findings that the total area of drained forest on organic soil is a net source of GHG. Consequently, it may be better to use peat from drained forestland for energy production than using coal. In addition, I argue that if the wood on the drained peatlands in Sweden is to be used to substitute for fossil fuels it is better to use the peat directly as most of the tree carbon is to be defined as peat carbon. This would also reduce the CO₂ emissions from decomposition of organic matter stored before drainage, which corresponded to 15% of the national consumption of fossil fuels.

MAIN CONCLUSIONS AND FUTURE RESEARCH
This thesis contributes to the knowledge of how forests on drained organic soils function in the context of GHG exchange. Results are presented on GHG exchange in a peat-covered forest with a peat layer thinner than 30 cm and at sites dominated by deciduous trees, i.e. systems not previously well studied, and provides an estimate of the total emissions of GHG from the drained forestland in Sweden.

The most important conclusions were:
- Temporal variations in forest floor CO₂ release from poorly drained forest soils can to a large extent be explained by air temperature and groundwater table.
- All soil fluxes of GHG were to some extent affected by distance to trees. Although, distance to tree stems is not a good predictor for spatial variations in soil GHG fluxes, distance to trees should be taken into consideration when planning sampling schemes for poorly drained organic soils.
- The CN ratio is the most important factor affecting between-site spatial variation in N₂O fluxes, both for poorly and well-drained forests on organic soil. Groundwater table may be the most important factor determining the size of soil CO₂ and CH₄ fluxes.
• Most poorly drained sites in Sweden are probably net sinks for GHG. A large part of the tree biomass carbon could be regarded as peat carbon, which has profound political implications for the suitability of using the wood for fuel.
• If the main objective is to reduce the concentrations of GHG in the atmosphere, the results indicate that the poorly drained forested area should not be subjected to remedial drainage.
• The net impact of drained forestland in Sweden on the national GHG budget is small at present. However, the CO₂ release from decomposition of soil organic matter is significant.

In my opinion more research is needed in several fields. One of the most important concerns is the GHG fluxes from drained soils with a peat layer thinner than 30 cm. The different fluxes of CO₂ also need further resolution. The estimates in this thesis are based on several assumptions about the different sources and sinks of CO₂ (e.g. the carbon accumulation in fine roots and forest floor vegetation), which may not be accurate. Furthermore, the contribution to soil CO₂ release of different sources needs much more attention. Finally, I think that the strong correlation between soil CN ratios and N₂O fluxes needs to be studied more thoroughly, especially at sites with low CN ratios.

REFERENCES
Borken W, Brumme R (1997) Liming practice in temperate forest ecosystems and the effects on CO₂, N₂O and CH₄ fluxes. Soil Use and Management 13, 251-257.


Gundersen P, Callesen I, de Vries W (1998b) Nitrate leaching in forest ecosystems is related to forest floor C/N ratios. Environmental Pollution 102, 403-407.


Marklund LG (1988) Biomassfunktioner för tall, gran och björk i Sverige. SLU, Umeå. 73 pp. (In Swedish)


ACKNOWLEDGEMENTS
Detta arbete skulle inte ha varit möjligt att utföra utan hjälp från en mängd personer. Jag vill speciellt tacka:

Min handledare, Leif Klemedtsson, som alltid varit inspirerande och full av idéer och som brytt sig om mig som person och inte bara som doktorand;

Mina båda bihandledare Mats Nilsson, som kommit med kloka synpunkter på manuskript, och Björn Hånell, som kommit med uppmuntrade tillrop och självförtroendehöjande kommentarer;

Bo Svensson, som alltid ställt upp och funnits tillgänglig under doktorandutbildningens mindre roliga skeden;

All personal på Asa försökspark som bidragit till att fältarbetet blivit avsevärt mycket enklare och roligare. Speciellt vill jag tacka Stefan Eriksson som med aldrig sinande energi och kompetens utfört alla konstiga uppgifter som dykt upp under projektets gång. Jag vill även tacka Pia Rickardsson och Karin Magnusson som under perioder hjälpt till med fältarbetet och Magnus Ripström, ägaren av Asa vandrarhem, som varit mycket flexibel med bokningar;

All teknisk och administrativ personal på Tema Vatten som tålmodigt hjälpt mig lösa alla möjliga olika problem;

Alla andra på Tema Vatten som bidragit till att skapa en miljö i vilken det varit roligt att arbeta. Flera av er har även kommenterat manuskript och på andra sätt hjälpt mig att få min avhandling bättre. Jag vill speciellt tacka David Bastviken, Andreas Berg, Gunnar Börjesson, Åsa Danielsson, Jenny Grönwall, Elisabeth Johansson, Lena Lundman, Malin Mobjörk, Peter Wihlborg, Julie Wilk och Mats Öquist;

Per Weslien, Maria Gustafsson, Josefin Norman och Robert Björk som analyserat prover och kommenterat manuskript;

Hans Toet, som gjort alla kartor;

John Blackwell som språkgranskat artiklar och kappa;

Mina föräldrar, mor- och farföräldrar och alla vänner utanför universitetet som stöttat mig, låtit mig invadera sina hem när jag haft möten i olika delar av landet samt, framför allt, påmint mig om att det finns viktigare saker i livet än att bli doktor;

Mattias som ställt upp för mig under denna period både genom att läsa, kommentera och diskutera texter och ta hand om allt hushållsarbete. Jag lovar att försöka återgåda dig!