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NORTHERN PEATLANDS IN GLOBAL CLIMATIC CHANGE

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Preface

The high water table of undisturbed peatlands shifts the natural balance of growth and decay towards the accumulation of partially decomposed organic matter as peat. Peatlands form an extensive biome in the boreal and temperate zones and their estimated carbon store is 455 Pg, appproximately one third of the total world pool of soil carbon.

Northern peatlands are important in regulating the global climate. While sequestering carbon dioxide, these peatlands release ca. 24-39 Tg methane annually to the atmosphere. This is 5-20 % of the annual anthropogenic methane emissions to the atmosphere. The greenhouse gas balance of peatlands may change as a consequence of water level draw-down after land-use change, or if summers become warmer and drier, as has been predicted for high latitudes after climatic warming. Subsequent emissions of methane would decrease, whereas emissions of carbon dioxide and nitrous oxide would increase.

Within the Finnish Research Programme on Climate Change (SILMU), the research project "Carbon Balance of Peatlands and Climate Change" (SUOSILMU) has been under progress since 1990. It is a co-operative research project, with research groups from the Universities of Helsinki and Joensuu, the Finnish Forest Research Institute, the National Public Health Institute and the Finnish Environment Agency. The research consortium of this project organized a workshop entitled "Northern Peatlands in Global Climatic Change - Hyytiälä Revisited" October 8 - 12. 1995.

The main objective of the workshop was to review the state of the art of the carbon cycling research in natural and managed peatlands. The role of peatlands in the greenhouse effect, their response and feedback to the predicted climate change, and the consequences of land-use changes were assessed, and the future research needs were evaluated. The latest information on the role of peatlands in the atmospheric change was given in 50 posters and 4 key lectures. Results of SUOSILMU projects were demonstrated during a 1-day field excursion to one of the intensive study sites, Lakkasuo near Hyytiälä.

The workshop was hosted by the Department of Forest Ecology, University of Helsinki, in cooperation with The Finnish Research Programme on Climate Change. The material support from the Academy of Finland, Vapo Ltd. and the staff of the Hyytiälä Forestry Station was invaluable, and is highly appreciated.

On behalf of the organizers, we wish to thank all those who participated in the workshop. This proceedings volume contains the latest information in the field of peatland ecosystem - atmosphere interaction, and we feel that it is of value to all those who are interested in these questions.

Helsinki, January 3, 1996

Jukka Laine   Harri Vasander

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Vegetation responses

The effect of elevated CO_2 concentration on photosynthesis of *Sphagnum fuscum*

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Objectives

To measure net photosynthesis of *Sphagnum fuscum* in long term exposure to four CO_2 levels at semi-natural conditions.

To find out if there is an acclimation of net photosynthesis into prevailing CO_2 concentrations.

To measure the moisture dependent net photosynthesis at various CO_2 concentrations of samples grown at different CO_2 concentrations.

Methods

Sphagnum fuscum samples were grown on petri dishes (15 cm diam. and 1.5 cm height) at CO_2 concentration of 350, 700, 1000 or 2000 ppm for three months in greenhouses (n=6).

The samples were irrigated daily by deionized water. CO_2 exchange was measured by a infra-red gas analyzer (IRGA, URAS 2 T, Hartman & Braun) connected to data acquisition system (Hewlett-Packard 3052A). Pneumatically regulated trap-type plexi glass gas exchange chambers, shaped and sized to a petri dish, were used.

During the growth period, net photosynthesis of *Sphagnum fuscum* was measured under varying light, at various water contents and temperature conditions (Fig 1).

After the growth period, photosynthesis measurements were made in a phytotron at constant light (170 W m^{-2}) and temperature (22 °C) at four CO_2 concentrations. The samples were let gradually dry and simultaneously measured at various water contents (WC % dry weight), Fig.1.

Growth chambers

350 ppm	700 ppm

1000 ppm	2000 ppm

- a long term exposure to a constant CO_2 level.
- circumstances:
  * variable light
  * variable temperature
  * adequate water content
  * constant CO_2 level.

350	700
1000	2000

Phytotron
- a short term exposure to four CO_2 levels
- circumstances:
  * constant light $(170\ W\ m^{-2})$
  * constant temperature $(22\ ^{o}C)$
  * variable water content
  * four CO_2 levels

Fig. 1. Arrangement of the measurement of photosynthesis and the appplied conditions in the growth chambers and in the phytotron.

Results

Photosynthesis in long term CO_2 exposure

Elevated carbon dioxide concentration caused clear increase in the steepens of the response curve at the same radiation compared to lover CO_2 level. The saturation level of the net photosynthesis was elevated by increased carbon dioxide concentration, too (Fig. 2).

The acclimation of photosynthesis

During the growth period, there had been clear acclimation to the prevailing CO_2 concentrations. The cores grown at higher concentrations had lowered CO_2 net exchange capacity at all lower CO_2 concentrations measured (Fig.3)

At elevated CO_2 concentration, the increase of water content above the optimal range does not decrease net photosynthesis like at lower CO_2 concentrations (especially at 350 ppm). The long term CO_2 concentrations during the growth period does not seem to have an effect on the net photosynthesis at various water contents at increased CO_2 (Fig. 4).
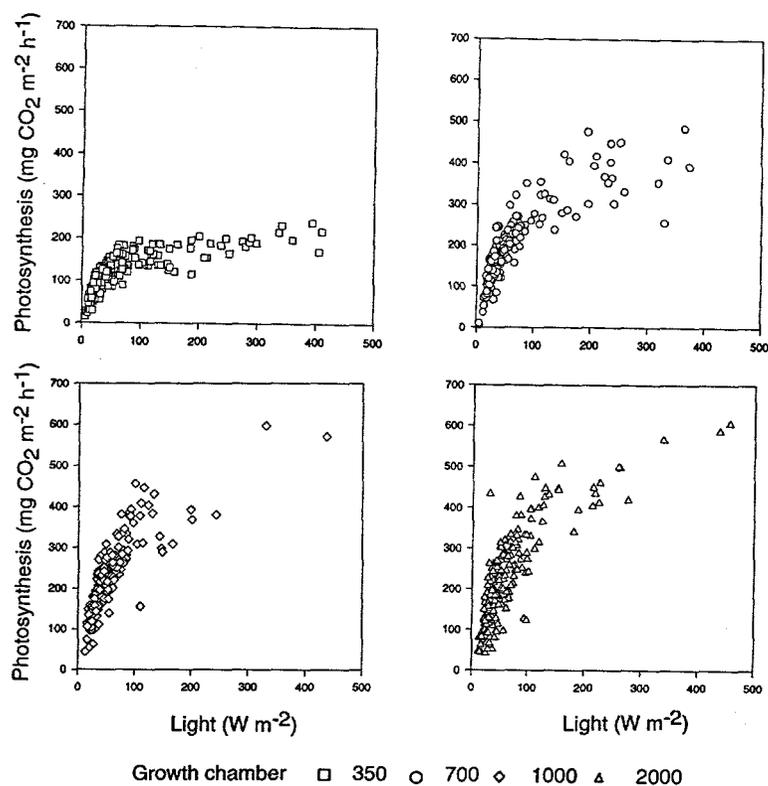
Fig. 2. Net photosynthesis (mg CO_2 m^{-2} h^{-1}) of *S. fuscum* at four CO_2 levels. Figures 350 ppm (upper left), 700 ppm (upper right), 1000 ppm (lower left) and 2000 ppm (lower right) chamber. Temperature $\geq$18 - <25 °C, WC $\leq$2200 %DW, radiation 0-500 µmol m^{-2} s^{-1}.
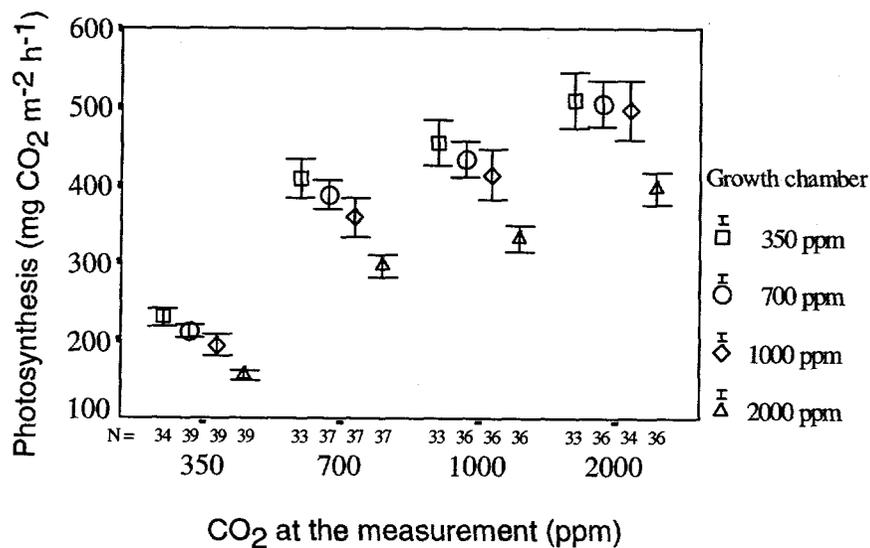
Fig. 3. Net photosynthesis of *S. fuscum* at four ambient CO_2 levels. At the figure, samples were grown at either 350 ppm, 700 ppm, 1000 ppm or 2000 ppm chamber. Temperature 22 °C, radiation 170 W m^{-2} and WC 800-1500 %DW. Mean ± SE presented.

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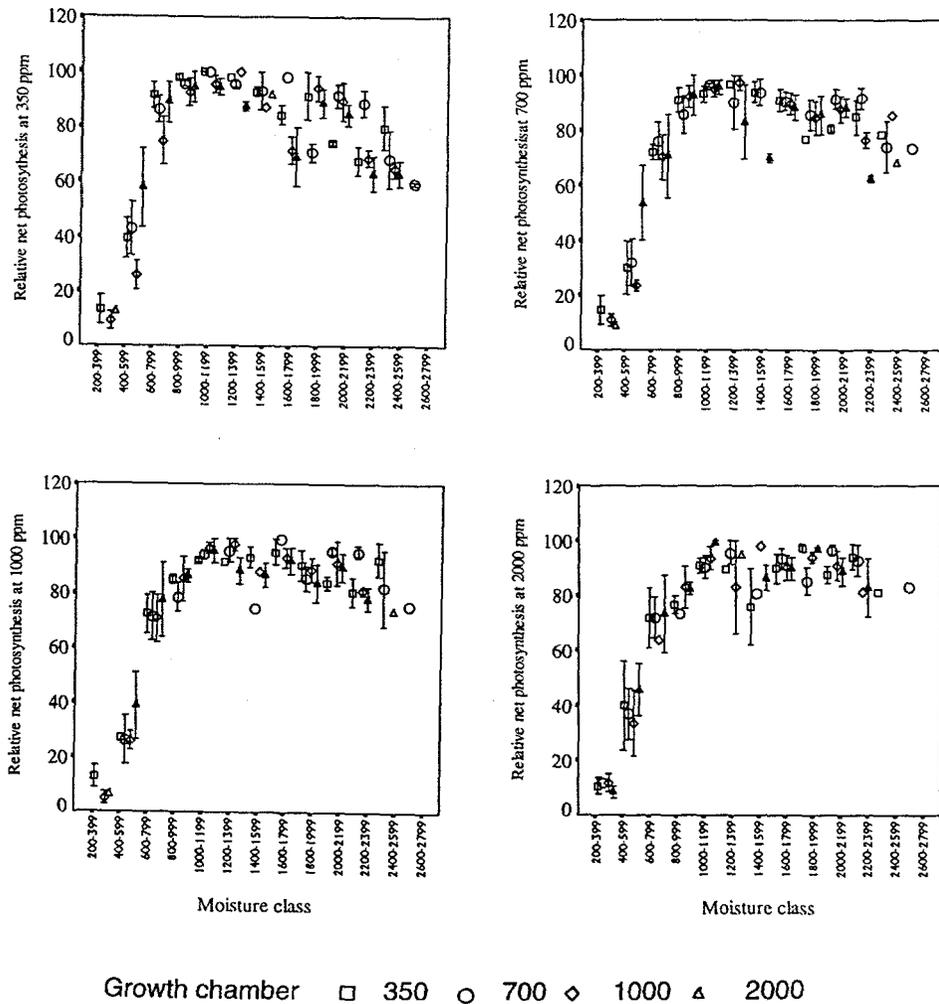
Fig. 4. Relative net photosynthesis (% of maximum) of *S. fuscum* at various water contents and at four ambient CO$_2$ levels. At the figure, samples were grown at 350 ppm (upper left), 700 ppm (upper right), 1000 ppm (lower left) and 2000 ppm (lower right) growth chamber. Temperature 22 °C, radiation 170 W m^{-2}. Mean ± SE presented.

Conclusions

1. Increased ambient CO$_2$ level causes higher net photosynthesis at the optimal water content in *Sphagnum fuscum*. Photosynthesis at elevated CO$_2$ is enhanced, especially at high light flux plants could utilize CO$_2$ very effectively .

2. *Sphagnum fuscum* clearly acclimates to ambient long term CO$_2$ levels and a short term elevation of available carbon dioxide increases net photosynthesis. Equally, decrease of available concentration decreases net photosynthesis.

3. At elevated CO$_2$ concentration, the maximal level of net photosynthesis is gradually shifted at higher water contents. At high CO$_2$ concentration there is no decrease in net photosynthesis at high water contents. Former ambient CO$_2$ concentration does not have an effect on the level of photosynthesis at various water contents.

The effect of elevated CO_2 and N-input on *Sphagna* with different trophy

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Introduction

During recent decades the amount of nitrogen deposition in the northern hemisphere has increased markedly. This deposition may have significant impact on species composition or vitality in future. It can be hypothesized that at increasing rates of nitrogen deposition *Sphagna* representing various levels of trophy (ombrotrophy - meso-eutrophy) react dissimilarly. Ombrotrophic mosses might lose in competition to more nutrient tolerating or demanding species.

Carbon dioxide is one of the main factors influencing plant photosynthesis and productivity. The atmospheric concentration of CO_2 has increased since the pre-industrial age, and this trend is predicted to continue till the next century. *Sphagnum* is an important CO_2 sink via peat formation and one of the objectives of this study was to find out possible growth response of four Sphagnum species to increased CO_2.

This experiment focused on measurements length increment and production at four N deposition and CO_2 levels on an ombrotrophic *Sphagnum fuscum* (Schimp.) Klinggr., indifferent *S. angustifolium* (Russ.) C. Jens., indifferent *S. magellanicum* Brid. and meso-eutrophic *S. warnstorfii* Russow.

Material and methods

Sixty four 15 cm thick cores for each species of living *Sphagnum* were taken for growth measurements from sampling sites located in eastern Finland. The sets of samples were grouped in series of four sample each. The plastic rings containing samples were placed in trays in which the water level was maintained at 10 cm for *S.* fuscum and at 7 cm for all the other species below the capitulum. The samples were fertilized weekly with new NH_4NO_3 enriched Rudolph's nutrient solution (Rudolph & Voigt 1986), so they received equivalent of either 0, 10, 30 or 100 kg N ha^{-1}a^{-1}. The trays were placed in four growth chambers in which the ambient CO_2 concentration was maintained automatically at different concentration.

Using the plastic strip method (Lindholm 1990), the length increment observed was at least three months. Production was calculated using the total length increment during the study period, the stem biomass values, and information about the surface area of the samples.

Results
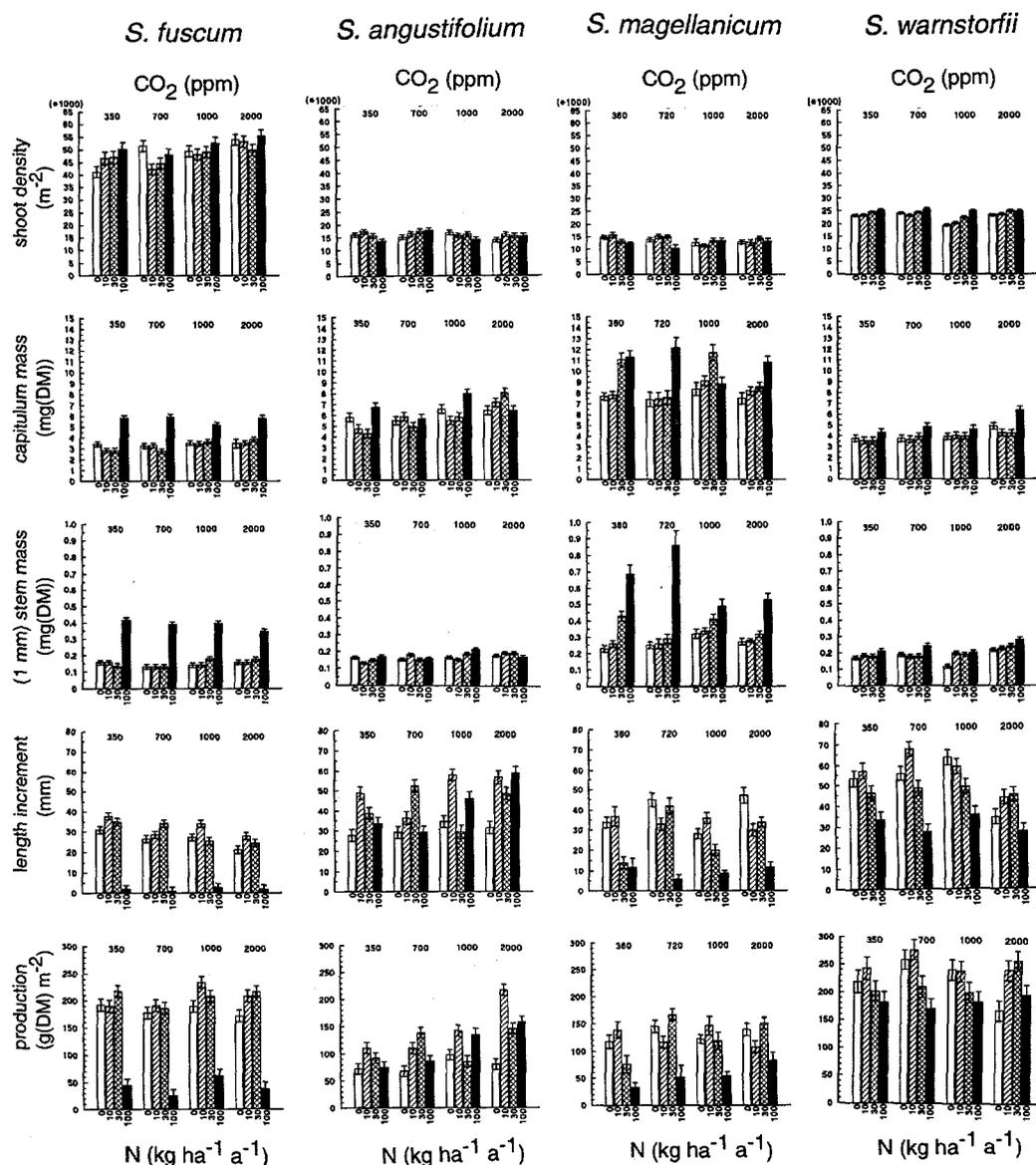
The shoot density (m^{-2}), dry mass of capitulum (g(DM) $1cm^{-1}$), dry mass of stem (g(DM) $1mm^{-1}$), length increment (mm per experimental time) and production (g(DM) m^{-2} per experimental time) of *S. fuscum*, *S. angustifolium*, *S. magellanicum* and *S. warnstorfii* at the four N deposition levels and at four CO_2 concentrations. Mean ± S.E presented. Note! The growing of these four species varied from 70 days to over 100 days *i.e.* over a three months period.

Conclusions

1. These four *Sphagnum* species differed clearly in the response of the production parameters to the experimental treatments, though the great amount of variation in the response was surprising. The overall response, like production, to treatments may be quite similar but the causes may be different.

2. Response to N deposition was found to reflect the ecological background of the *Sphagna*. Ombrotrophic *S. fuscum* clearly suffered from increased N load in every aspect. Indifferent *S. angustifolium* is widely spread at various growth habitats and it was least affected. *S. magellanicum* is also an indifferent species, but it is more restricted at lawns and specialized, this can also been seen in lowered tolerance of increased N. Though the availability of N was practically unlimited for the meso-eutrophic *S. warnstorfii*, character of this species is probably more connected with the pH and Ca content than available N content.

3. The effect of CO_2 on measured factors were lower than expected. In studies of photosynthesis of *Sphagna* at increased CO_2 (Silvola 1985, 1990), increase of photo-synthesis in higher CO_2 has been clearly shown. In this study the effect was quite variable in growth, but some evidence of the effect of CO2 was gained. The effect was species dependent and had an almost opposite effect on various parameters of production. However, further studies are needed to reveal the fate of assimilated carbon at increased CO_2.

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Potential NH_4^+ and NO_3^- uptake on various *Sphagnum* species

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Objectives

To test differences in NH_4^+ and NO_3^- uptake on various *Sphagnum* species under laboratory conditions. The studied species represent two gradients: the minerotrophy-ombrotrophy (mire margin - mire expanse) gradient, and the hummock-hollow gradient.

Material and Methods

The samples were collected from Åkhultmyre (57°06'N, 14°33'E) in the beginning of October 1995. The number of capitula per unit area was counted for each of the species. Cores of living *Sphagnum* were kept at a constant temperature and light with diurnal cycle in a laboratory for two weeks before the experiment. The water was maintained at a level of 7 cm below the capitulum with a water collected from the bog.

A test-solution containing a 100 µmol l^{-1} of N was prepared by dissolving an appropriate amount of NH_4NO_3 in distilled water. The solution pH was adjusted to the level of 5 - 5.5 by adding mild hydrochloride acid to the solution. A volume of 100 ml of the test-solution was poured in open 10 x 10 x 3,5 cm containers.

Capitula of *Sphagnum, i.e.* the uppermost 1 cm of shoot were sampled and mounted in conical holes in cellular plastic fleets (30 holes/fleet) so that the distal ends of the capitula came into contact with the solution below. Three fleets were prepared for each species. The fleets were set in the test-solution and the containers were set mechanically in a slight horizontal motion on a worktop.

Samples, volume of 5 ml, were sucked from the solution first 5 minutes after the start. Sampling was repeated after 2 h, 4 h and 6 h. After the last samples were taken, the capitula were collected and dried (24 h/85 °C) before weighing.

Decrease in concentrations of NH_4^+ and NO_3^- in the test solution was measured and the rate of uptake in *Sphagnum* of NH_4^+ and NO_3^- was calculated on the basis of ion concentration and the dry mass of the capitula or capitulum density (Twenhöven 1992).

NH_4^+ and NO_3^- concentrations of the liquid samples were analyzed by Flow Injection Analysis (FIA) and for the uptake of NH_4^+ and NO_3^-, one way analysis of variance (ANOVA) was used.

The concentration of N in solution

It was noticed that the amount of NH_4^+ left in the solution after 6 hours was distinguishably higher in some of the *Sphagnum* species. In order to compare the uptake of NH_4^+ of different species in a approximately same concentration in the test solution, the species were grouped in two groups - species with capitulum mass exceeding 7 mg (*S. fallax, S. pulchrum, S. papillosum* and *S. magellanicum.*) and species with lower capitulum mass (*Sphagnum fuscum, S. rubellum* and *S. cuspidatum*). For species with high capitulum weight, results were calculated on the basis of samples taken at 4 hours after the start and for the other group from samples taken at 6 hours, respectively.

Results

Capitulum dry mass and capitulum density

The dry mass of top 1 cm and the capitulum density differed with a high statistical significance ($p<0.001$) (Fig. 1). Dry masses were low at hummock forming species and increased in species considered to be lawn species. The shoot density was highest on hummock species *S. fuscum* and *S. rubellum*. The biomass of capitula (g m^{-2}) arranged according to the hummock - hollow gradient as: *S. fuscum* > *S. rubellum* > the lawn *spp.* > *S. cuspidatum*.

Nitrogen uptake in relation to mass

The rate of uptake of NH_4^+ (μmol g^{-1} h^{-1}) differed significantly between the studied *Sphagnum* species ($p<0.001$). The highest uptake rate was found on hummock forming species (Fig. 2). Lawn species showed considerably lower uptake rates and the rate of uptake of NH_4^+ in *S. cuspidatum* was a bit lower than in hummock species, thus near the highest values of lawn species.

The rate of uptake of NO_3^- (Fig. 2) differed also significantly ($p.<0.001$) among the studied species. Compared to the uptake rate of NH_4^+, the uptake of NO_3^- was considerably lower and the variation within a species could be quite high. The highest uptake of NO_3^- was on *S. cuspidatum* and *S. fallax* which differed significantly from most of the other *Sphagnum* species.

Nitrogen uptake in relation to individual

The highest individual uptake capacity of NH_4^+ (μmol $capitulum^{-1}$ h^{-1}), was on the lawn species *S. fallax, S. pulchrum* and *S. papillosum* (Fig. 3). Although the capitulum mass of *S. magellanicum* was highest of all the species, the individual uptake capacity of NH_4^+ was surprisingly low due to the low uptake efficiency of this species.
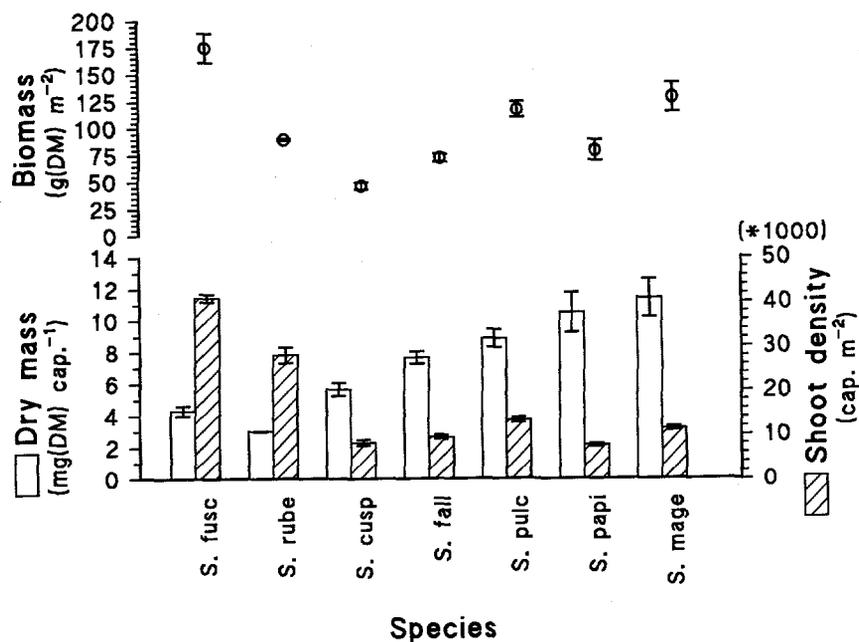
Fig. 1. Dry mass of the uppermost part of shoot (mg(DM) cap.$^{-1}$, open bars), number of capitula per unit area (cap. m^{-2}, filled bars) and biomass of uppermost parts in a sq. meter (g(DM) m^{-2}) of *Sphagnum*. Mean ± S.E presented.
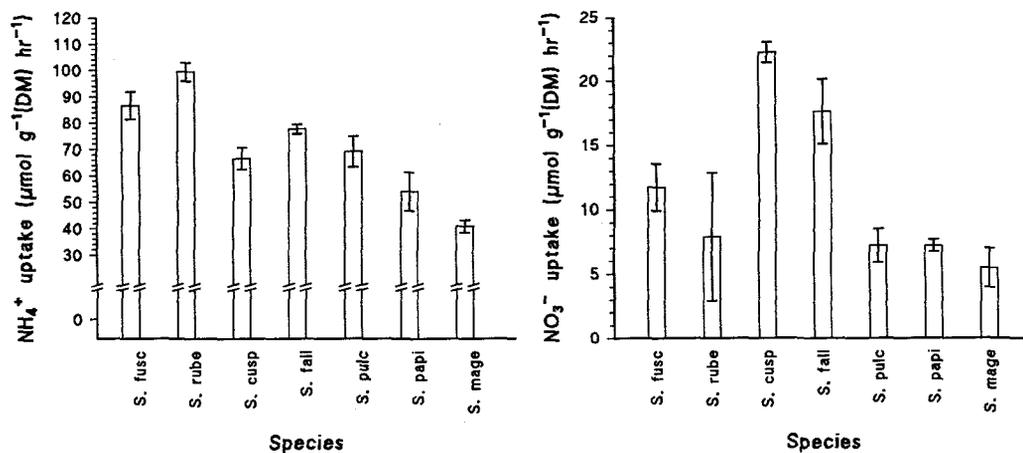
Fig. 2. The rate of uptake of NH$_4^+$ (left) and NO$_3^-$ (right) as (μmol g^{-1}(DM) h^{-1}) in seven *Sphagnum* species. Mean ± S.E. presented (n=3).
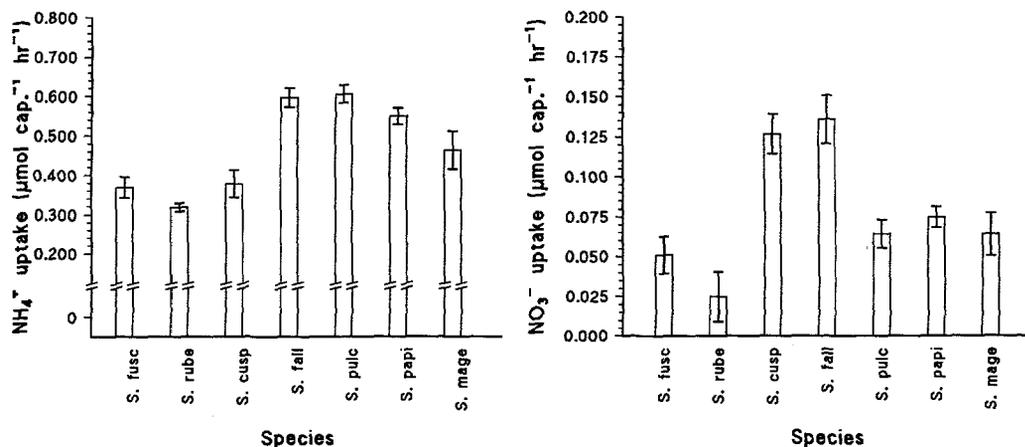
Fig. 3. The individual uptake rate of NH_4^+ (left) and NO_3^- (right) as (μmol cap.$^{-1}$ h^{-1}) in seven *Sphagnum* species. Mean $\pm$ S.E. presented.
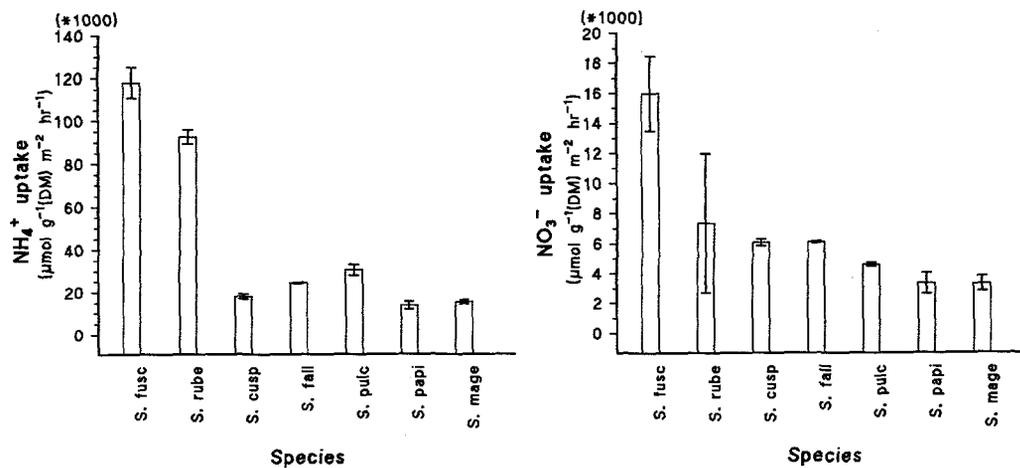
Fig. 4. The uptake rate of NH_4^+ (left) and NO_3^- (right) as (μmol g^{-1}(DM) m^{-2} h^{-1}) in seven *Sphagnum* species. Mean $\pm$ S.E. presented.

The highest individual uptake rate of nitrate (μmol capitulum^{-1} h^{-1}) was on *S. fallax* and *S. cuspidatum* and they differed significantly from the other species ($p<0.001$). The remaining three lawn forming *Sphagnum* species had a bit lower uptake efficiency, but significantly higher than that of the hummock species (Fig. 3).

Nitrogen uptake in relation to cover

The hummock species were most efficient in filtering NH_4^+ (Fig. 4) and could be distinguished from the other. The lawn species and *S. cuspidatum* were about 80% less effective.

In the uptake efficiency of NO_3^- (μmol g^{-1}(DM) m^{-2} h^{-1}, Fig. 4), the differences between species were not so pronounced due to high variation in hummock species. Of the studied *Sphagnum,* only *S. fuscum* could be easily distinguished from the other species with reasonably high uptake rates.

Conclusions

There are distinct differences in the uptake rate between various *Sphagnum* species and these differences seem to be due to both structural and environmental factors:
1) on individual basis the highest uptake rate is in species with large capitulum and high number of ion exchange sites *i.e.* lawn species,
2) on dry mass basis the most effective species are the hummock species though they have a low DM,
3) from the ecosystem point of view, hummock species with high number of shoots per unit area and high uptake rate on dry mass basis, are the most effective species in filtering available nitrogen.

References

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Effect of N deposition on free amino acid concentrations in two *Sphagnum* species

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Introduction

Nitrogen saturation of ecosystems occurs when the availability of nitrogen is not a growth limiting factor. This situation can be reached through fertilisation or by nitrogen deposition. Prolonged nitrogen saturation may overload the ecosystem and cause changes in the vigour and eventually in the composition of plant communities. But before this stage is reached, certain changes in nitrogen metabolism occur. These changes can be used as an early warning of nitrogen overload to ecosystems (Näsholm.et al. 1994).

A change in the amino acid pool of plants has been used as an indication of various kind of stress, including, temperature, nutrient imbalance, shading, drought or excess water (e.g. Singh *et al.* 1973; Steward and Lahrer, 1980; Rabe, 1990). It has been postulated that such stresses have an effect on protein synthesis but not on the nitrogen uptake of plants (Rabe, 1990). The result is an increase in the concentration of NH_4^+ ions in plant cells, which may have toxic effects to the plant and the processes that assimilate the free NH_4^+ ions. One of such process is the synthesis of amino acids, especially those containing a significant proportion of nitrogen, e.g. arginine, glutamine and asparagine (Rabe and Lowatt, 1986; Miflin and Lea; Givan, 1979).

This study about the quantification of amino acids in two species of *Sphagnum* mosses is a part of a larger study, the aim of which is to clarify how a number of *Sphagnum* species will cope with climatic change and nitrogen deposition. *Sphagna* are the most important members of the peat forming communities in the boreal zone. *Sphagnum* communities are formed by species specialised to grow in conditions of low nutrient availability, mainly provided via deposition. The present structure and composition of mire communities may be endangered due to elevated levels of nitrogen deposition that have persisted over the last few decades.

Present knowledge about the quantities of amino acids in *Sphagna* and the effects of nitrogen deposition is quite limited (e.g. Press and Lee 1982, Woodin and Lee 1987). Most such studies have been made on vascular plants, especially trees.

Methods

Samples of two *Sphagnum* species - *S. fuscum* (Schimp.) Klinggr. and *S. angustifolium* (Russ.) E. Jens. - were collected from undisturbed stands from an area of low nitrogen deposition in Northern Carelia (Finland). The samples in plastic containers (6 × 15 cm) were placed in trays in which the water level was maintained at 10 cm (*S. fuscum*) and 7 cm (*S. angustifolium*) below the capitulum. The samples were irrigated weekly for three months with Rudolph's nutrient solution (Rudolph and Voigt 1986) containing various concentrations of NH_4NO_3.

A the end of the three month period, a fresh sample (300 mg) of the capitulae (top 1 cm of shoot) was cut and frozen in liquid nitrogen and homogenised (Mikro-Dismembrator II). The homogenised samples were then extracted with 4 ml 0.001 M K_2HPO_4 + 1 ml ethanol for 30 min in an ultrasound bath and centrifuged (ca. 5 min at 5000 rpm). The concentration of amino acids in the clear extract were determined using HPLC analysis (pre-column derivation with FMOC and fluorescence detector, $em\lambda$ = 313 nm, $ex\lambda$ = 260 nm) (Näsholm *et al.* 1987).

Results

The total free amino acid concentration in both moss species increased with increasing N additions (Fig. 1).
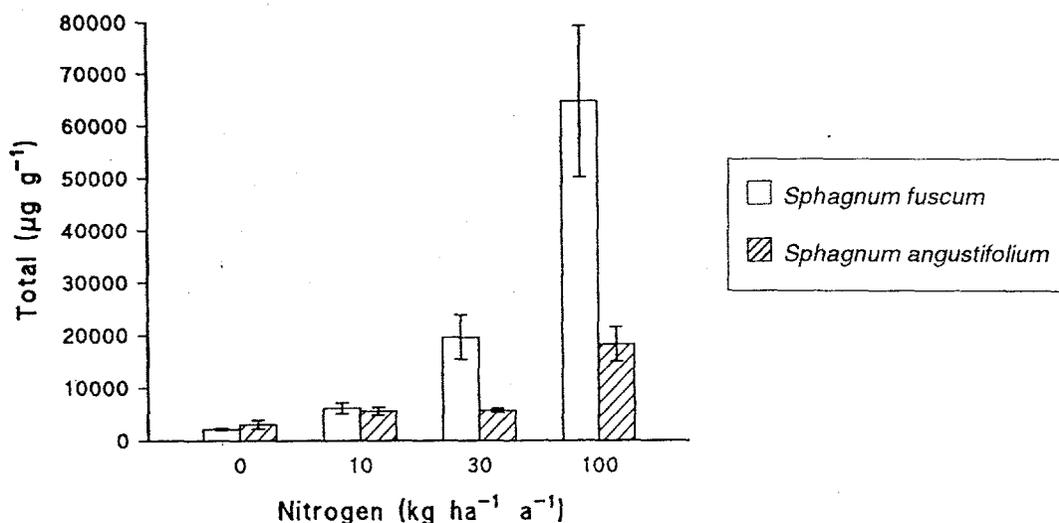
Fig. 1. Concentrations of total free amino acids (µg/g dry weight) at different nitrogen deposition levels in *S. fuscum* and *S. angustifolium*.
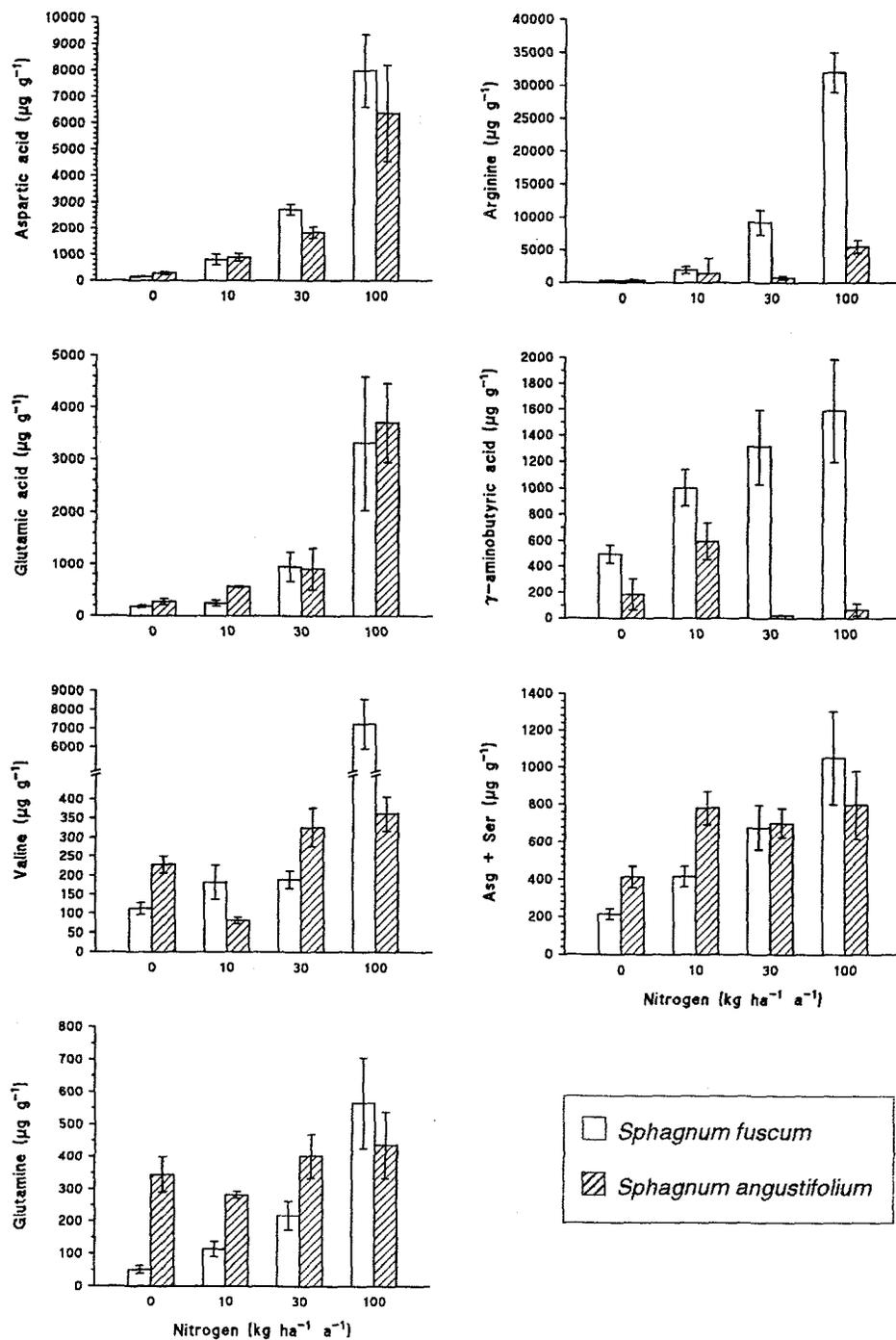
Fig. 2. Behaviour of dominant free amino acids (µg/g dry weight) in *S. fuscum* and *S. angustifolium* at different nitrogen deposition levels.
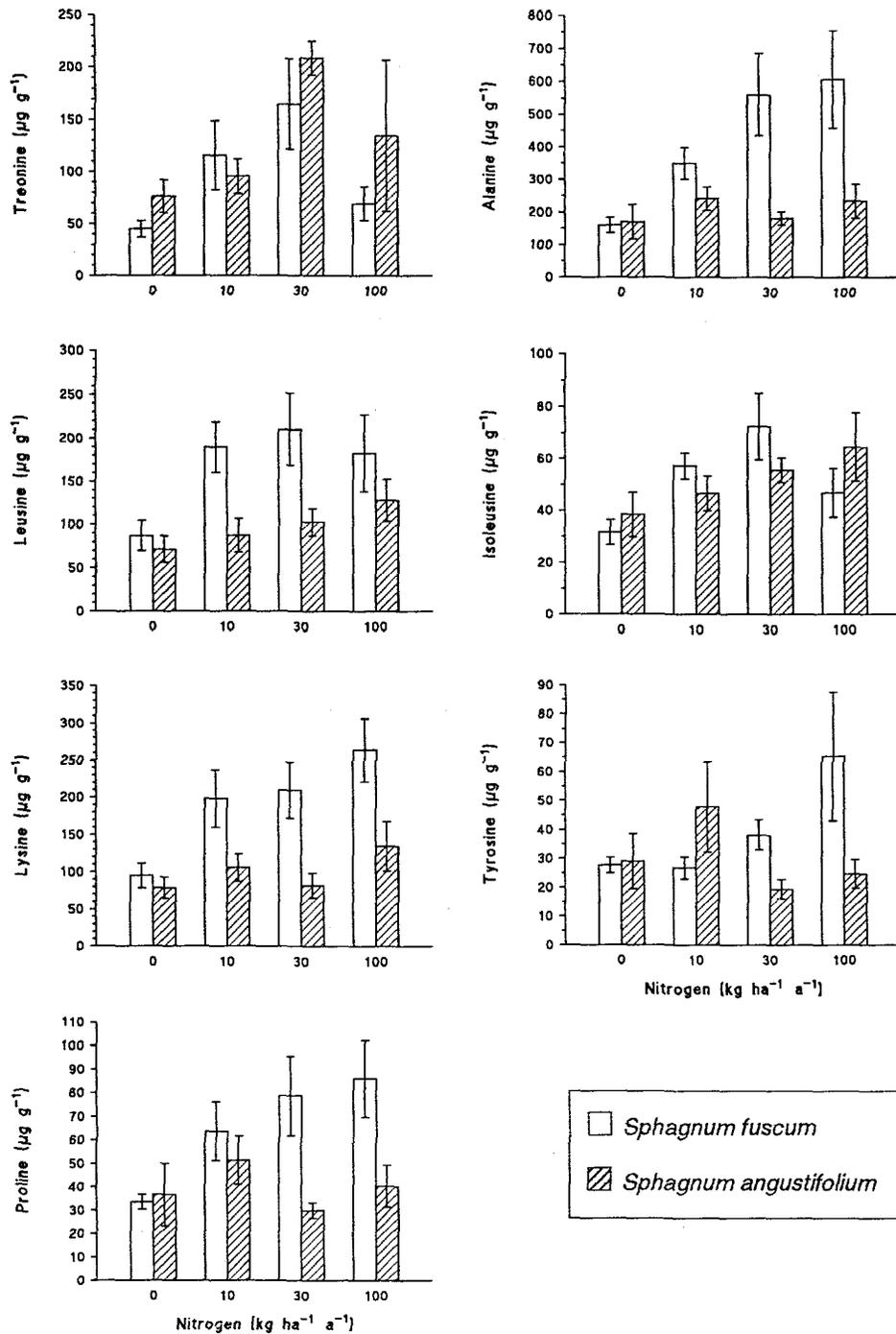
Fig. 3. Behaviour of less dominat free amino acids (µg/g dry weight) in *S. fuscum* and *S. angustifolium* in different nitrogen deposition levels.

In both species, the range of main three amino acids present in mosses formed at N applications, was limited. For S. *fuscum* the main amino acids were arginine, aspartic acid, glutamic acid, γ-aminobutyric acid, valine, glutamine and asparagine/serine (incomplete separation), and alanine (Fig. 2). Arginine was the most dominant, accounting for more than half of total amino acid concentration at higher nitrogen deposition levels. Less important amino acids were e.g. leusine, lysine, isoleusine, proline and threonine, but their concentrations also increased with increasing nitrogen deposition (Fig. 3).

For S. *angustifolium*, the main amino acids were aspartic acid, arginine, glutamic acid, asparagine/serine and glutamine, respectively (Fig. 2). The aspartic acid and glutamic acid concentrations clearly increased with increased nitrogen deposition. However, arginine, γ-aminobutyric acid and also the concentrations of some of the minor constituents (e.g. glycine, lysine, alanine and tyrosine, Fig. 3) were lowest at the deposition level of 30 kg N ha^{-1}. The highest quantity of γ-aminobutyric acid was found at the deposition level of 10 kg N ha^{-1} in S *angustifolium*.

The length growth of S. *fuscum* was highest at 10 kg N ha^{-1} a^{-1} N deposition, biomass production at 30 kg N ha^{-1} a^{-1}, but the plants were almost moribund at 100 kg N ha^{-1} a^{-1} deposition (Jauhiainen 1994). S. *angustifolium* showed a small but clear increase in length growth and biomass production with 10 and 30 kg N ha^{-1} a^{-1} depositions (Jauhiainen *et. al.* 1995). However, the highest N deposition level did not have a clear effect on growth, as happened with S. *fuscum*.

Discussion

Several stresses can promote the accumulation of free amino acids in plants. Nitrogen fertilisation has been shown to increase the concentration of the free amino acid pool in tree foliage, in particular arginine, glutamine and proline (Durzan and Stewards 1967, Ebell and McMullan 1970, Kim *et al.* 1987). Our results for *Sphagnum* mosses showed that the increase in the concentration of total free amino acids pool(Fig. 1.) was also due to a few dominant amino acids, in particular arginine, glutamic acid and aspartic acid (Fig. 2).

It is known that glutamic acid and aspartic acids play a major role in the metabolism of nitrogen in plants (Näsholm and Ericsson, 1990). Our results suggest that stress in Sphagnum species arising from nitrogen deposition is handled by diverting the nitrogen into a few specific amino acids for storage for later protein synthesis, which has been the suggested mechanism also in vascular plants (Rabe and Lovatt 1986).

The formation of γ-amino butyric acid is mainly through the decarboxylation of glutamic acid (Streeter and Thompson, 1972). Therefore the concentration of γ-amino butyric acid has been found to be inversely correlated to that of glutamic acid (Näsholm and Ericsson 1990). In our experiment the concentration of γ-amino butyric acid increased with N deposition in S. *fuscum* but not S. *angustifolium*. However, the concentration of glutamic acid increased with N deposition in both species (Fig. 2.).

Arginine and its intermediates and other basic amino acids, asparagine and glutamine, may have a significant role in the assimilation of free NH_4+ (Miflin and Lea, 1977; Givan, 1979; Rabe and Lovatt, 1986). For *S. fuscum and S. angustifolium*, arginine accounted for a large part of the increase in total free amino acids related to N deposition. The concentrations of asparagine and glutamine also increased with N deposition level, particularly in the case of *S. fuscum*.

Proline has been shown to be an indicator of stress, for example drought (Singh et al. 1973), frost (Sakai and Larcher, 1987) and may function as a transport and storage form of nitrogen (Sagisaka and Araki, 1983). According to our results, the concentration of proline (Fig. 3) was very low compared to the dominant amino acids. In the case of *S. angustifolium*, nitrogen deposition had no effect on proline concentrations but in *S. fuscum* the concentration of proline was roughly doubled at the highest nitrogen deposition level.

While amino acids such as alanine, threonine, leusine, isoleusine and lysine tended to increase in response to N deposition levels in the case of *S. fuscum*, such trends were less evident in the case of *S. angustifolium*. This difference may reflect a difference in protein synthesis between the two species (Rabe, 1990).

In conclusion, N deposition increased the production of amino acids in the two *Sphagnum* species studied; asparagine, glutamic acid and aspartic acid in particular. The assimilation of NH_4^+ into the production of these amino acids can be considered as a detoxification reaction by the plants to help cope with high N deposition. Elevated concentrations of these amino acids in Sphagnum mosses may therefore be used to indicate a metabolic response to N deposition in advance of physiological changes.

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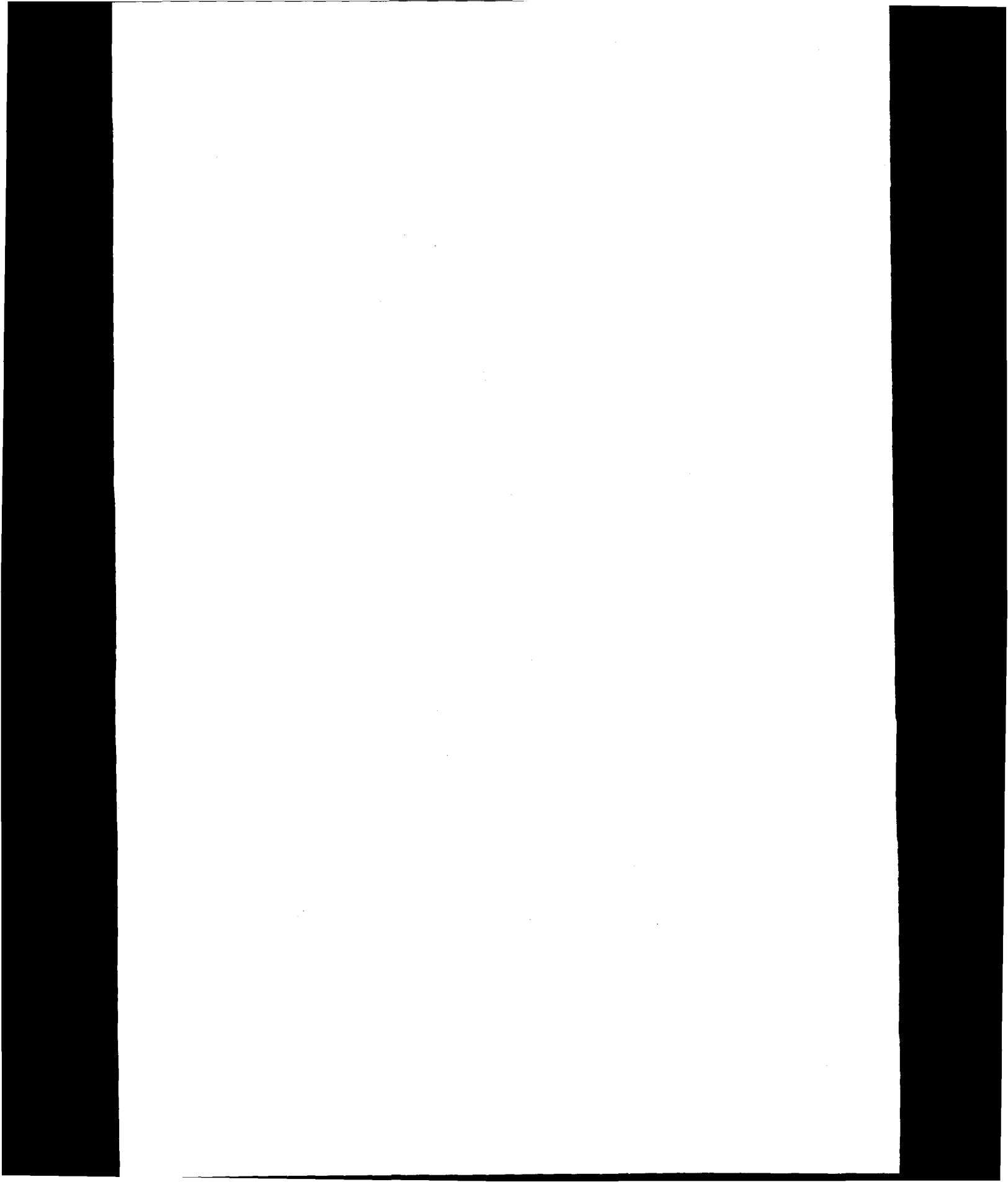
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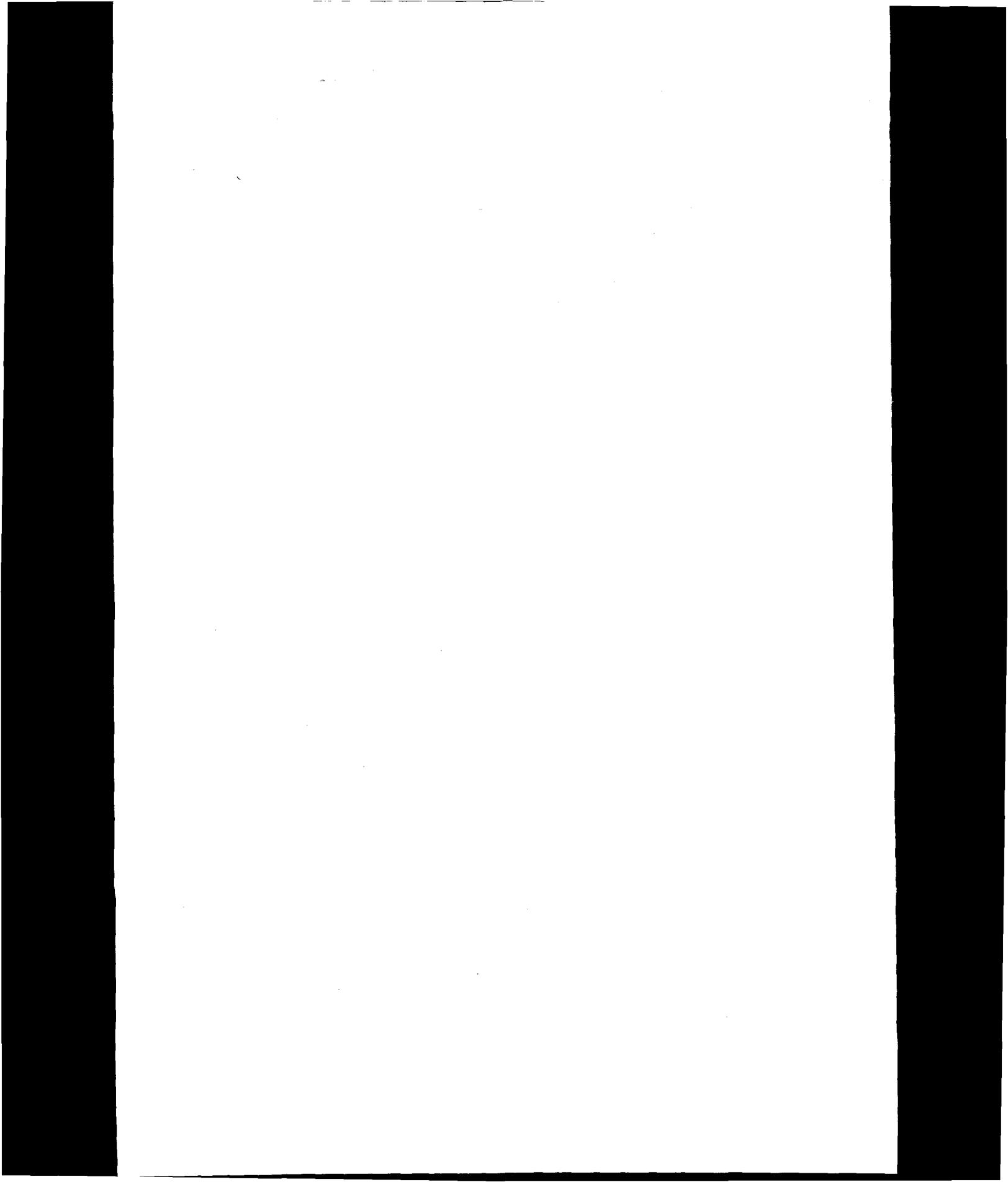
Relocation of carbon from decomposition of ^{14}C-labelled needle and fine root litter in peat soil

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Introduction

Drainage of peatlands promotes a shift of biomass and production from the ground vegetation to the trees. Thus, the above-ground (e.g. needles) and below-ground (roots) litter production of trees increases (e.g. Vasander 1990). Fine roots in particular are an important factor in the carbon and nutrient cycle in forest ecosystems (e.g. Vogt et al. 1991). A major part of the annual net primary production of trees may be allocated below ground (e.g. Ågren et al. 1980, Grier et al. 1981, Keyes & Grier 1981), the relative proportion being smaller on fertile sites than on less fertile ones (e.g. Finér & Laine 1994).

For modelling the carbon balance of drained peatlands, it is important to know the fate of carbon from newly introduced and decomposing litter. Newly added and fertilised tree litter material may be decomposed at a rate different than litter from the ground vegetation (e.g. sedges and mosses).

The objectives of this study are to study the pathways of decomposing litter carbon in peat soil and to evaluate the use of the litterbag method in a controlled environment.

Material and methods

In the beginning of summer 1994, one-year-old nursery grown Scots pine seedlings were labelled with $^{14}CO_2$. They were grown for one growing season to allow the ^{14}C to permeate into the new structures of the needles and the roots. In order to produce material with two different nitrogen levels, two different fertilising programmes were used: half of the seedlings received 1.1 kg N m^{-2} in June, 1.1 kg N m^{-2} in July and 1.9 kg N m^{-2} in August. The nitrogen dose for the other half of the seedlings was 1.1 kg N m^{-2}, 0 kg N m^{-2} and 1.9 kg N m^{-2}, respectively. The seedlings were fertilised 4 times in June, 3 times in July and twice in August.

In the autumn, the new needles and living fine roots (diam. < 2mm, in this case all lateral roots) were harvested and dried for some weeks at room temperature. Before weighing, the material was dried at 60°C to a constant weight. Root and needle litterbags were prepared by weighing exactly 1g of litter into 10x10 cm^2 nylon bags with 1 mm mesh size. Subsamples of the root and needle litter material were analysed to determine the average ^{14}C-activity. The litter samples were combusted in a Junitek oxidiser and the ^{14}C-activity was measured in a Wallac liquid scintillation counter.

For the incubation experiment, undisturbed Sphagnum- and Carex-peat cores with the living top moss layer were taken from the Lakkasuo mire complex in Orivesi, central Finland (61°48'N, 24°19'E, 150 m above sea level). The cores were taken from minerotrophic (Carex-peat) and ombrotrophic (Sphagnum-peat) drained peatland sites. The peat cores were taken into plastic tubes about 12 cm in diameter and 60 cm long.

In January 1995, the incubation experiment was started and one litterbag was placed horizontally into each peat core. The needles were placed under the 1-2 cm thick living moss layer and the roots were placed 4-5 cm below the surface. The incubation temperatures used were 15°C and 25°C. Half of the cores were incubated for 4 months, representing one growing season, and the rest for 8 months. There were four replicates for each fertilisation treatment, temperature treatment, peat type and litter type. Thus the experiment consisted of a total of 128 cores.

The ground water table was maintained at 40 cm in all tubes. The relative humidity of air was kept at 80% during the day (8.00 - 18.00) and at 90% during the night (18.00 - 8.00). The lights were switched on from 8.00 to 16.00. The light intensity was kept constant at about 650 - 720 $\mu mol\ m^{-2}\ s^{-1}$ throughout the experiment. The cores were watered (="rainfall") once a week with water which mimicked the amount and chemical composition of the local rain water. The incubation was carried out in growth chambers.

The $^{14}CO_2$-flux from 32 cores (one of each replication) was analysed. Pressurised air was directed through a cuvette and CO_2 trapped with a trapping agent (LUMASORB II). The trapping agent samples were analysed once a week in a Wallac liquid scintillation counter for ^{14}C-activity. The quantity and quality (TOC and ^{14}C-activity) of the percolating water was also measured.

At the end of the experiment, the mass and ^{14}C-activity of the remaining litter in each litter bag was analysed. The moss layer was harvested and the peat cores were cut into different layers which were dried at 105°C to a constant mass before weighing and milling. Subsequently, three subsamples of each peat and moss layer were analysed for ^{14}C-activity by combustion in a Junitek oxidiser and activity counting in a Wallac liquid scintillation counter.

Preliminary results and discussion

Only the results of the 4-month-experiment will be shown in this paper because the preparation and processing of the 8-month-material and data are still ongoing.

The $^{14}CO_2$-emission showed a maximum in the first few weeks of the experiment, thereafter the emission decreased and levelled off (Figures 1a and 1b). The emission peak might have been due to decomposition of soluble and easily degradable constituents.

Fig.1a.
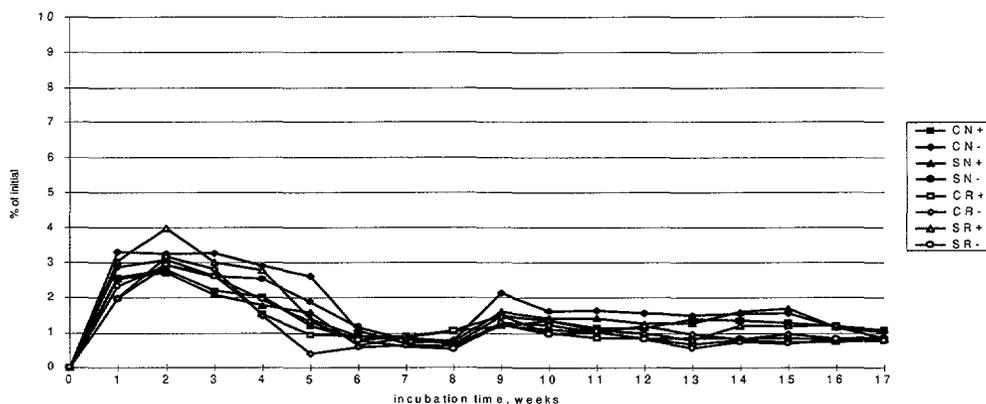
Fig. 1b.
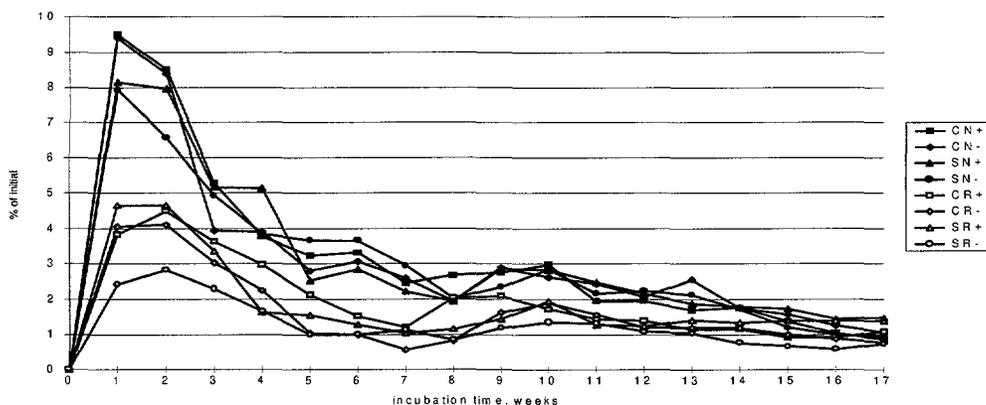
Fig. 1a and 1b. The ^{14}C-flux from labelled litterbags during the 4-month-experiment at 15°C (Fig. 1a) and 25°C (Fig. 1b). The results are expressed as % of the initial activity estimated from subsamples.
C Carex-peat
S Sphagnum-peat
N needles
R roots
+ litter from seedlings fertilised with more nitrogen
- litter from seedlings fertilised with less nitrogen (see text for details)

The mass and ^{14}C-activity loss from the needles was greater than that from the roots at both incubation temperatures (Figures 2a and 2b). The needles decomposed faster at 25°C than at 15°C, whereas the effect of temperature on the decomposition of the

fine roots was not as clear. At 15°C the needles fertilised with the higher nitrogen dose decomposed faster than those fertilised with the lower dose; at 25°C no such differences were found. In all cases the ^{14}C-activity loss from needles was lower than the corresponding mass loss. The fine roots of the less fertilised seedlings were decomposed faster (i.e. mass loss was greater) but the ^{14}C-activity loss was smaller than that of the roots which had received more nitrogen. The decomposition of needles was faster in Carex-peat than in Sphagnum-peat at 15°C. At the higher temperature there were no differences between peat types. Peat type did not affect the decomposition of root litter at either temperature. The results indicate that the decomposition of needle and root litter differs in peat soils, probably due to the differences in the chemical composition of the litter type. In earlier studies, the decomposition of fine root litter of trees, like that of leaf litter, has been related to the chemical composition of the decaying material (Berg & Staaf 1980, Berg 1984). In this study the decomposition of needle litter was more affected by temperature and peat type than the root litter.

From 30 to 50% of the total initial ^{14}C-activity of needle litter and 20-40% of that of root litter was released into the atmosphere at 15°C. The percentages at 25°C were 60% and 20-40% respectively (Figures 3a and 3b). The rest of the label not remaining in the litterbags was distributed between the peat and moss layers or leached into the ground water.

From 10 to 30 % of the labelled carbon lost from the litterbags was still found in the peat soil cores (Figures 4a and 4b). This proportion was highest in the case of the less fertilised roots. About 10% of the labelled carbon lost from the litterbags was trapped by the photosynthesis of the mosses. That percentage was not much affected by temperature, although in many cases the mosses seemed to grow badly at the higher temperature. The labelled carbon retained was found mostly in the upper layers, especially in the moss layer above the litterbag and in the layers directly below the litterbag (0-1 cm and 1-6 cm) (Figures 5a and 5b). However, at least small amounts of labelled carbon were found from the lower peat layers and from the ground water.

Earlier litterbag decomposition studies presupposed that the mass and carbon lost from litterbags is released to the atmosphere. The preliminary results of this study show that after 4 months decomposition in a controlled environment, from 10 to 30 % of the carbon lost from the litterbags is still found in the system. The litter material used in this study probably had a greater proportion of carbon in soluble constituents than true litter would have had. That may have affected the amount of decomposition and the flux of labelled carbon to the atmosphere, moss layer, peat and ground water. However, these preliminary results suggest that we should re-evaluate the results of litterbag decomposition experiments and bear in mind that not all the carbon is necessarily lost to the atmosphere when calculating the carbon balances of drained peatlands.

Fig. 2a: 15°C.

Fig. 2b: 25°C.
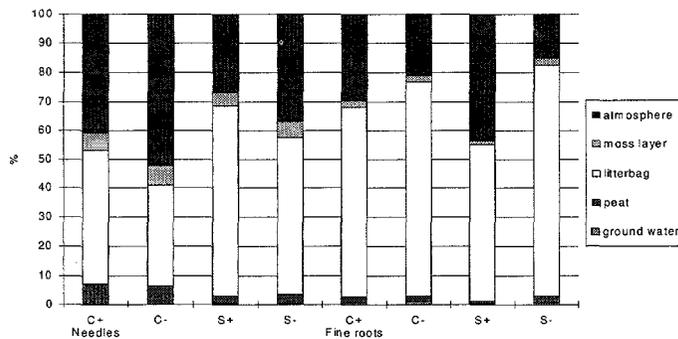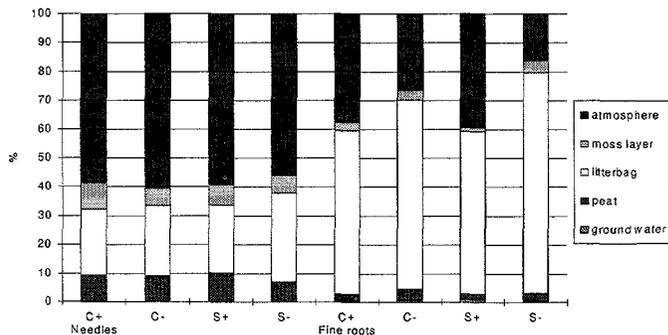
Figures 2a and 2b.
The average mass and ^{14}C-activity losses from the litter material at the end of the 4 month incubation period at 15°C and 25°C temperature. Each bar represents the mean of 4 replications.

Fig. 3a: 15°C.

Fig. 3b: 25°C.

Figures 3a and 3b.
The relative distribution of the estimated total initial ^{14}C-activity between the atmosphere, moss layer, litterbag, peat and ground water at the end of the 4 month incubation period at 15°C and 25°C temperature.
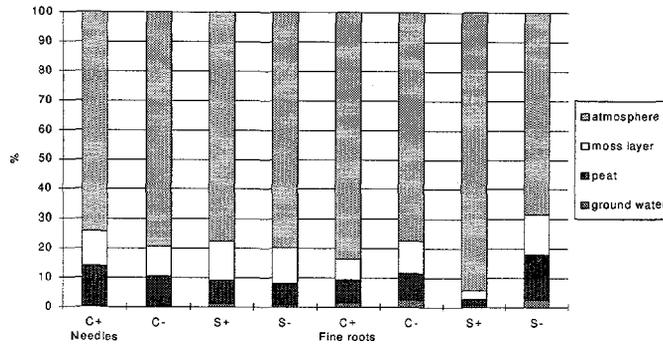
Fig. 4a: 15°C.
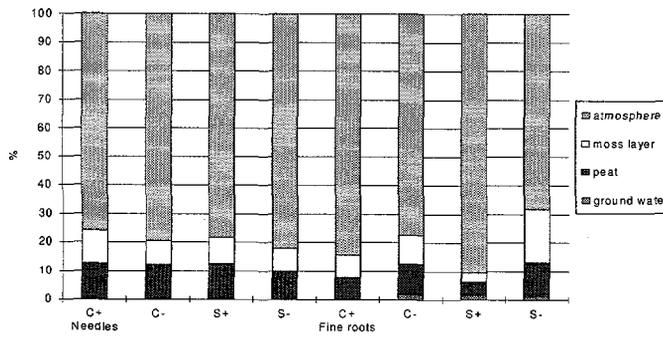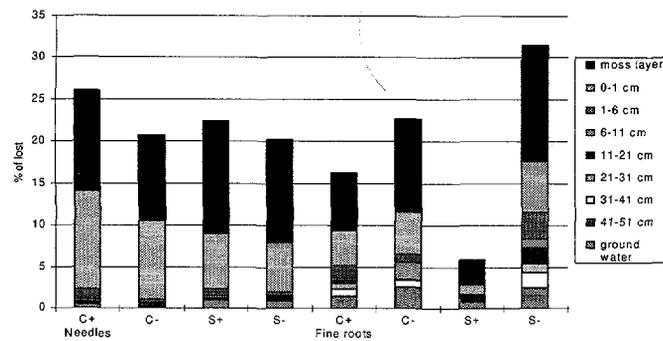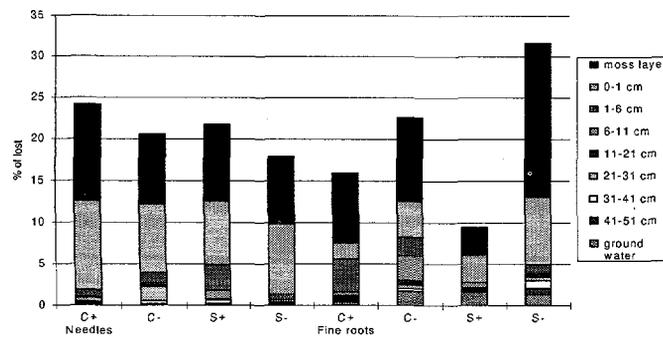
Figures 4a and 4b.
The relative distribution of the ^{14}C-activity lost from the litterbags to the atmosphere, moss layer, peat and ground water.

Fig. 4b: 25°C.

Fig. 5a: 15°C.

Figures 5a and 5b.
The distribution of the ^{14}C-activity lost from the litterbags to the moss layer and peat layers and ground water.

Fig. 5b: 25°C.

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Fine root production at drained peatland sites

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Introduction

The preliminary results of the Finnish project "Carbon balance of peatlands and climate change" show that fine roots play an important role in carbon cycling on peat soils (Finér et al. 1992ab, Finér & Laine 1994, Saarinen et al. 1994, Laiho & Finér 1995). After drainage the roots of mire species are gradually replaced by the roots of trees and other forest species. Pine fine root biomass reaches a maximum level by the time of crown closure, some 20 years after drainage on pine mires (Heikurainen 1955, Laiho & Finér 1995).

The studies on drained peatlands suggest that fine root production accounts for 10-40% of the net primary production of Scots pine (Finér et al. 1992b). The proportion of 15-30% has been reported for young Scots pine stands growing on mineral soil sites in Sweden (Axelson & Axelsson 1986). The annual detrital input to peat by Scots pine roots can be equal to the needle litterfall (Finér et al. 1992b).

The measurement of fine root production is susceptible to many errors. The ingrowth bag method is less laborious than the sequential coring method. On mineral soil sites the two methods have given similar quantitative results, when the time elapsed from the installation of the bags to the sampling has been long enough; two years in Scots pine stands (Persson 1979). On peat soil the ingrowth bag method has been used by Backéus (1990) for studying fine root production of boreal open bog vegetation. No comparison between the sequential coring method and ingrowth bag methods has been done on peat soil so far.

The aim of this study is to compare the results of the sequential coring method and the ingrowth bag method used for estimating fine root production on three drained peatland sites of different fertility. The results are preliminary and continuation to the work done in the study Pine root production on drained peatlands, which is part of the Finnish project "Carbon cycling on peatlands and climate change".

Material and methods

Study sites

The material for this report was collected from three sample plots, two of them were located in Orivesi, central Finland (61° 48' N, 24° 19' E, 150 m a.s.l.) and one plot in Rääkkylä eastern Finland (62° 14' N, 29° 50' E, 81 m a.s.l.). The sample plots in Orivesi were drained in 1961 and the sample plot in Rääkkylä 1967. The sample plot on tall sedge fen (VSN) in Orivesi was fertilized and planted with Scots pine (*Pinus sylvestris*) seedlings after drainage. The Scots pine stand on tall sedge pine fen (VSR) in Orivesi and that on low-shrub pine bog (IR) in Rääkkylä were naturally afforested before drainage. Phosphorus and calcium concentrations were highest in the surface peat of VSN site and lowest on IR site (Table 1). The tree stand volume was 123 m^3/ha on VSN site, 150 m^3/ha on VSR site and 134 m^3/ha on IR site. The corresponding stem numbers/ha were 3860, 2370 and 2060.

Table 1. Nutrient concentrations in the surface peat layers on study sites.

		VSN 0-20 cm	20-40 cm	VSR 0-20 cm	20-40 cm	IR 0-20 cm	20-40 cm
N	%	2.120	1.850	2.340	2.240	1.150	1.230
P	%	0.120	0.068	0.083	0.055	0.063	0.043
K	%	0.031	0.009	0.026	0.008	0.038	0.009
Ca	%	0.356	0.294	0.277	0.317	0.232	0.232
Mg	%	0.027	0.019	0.021	0.023	0.041	0.033
S	%	0.221	0.175	0.199	0.218	0.155	0.189
Fe	%	0.701	0.449	0.355	0.307	0.127	0.090
Mn	ppm	34.3	23.4	13.8	5.8	65.0	7.7
Zn	ppm	14.2	4.8	12.9		25.4	7.4
Cu	ppm	4.8	2.7	3.2		6.5	8.1
B	ppm	4.8	4.9	16.1	17.8	2.1	1.3

The sequential coring method

Before starting the production measurements the depth distribution of root systems was investigated. It was found that there were only a few roots growing deeper than 20 cm from the peat surface (Finér et al. 1992a). Thus the sampling was restricted to the uppermost 20 cm-peat-layer.

Living and dead root biomasses were determined once a month during the summer using the core method. Forty peat cores (24 cm^2) were taken systematically on each plot from the 0-20 cm peat layer. Sampling started in May 1991 and continued to September 1993. All material was not available for this report.

Root production was calculated by balancing transfers in living and dead roots biomass compartments according to the decision matrix presented by Fairley and Alexander (1985). The estimates were calculated in two different ways. In the first case only the statistically significant differences were used (MANOVA, Duncan's test, $p < 0.05$, N=40) and in the second case all the differences between the sequential samplings were included.

The ingrowth bag method

At the beginning of June 1993 on each plot eighty ingrowth bags (15 cm^2) were systematically inserted into the peat to the depth of 22 cm starting from the surface. The mesh size of the nylon bags was 6 mm. Bags were filled with commercial milled fertilized peat. The aim was to fill the bags to the same bulk density as the surrounding peat, however, the peat was somewhat compacted while installing the bags in soil. Twenty bags were recovered in September 1993 and the next 20 at the beginning of June 1995. The rest were taken up in July, August and September 1995 to study the sesonality in fine root production. In laboratory the biomass of pine and field layer roots found from the bags was determined. All roots were regarded living. The results of the first two samplings were available for this report.

Results

Biomass

The living and dead fine root biomass of Scots pine and field layer vegetation did not much fluctuate between years on the study sites (Fig. 1). The fine root biomass increased with the decrease in the fertility of site. The proportion of dead pine roots out of the total root biomass was greater on IR site than on VSR or VSN sites, and the proportion for the field layer vegetation roots was the opposite. This indicates differences in root decomposition and production between sites.

Only very few roots had grown into the ingrowth bags during the first growing season (results not shown). The same phenomenon is known from mineral soil sites (Persson 1979). In this study that could result from the differences in physical and nutritional properties between the bags and surrounding soil and the slow recovery of roots from the disturbance due to the installation of bags. In peatlands it is impossible to fill the ingrowth bags with the similar soil as the soil surrounding the bags.

At the end of the second growing season the field layer vegetation root biomass was greater in ingrowth bags than that found in cores at VSN and VSR sites (Fig. 2). At the IR site there were much less field layer vegetation roots in ingrowth bags than in cores. The field layer on IR site was dominated by dwarf shrubs, whereas on VSR and VSN sites vegetation consisted more of birch, herbs and sedge like species. Pine root biomass was the lowest on IR site in ingrowth bags and on all sites there were less pine roots in ingrowth bags than in cores. All pine roots found from the ingrowth bags

were thinner than 2 mm, and only a few roots of field layer species had grown thicker than that.
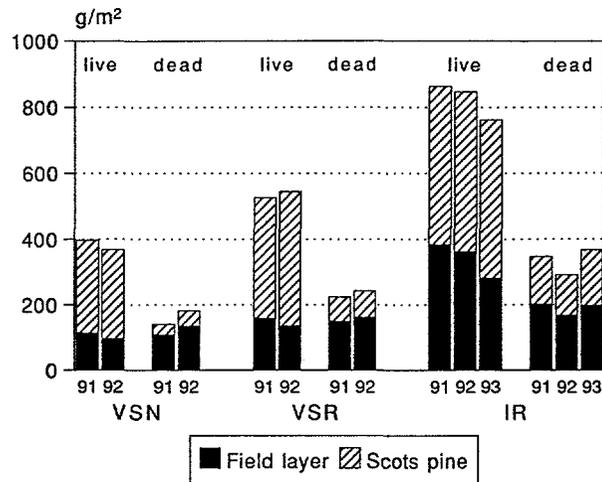
Fig 1. Mean living and dead fine root (∅ ≤ 10 mm) biomass of Scots pine and field vegetation on the study sites.
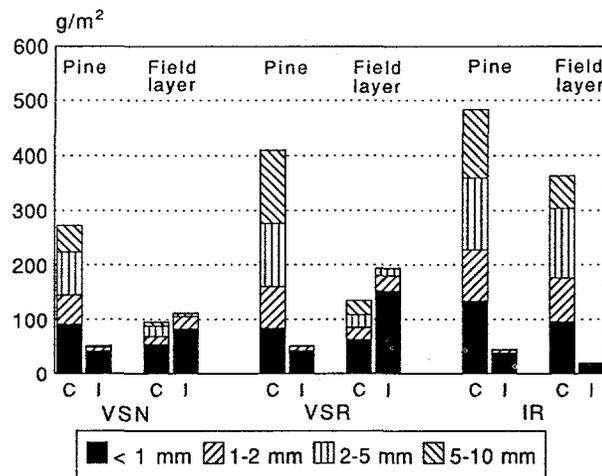
Fig 2. Mean living fine root biomass of Scots pine and field layer vegetation on the sites during 1991-1992 measured with the coring method (C) and the root found from the ingrowth bags (I) after two growing seasons.

The results indicate that the root biomass in the ingrowth bags was not in the balance with that in the soil around the bags even after two growing seasons. The results suggest differences in growth rate between species. The roots of sedge like species and birch grew faster into the bags than those of pine or dwarf shrubs. Pine root

biomass has stabilized in ingrowth bags within two years on mineral soil sites, but two years was not long enough for forest dwarf shrubs (Persson 1979).

Root production and turnover

Except for field layer vegetation on VSN and VSR sites, the ingrowth bag method gave lower estimates for fine root production than the coring method (Tables 2 and 3). The production estimates based on the coring method suggested that pine root production increased with the decrease in the fertility of site, thus supporting the results, which show that poor nutrient status in soil increases the accumulation of biomass into the root systems (Keyes & Grier 1981). The ingrowth bag method indicated smaller variation in pine root production between sites and the production decreased along with the decrease in fertility. This is contradictory to the results of the coring method. The ingrowth bag method and the coring method (statistically significant differences) gave consistent relative differences for the magnitude of production of the field layer vegetation roots. Thus the root production was the lowest on IR site and the greatest on VSR site. Since the root biomass had not reached balance in ingrowth bags, the results of coring method were probably more realistic than those of the ingrowth bag method.

Table 2. Fine root production g/m^2 in ingrowth bags (n=20) during two growing seasons on the study sites. Standard deviation in parenthesis.

	Pine		Field layer		Total	
VSN	51.7	(72.7)	111.8	(172.1)	163.5	(171.2)
VSR	51.3	(70.8)	193.4	(216.9)	244.8	(226.8)
IR	44.3	(51.4)	19.3	(39.3)	63.6	(60.0)

Table 3. Fine root production (g/m^2). In the first (I) case only the statistically significant differences in live and dead root biomass were used and in the second case (II) all differences.

		Pine		Field layer		Total	
		I	II	I	II	I	II
1991	VSN	97	210	95	176	192	386
	VSR	121	255	135	252	256	507
	IR	94	368	28	320	122	688
1992	VSN	29	271	36	170	65	441
	VSR	56	274	204	379	260	653
	IR	106	276	80	302	187	578
1993	IR	136	148	0	284	136	432
$\bar{x}$ 91-92	VSN	63	241	65	173	129	414
	VSR	88	265	170	316	258	580
	IR	100	322	54	311	154	633

The results of the ingrowth bag method were in accordance with the root turnover estimates (Table 4). The turnover rate was greater on more rich VSN and VSR sites

than on poor IR site and it was greater for field layer vegetation than for pine on rich sites.

Table 4. Fine root turnover (production/living biomass). In the first (I) case only the statistically significant differences in live and dead root biomass were used and in the second case (II) all differences.

		Pine		Field layer		Total	
		I	II	I	II	I	II
1991	VSN	0.34	0.74	0.83	1.56	0.48	0.97
	VSR	0.33	0.69	0.85	1.58	0.48	0.96
	IR	0.20	0.77	0.07	0.83	0.14	0.79
1992	VSN	0.11	0.99	0.38	1.77	0.18	1.20
	VSR	0.14	0.68	1.50	2.79	0.48	1.20
	IR	0.22	0.57	0.22	0.83	0.22	0.68
1993	IR	0.28	0.31	0	1.01	0.18	0.57
$\bar{x}$ 91-92	VSN	0.23	0.87	0.61	1.67	0.33	1.09
	VSR	0.24	0.69	1.18	2.19	0.48	1.08
	IR	0.21	0.67	0.14	0.83	0.18	0.74

Conclusions

In this study the fine root biomass was greater on the poor site than on the rich sites. Pine fine root production increased with the decrease in fertility. The production of field layer species was greater on the rich sites than on the poor site. Root turnover was also greater on the rich sites than on the poor site.

The results suggested that the ingrowth bag method measured more root activity than the magnitude of production. More than two growing seasons would have been needed to balance the root dynamics in the ingrowth bags with the surrounding soil. That time would probably have been longer on the poor site than on the rich ones and longer for pine and field layer consisting of dwarf shrubs than for field layer consisting of sedge like species and birch.

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Decomposition of mosses in different peats

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Introduction

Decomposition of organic matter is influenced by the properties of plant material and by the environmental factors. The plant properties are nutrient content, lignin content, age of the plant, structure, amount of readily degradable compounds and resistance to different substances. The environmental factors are water chemistry, soil moisture (climate, drainage), animals, pH and temperature. (Mikola 1954, Lieffers 1988, Clymo 1965, Verhoeven and Toth 1995, Nilsson 1992, Coulson and Butterfield 1978, Berg 1984, Rochefort et al. 1990)

Lignification seems to protect plant tissues to a certain extent against decomposition (Eggelsmann et al. 1993). As analyzed by Fyles and McGill (1987), the lignin content of *Pleurozium schreberi* is 22.6 % and *Hylocomium splendens* is 25.5 %. According to Bartsch and Moore (1984) the lignin content of *Sphagnum angustifolium* in lawn is 10.0 % and in hummock 7.5 %.

Decomposition under controlled laboratory conditions allows comparison of litter types on the basis of inherent properties which influence decomposition, irrespective of climatic controls (Fyles and McGill 1987).

[14]C-labelling technique has been used before in plant material decomposition studies; e.g. roots and litter (Jones and Darrah 1994, Scheu and Wolters 1991). The advantage of the ^{14}C labelling is that it allows a more complete analysis of the decomposition system than the analyses by other techniques in field study (Cogle et al. 1987).

Our objectives are:

1. to determine the decomposition rate of mosses in different peats,
2. to estimate the carbon flow from decomposed mosses to the microbial biomass carbon in different peats,
3 to determine the ^{14}C flow from labelled mosses to the environment: peat and CO_2.

Sites

The material for this experiment was collected on June 16th, 1994, from four sample plots on the Lakkasuo mire complex in the central Finland (61°48'N, 24°19'E). Two of the sample plots are located in the virgin part and two in the drained (1961) part of the mire. The site types were mesotrophic flark fen and tall sedge fen. The sampling depth was 0 - 10 cm.

The moss types selected for the experiment were three of the most common ones in the Finnish forests. The mosses used were *Pleurozium schreberi*, *Sphagnum angustifolium* and *Hylocomium splendens*.

Experiment

^{14}C-labelled moss was added to different peat materials in nylon bags and incubated at 25 °C for 70 days. Respired $^{14}CO_2$ was measured weekly. After incubation, the ^{14}C remaining in peat and moss was measured. Also the ^{14}C extracted before and after fumigation was analyzed. Microbial biomass carbon in the peat was measured with fumigation extraction method (Martikainen and Palojärvi 1990) before and after the experiment using the specific k_{EC} factor estimated for these peats (Karsisto and Varjola, this volume).

Results and discussion

The loss of mass and ^{14}C were highest for *Sphagnum angustifolium* in both virgin and drained peat. Decomposition was highest in order *Sphagnum angustifolium* > *Pleurozium schreberi* > *Hylocomium splendens*.(Figs. 1 and 2).This result correlate the lignin content of mosses
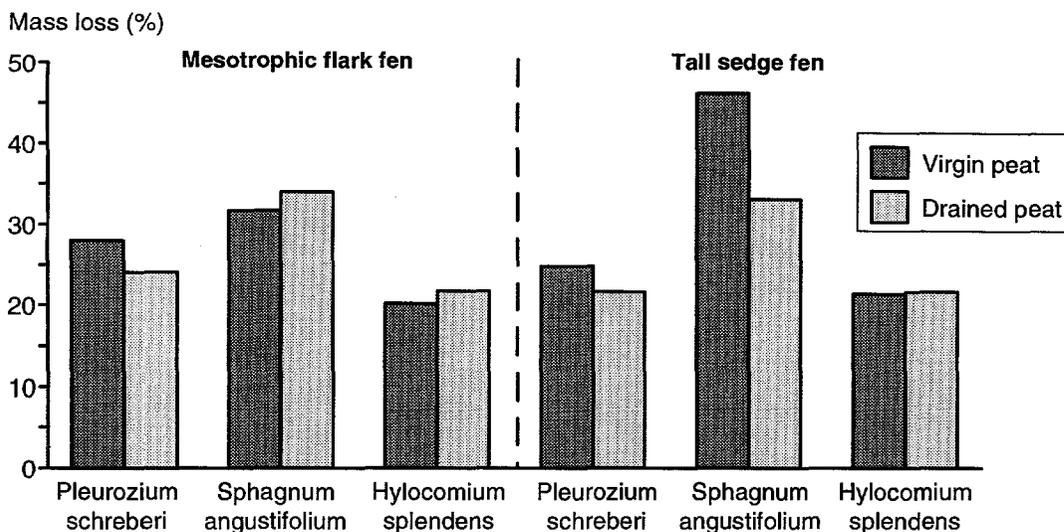
Fig. 1. Mass loss of the mosses in different peats.
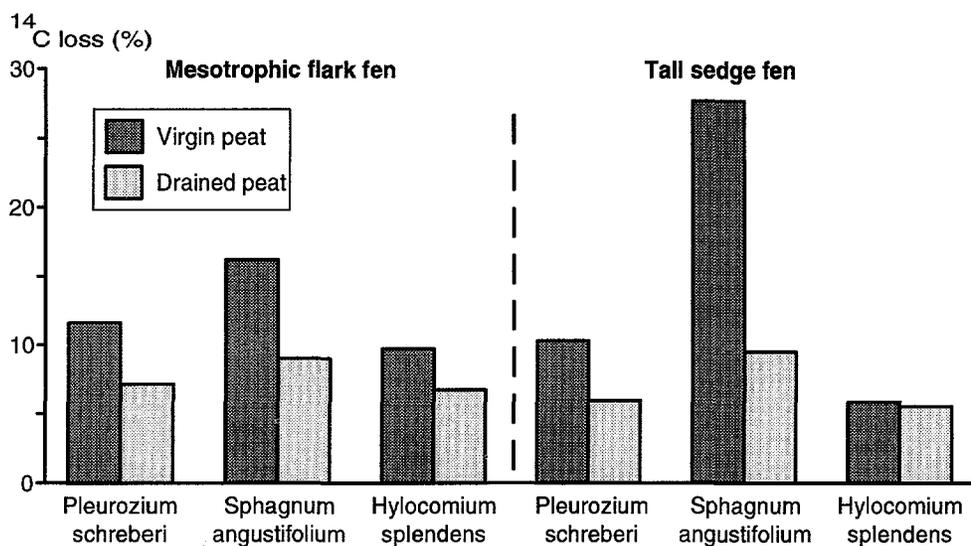
Fig. 2. ^{14}C loss of the mosses in different peats.

In this experiment of all mosses the respired $^{14}CO_2$ was higher in virgin peat than in drained peat. Respiration was highest in the experiment where ^{14}C-labelled *Sphagnum angustifolium* was added to the peat. In the peat of mesotrophic flark fen, *Sphagnum angustifolium* respired less than in the peat of tall sedge fen. The other mosses did not show any noticable differences in decomposing in either peat (Fig. 3a and 3b).

Fig. 3a. Respiration of $^{14}CO_2$ in the peat with different mosses.

Respired $^{14}CO_2$ of the total ^{14}C (%)

Tall sedge fen

Legend:
- ■ Drained, mesotrophic flark fen, 0-10 cm
- ● Virgin, mesotrophic flark fen, 0-10 cm
- ▲ Virgin, mesotrophic flark fen, 10-20 cm
- □ Drained, tall sedge fen, 0-10 cm
- ○ Virgin, tall sedge fen, 0-10 cm
- △ Virgin, tall sedge fen, 10-20 cm
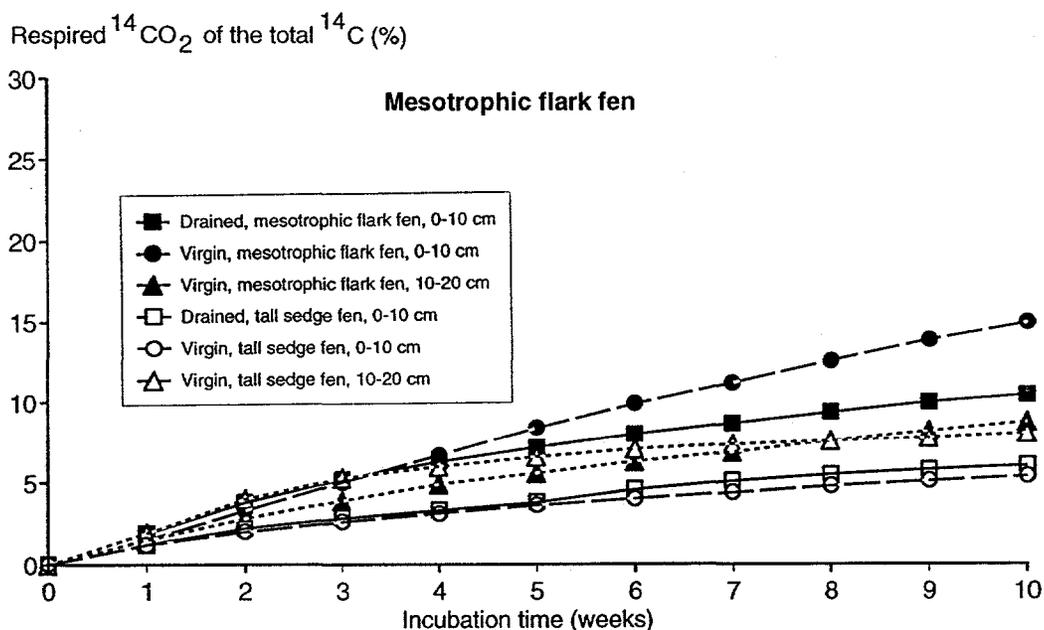
Incubation time (weeks)
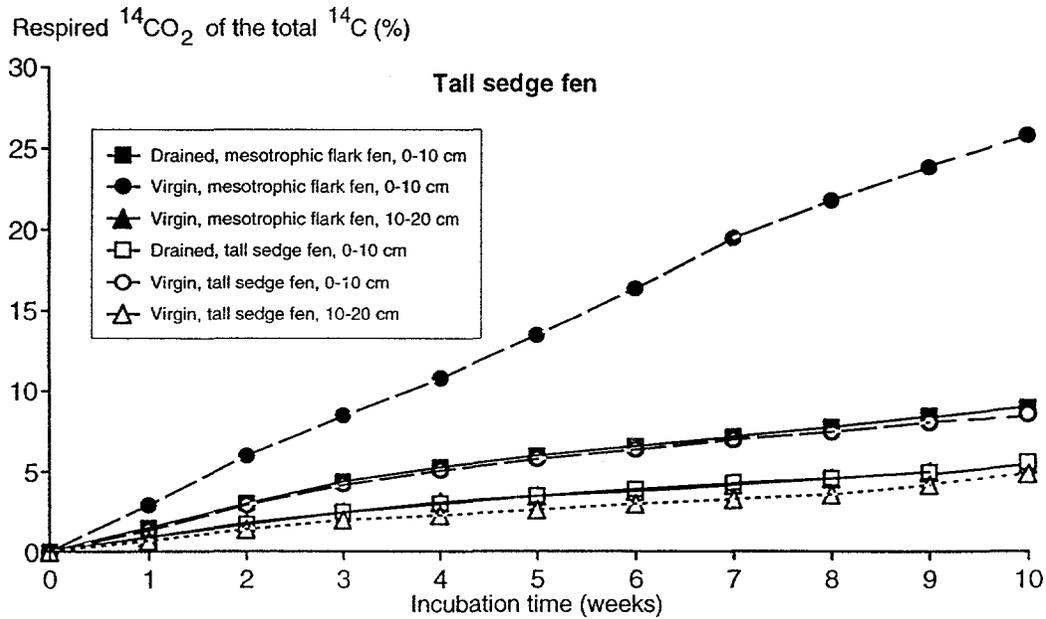
Fig. 3b. Respiration of $^{14}CO_2$ in the peat with different mosses.

In this experiment 54-80% of the labelled ^{14}C remained in moss. This can be seen in Fig. 4a and 4b with *Sphagnum angustifolium*. It seems that during decomposition process ^{14}C is not only respired as $^{14}CO_2$, but immobilized in peat (0,7-3,4%) as well. No water-soluble ^{14}C was found.

Recovered ^{14}C (%)   **Mesotrophic flark fen**

Pleurozium schreberi in virgin peat	Sphagnum angustifolium in virgin peat	Hylocomium splendens in virgin peat	Pleurozium schreberi in drained peat	Sphagnum angustifolium in drained peat	Hylocomium splendens in drained peat

Legend:
- ■ ^{14}C in peat
- □ Respired $^{14}CO_2$
- ▨ ^{14}C remained in moss

Fig. 4a. Turnover of ^{14}C.

Recovered ^{14}C (%)
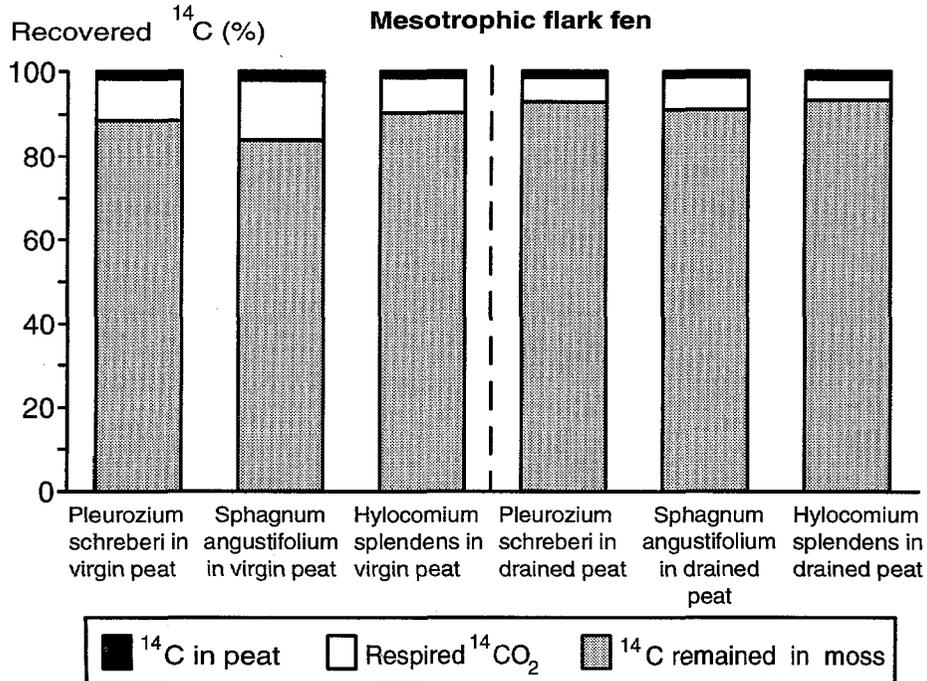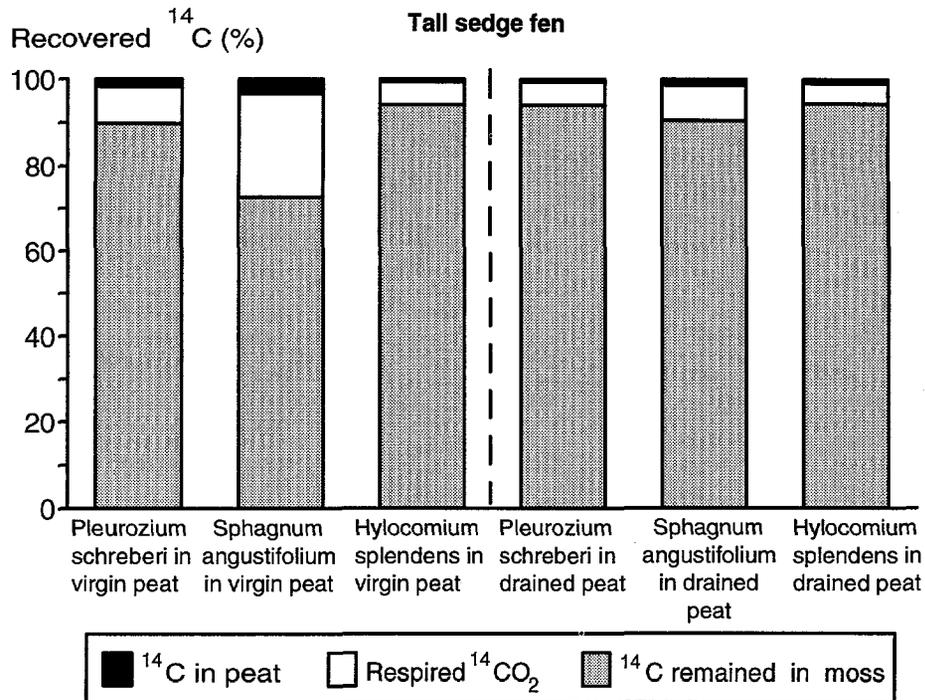
Tall sedge fen

Fig. 4b. Turnover of ^{14}C.

Microbial biomass carbon of mesotrophic flark fen peat were lower than for tall sedge fen peat. The microbial biomass carbon of virgin peat was lower than in drained peat for both fens. Microbial biomass carbon in virgin peat incubated with *Sphagnum angustifolium* was higher than with other mosses on both fens. (Fig. 5)

Microbial biomass carbon (mg/g)
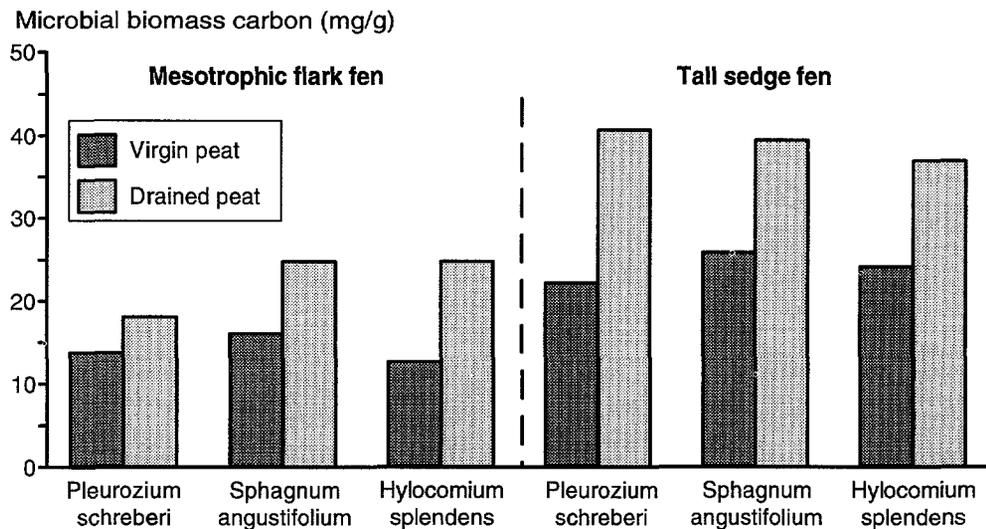
Fig. 5. Microbial biomass carbon in different peats with mosses.

Conclusion

1. The decomposition rate of ^{14}C-labelled *Sphagnum angustifolium* was highest.
2. The $^{14}CO_2$ respiration of virgin peat was higher than that of drained peat.
3. When the decomposition rate was high, ^{14}C was also immobilized in peat organic matter.
4. Although microbial biomass carbon in virgin peat was lower, the degrading capacity of the virgin peat was equal or even better than that in the drained peat.
5. Carbon flow from the decomposed mosses was higher in drained peat than in virgin peat. However, the loss of mass and ^{14}C was roughly the same or higher in virgin peat than in drained peat.

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Plant biomass carbon store after water-level drawdown of pine mires

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Introduction

Tall-sedge pine fen is the site type most commonly drained in Finland. In their natural undrained condition sites of this type are rather wet with sparse, Scots pine dominated forest growing on hummocks and with large lawns dominated by sedges, usually *Carex rostrata* and/or *C. lasiocarpa*. Most of the primary production takes place in the field and ground layers (Reinikainen et al. 1984). The major pathway for carbon accumulation in the system is via *Sphagna* and sedge roots, carbon accumulation by the tree stand being very slow. After drainage the situation changes radically as the sedges die out (Laine et al. 1995) and the tree stand growth increases considerably (e.g. Laiho & Laine 1994). The aim of this study is to produce means of estimating the post-drainage dynamics of the plant biomass carbon store. The study is based on the assumption that sites similar before drainage will change in a similar manner following drainage.

Material and methods

For the study, six biomass sample plots (Table 1) were established on tall-sedge pine fens in central Finland (ca. 61°35' - 61°52'N, 24°05' - 24°25'E). Two of the sites were undrained and the rest had been drained 8, 22, 30 and 55 years earlier. They were selected from a larger group in a study of post-drainage nutrient stores in peat (Laiho & Laine 1994).

Table 1. General features of the sites sampled.

	Site					
	1	2	3	4	5	6
Year of drainage	-	-	1984	1970	1961	1937
Tree stand volume, m^3ha^{-1}	6	35	10	76	102	150
Proportion pine/birch/spruce	100/-/-	95/5/0	100/-/0	69/31/0	83/15/2	88/9/3
Number of stems, ha^{-1}	878	1700	594	5320	2328	2982
Sample plot size, m^2	900	600	1263	1158	1190	1328

Ground vegetation was inventoried in the area of the whole sample plot. Tree stands (all trees with a $d_{1.3} \geq 1.5$ cm) were measured for $d_{1.3}$, d_6, height, crown height and bark

thickness. On each plot, 10 - 20 sample trees were chosen for destructive harvesting. The carbon stores of the above-ground tree stand components were calculated using equations relating sample tree component dry masses to characteristics measured from all trees, and component carbon concentrations measured from a subset of samples. For stumps and coarse roots, the equation was derived from the literature, and an estimated carbon concentration of 50% was used. At 16 - 20 systematically chosen points, peat was sampled to a depth of 30 cm for determination of the fine root dry mass. At the same points, the ground vegetation and litter layer were also harvested (0.5 m^2 for field layer vegetation, 0.2 m^2 for ground layer and litter). For fine roots, ground vegetation and litter, a carbon concentration of 50% was also assumed.

Results

The total carbon store of the tree stand increases after drainage over the whole range of drained sites (Fig. 1). The increase in the stemwood carbon continues during the whole post-drainage period under observation, but the increase in the branches and foliage appears to have already levelled out at the 22-year-old drained site.

The carbon store of the cyperaceous and herbaceous field layer decreases steadily after drainage (Fig. 2). Shrub carbon has its maximum value at the most recently drained site 3 (8 years), and after a temporary decrease it shows the second biggest value on the site drained earliest. Accordingly, the moss carbon also reaches its maximum on the most recently drained site, and increases again on the earliest drained site, but only to a value similar to that on the undrained sites.
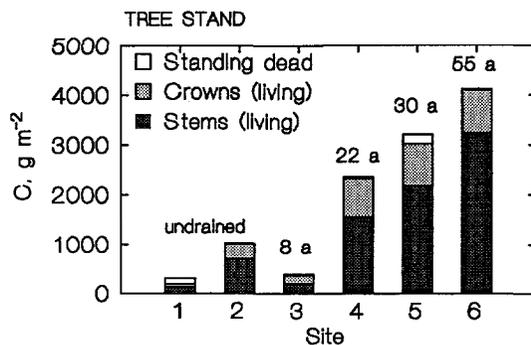
Fig. 1. Carbon fixed in the above-ground parts of the tree stands. Drainage age (time since drainage, years) in the year of measurements indicated above the bars.
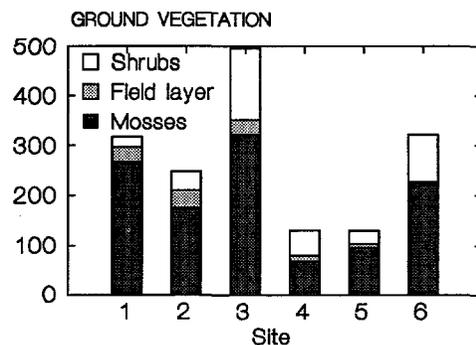
Fig. 2. Carbon fixed in the above-ground parts of the field layer vegetation (dwarf shrubs separately) and mosses above the rooting zone. Drainage ages as in Fig. 1.

The living root carbon store increases after drainage (Fig. 3). At the beginning of the post-drainage succession, the increase is mainly due to the increase in the tree and shrub fine roots which replace the roots of mire species like *Carex* spp. sedges. Later on, the increase is due to the long-term carbon accumulation in coarse roots and stumps.

The litter layer carbon store on the earliest drained sites is twofold that of the undrained sites (Fig. 4). The difference in annual litterfall is even bigger (Fig. 5).
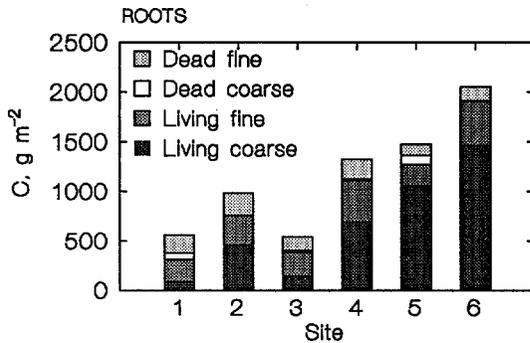
Fig. 3. Carbon fixed in the root systems. Drainage ages as in Fig. 1.
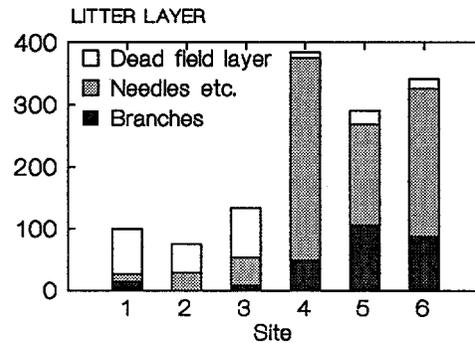
Fig. 4. Carbon fixed in the litter layer above the rooting zone. Drainage ages as in Fig. 1.

The total plant biomass plus litter layer carbon store increases considerably after drainage (Fig. 6). Most of this increase is caused by the increase in tree stand biomass, both above and below ground, and has a very close relation to the tree stand volume (Fig. 7).
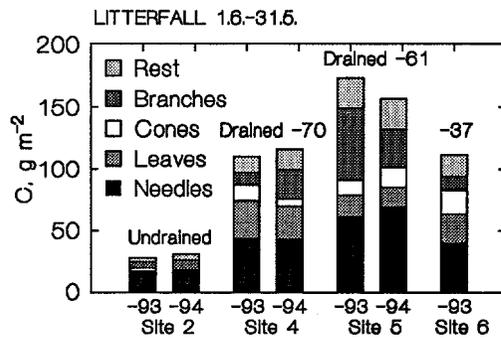
Fig. 5. Carbon in the litterfall during 1.6.1993-31.5.1994 and 1.6.1994-31.5.1995.
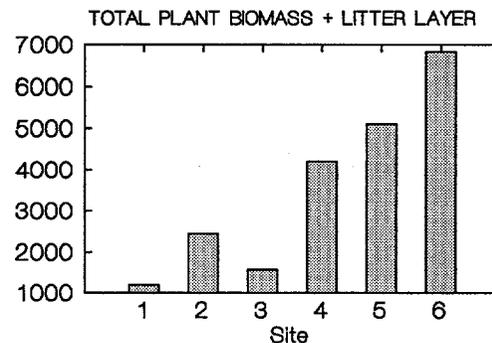
Fig. 6. Total carbon fixed in the plant biomass including the litter layer. Drainage ages as in Fig. 1.

Discussion

The equations presented in Fig. 7 can be used in estimating the plant biomass component of the carbon balance of pine mires, and the change caused by drainage for forestry. If possible, a site drained earlier should still be measured to ensure the extrapolation to bigger tree stand volumes. Within the drainage age range studied, the carbon pool in the plant biomass and litter layer had increased by ca. 5000 g m^{-2}. The inclusion of the carbon

released from decaying litter but still retained in the soil would probably add further to the increase.

TREE STAND, ABOVE-GROUND

TOTAL PLANT BIOMASS + LITTER LAYER

$y=33.2x-0.038x^2$

$y=1014.8+43.9x-0.034x^2$

Tree stand volume, m^3ha^{-1}
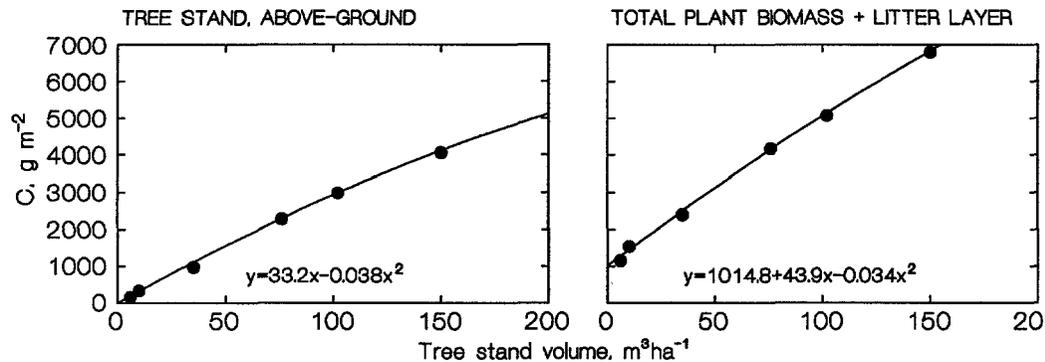
Fig. 7. Carbon fixed in the above-ground parts of the tree stands and the total plant biomass in relation to the tree stand stem volume.

The secondary succession initiated by forest drainage is probably much faster than that caused by the possible gradual drying resulting from predicted climatic warming. On the other hand, the equations are based on tree stand growth in the present environment, and in a warmer climate and with increased atmospheric CO_2 concentration, the growth may be faster than at present (Väisänen et al. 1994, Kellomäki & Väisänen 1995). However, as the equations are based on the tree stand stem volume, the only assumption that has to be made is that the relations between the biomass components will not change with the changing conditions. As long as more accurate estimates of the changes in tree stand growth induced by climatic warming are lacking, using data from drained peatlands is probably the best way to estimate the changes in carbon balance brought about by the climatic change.

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Carbon-nitrogen interactions and biomass partitioning of *Carex rostrata* grown at three levels of nitrogen supply

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Introduction

Biomass and production of vascular plants constitutes a major source of carbon input in peatlands (Wallén 1986, Sjörs 1991). As rates of decomposition vary considerably with depth, the vertical distribution of biomass may substantially affect accumulation of carbon in peatlands. Therefore, allocation patterns between shoot and roots are particularly important when considering carbon balance of peatland ecosystems.

The stimulatory effect of increasing atmospheric concentration of CO_2 on photosynthesis may increase availability of carbon to most C3 plants. Availability of nitrogen may also altered both due to increased atmospheric deposition and changes in mineralization rates associated with climate change (Sinclair 1992).

Most root-shoot partitioning models predict that allocation of biomass is dependent on the availability and uptake of carbon and nitrogen. A decrease in supply of carbon would favour allocation to shoots and a decrease in supply of nitrogen would increase allocation to roots (Wilson 1988; Thornley 1991; Dewar 1993). At a cellular level, non-structural carbohydrates and free amino acids are thought to represent the biochemically available fraction of carbon and nitrogen, respectively (Campagna & Margolis 1989).

The aim of this work is study the long-term growth responses of *Carex rostrata* to changes in the availability of nitrogen. Special attention is paid to soluble sugars and free amino acids, which may control partitioning of biomass.

Material and Methods

Tillers of *Carex rostrata* were collected from an oligotrophic fen in May 1993 and planted in buckets (65 l, Ø 44 cm) filled with sedge-*Sphagnum* peat. The original density of shoots was 58 shoots/m2. Sedges were cultivated in a greenhouse at the Lammi biological station from May 1993 to August 1994. The greenhouse was not heated during the winter. Containers were watered with rain water and eutrophic lake water. Additional nitrogen was added with the water as ammonium nitrate five times during both summers. The amount of nitrogen added was 2 , 5 and 10 g N m^{-2} a^{-1} (referred to as "low N", "medium N" and "high N") later in the text.

During the summer 1994, the length and density of shoots was measured five times, and the biomass of shoots was calculated using the regression between the length and the dry weight of shoots. At the end of the experiment, below-ground biomass and specific root length (SRL, mg cm^{-1}) were also measured from 10x10 cm cores.

Leaf samples were collected three times (25 May, 30 June and 4 August) and root samples once at the end of the experiment (4 August). Samples were frozen with liquid nitrogen and homogenised in 80 % (v/v) ethanol. Soluble sugars (reducing sugars and sucrose were measured with a p-hydroxybenzoic acid hydrazide method (Blakeney & Mutton 1980) and total free amino acids with a modified fluorometric o-phtalaldehyde method (Dawson & Liebezeit 1981). Total nitrogen was analysed using Kjeldahl method.

Results

No differences were observed in the concentration of soluble sugars and free amino acids in the shoots of Carex in the beginning of the second growing season. At the end of the summer, however, the concentration of free amino acids was significantly higher (both shoots and roots) and the concentration of soluble sugars lower (roots only) in the "high N" treatment (data not shown). At the same time, the concentration of total nitrogen increased with increasing supply of nitrogen (data not shown). Increasing ratio of sugars to amino acids with decreasing supply of nitrogen indicates that low long-term supply of nitrogen may shift the internal carbon:nitrogen balance of Carex towards relatively higher concentration of soluble carbohydrates (Fig. 1).

Changes in allocation patterns became evident only towards the end of the second growing season. The biomass of shoots was highest in the "high N" treatment resulting in a high above/below ground biomass ratio (Fig 2). Density of current-year shoots increased with increasing supply of nitrogen (Fig. 3). No differences were observed in the below-ground biomasses (data not shown). Specific root length (m/g), however, was highest in the "high N" treatment (data not shown).

Discussion

Low supply of nitrogen may increase the ratio of soluble sugars to free amino acids in C. rostrata, which indicates increased availability of carbon in relation to nitrogen. Comparable changes in the internal carbon:nitrogen balance of plants have also been observed in CO_2 enrichment experiments (Campagna & Margolis 1989).

Low ratio of soluble sugars to free amino acids at high supply of N was accompanied by a high above/below ground biomass ratio. This supports the view that (A) allocation of biomass to shoots and roots is dependent on the availability and uptake of carbon and nitrogen (Wilson 1988; Thornley 1991; Dewar 1993) and (B) non-structural carbohydrates and free amino acids may be associated the physiological control of biomass partitioning (Campagna & Margolis 1989).

Sugars / free amino acids
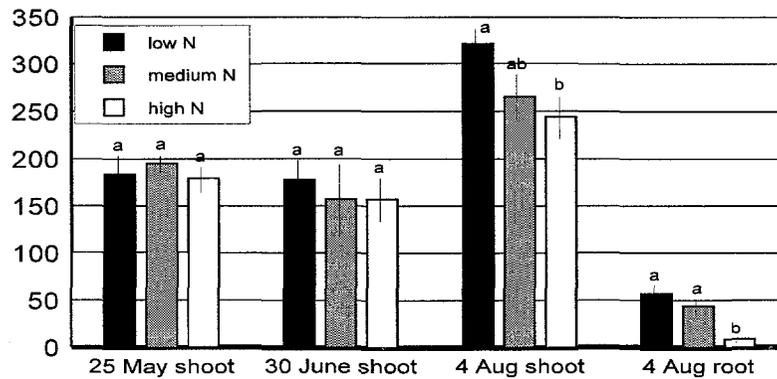
Fig. 1. Ratio of soluble sugars to free amino acids in the shoots and roots *of Carex rostrata* grown at three levels on nitrogen supply for two growing seasons. Data presented is from the second growing season. n=4 in each treatment. Different letters in bars indicate statistical difference at 10% level in Tukey's test a posteriori.

Above/below ground biomass
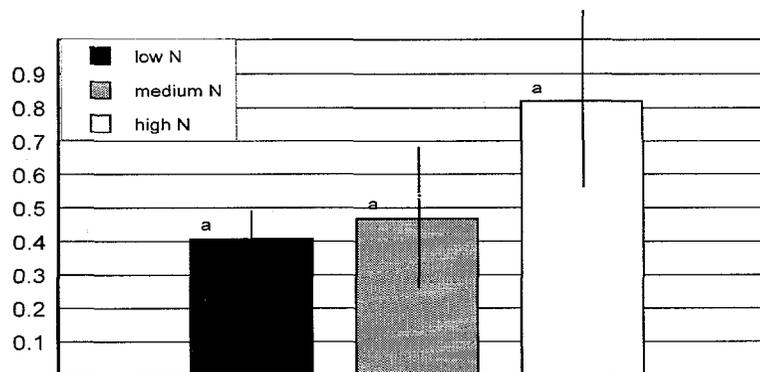
Fig. 2. Above/below ground biomass ratio of *Carex rostrata* grown at three levels on nitrogen supply for two growing seasons. Data presented is from the second growing season. n=4 in each treatment. Different letters in bars indicate statistical difference at 10% level in Tukey's test a posteriori.

Density of shoots
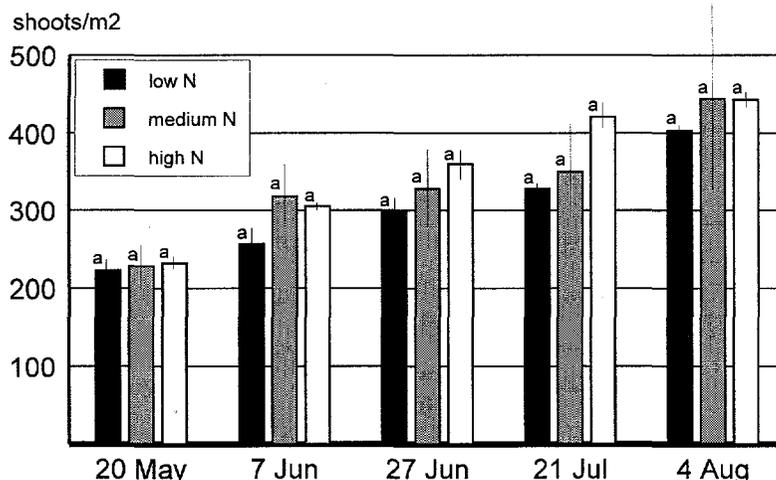
shoots/m2

Fig. 3. Density of current-year shoots of *Carex rostrata* (shoots m^{-2}) grown at three levels on nitrogen supply for two growing seasons. Data presented is from the second growing season. n=4 in each treatment. Different letters in bars indicate statistical difference at 10% level in Tukey's test a posteriori.

As noted by Aerts et al. (1992), shoot:root biomass ratios should be interpreted with great care. At high N supply the high specific root length may partly compensate for the low proportion of below-ground biomass. Increasing concentration of atmospheric CO_2 and changes in the availability of nitrogen due climatic change and atmospheric deposition may affect root-shoot partitioning in sedge-dominated plant communities.

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Root exudation of *Eriophorum angustifolium* grown in nutrient solutions simulating oligotrophic mire waters

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Introduction

Although peatlands are significant sources of the two greenhouse gases, CH_4 and N_2O, surprisingly little research has been done on the specific biogeochemical and ecophysiological processes affecting the production of these gases in peatlands.

There are studies indicating indirectly a close connection between vascular plant activity, methane emission (Whiting and Chanton 1993, Chanton et al. 1993, Charman et al. 1994), and nitrous oxide emission (Martikainen et al. 1993). The decomposition of organic matter in anoxic conditions is a concerted action of different anaerobic bacteria, with methanogenic bacteria reducing CO_2 with the supply of hydrogen to CH_4. The methanogenic bacteria have to rely upon other anaerobic organisms for their substrate-supply, and the quality of the organic matter may therefore be of utmost importance for the process. Thus, the highest rates of methane production is found where the highest contents of easily available carbon compounds and mineral nutrients can be expected to be found (Clymo and Pearce 1995)

Root exudates, normally including soluble sugars and amino acids (Rovira 1969, Russell 1977), provide a readily available substrate for a variety of soil micro-organisms. Therefore, exudates are regarded as a major reason for incresed microbial activity in the rhizosphere. However, the whole rhizosphere effect is poorly documented in peatlands (Dickinson 1983).

The objective of this work was to study root exudation in Eriophorum angustifolium grown in nutrient solutions simulating a nutrient gradient from ombrotrophic to weakly minerotrophic mires, using a ^{14}C pulse labelling technique.

Material and methods

Plant material
Plant material was collected from a small oligotrophic kettle hole mire outside Lund, S. Sweden. To obtain homogeneous material, all the old roots were removed and the leaves were cut at a length of 15 cm.

Plants were grown in a greenhouse in three different nutrient solutions. One was simulating ombrotrophic bog water and the other two different poor fen waters (Table 1). The basic cation concentrations and pH were close to those measured in mire waters in southern Sweden (Malmer 1962). NH_4^+ was used as the only nitrogen source, and both $NH_4^+/H_2PO_4^-$ and SO_4^{2-}/Cl^- ratios were similar in all the solutions. Micronutrients were included in equal concentrations to all solutions. The pH of the solutions was adjusted to 4.0 (bog) or 5.5 (poor fen) with HCl and NaOH, respectively. The treatments are referred to as "bog", "poor fen 1" and "poor fen 2".

Table 1. Composition of nutrient solutions simulating ombrotrophic bog water and minerotrophic poor fen waters. NaOH and HCl used for adjusting pH (4.0 and 5.5 for bog and poor fen, respectively) are excluded.

Element	Bog (μM)	Poor fen 1 (μM)	Poor fen 2 (μM)
Ca^{2+}	146	175	175
K^+	27	46	46
Mg^{2+}	49	73	73
NH_4^+	25	25	50
$H2PO_4^-$	5	5	10
SO_4^{2-}	89	114	121
Cl^-	442	566	603

The plants were pre-grown for 16 days before the start of the experiment in 6l containers, mounted in a non-transparent lid, with the base of each shoot adjusted to reach the solution. 2 l of the solution was changed three times a week. Thereafter, the plants were transferred to 350 ml aluminium-wrapped Erlenmeyer bottles containing 300 ml nutrient solution. The solution in the bottles was changed three times a week.

Additional light was provided with Philips HPLR 400 W lamps using 14/10 h light/dark period. The average light intensity at the level of the leaves was 9.2 klux. Average air temperature was $17.9 \pm 2.8/11.5 \pm 2.6$ °C (light/dark, average ± SD).

^{14}C labelling and analyses

Exudation rate of roots was studied by measuring ^{14}C activity in nutrient solutions after a ^{14}C pulse labelling. All the nutrient solutions were changed two hours before the beginning of the labelling. Plants were labelled in transparent 32x32x39 cm cuvettes (12 plants/cuvette) for 4 hours. Each cuvette was provided with 200 μCi 14CO2 by acidifying sodium-^{14}C-bicarbonate with hydrochloric acid.

The plants were harvested 48 h after the labelling. Shoot height, mean root length and number of roots and growing leaves were measured. The plants were dried (48 h, 85°C) and the weight of shoots and roots was measured. ^{14}C activities were analysed in shoots, roots and nutrient solutions. Dried root and shoot samples were oxidised in a Packard Tri Carb Sample Oxidiser with Carbosorb as a CO_2 trap. An aliquot of 6 ml of the nutrient solution was acidified with a drop of phosphoric acid and allowed to stand in open scintillation bottles for 24 hours in order to remove dissolved inorganic ^{14}C released through root respiration. ^{14}C activities were measured with a Packard Tri Carb Scintillation counter.

Results

Both length of shoots and number of growing leaves increased slightly with increasing nutrient availability (Table 2). In the poor fen treatments the plants had a higher number of roots than the plants grown in bog water.

Table 2. Growth parameters of Eriophorum angustifolium after 34 days in different nutrient solutions simulating different mire waters. Mean ± SE, n=11. Different letters in each row indicate statistical difference at the 10 % level tested using Tukey's test a posteriori.

	Bog	Poor fen 1	Poor fen 2
Shoot lenght (mm)	187 ± 8a	192 ± 9a	208 ± 11a
Number of growing leaves	1.2 ± 0.2a	1.6 ± 0.3ab	1.9 ± 0.2a
Mean root lenght (cm root^{-1})	9.4 ± 0.7a	9.3 ± 0.7a	7.9 ± 0.7a
Number of roots	6.1 ± 0.7a	8.7 ± 0.9b	8.5 ± 0.6ab
Shoot weight (mg dw.)	322 ± 49a	412 ± 67a	384 ± 40a
Root weight (mg dw.)	37 ± 9a	37 ± 6a	27 ± 3a
Shoot/rootweigth ratio	14 ± 2a	13 ± 2a	14 ± 1a

There was an increasing allocation of carbon to roots with decreasing nutrient availability of the mire water (Fig. 1). Root exudation - whether expressed as the percentage of the total ^{14}C recovered in the solutions (Fig. 2) or as ^{14}C activity per root length or number of roots (Fig. 3) was higher in the bog treatment than in the two fen treatments.
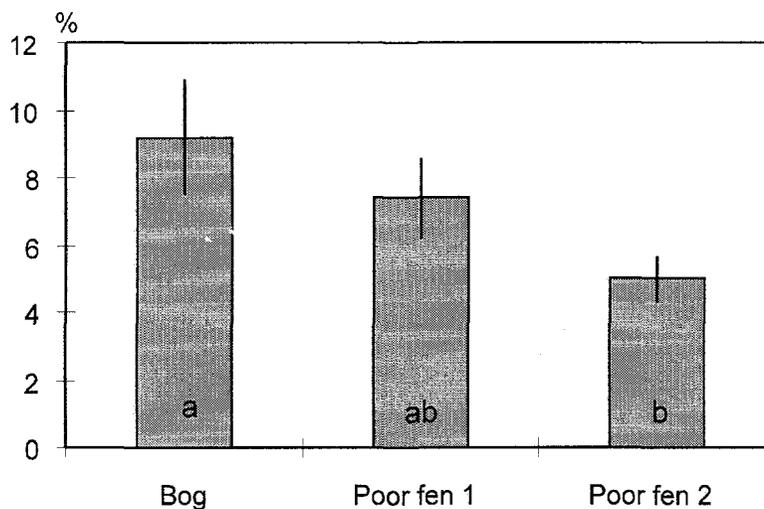
Fig. 1. Recovery of ^{14}C (% of the total ^{14}C recovered) in the roots of Eriophorum angustifolium grown in nutrient solutions simulating mire waters. The plants were pulse labelled with ^{14}C and activies were measured in the roots 48 h after the labelling. Mean±SE, n=11. Different letters in bars indicate statistical difference at 10% level in Tukey's test a posteriori.
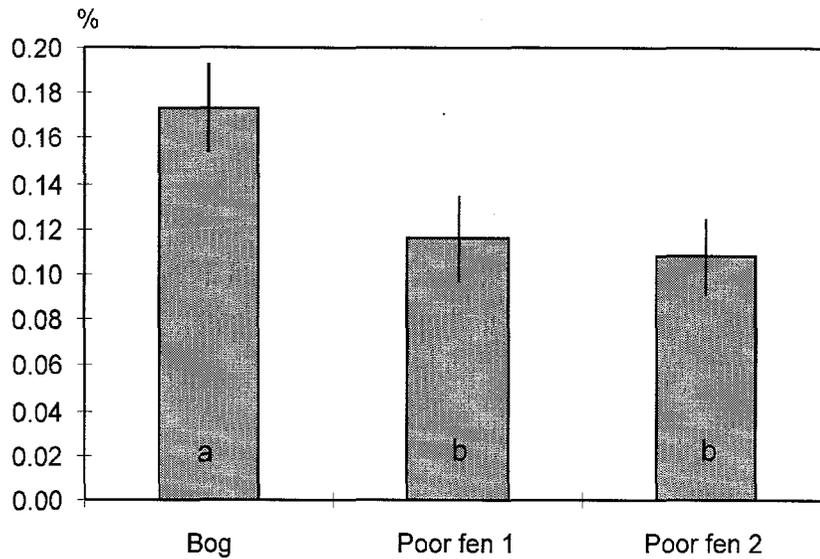
Fig. 2. Recovery of ^{14}C root exudates(% of the total ^{14}C recovered) of *Eriophorum angustifolium* grown in nutrient solutions simulating mire waters. The plants were pulse labelled with ^{14}C and activies were measured in the nutrient solutions 48 h after the labelling. Mean±SE, n=11. Different letters in bars indicate statistical difference at 10% level in Tukey's test a posteriori.
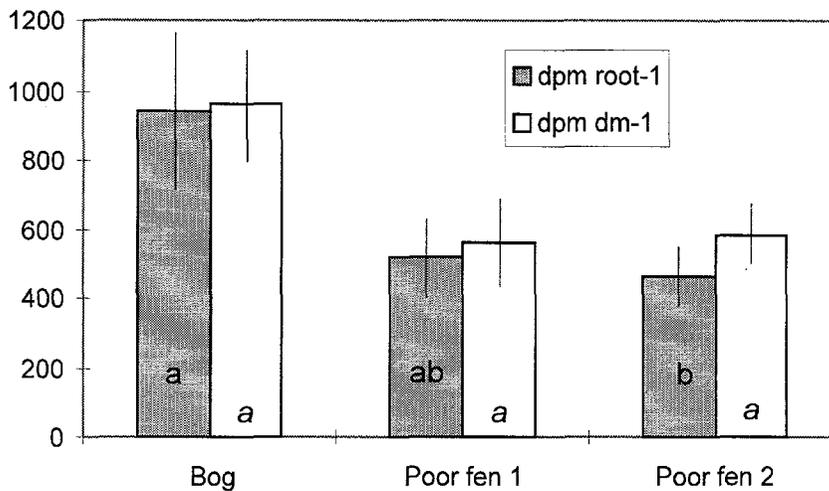
Fig. 3. The amount of ^{14}C exuded into the nutrient solutions in relation to number of roots (shaded bars) and length of roots (open bars) of *Eriophorum angustifolium*. The plants were pulse labelled with ^{14}C and activies were measured in the nutrient solutions 48 h after the labelling. Mean±SE, n=11. Different letters in bars indicate statistical difference at 10% level in Tukey's test a posteriori.

Discussion

Root exudation was highest in the bog treatment, but hardly any differences could be observed between the two fen treatments containing different concentrations of ammonium and phosphate. Low concentration of basic cations and/or low pH may have increased exudation. Differences between the treatments were most prominent when exudation was calculated per number of roots. This is in agreement with the fact that exudation of soluble organic compounds is greatest at root tips (Trofymow et al.. 1987).

Net exudation of organic compounds is dependent on both passive diffusion of molecules across the plasma membrane into the apoplast and active re-sorption against concentration gradients (Jones and Darrah 1992, 1993). The driving force for the active uptake is mebrane potential, which is generated by H^+-ATPases pumping protons from cytosol into the apoplast (Briskin and Hanson 1992). Potassium ions are known to be essential for the functioning of plasma membrane H^+-ATPases (Briskin and Hanson 1992), and potassium deficient plants have shown to exudate increased amounts of organic compounds (Kraffzcyk et al.. 1984). Hence, low potassium concentration is a potential factor which may have increased exudation in our bog treatment.

Accumulation of soluble sugars has been observed in roots of *Carex rostrata* grown with a low supply of nitrogen (Saarinen, in preparation). High sugar content of roots increases concentration gradient between cytoplasm and apoplast, which would increase passive diffusion of sugar into apoplast. Therefore, increased passive diffusion could partly explain increased exudation associated with low nutrient availability.

Root exudation has been shown to increase the availability or uptake of many nutrients, e.g. phosphate (Hoffland 1992), iron (Römheld 1991), manganese and copper (Mench and Martin 1991) through formation of complexes with nutrient ions. Increased microbial activity in the rhizosphere may also enhance the mineralization of nutrients. Therefore, high rate of root exudation may also have an adaptive function by enhancing uptake of nutrients in oligotrophic mire ecosystems.

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Gas fluxes
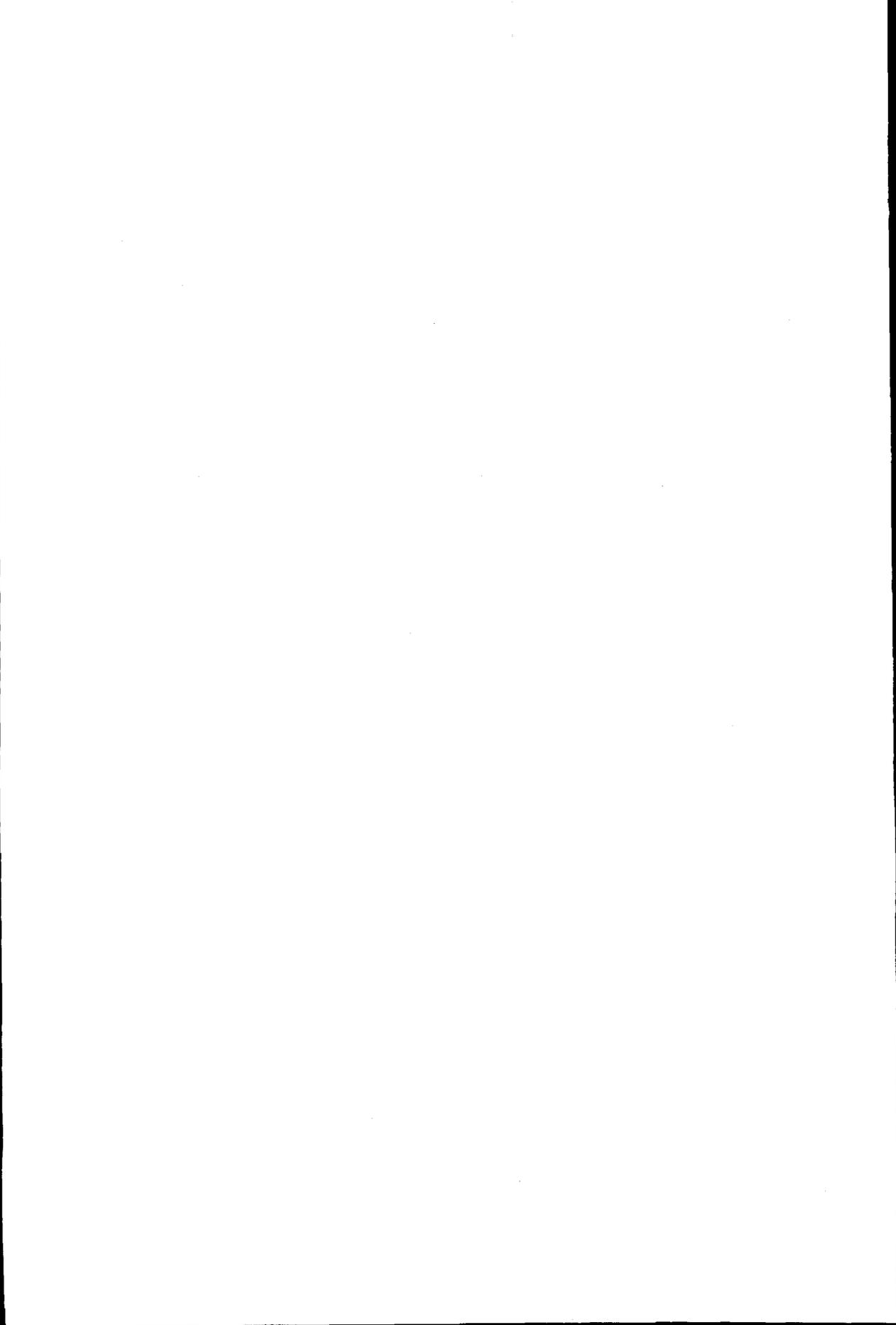
Temporal and spatial variations in CH$_4$ emissions of flooded meadows and vegetated hydrolittoral

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Introduction

Methane (CH$_4$) is formed in soils from organic matter in anoxic conditions by fermentation. Such conditions occur in waterlogged enviroments such as mires, but also in sediments of various waterbodies. High salinity and sulphate concentration are known to inhibit marine CH$_4$ formation (Bartlett et al. 1987), but in this respect freshwater sediments are more feasible to the methanogens. Due to brackish water of Baltic Sea, the shores of Finnish Bay and Bothnian Bay are also possible methane sources. CH$_4$ release from tundra lakes has been studied in Alaska and Canada (Harriss et al. 1993), but little is, however, known on CH$_4$ emissions from boreal freshwater systems. Large emissions have been reported in artificial lakes (Rudd et al. 1993) and beaver ponds (Yavitt et al. 1990, Bubier et al. 1993).

Vascular plants have an important role in CH$_4$ formation and transport from the sediment. They provide easily decradable substrates for methanogens in form of root exudates and fine root litter (Lynch and Whipps 1991). Some aquatic helophytes and floating-leaved plants are adapted to oxygenate their root systems through aerenchymate tissues. *Nymphaeids* and helophytes has been shown to vent sediment methane to the atmosphere through the aerenchyma (Dacey & Klug 1979). Rhizodeposition (exudates and fine root litter) into the sediment may provide much of the substrate needed in methanogenesis. The possible changes in lake water level affect the moisture of adjacent upland soils thus controlling the processes of CH$_4$ formation and oxidation on the areas associated to the watercourse.

We measured CH$_4$ release from different shore types both in nutrient poor and mesotrophic waterbodies. Data was collected during the open-water season from transects along the moisture gradient, in some sites extending from only temporarily flooded upland soils to lake hydrolittoral through the helophyte stands. Diurnal variation in total dark respiration and CH$_4$ release was monitored in a reed (*Phragmites australis*) stand using an automated gas chromatograph system to measure same plots after every 4-5 hours. As most of the manual sampling was made during daytime, diurnal variation was studied also as a campaign in two flooded sedge meadow sites by collecting data repeatedly

repeatedly from same plots through a period of two days and nights. Sediment CH_4 production and oxidation potential was also tested in a pre-analysis.

The purpose of this work was to study both the dynamics and spatial variation of CH_4 release from lake associated wetlands and vegetated littoral systems, and compare those results with emissions from Finnish mire ecosystems measured with similar techniques. Some preliminary results are shown in this paper.

Material and methods

CH_4 emissions from vegetated littoral zone at two freshwater bodies Mekrijärvi, Ilomantsi (62°45'N, 31°00'E) and Heposelkä, Siikasalmi (62°30'N, 29°30'E) were measured during summer 1995. To assess spatial variability, different shore types were sampled at the oligotrophic humic lake Mekrijärvi and the mesotrophic water body (Heposelkä, Siikasalmi), which is a part of the large watercourse Saimaa. The shore types included a flooded margin bank of an agricultural field, edge of an oligotrophic mire (with dwarf shrub pine bog with lake margin), flooded sedge marshes or meadows on peat or mineral bottom and a mixed *Carex-Phragmites* stand on soft mineral bottom. The mire edge and the sedge meadows dried to various degree when lake water level descended in the course of summer.

Closed aluminium chambers (volume 54 dm^3 or 90 dm^3) and 4-6 pre-assembled 60x60 cm collars per site with a groove for gas-tight water sealing were applied. In higher stands, several collars were placed upon each other and a 216 dm^3 transparent chamber was applied. Chamber air was mixed with a battery-operated fan. Four 30-40 ml samples were drawn into 50 ml polypropylene syringes from the chamber headspace during the 20-30 min incubation and the fluxes were calculated from the linear change of the gas concentration. Soil temperature profile (0-30 cm) and depth of the water table was measured in connection with the gas sampling.

An automated gas chromatograph system (Silvola *et al.* 1992) was used to monitore temporal variability in CH_4 from 6+6 collars was established at Siikasalmi. Manual gas collections from additional 16 collars at Siikasalmi area and 23 collars at Mekrijärvi were made 1-4 times a week. Sample CH_4 concentrations were determined using gas chromatography (Shimadzu GC-14 A, 1.8 m column filled with Haye Sep Q, 80/100 mesh) and FI-detector.

Methane production and oxidation potential was examined in littoral sediment cores, the data from a *Phragmites australis* stand is presented here. In the oxidation experiment, the initial CH_4 concentration was set to about 100 ppm. Sediment samples for the CH_4 production experiment were deoxygenated in field using N_2. CH_4 concentration change in the 600 ml sample bottles was followed during incubation in 20°C for two weeks.

Results and discussion

Temporal variations

Measurements were started in late May or early June before the emergence of aquatic vegetation. The emergence of vegetation evidently increased the basic level of CH_4 emission. Maximum daily emissions, 20-200 mg CH_4 $m^{-2}d^{-1}$ in Mekrijärvi and 250-400 mg in Heposelkä flooded sedge marshes, were obtained during July-August in the sites where the sediment was still submerged (Fig. 1a-c). The largest fluxes, however, were measured in a *Phragmites australis* stand. There the maximum daytime emissions near the waterline were 200-250 mg CH_4 m^{-2} per hour, giving 4800-6000 mg CH_4 $m^{-2}d^{-1}$, if the maximum hourly values were directly converted to daily estimates.
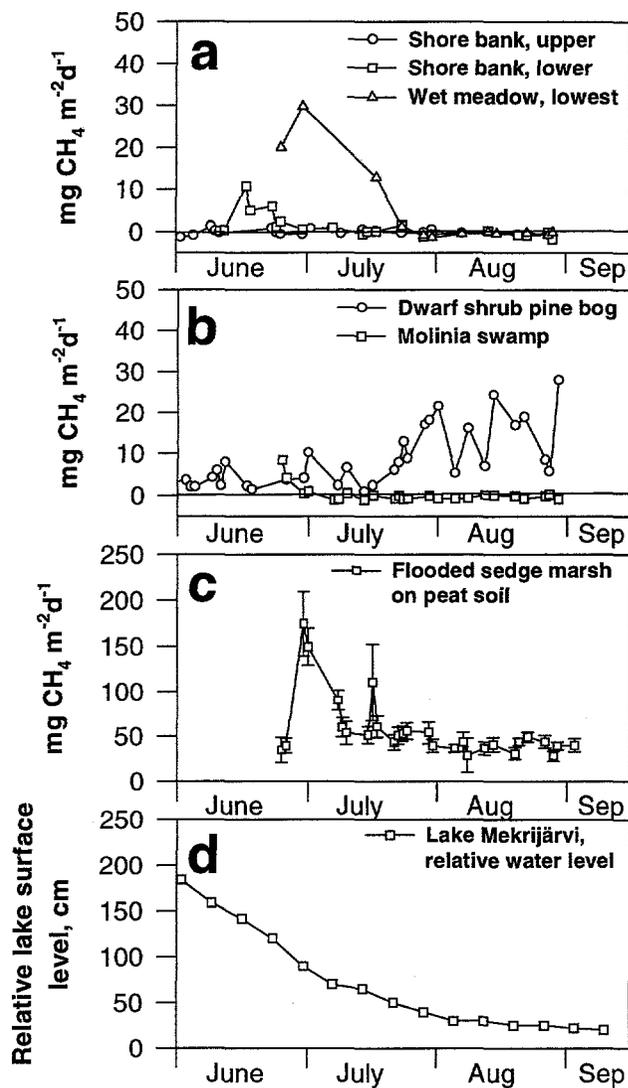
Fig. 1. Examples of methane fluxes (mg CH_4 $m^{-2}d^{-1}$) in different site types in Mekrijärvi, Ilomantsi. Data from separate collars (a, b) or averaged from 6 collars on similar subsites (± S.E.) (c). Flooded bank of an agricultural field on mineral soil from upland to moist grassy meadow (a), edge of a bog from dwarf shrub pine bog to *Molinia caerulea* meadow (b) and flooded sedge marsh on peat soil (c). Relative lake water level (weekly measurements) also indicated (d).

The emissions in flooded shores above the hydrolittoral decreased rapidly following the descending water level. The drawdown of lake water level was especially notable in Mekrijärvi, where the water level decreased from June to September by about 1.5 m (Fig. 1d). Drying of the flooded shore banks turned CH_4 fluxes from net emission (2-10 mg CH_4 $m^{-2}d^{-1}$) to net uptake of 1-2 mg CH_4 $m^{-2}d^{-1}$.

Strong (2-5 x) diurnal fluctuation in CH_4 emission was observed in the *Phragmites* stand. The emissions inreased shortly after sunrise, peaked before and decreased shortly after midday, and by night descended to a level specific to the collar (Fig. 2). This phenomenon is possibly explained by mass flow through the reed stems, initiated by the morning rise in temperature and the opening of stomata in light. There is a continuous gas space connection at least between the pith cavity and cortical aerenchyme in the rhizomes of *P. australis* (Armstrong & Armstrong 1988).The rapid reduction in gas emission, while there still was plenty of daylight in the afternoon, indicated the exchaustion of CH_4 in the plant. Methane must thus enter the plant aerenchyme system also during the nighttime, when the gas flux to the atmosphere was at minimum. Similar fluctuations were observed in floating-leaved *Nuphar luteum* and *Nelumbo*, where the gas transport in the porous aerenchyma system was shown to be driven by heat (Dacey 1981, 1987).
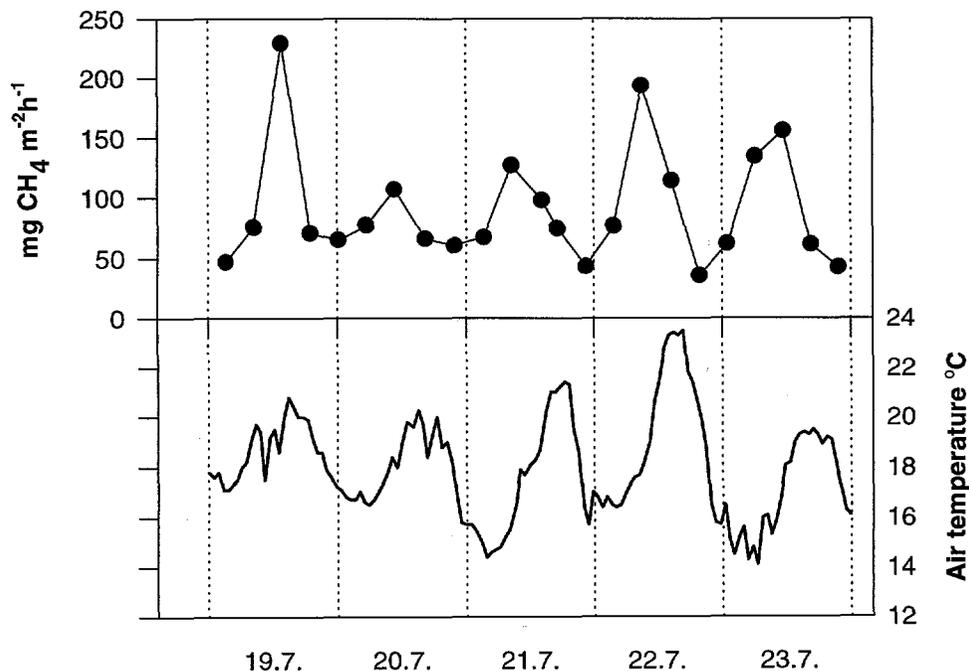
Fig. 2. Example of diurnal fluctuation in CH_4 emission (mg CH_4 $m^{-2}h^{-1}$) from a *Phragmites australis* stand during five days in July 1995 (upper). Hourly course of air temperature (°C) above the stand (lower) is also indicated.

Diurnal fluctuation was observed during the two day campaign also in the sedge marsh sites, but the maximum emissions occurred there late in the evening (data not shown). Also, the magnitude of fluctuation was much smaller than that in *Phragmites* stand. In *Carex* stands the release of CH_4 may be controlled differently, perhaps by variation in sediment substrate concentration after photosynthesis during daytime.

Sediment CH_4 production and oxidation

In a pre-analysis, CH_4 production was clearly strongest in 0-20 cm layer (top sediment). No methane production, however, was found in deeper layers (20-50 cm). Oxidation was also strongest in surface layers (0-10 cm, data not shown). The density of *Phragmites* roots seemed to be clearly largest in the top 30 cm of the sediment, perhaps affecting both production and oxidation of CH_4.

Spatial variations and comparison with Finnish mires

A gradient from net CH_4 oxidation in dry shore bank to net flux in vegetated littoral was observed in different site types. Open-water period average CH_4 fluxes from the scantily vegetated oligotrophic Mekrijärvi littoral zones seemed to be generally lower than those measured at the mesotrophic Siikasalmi area.
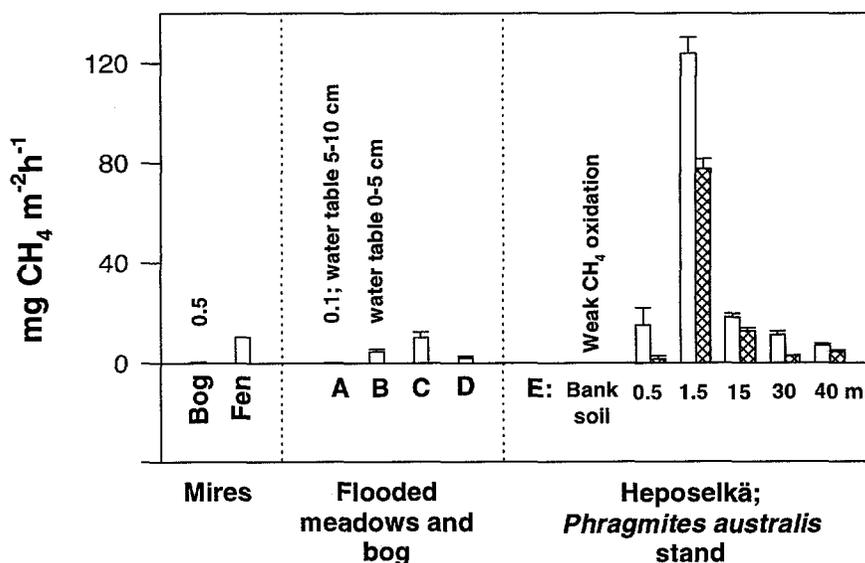
Fig. 3. Comparison of average July-August CH_4 fluxes (error bars=S.E.) from different wetland ecosystems in eastern Finland. Mires are represented by a *Sphagnum fuscum* bog (Bog) and a low-shrub *Sphagnum papillosum* pine fen (Fen). Flooded meadows and bog are lake associated wetlands: dwarf shrub pine bog edge (A), flooded sedge marsh on peat soil (B), flooded sedge marsh sites on mineral soil (C, D), and *Phragmites australis* hydrolittoral stand on soft mineral bottom (E) with distances 0.5-40 m accross the stand from the water-line. Double bars in E indicate two replicate chamber positions.

Average July fluxes from the flooded sedge meadows seem to fall within the range obtained for Finnish virgin mires (Nykänen et al. 199x). Flux rates 1.5 m from the waterline in the *Phragmites* stand, however, greatly exceeded the mire emissions (see Fig. 2 and 3), showing emissions comparable to beaver ponds in North America (Bubier *et al.* 1993). The emissions were instantly lowered, when the reed stems were removed from the collar area, indicating that most of the methane was transported through the plants to the atmosphere. In the same reed stand, only a few meters apart from the waterline, the fluxes were much lower, being comparable to those in flooded sedge marshes. One of the reasons causing such large emissions may be the accumulation of reed stem litter in the sediment near waterline.

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Reconstruction of annual carbon dynamics and balance for an oligotrophic pine fen

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Introduction

Atmospheric carbon dioxide (CO_2) is bound by mire vegetation in photosynthesis during the growing season, and is re-released by respiration of plants, soil animals and micro-organisms consuming dead organic matter. A small proportion of annual primary production may fall below the water table to anoxic conditions and thus escapes the oxidative decomposition. Also from anoxic peat, carbon is released with clear seasonal and spatial variation as methane (CH_4) (Clymo & Reddaway 1971). The rate of carbon accumulation in peat depends on the annual inbalance of plant production and litter decomposition. Exchange of CO_2 and CH_4 between peat, vegetation and the atmosphere thus reflects the dynamics of carbon flows in the ecosystem.

Nutrition, water flow and depth of groundwater table affect the distribution of vegetation in mire surface (see Malmer et al. 1994). As a consequence, mires are characterized by a mosaic of microsites with distinctive vegetation (hummocks, lawns and hollows). Microsite (e.g. hummock-hollow) variation results in large spatial heterogeneity in the rate of gas exchange across the mire surface. Processes of photosynthesis and respiration are greatly influenced by variations in solar irradiation, soil temperature and groundwater table, making the annual carbon balance a dynamic system. Components of annual carbon accumulation (A) in a mire ecosystem can be expressed as follows (modified after Pakarinen 1975):

$$A = P_G - L, \text{ where } L = R_{TOT} + D \pm W + F.$$

P_G in the equation is gross primary production and L denotes the various biological and physical carbon losses from the peat-forming ecosystem. The biological losses ($R_{TOT} = R_D + R_C$) include CO_2 release in dark respiration of plants (R_D) and in respiration of con-

sumers (R_C) (herbivores, aerobic microorganisms, and soil animals) and as CH_4 in anaerobic decomposition (D), while weathering (W) and fire (F) are the main physical loss factors. Weathering in our study site includes mainly leaching and the effect of visiting herbivory, and causes addition or removal of matter to and from the system, while fire was absent in the study year.

The accumulation equation can be considered as an integral of the component processes over a year. It could be solved by reconstructing the annual course of its components. The biological net CO_2 exchange, the difference of gross production and soil respiration, ($P_N = P_G - R_{TOT}$), is highly time dependent over the growing season, and should be determined with great temporal and spatial accuracy. P_N is affected both by the physiological status of plants and the changes in environmental conditions such as illumination, temperature and moisture.

We measured P_N, R_{TOT} and D in different conditions for hummock, lawn and flark communities over one growing period, and also recorded time series of what we considered to be the most important environmental variables. Emissions of CO_2 and CH_4 were determined also during wintertime. Statistical response functions between P_N and the environmental factors were defined, and a tentative estimate of annual carbon accumulation for the mire is presented here on the basis of the reconstructed P_N and the loss components. The component of leaching was collected from literature (Sallantaus 1992).

Material and methods

Net ecosystem CO_2 exchange (P_N), total CO_2 release (R_{TOT}) and CH_4 release (D) from different treeless surfaces of low-sedge *Sphagnum papillosum* pine fen was studied in eastern Finland (Salmisuo, Ilomantsi 62°47'N, 30°56'E).

Net exchange was measured using a thermostated transparent, vented static chamber system (25 pre-assembled 60x60 cm aluminium collars, chamber volume 72 dm^3). Concentration of CO_2 was monitored with a portable infrared gas analyzer LCA-2, Analytical Development Company Ltd., England, equipped with a pump (suction from chamber headspace 100-150 ml min^{-1}). The measurement period lasted usually 150-180 s, and the sampling was performed under stable irradiation conditions. During the incubation period, solar irradiation was monitored constantly to avoid large changes in its level to disturb the measurement. For R_{TOT} measurements, the chamber was covered with an opaque lid made of 1 mm aluminium plate. CO_2 fluxes were calculated from the linear change of the headspace CO_2 concentration, recorded at every 30 s. Each collar was visited 1-3 (occasionally more) times weekly.

Summertime CH_4 emission measurements from the same collars are described in more detail in Saarnio *et al.* 199x. Dark static chamber techniques (Crill 1991) were applied to obtain wintertime CO_2 and CH_4 emission data, with 1 hour incubation period. Summer measurements were made during the growing period 1993, starting in late May and continuing until late October and the winter measurements (n=8) once a month from November 1994 to May 1995. The mire surface types sampled included wet hollows, intermediately watered *Sphagnum papillosum-S. balticum* lawns with some *Eriophorum*

vaginatum, lawns dominated by sedges and hummocks of varying heights above the water level. For more information on vegetation, see Saarnio *et al.* 199x.

During the gas exchange measurements, time series of solar irradiation (I, PAR μmol $m^{-2}s^{-1}$), depth of water table (WT) and temperature (T) in air and in soil profile were registered. Hourly time series on temperature and irradiation (integrated over every 7.5 min and averaged for the hour) were collected at nearby research station over the snow-free season. An estimate of gross photosynthesis (P_G) in varying conditions of irradiation, moisture and temperature was calculated as a sum of the chamber CO_2 fluxes measured in light (observed P_N) and dark (observed R_{TOT}) conditions.

Environmental response functions for P_G and R_{TOT}, depending on solar irradiation, soil surface temperature, effective temperature sum index (ETI) and depth of water table were constructed for the different mire surfaces. ETI was calculated as a quotient of effective temperature sum (treshold +5°C) and day number, and was used to model the presumed change in plant photosynthesis activity (a curve rising to its maximum in July and descending towards autumn). The response fuctions were of general form

$P_G = f (I) + g (T, ETI, WT)$ and
$R_{TOT} = f (T, WT)$,

where f (I) is a rectangular hyperbola and the other functions are polynomials. Parameter values for the environmental variables in the functions were estimated using nonlinear (P_G) or linear (R_{TOT}) regression techniques. P_N was calculated for every hour as a result of P_G - R_{TOT}. Respective functions were estimated for summer CH_4 emission (see Saarnio *et al.* 199x). The functions were used as transfer functions to reconstruct CO_2 exchange and CH_4 emission dynamics over the growing season according to the time series of irradiation, temperature and water table data monitored at the site. The hourly estimates were integrated to obtain estimates for the whole growing season.

Results and discussion

The study plots represented different classes of microsites, flarks, *Sphagnum*- and *Carex*-lawns and hummocks. Depth of water table in Salmisuo was rather stagnant over the study period, leaving the variation in solar irradiation and temperature the most decisive factors for P_N at *Carex*-lawn and flark microsites in the statistical environmental response functions. Data from *Sphagnum*-lawns and hummocks, however, originated from study plots having more or less different heights relative to the water table. In these, a parameter for depth of the water table was included in the response functions. An example of model predictions with observed P_G and a reconstruction of the course of P_N for different microsites over a week in August is given in Fig. 1 and 2.

During warm summer days, the highest hummocks in many occasions released more CO_2 than was bound in the same time by photosynthesis, but the instantaneous daytime balance of lawns and flark plots usually indicated net carbon binding to vegetation. In dim light, especially during the nighttime, all plots released CO_2, but some photosynthesis was measured even below irradiation level of 50 μmol $m^{-2}s^{-1}$. Fig. 3 summarizes (on monthly basis) the reconstructed P_N over the growing season at
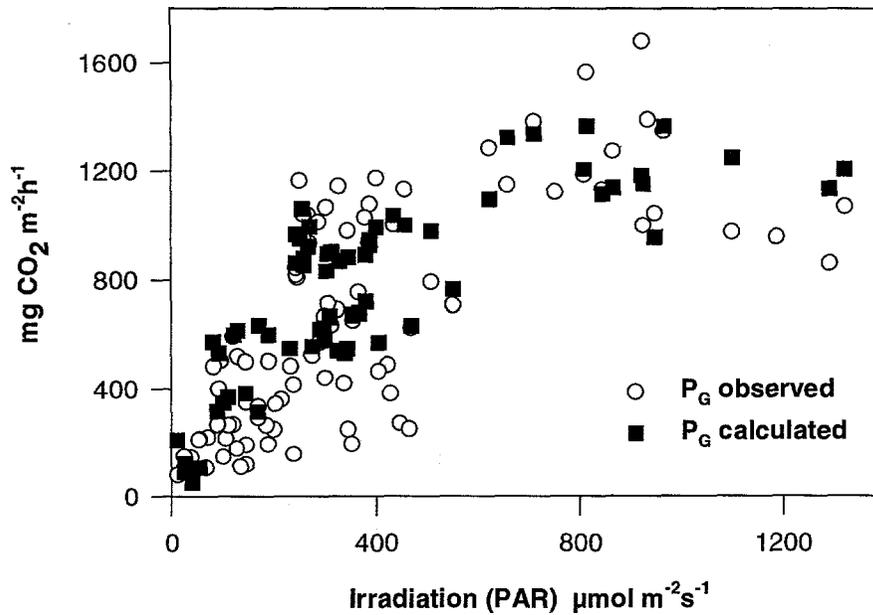
Fig. 1. Gross CO_2 exchange (P_G,, mg CO_2 m^{-2} h^{-1}) in *Carex*-lawn sites in Salmisuo with model predictions, plotted against solar irradiation (PAR, μmol m^{-2} s^{-1}).
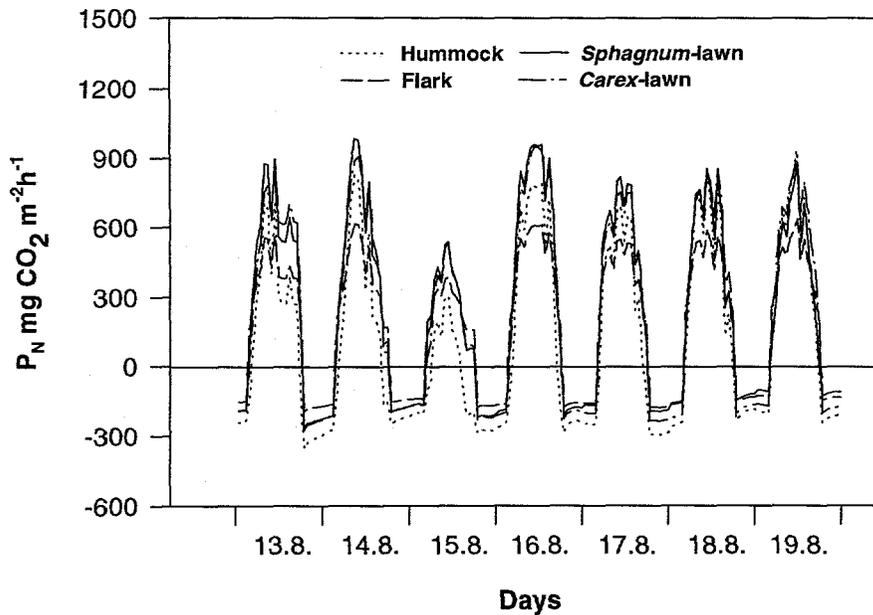
Fig. 2. An example of reconstructed hourly dynamics of P_N on different microsites during one week in August 1993. Positive values indicate binding and negative values the release of carbon.
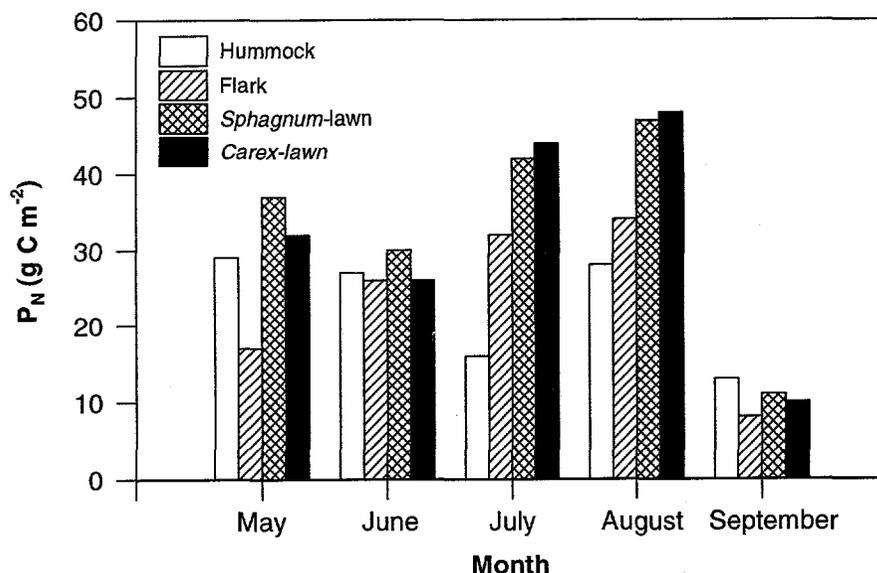
Fig. 3. Recostructed monthly sums of P_N (g C m^{-2}) in different microsites in Salmisuo, Ilomantsi 1993.

different microsites. In this reconstruction, hummocks have their lowest rates of carbon sequestration during the mid-season period. Net ecosystem CO_2-exchange integrated over the growing season was largest (167 g CO_2-C m^{-2}) in *Sphagnum* lawns, lowest in hummocks and wet hollows (114 g and 118 g, respectively) and rather high (160 g) in *Carex rostrata* lawns. It also seemed that the highest (and oldest) hummocks were the most inefficient carbon binders. Temperature, water table and thus the rate of soil respiration seem to constrain the daily ecosystem net CO_2 exchange.

Compilation of carbon budget for a mire ecosystem on the basis of gas exchange results is difficult due to the great spatial microsite variability in gas exhange, together with other potential sources of error. The stunted 2-3 m high pines, growing scarcely on the highest hummocks, were omitted from the analysis because of methodological difficulties. The reconstructed figures for CO_2 exchange and CH_4 emission represent the conditions of the study year only. Also, we did not measure carbon leaching (DOC) from Salmisuo, and use here a value obtained for virgin minerotrophic catchment of Lakkasuo (Hyytiälä), 8 g m^{-2} a^{-1} (Sallantaus 1992).

The accumulation rates obtained for different microsites in Salmisuo ranged from 37 g C m^{-2} a^{-1} (flarks and hummocks) to 79 g C m^{-2} a^{-1} (*Sphagnum* lawns). An overall estimate for the mire can be obtained by using the distribution of hummocks (14 %) and lawns (86 %) as weight coefficients, leading to annual peat deposition of 73 g C m^{-2}. This is more than is usually obtained for Finnish mires according to peat-core estimates (long-term rate of carbon accumulation 4-50 g m^{-2}, Tolonen *et al.* 1994, 199x).

Carbon efflux as methane is an important component of mire carbon balance. According to Saarnio *et al.* 199x, the differences in CH_4 emission from the different microsites were

considerable, the hummocks releasing least and *Carex*-lawns most methane. There were no marked differences in the winter fluxes of CH_4 and CO_2 (3 g CH_4-C m^{-2} and 55 g CO_2-C, respectively) between hummock and lawn surfaces.

The proportion of CH_4 release of the annual gross CO_2 exchange (P_G of the growing season minus winter CO_2 release) ranged 3 - 9 %, being larger in moist microsites with sedges and smaller in the drier hummocks. When the annual CH_4 release rates are compared with the annual carbon accumulation figures, the proportions are much higher, ranging from 32 % in *Sphagnum*-lawns to as high as 52 % in *Carex*-lawns.

Our result should be considered as the actual rate of carbon accumulation for the year 1993. Altough the present results seem large, they are of reasonable magnitude. It can be expected that the rate of peat accumulation in this kind of mire is high in a season with persisting high water tables, and would be much lower in drier seasons. The statistical dependensies of CO_2 exchange dynamics, modelled in this study, suggest that carbon balance in a peatland ecosystem is highly dependent on daily climatic events during the growing season. In Salmisuo, even a temporal summertime drawdown of water table should reduce carbon sequestration in hummocks and lawns with current vegetation assemblages.

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Possible responses of northern peatlands to climate change in the zone of discontinuous permafrost, Manitoba, Canada

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Introduction

Over half of the world's peatlands occur in the boreal zone (45 - 60° N. lat), a region that global climate models predict will experience large changes in temperature and precipitation with increasing atmospheric CO_2 concentrations (Hansen et al. 1988). The northern part of the boreal zone is characterized by discontinuous permafrost, an area that is particularly sensitive to climate change with the possible degradation and thawing of frozen peat. Peatlands are large sources of atmospheric methane (CH_4), an important greenhouse gas. Yet few measurements of methane have been conducted in discontinuous permafrost environments. As part of the Boreal Ecosystem-Atmosphere Study (BOREAS)(Sellers et al. 1995), we measured CH_4 flux in a diverse peatland complex (bogs, fens, peat plateaus, and collapse scars), representing the complete range of temperature, moisture, and plant community gradients found in northern peatlands. The measurement period May to September 1994 was one of the warmest and driest seasons on record, which provided an opportunity to observe the short-term responses of different parts of the peatland ecosystem to a warmer and drier climate as an analog to predicted climate change in the region.

Results and Discussion

Peak seasonal CH_4 fluxes ranged from 0 in the frozen peat plateaus and palsas to 1200 mg m^{-2} d^{-1} in the open sedge-rich fens and saturated lagg areas of collapse scars, where the permafrost was degrading. Continuous measurements of water table and peat temperature showed that hummocks were usually warmer than hollows at a peat depth of 10 cm even though CH_4 flux from the hummocks was significantly lower. However,

the peat temperature at the average position of the water table was higher in the hollow, suggesting that the temperature at the most active zone of CH_4 production and oxidation is a better predictor of CH_4 flux than a standard depth from the peat surface. The seasonal average peat temperature at the water table explained most of the variability in mean CH_4 flux ($r^2 = 0.63$)(Fig. 1).
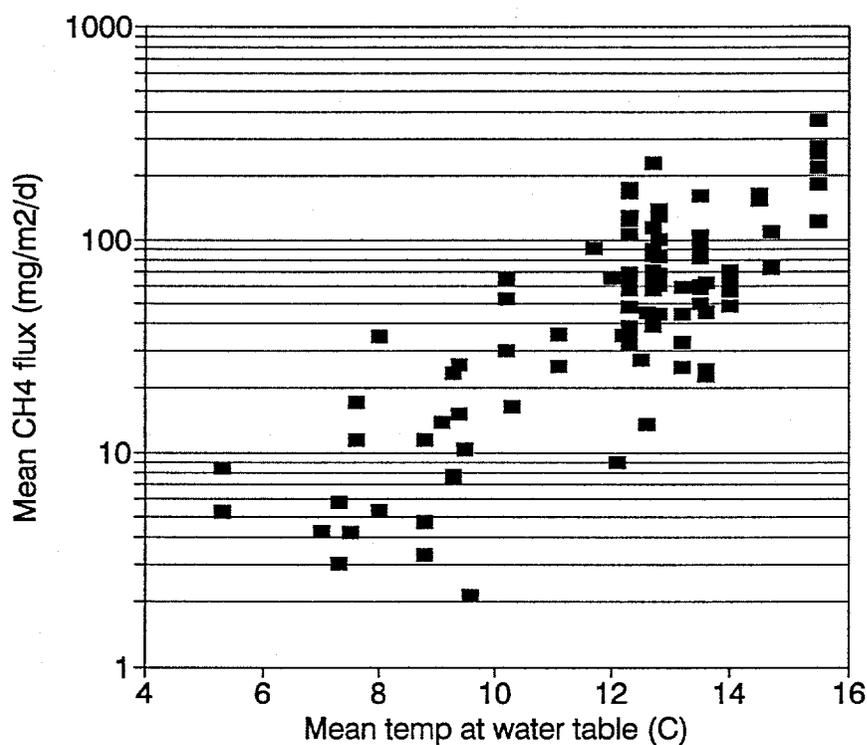
Fig. 1. Seasonal mean CH_4 flux versus mean temperature at mean position of water table (regression). Each point represents an individual collar. Log CH_4 flux = 0.16 temp - 0.35. r^2 = 0.63; p < 0.001; standard error of the estimate = 0.28; standard error of x coeff = 0.0003.

At the wet end of the moisture gradient, where fluxes were the highest, a variety of complex processes may explain the variability in CH_4 emission. Laboratory incubations of the different peat types in flooded areas showed that methane production potentials did not correlate well with CH_4 flux. But we suggest that a variety of mechanisms may be important: 1) ebullition, 2) floating peat, 3) ecosystem productivity and 4) degrading peat.

1) Ebullition: In areas of open water with thick ice cover, a large flux of methane was observed in May as the ice melted. In some sites, ebullition occurred sporadically throughout the season. These large emissions often dominated the total seasonal flux. 2) Floating peat: Even though 1994 was the third driest year on record, areas of floating peat maintained high CH_4 emissions throughout the season. Continuous measurements of the peat surface and water table showed that the peat surface adjusted to the falling water table, maintaining anaerobic conditions for CH_4 production and minimizing oxidation in the upper layers of peat. 3) Ecosystem productivity: When the sites were flooded, a strong relationship between net ecosystem productivity (NEP) and CH_4 flux was observed (Bellisario, unpub. data), perhaps due to the production of root exudates providing substrates for methanogenesis. However, as the season progressed and the water table dropped below the rooting zone of the dominant vascular plants, the relationship weakened. When the peat became drier, oxidation probably dominated the flux and the production of root exudates was no longer below the water table in the zone of CH_4 production. The most productive sites were dominated by sedges, esp. *Carex* and *Eriophorum* species. 4) Degrading pools: In many areas of northern peatland, pools develop in later successional stages and emit large amounts of CH_4 (Moore et al. 1994). Other types of open water areas with high CH_4 emissions include lagg areas of collapse scars in degrading permafrost and beaver ponds (Roulet et al. unpub. data). Little or no vegetation occurs in these areas and the temperature, composition and age of the peat may be important in determining CH_4 flux.

Because of the different processes controlling CH_4 emission from these ecosystems, we would expect a variety of reponses to climate change (Table 1). A warmer and drier climate would potentially affect these various types of peats differently, resulting in higher CH_4 flux in some cases (floating and degrading peats, collapse scars), lower flux in others (non-floating peat). The effect on sedge-dominated sites is ambiguous because of the competing processes of increased plant productivity and increased microbial oxidation of CH_4.

Table 1. Climate change scenario: If the climate became warmer and drier in the future, we would expect different short-term responses from the various types of peatland environments as observed in the abnormally dry and warm 1994 summer season. Some areas would probably experience higher CH_4 emissions, some lower. In the long run, plant community structure and hydrological function would change in each of these systems resulting in a different set of ecosystem responses.

Peatland environment	Predicted change in CH_4 flux	Rationale
Non-floating peat	decrease	Decrease in water table would have a greater effect on CH_4 oxidation than increase in temperature would have on production (Roulet et al. 1992).
Floating peat	increase?	Peat surface would adjust to falling water table, but higher temperatures would increase CH_4 production. However, there may be a limit to peat surface adjustment.
Sedge-dominated sites	decrease?	Higher temperatures may increase productivity, but lower water table would decouple rooting zone from anaerobic zone of CH_4 production.
Degrading pools	increase	Higher temperatures would increase decomposition and a drop in water table would create an aerobic zone only in very shallow pools.

Plant communities indicate many of these ecological processes that control CH_4 flux (Bubier et al. 1995, in press): They are indirect indicators of mean water table position, which is a good predictor of seasonal CH_4 emissions, and may be used to extrapolate flux from the chamber to the landscape scale using remote sensing. Canonical correspondence analysis (CCA)(Fig. 2) shows that treed sites have low CH_4 flux because they correlate with dry (height above mean water table - HMWT), aerated conditions. Open graminoid sites are associated with high CH_4 flux because sedges indicate flooded

anaerobic conditions, highly productive ecosystems, and areas of vascular plant transport enhancing CH_4 emissions. Floating peats, typically near the laggs of collapse scars, are characterized by certain vascular and bryophyte species, especially *Carex rostrata* and *Sphagnum riparium*. In the long-term, plant communities and CH_4 emissions will change in response to a warmer and drier climate as the hydrology and thermal regime of the peatlands evolve.
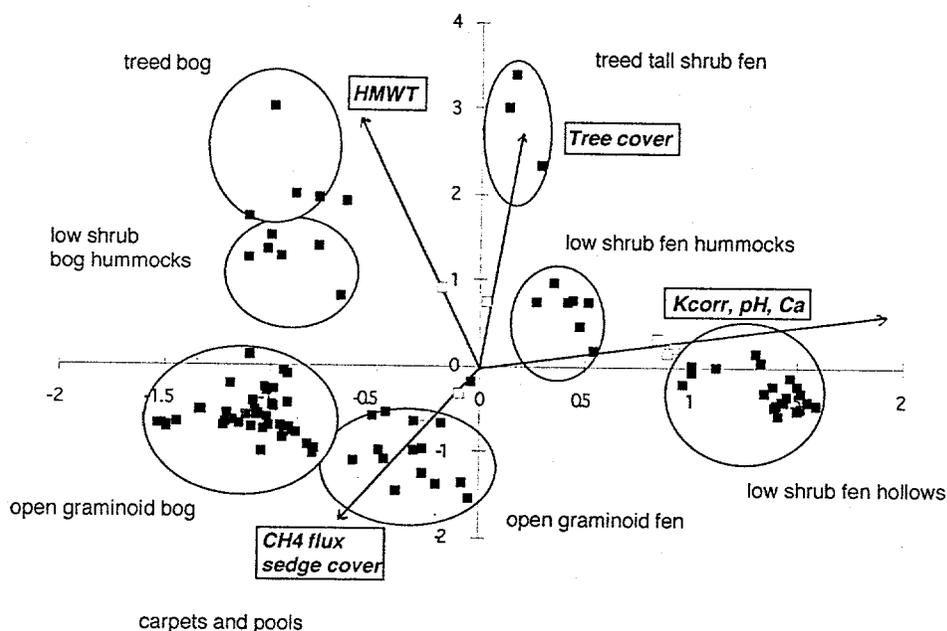
Fig. 2. Canonical correspondence analysis (CCA) of vascular and bryophyte assemblages and environmental variables. Solid boxes on the sample biplot represent individual collars; vegetation classes and microtopographic groupings are circled. The vectors (drawn from open boxes) represent environmental variables increasing in the positive direction from the center (0,0 point) of the biplot, which represents the mean value of all environmental variables. Negative values are by extending the vectors in the opposite direction. The position of each collar along an environmental gradient can be inferred by drawing a perpendicular line from the collar to the vector of interest. Eigenvalues of axes 1,2 and 3 are 0.72, 0.54, and 0.43. Interset correlations for pH, calcium, and corrected conductivity (Kcorr) with axis 1 are 894, 861, and 814 respectively. Interset correlations for height above mean water table (HMWT), tree cover, sedge cover, and CH_4 flux with axis 2 are 837, 687, -633 and -546 respectively.

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Carbon dioxide emissions and nutrition on a drained pine mire - a case study

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Introduction

Drainage of boreal peatlands intensify aerobic decomposition and carbon dioxide emission from the peat substrate and increase tree growth. CO_2 emission rates depend on the ground water level and the soil temperature. Predicted rises in mean air temperatures due to anthropogenically induced climate change are expected to further increase carbon dioxide emission from drained boreal peatlands. The role of added nutrients is somewhat vague.

The purpose of this paper is to present some preliminary results on microbial biomass carbon and on carbon dioxide output / input relationship on a pine mire.

Material and methods

The experiment is located on the mire Liesineva in Parkano, western Finland, 61° 59' 15" N, 23° 14' 15", altitude 159-162 m (Fig. 1). It is a part of an eccentric raised bog, which originally consisted of both ombrotrophic and oligo-minerotrophic sites. It was drained for forestry in 1915 with very wide ditch spacings (> 200 m). In the middle of the 1950s a ditch spacing experiment was established in the area (Huikari 1959). The area was then partly treeless, partly sparsely stocked with 1-5 m tall, poorly growing Scots pines (*Pinus sylvestris* L.). Tree growth improved somewhat after this additional ditching, but more vigorously only after fertilization in the middle of the1960s.

The experiment consists of two blocks (2 and 3). Block 3 was drained with covered ditches in 1954 and block 2 with open ditches in 1955. The open ditches were cleaned and some new border ditches were dug in autumn 1992. The total area is about 17 ha. Presently, the dominant tree species is Scots pine, with downy birch (*Betula pubescens* Ehrh.) mixture on some plots.

The ditch spacings examined for this paper were 20, 40, 60 and 100 m. There were three fertilization treatments: control, PK/NPK application in 1965 and additional NPK application in 1977 (Fig. 1). There were 30 plots of which 20 (controls and additional

NPK 1977) were investigated for microbial biomass carbon. Peat samples were taken in autumn 1992. The tree stands were measured for stem volume in summer 1993. During the growing seasons (May to September/October) of 1992 and 1993 carbon dioxide emissions were regularly measured on every plot (intervals 8 to 11 days) with a portable infra-red gas analyzer. The air temperature, the soil temperatures at the depths of 0 cm, 5 cm and 10 cm and the ground water level were measured simultaneously.
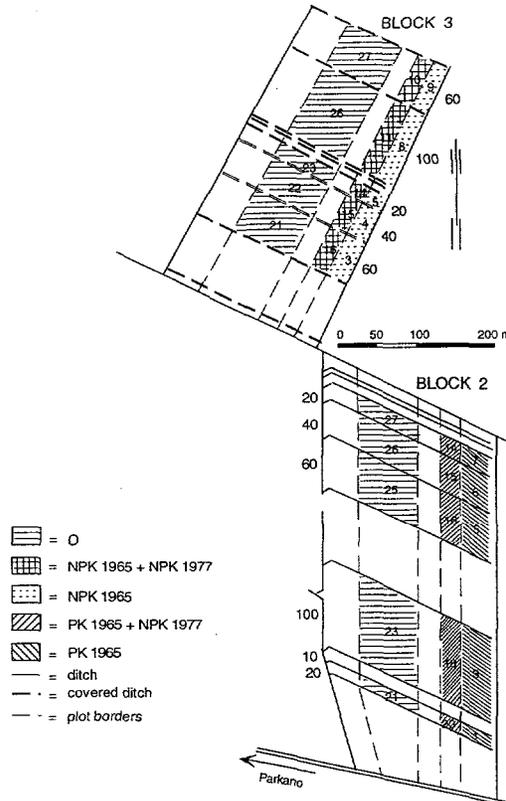
Figure 1. Layout of the experiment. The approximate location of the observation spots for CO_2 emission, ground water level and soil temperature (n = 6 per plot) measurements is indicated by the plot number. See text for further details.

The CO_2 emission rates were estimated by fitting the measured plotwise CO_2 emission and the microclimate data into a regression model (Mead *et al.* 1993, Silvola *et al.* 1994). The climatic data collected from a weather station about 5 km from the experiment were applied to interpolate the daily and monthly CO_2 emission sums.

The tree stand biomass was calculated according to Finér (1991). The carbon content of tree biomass was calculated using coefficient 0.519 (Laine *et al.* 1995). There were no recent results of annual tree growth available. In order to get a rough estimate on

the relationship between the effect of fertilization on the CO_2-C output from peat in 1992 and 1993 and the carbon uptake by tree biomass, the estimated annual carbon uptake rates were calculated by dividing the estimated tree biomass carbon by drainage age (38 and 39 years) .

The microbial biomass carbon of the peat samples was measured with a fumigation extraction method (Martikainen and Palojärvi 1990).

Results

Carbon dioxide emissions

The total CO_2-C emission rates varied between 240 and 570 g C m^{-2} a^{-1} in 1992 and 120 and 560 g C m^{-2} a^{-1} in 1993. The results agree with the published results from similar sites (Silvola et al. 1994).

The variables which explained the CO_2 emission rates best were the ground water level and soil temperature (Fig. 2).
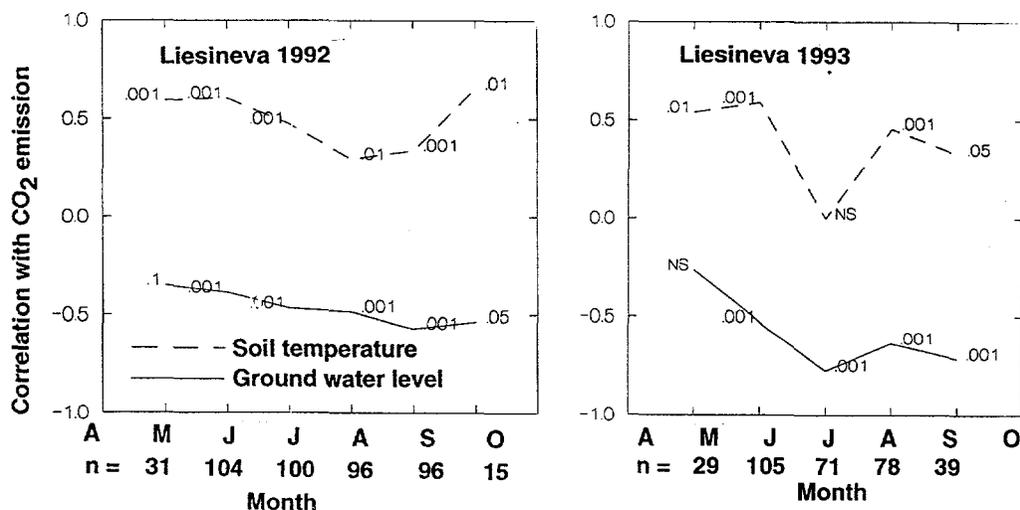
Fig. 2. Monthly correlations of soil temperature (positive) and ground water level (negative) with CO_2 emissions during 1992 (May to October) and 1993 (May to September). The numbers close to the lines indicate the risk level (NS = no significance, 0.05 = 5% risk, 0.01 = 1% risk and 0.001 = 0.1% risk) of the correlation coefficient for the corresponding month. The total amount of observations are given below the month abbrevations.

The mean CO_2-C emission rate was higher in 1992 than in 1993. The low correlation between CO_2 emission and the soil temperature during July in 1993 was (indirectly) caused by an exceptionally high ground water level. Because the ground water level

determines the aerobically reactive belowground surface area, it is the primary factor in organic matter decomposition while the soil temperature is only a secondary one.

Carbon uptake and release

In 1993 the estimated mean annual tree growth ranged between 0.3 and 7.3 m^3/ha. The corresponding estimated mean post-drainage carbon uptake of the tree biomass varied between 12 and 217 g m^{-2} a^{-1}.The estimated annual carbon emission from soil was subtracted from the mean annual carbon bound by trees and plotted against the former (Fig. 3). The total CO_2-carbon output from the peat was greater than the carbon uptake by the tree biomass on all plots. When comparing plots at the same carbon dioxide emission level the remainder (tree biomass carbon - carbon emitted from soil) was higher on the fertilized than unfertilized plots.

The differences between the years 1992 and 1993 is accentuated especially in the unfertilized plots as smaller relative CO_2-carbon output rates from the peat due to a higher mean water level in mid-summer.
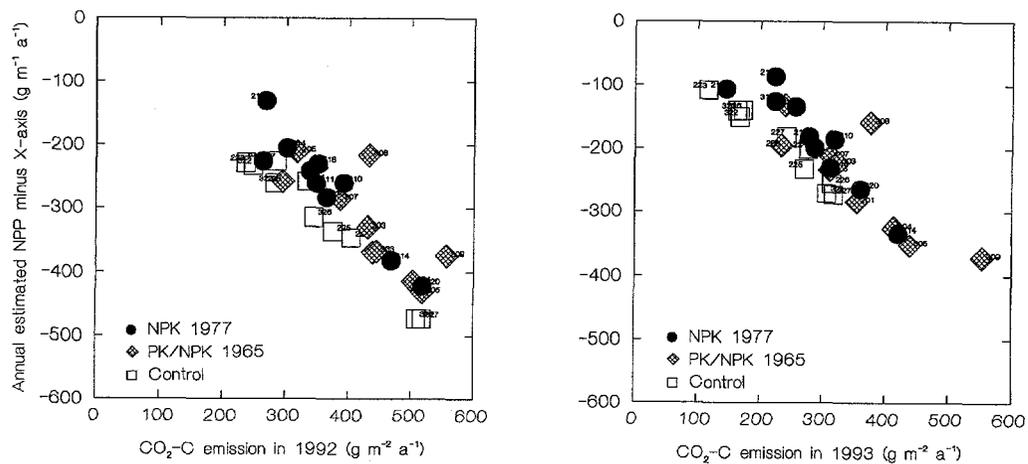
Fig 3. The estimated plotwise carbon bound into tree biomass (net primary production, NPP) minus the estimated plotwise carbon emitted from the substrate (soil respiration) (Y-axis) plotted against the former (X-axis) in different fertilization treatments.

Microbial biomass carbon

The amounts of microbial biomass carbon were 2.9 to 10.8 mg/g in the raw humus layer, 1.5 to 7.7 mg/g in the 0-10 cm layer and 0.7 to 4.1 mg/g in the 10-20 cm layer. In this study there occurred a slightly negative relationship between tree biomass and microbial biomass carbon (Fig. 4).
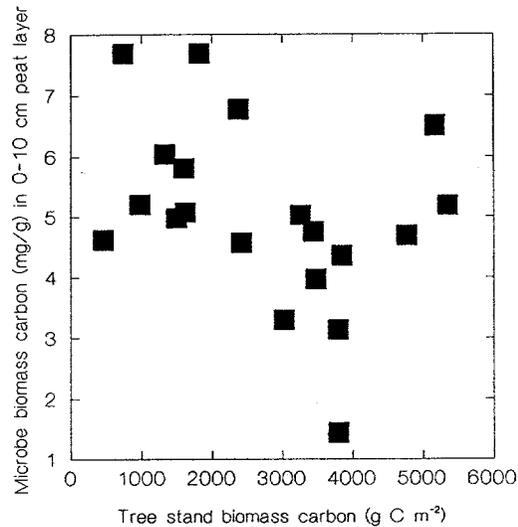
Fig. 4. Plotwise microbial biomass carbon in the 0-10 cm peat layer versus tree stand biomasses.

Discussion

The amount of microbial biomass carbon in the peat layers had a slightly negative correlation with tree stand biomass, but the data is insufficient for any further conclusions.

It was not feasible to try to estimate the carbon balance in the stands, because root respiration and the amount of carbon bound by trees at the time of emission measurements were not known. However, the mean annual carbon fixed into tree biomass after intensified drainage gives a measure for estimating the effect of fertilization on the proportion of carbon emitted from soil in relation to carbon bound into trees, because fertilization had been the main reason for enhanced tree growth.

When regarding the estimated plotwise tree stand development, it is seems that the nutrient applications slightly enhanced the fixation of carbon by the tree biomass in relation to CO_2 emission rates. These preliminary results give reason for further, more detailed studies on the relations between stand growth and CO_2 emission sources on these sites

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Carbon cycling in restored peatlands: results from calibration, summer 1994

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Introduction

The most extensive use of peatlands in Finland is for forestry. Some 5.3 million hectares of the original peatland area of 10.4 million hectares have been drained for forestry purposes (Laine & Päivänen 1992). Now many peatland forests are coming to the end of the first rotation and a decision must be made on the land-use of these areas in the near future. Restoration by rewetting is one alternative, especially in those areas where forestry has failed. In nature conservation areas, restoration of drained peatlands is also sensible and has already been done to some extent.

Before large scale rewetting programs are started, it must be determined how these activities will affect the structure and functioning of previously changed peatland ecosystems. The aim of this study is to discover if it is possible to restore a part of peatlands previously drained for forestry use back to functioning mire ecosystems, in which a substantial part of the organic matter produced mainly by bottom and fieldlayer vegetation is accumulated as peat due to the low level of microbial activity in the soil. The main hypotheses of this study are, that after rewetting, vegetation formed after secondary succession will start to develop towards mire vegetation and that net ecosystem production will become positive as the rate of aerobic decomposition decreases.

Material and methods

Study sites

The study sites are located in the communes of Orivesi and Ruovesi in central Finland. Konilamminsuo mire (61°48'N, 24°17'E) was drained for the first time in 1955 and in its virgin stage the site type in the study area was minerotrophic tall-sedge pine fen. Viheriäisenneva mire (61°51'N,24°14'E) was first drained in 1967 and before drainage the

site type was ombrotrophic low-sedge bog (Heikurainen & Mannerkoski 1982). In summer 1994 bottom and field layer vegetation of both mires were in a transition stage. In the Konilamminsuo mire there was a 70-year-old Scots Pine stand (130 m³/ha), while the Viheriäisenneva mire was almost treeless. The restoration by clearcutting and blocking the drainage ditches was started in February 1995.

Methods

At the study sites, 9 sample plots in Konilamminsuo mire and 8 in Viheriäisenneva mire were established on subjectively chosen surfaces that represented different kinds of vegetation. Sample plots were delimited with 0.36 m² (60 x 60 cm) aluminium collars, inserted into the peat to a depth of 30 cm. Vegetation of the sample plots was analysed using the percentage cover estimation method. Carbon dioxide and methane fluxes from these sample plots were measured by using two kinds of static chamber (aluminium and plastic) techniques.

When measuring CH_4 emission (released in anaerobic decomposition) and respiration (CO_2 released from decomposition and respiration of vegetation), an aluminium chamber was placed into the collar and a series of air samples was taken from the chamber with 60 ml plastic syringes 5, 15, 25 and 35 min. after closing the chamber. CH_4 and CO_2 concentrations of the samples were determined in the laboratory by gas chromatography (HP 5890 Series II, for the analysis method, see Nykänen et al. 1995) within 24 h of sampling. Simultaneously with air samples, the temperatures in peat profiles and the groundwater level were measured.

A transparent plastic chamber technique was used in the CO_2 balance and respiration measurements. The CO_2 concentration of the air in the chamber was measured with a portable infrared gas analyser (ADC LCA 2) at intervals of 30 seconds after closing the chamber. The first measurement was carried out in sunlight over a 3-minute period (CO_2 balance measurement). After that the chamber was removed and aerated. The second measurement (also taken over a 3 minute period) was done with the chamber covered with a light-tight sheet (respiration measurement). The chamber was fitted with a cooling system to keep the air temperature inside the chamber the same as the ambient temperature. On the top of the chamber there was a vent to equalise the pressure inside the chamber. Solar irradiation, water table level and soil temperatures in peat profiles were measured at the same time as the CO_2 balance.

Carbon dioxide balance and respiration and methane emission and uptake were calculated from the linear change of CO_2 and CH_4 concentrations inside the chambers as a function of time. Data from the aluminium and the plastic chamber measurements were used for respiration, CH_4 emission and uptake calculations, but for CO_2 balance calculation only data from the plastic chamber measurements were used. CO_2 fixation in photosynthesis was calculated as the resultant of CO_2 balance and respiration values.

Results and discussion

Soil and vegetation respiration

The rates of respiration in Konilamminsuo mire varied from 180 to 860 $mgCO_2$ m^{-2} h^{-1} and in Viheriäisenneva mire from 100 to 830 $mgCO_2$ $m^{-2}h^{-1}$. The highest rates were measured in midsummer and the lowest early in the summer and in the autumn. Spatial variations of respiration in both mires were small, only hummocks in the Viheriäisenneva mire being separated on the basis of the higher rate of respiration. In the more nutrient-rich Konilamminsuo mire where the groundwater level was deeper, the average rates of respiration were higher than those in Viheriäisenneva. The groundwater level in the Konilammensuo mire varied between 15 and 73 cm and in the Viheriäisenneva mire between 13 and 53 cm. (Fig. 1.)
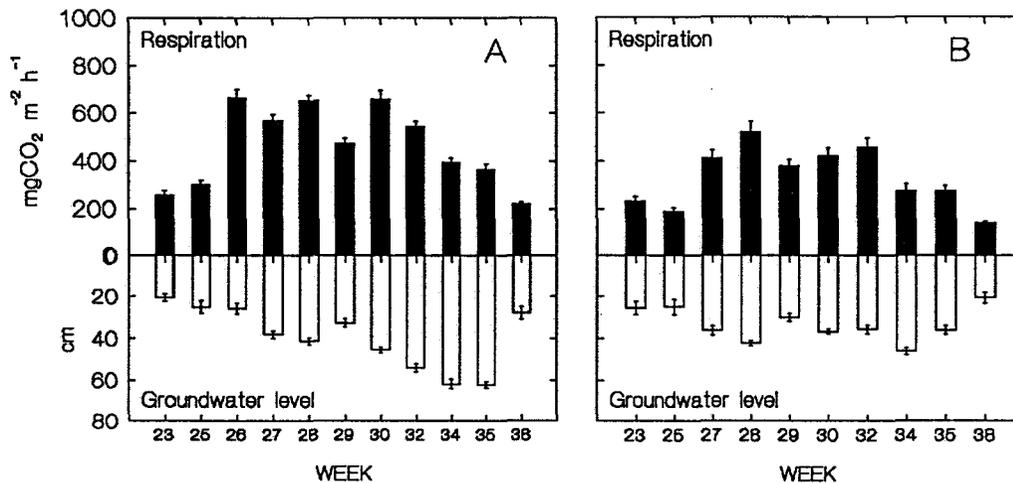
Fig. 1. Average rates of respiration and groundwater levels in Konilamminsuo mire (A) and in Viheriäisenneva mire (B) during summer 1994. (S.E. on the bars)

There was a positive correlation between rates of respiration and soil temperature in both mires. Normally biochemical processes have an exponential relationship with temperature increase, but in this study the best equation could be made by using a linear regression (equation not shown). The main reason for the linear regression is that during the measurements the soil temperature was always higher than 8.4°C (Fig. 2.). The level of respiration was similar to those results measured earlier in the SUOSILMU subprojects in the same kind of site types (Silvola & Alm 1992, Silvola & al. 1994, Martikainen & al. 1992).
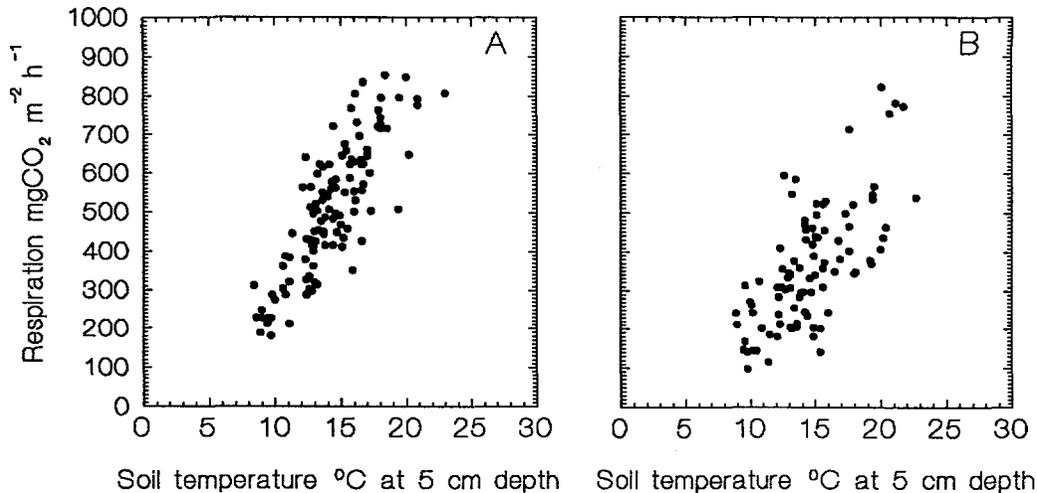
Fig. 2. Scatter plots of respiration rates dependence on soil temperature at 5 cm depth in Konilamminsuo mire (A) and in Viheriäisenneva mire (B).

Carbon dioxide fixation and balance

The most important factors controlling CO_2 fixation in photosynthesis are solar irradiation, temperature, nutrient and water availability. The summer of 1994 was not an ideal one for do *in situ* measurements of photosynthesis, because in midsummer when most of the measurements were carried out, the vegetation in the sample plots was suffering from drought. The CO_2 fixation rate varied in Konilamminsuo mire from 110 to 1050 $mgCO_2$ m^{-2} h^{-1} and in Viheriäisenneva mire from 40 to 1220 $mgCO_2$ m^{-2} h^{-1}. During measurements, irradiation of the plastic chamber was kept as constant as possible and the total range of irradiation was from 50 to 900 μmol m^{-2} s^{-1}. The highest rates of CO_2 fixation in Konilamminsuo mire were in sample plots having vegetation cover with *Eriophorum vaginatum* L., *Vccinium myrtillus* L., *Vaccinium uliginosum* L., *Ledum palustre* L. and *Sphagnum angustifolium* (Russ.) C. Jens. and in Viheriäisenneva mire, in hummocks with *Calluna vulgaris* (L.) Hull, *Empetrum nigrum* L., *Vaccinium uliginosum* L. and *Betula nana* L.. The lowest rates of CO_2 fixation were measured in *Pleurozium schreberi* (Brid.) Mitt. (Konilamminsuo) and *Cladonia spp.* Hill ex Browne (Viheriäisenneva) vegetation.

The CO_2 balance of an ecosystem is dependent on total respiration and total fixation in photosyntesis. Because automatic measuring systems for environmental data were not available, it is difficult to calculate or simulate daily CO_2 balance. When results of respiration and fixation from the measurements are combined, it can be said that most of the CO_2 released in daytime respiration could be bound in photosynthesis in Konilamminsuo and all in Viheriäisenneva (Fig. 3.). It can be concluded that carbon cycling is fast; the emissions from the peat (total respiration) are rapidly returned to the peat by the vegetation (photosynthesis).
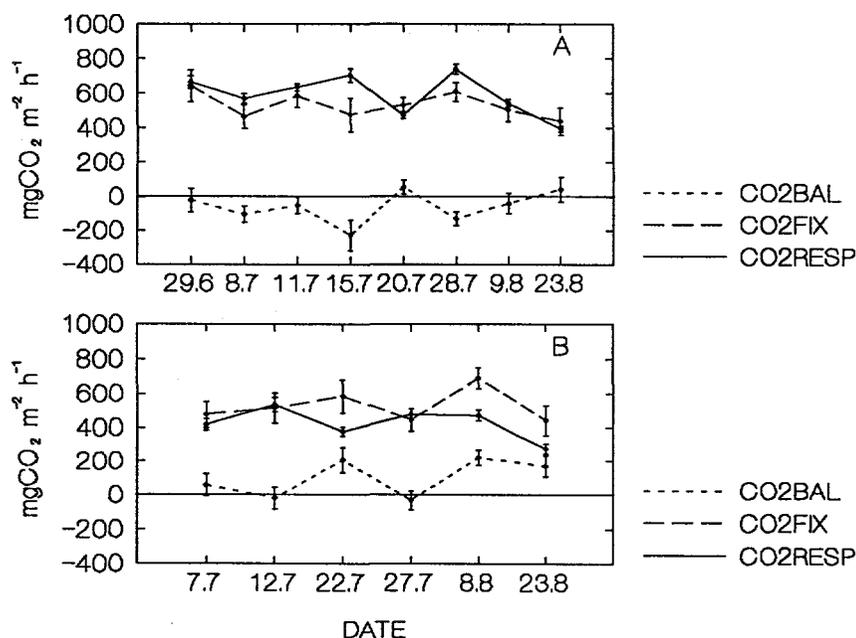
Fig. 3. Average results of respiration (CO2RESP), CO_2 fixation (CO2FIX) and CO_2 balance (CO2BAL) measurements in Konilamminsuo mire (A) and in Viheriäisenneva mire (B), during summer 1994. (S.E. represented by bars.)

Methane emission and uptake

Methane emissions were low in both mires. In the Konilamminsuo mire, methane emissions varied from 0.1 to 7.6 $mgCH_4$ m^{-2} d^{-1} and in the Viheriäisenneva mire from 0.3 to 18.1 $mgCH_4$ m^{-2} d^{-1}. Groundwater level varied between 15 and 66 cm, soil temperature at 5 cm depth between 8.4 and 19.4 °C and at 30 cm depth between 6.6 and 13.7 °C in the Konilamminsuo mire. In the Viheriäisenneva mire groundwater level varied between 13 and 41 cm, soil temperature at 5 cm depth between 9.5 and 16.0 °C and at 30 cm depth between 6.4 and 14.7 °C. The relationship between temperature or groundwater level and methane emissions were not clear. In Viheriäisenneva mire, where peatlayer was deeper and groundwater level shallower the average methane emission was higher than in Konilammensuo mire (Fig. 4.).

In each mire only one of the sample plots, dominated by *Sphagnum angustifolium* (Russ.) C. Jens. in the Konilamminsuo mire and by *Sphagnum balticum* (Russ.) C. Jens. in the Viheriäisenneva mire, acted as a methane source at every sampling time (Fig. 5.). It was assumed that vegetation surfaces dominated by *Sphagnum* mosses have low oxygen diffusion rates despite the rather deep groundwater level, and thus a low methane oxidizing capacity. In all other sample plots methane uptake from air to peat caused by methane oxidizing bacteria was observed. In Konilamminsuo, rates of methane uptake varied from 0.1 to 1.8 $mgCH_4$ m^{-2} d^{-1} and in Viheriäisenneva mire from 0.05 to 1.5 $mgCH_4$ m^{-2} d^{-1}.
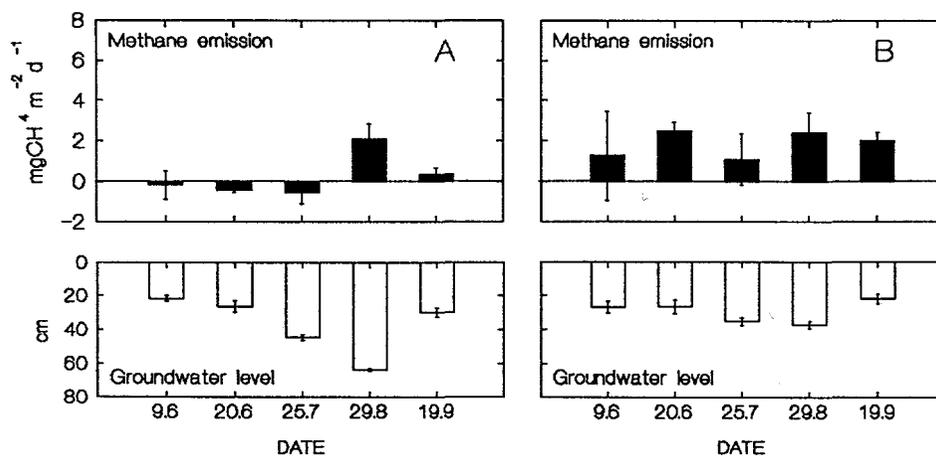
Fig. 4. Average rates of CH$_4$ emissions and groundwater levels in Konilamminsuo mire (A) and in Viheriäisenneva mire (B) during the summer 1994. (S.E. on the bars.)
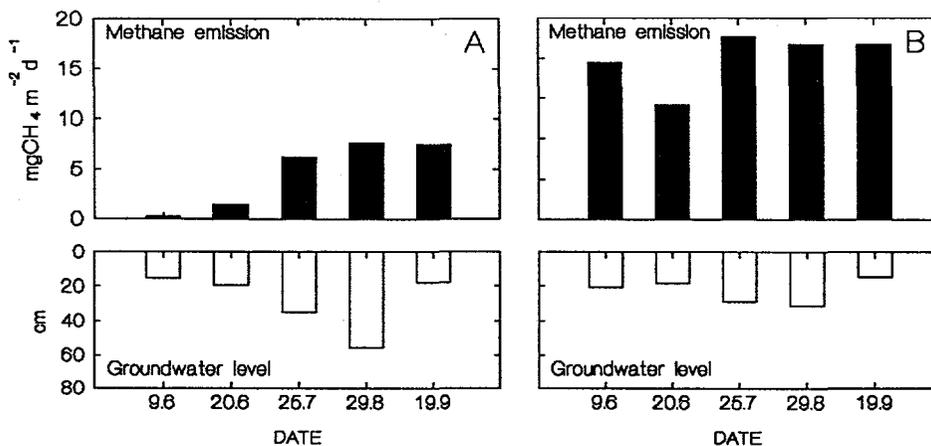
Fig. 5. Rates of CH$_4$ emissions and groundwater levels in one sample plot dominated by *Sphagnum* mosses in Konilamminsuo mire (A) and in Viheriäisenneva mire (B) during summer 1994.

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Methane emissions from an ombrotrophic mire in southern Finland receiving experimental nitrogen load

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Introduction

Release of methane (CH_4) from peatlands is the difference between its production and oxidation in peat profile. The bacteria producing CH_4 in peat depend energetically on the organic compounds produced by mire vegetation. Methane production is thus associated with primary production (Dacey et al. 1994). Increase in primary production can be expected to enhance production of CH_4. Low availability of nitrogen on ombrotrophic peatlands could well limit the primary production. Nitrogen load can enhance methane release not only via photosynthesis but also by inhibiting oxidation of CH_4 in peat (Crill et al. 1994; Martikainen et al., this volume). Load of nitrogen could also disturb methane production if *Sphagnum* vegetation does not tolerate the surplus nitrogen or if microbial reduction of nitrate consumes substrates needed in methanogenesis (Boone 1991).

We studied whether an experimental nitrogen deposition on ombrotrophic nitrogen–poor peat would have any effect on the methane effluxes.

Material and methods

The study was made on the ombrotrophic southern part of Lakkasuo mire in Orivesi, central Finland ($61°$ 48'N, $24°$ 19' E, 150 m a.s.l.). The mire site type is *Sphagnum fuscum* pine bog with some features of hummock and hollow pine bog. The thickness of peat is appr. 3 m of which the most part is ombrotrophic *Sphagnum* peat. Vegetation

on hummocks consists of dwarf shrubs (*Calluna vulgaris, Empetrum nigrum, Andromeda polifolia*) and bog mosses (*Sphagnum fuscum, S. angustifolium, S. rubellum*). On hollows, there exists a sparse cover of dwarf shrubs (*Andromeda polifolia, Vaccinium oxycoccos*) and bog mosses (*S. balticum, S. angustifolium*). The nitrogen content in the surface 30 cm of peat varies between 0.5 – 0.7 % and $pH(H_2O)$ between 3.8 – 4.1.

As the first aim of the study has been to investigate the growth and competition of *Sphagnum fuscum* and *S. angustifolium* in different nitrogen depositions, three sites where *S. fuscum* grew alone and three sites with *S. angustifolium* in a mixture were selected. Four 0.5 x 1.0 m plots were delimited at each of the six sites. The depth to the water table in each plot was measured weekly from a bore hole lined with a perforated plastic tube 2 cm in diameter.

Over the growing seasons (May – September 1992–1995) three of the four plots on each of the six sites received a total of either 0, 30 or 100 kg ha^{-1} N by irrigation. Nitrogen was given as NH_4NO_3 solution, and the 0 treatment was deionized water. The treatment solutions were applied on six occasions. The fourth plot at each site received no treatment and served as a control.

Aluminium frames of 0.5 m^2 (30 cm high) covering the whole study area were inserted to the peat. The fluxes were measured by filling the groove on the top of the frame with water and then putting an aluminium chamber on the frame (gas–tight connection with water seal). Four gas samples with plastic syringes were taken at intervals of 5, 15, 25 and 35 min and sent to Kuopio (Department of Environmental Microbiology) for gas analyses (Nykänen et al., this volume).

Results and discussion

During the study period in 1994 the fluxes of CH_4 varied from a slight uptake of –0.3 mg CH_4 m^{-2} d^{-1} to the emission of 36 mg CH_4 m^{-2} d^{-1}. As water level was generally 2–3 cm higher on sites where *S. angustifolium* grew in a mixture with *S. fuscum*, methane fluxes were a little higher from those sites except the highest nitrogen deposition level (Fig. 1). The fluxes on the experimental area were typical for ombrotrophic mires (see Nykänen et al., this volume).

The treatments from the year 1992 to 1994 have changed the CH_4 fluxes. The differences in the fluxes between different treatments on the *S. fuscum* plots were statistically significant. The N treatments had increased the emissions, most of all the treatment with 100 kg N ha^{-1} a^{-1} (Fig. 1). Also on the plots with *S. angustifolium/ S. fuscum* the positive effect of N on the CH_4 fluxes was evident (Fig. 1). The highest N treatment stimulated more the CH_4 fluxes from *S. fuscum* than from the plots with *S. fuscum* and *S. angustifolium*. Our treatments with NO_3^- showed no evidence for reduction in CH_4 emission as reported for sulphate load (Freeman et al. 1994; Nedwell & Watson 1995).

There are not yet any results published concerning the growth of different *Sphagnum*

species after 1992 which was the first measurement year (cf. Jauhiainen et al. 1992). It was noticed that length growth during the first year differed on different blocks but there was no trend concerning the nitrogen treatment. After four years of treatment, no changes in vegetation have been noticed. However, a difference between the reaction of *S. fuscum* and *S. angustifolium* was noted on the plots which had received 100 kg N ha^{-1} a^{-1}. *Sphagnum fuscum* was clearly suffering from the 100 kg deposition, and the amount of *Mylia anomala* was noticeable amongst the *Sphagnum fuscum* stems. This is in accordance with the results of a 120 days laboratory study performed (Jauhiainen et al. 1994) where a load of 100 kg N ha^{-1} a^{-1} inhibited elongation and biomass production almost completely. In this field experiment the reaction was more slowly as extra nitrogen could be leached downwards in the peat while in the laboratory experiment it was all the time in the living moss. *Sphagnum angustifolium* had increased growth on the plots which received extra nitrogen. This is also in accordance with the results of a laboratory study (Jauhiainen & Vasander 1994 and unpublished results). These changes in vegetation suggest that primary production was increased more on plots with *Sphagnum angustifolium* and *Sphagnum fuscum* than on those containing only *Sphagnum fuscum*. However, this did not correlate with the CH_4 release. Presently we do not know if there was any inhibition in CH_4 oxidation induced by the N treatments and if there were differences in the inhibition between plots. Reduction in oxidation of CH_4 would be a reason to the increase in CH_4 release after the N treatments. It could also be possible that the availability of peat derived organic compounds were increased by N favoring methane production. To explain the increase in the CH_4 emissions from the plots treated with N studies on processes in production and oxidation of CH_4 are required.
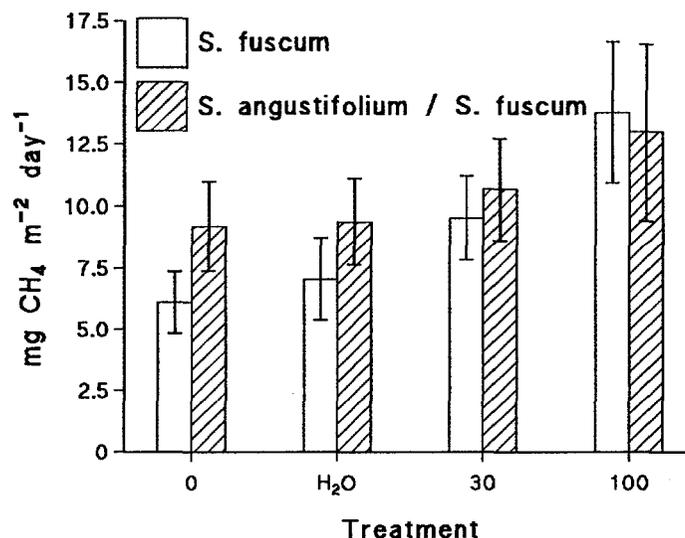
Fig. 1. Mean emissions of CH_4 in summer 1994 from the experimental plots.

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Methane fluxes in a drained and forested peatland treated with different nitrogen compounds

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Introduction

Methane (CH_4) fluxes in boreal mires correlate with level of water table. Therefore, draining of mires for forestry generally lowers emissions of CH_4 from peat surface. Mires emitting CH_4 at high rates can even become sinks for CH_4 after drainage (Martikainen et al. 1995; Roulet & Moore 1995; Nykänen et al., this volume). It has been shown that microbial oxidation of CH_4, a process responsible for CH_4 consumption in soils, is inhibited by mineral nitrogen (see Bedard & Knowles 1989). Decrease in the uptake of CH_4 in some forest soils by nitrogen has been found (Steudler et al. 1989; King & Adamsen 1992). There is no knowledge how nitrogen load would change the CH_4 dynamics in boreal mire ecosystems showing uptake of CH_4. Drained peat soils of high nitrogen concentration can have very different reactions to extra nitrogen than mineral forest soils of lower nitrogen concentration. We studied the effects of nitrogen addition on CH_4 uptake and CH_4 oxidation in a forested mire known to have a high CH_4 uptake rate. Furthermore, the effects of different nitrogen compounds on the CH_4 dynamic were compared.

Materials and methods

The experimental drained and forested peatland is located in eastern Finland ($62°51'N$ $30°53'E$). The mire was drained 40–50 years ago. Experimental plots of 3 x 3m were established in 1992. There were altogether 12 plots. The treatments with urea, NH_4Cl or KNO_3 were given on 17 September in 20 l of groundwater. There were 3 replicate control plots and 3 replicates for each treatment. The amount of N added was 100 kg ha^{-1}. The details of the experimental design are given by Crill et al. (1994).

The fluxes of CH_4 were measured with the closed chamber technique used in trace gas studies in mire ecosystems in Finland (Crill et al. 1994; Nykänen et al. 1995). The oxidation of CH_4 in peat profiles was studied in laboratory experiments (see Crill et al. 1994). The relationships between the CH_4 oxidation and nitrogen cycle were also considered. Intensive process studies on CH_4 and N cycles were done in 1993 (see Regina et al. for the N studies, this volume). The flux measurements were done at least once a month from autumn 1992 to autumn 1995.

Results and discussion

Short-term effects

Before the nitrogen additions the 12 plots had rather similar CH_4 uptake rates (Crill et al. 1994; Fig. 1). The first peat samples were taken about 3 weeks after the N treatments for the CH_4 oxidation experiments. The N additions had lowered the oxidation activity, NH_4Cl had the highest inhibition and urea the lowest (Fig. 1). The inhibition was not closely associated with concentration of NH_4^+ in peat. The differences in the inhibitory effects of the N compounds on the CH_4 uptake were similar to their effects on the CH_4 oxidation; NH_4Cl had the highest and urea the lowest inhibition (Fig. 2).
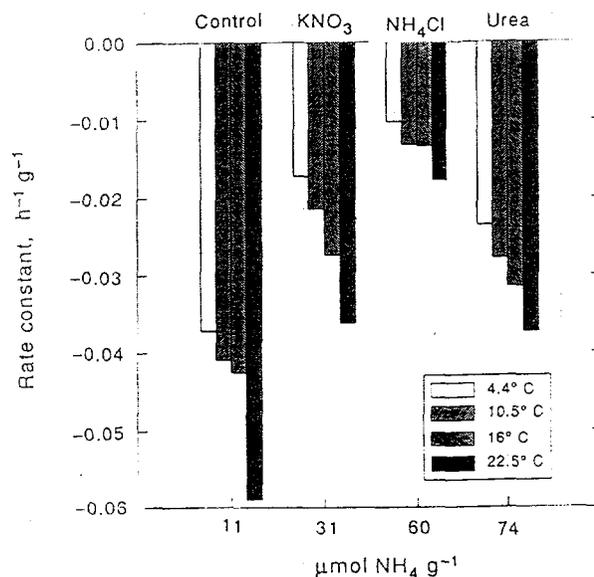
Fig. 1. CH_4 oxidation in soils treated with different N compounds. Samples were taken three weeks after the N additions (Crill et al. 1994). Experiments were done at four temperatures (4.4, 10.5, 16 and 22.5°C). The columns show the first-oder reaction rate constants, the greater the negative value the higher is the oxidation rate. Concentration of NH_4^+ shown below the columns.

The effects from the 2nd to the 4th growth season after the treatments

The microbial and chemical analyses to explain the long-term effects of the N additions on the CH_4 cycle are not yet ready. Here some preliminary observations. In laboratory experiments in 1993 (2nd year after the N treatments) urea had the highest inhibitory effect on the CH_4 oxidation (results not shown). The reducing effect of N compounds on the uptake rates was evident also in 1993; the mean CH_4 uptake rate was lowest in peat treated with urea (Fig. 2). Also NH_4Cl inhibited the CH_4 uptake more than KNO_3 (Fig. 2). The laboratory experiment releaved that the inhibition in CH_4 oxidation in 1993 was merely associated with NO_3^- concentration and nitrification activity, not with the concentration of NH_4^+ in peat (results not shown).

In 1994 (3rd year after the treatments) there still was evidence of the reduction in the CH_4 uptake in the N treated soils. The average uptake was lowest in peat treated with NH_4Cl and urea (Fig. 2). Also in 1995 the average CH_4 uptake was lowest in the N treated soils.
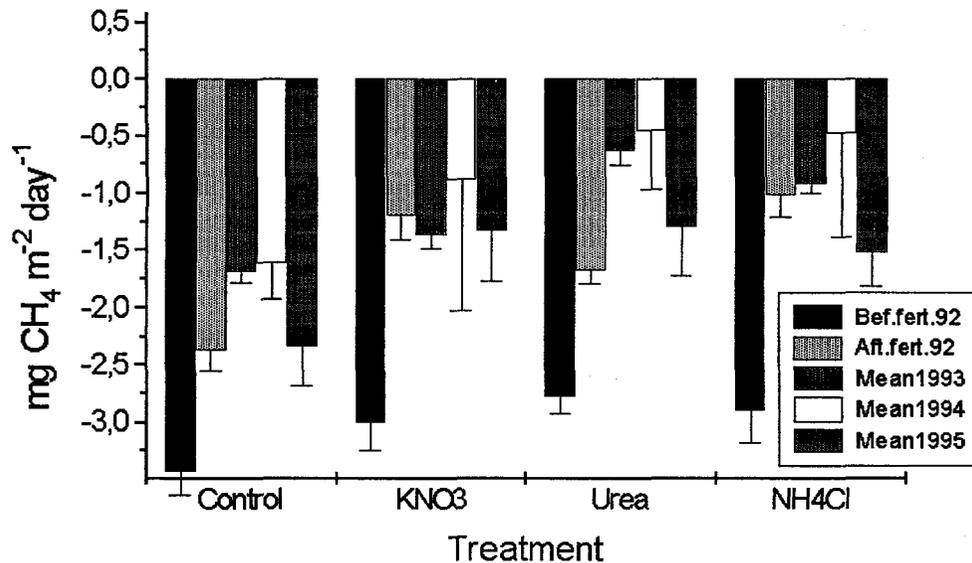
Fig. 2. Uptake of CH_4 *in situ* measured by the chamber technique from 1992 to autumn 1995.

The rates of CH_4 uptake into untreated peat in 1992 were higher than the rates in 1993 and 1994. However, in 1995 the uptake rates into untreated peat were higher than in 1993 and 1994 (Fig. 2). We can expect similar trend to take place also in the N treated soils. Therefore also the relative rates of the CH_4 uptakes were calculated by comparing the rates after the treatments to the rates before the treatments (Table 1). The relative uptake of CH_4 in soils treated with urea or NH_4Cl was weakest in 1994. Only urea showed in 1995 some indication about decrease in the CH_4 uptake. In laboratory experiments there were in autumn 1995 no statistically significant differences in the CH_4 oxidation (results not shown)

Table 1. The percentage rates of the CH_4 uptake after the treatments. The rates were compared to the rates measured from the plots in 1992 before the N additions

Plots	1992	Year 1993	1994	1995
Control	69	49	47	68
KNO_3	40	46	30	45
Urea	61	30	17	47
NH_4Cl	35	32	17	53

The experiments showed that also in peat soils with high nitrogen content, surplus N can reduce oxidation and uptake of CH_4. However, there are differences in the effects of various N compounds. The short-term inibitory effects differed from the long-term effects. For example, urea had a minor short-term inhibition but had the greatest inhibition later on. However, in 3 years the inhibition was almost over also in the urea-treated soil.

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Role of drainage ditches in emissions of methane from mires drained for forestry

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Introduction

Management of peatlands for forestry requires lowering of water table by ditching. It has been shown that the increase in aeration of the uppermost peat profile after lowering of the water table highly changes the fluxes of greenhouse gases from the peat surface. Methane (CH_4) emissions generally decrease whereas those of carbon dioxide (CO_2) (Silvola et al. 1996, Martikainen et al. 1995) and nitrous oxide (N_2O) (Martikainen et al 1993) increase after drainage.

However, the CH_4 -emissions from ditches have so far gained less attention. There is evidence from some forested mires in Canada that the ditches contribute significantly to the total fluxes (Roulet and Moore 1995). Similar observations are from mires used for peat harvesting (Svensson et al. 1995, Nykänen et al. 1995a). We studied the fluxes of methane from drainage ditches and peat surfaces on a forest drainage area in Lakkasuo mire complex, Central Finland.

Methods

Gas fluxes were measured at Lakkasuo mire complex in August and September 1995. The fluxes were measured weekly with modification (Nieminen et al. 1996) of the closed chamber method used for trace gas studies in SUOSILMU (Martikainen et al. 1995, Nykänen et al. 1995b). Six measuring sites were established and the samples were collected from the bottom of the ditch, the side of the ditch and the strip adjacent to the ditch (Fig. 1.).

The sites are described in Table 1. The fluxes from ditches containing water (sites 1 and 2) were measured with floating chambers constructed from the similar ones used for the other surfaces. The release of bubbles from the ditch bottoms was measured with inverted funnels (sites 1, 2, 4 and 5) using a collecting period of one week. The diluting of CH_4 during the week was corrected by comparing the CH_4-consentrations to those of fresh samples. The gas samples were taken with plastic syringes and the samples were analyzed for CH_4 within 24 hours by gas chromatographs equipped with FI- TC- and EC-detectors (Nykänen et al. 1995b).
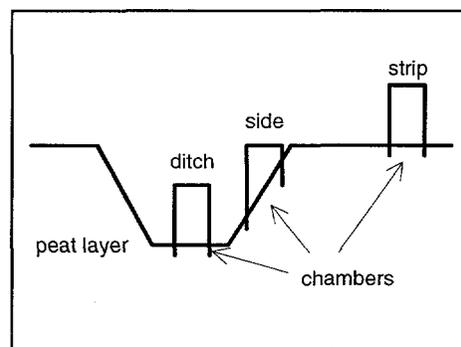
Fig. 1. Location of the chambers on the experimental sites.

Table 1. General description of the study sites.

Site	Trophy	Ditch description	Peat constit.	Vegetation					
				ditch bottom		ditch side		strip (peat surface)	
				mosses	other	mosses	other	mosses	other
1	minero-trophic	old ditch standing water	Carex	-	-	Pol str	-	Ple sch Pol com	-
2	minero-trophic	old ditch moving water	Carex	-	-	-	-	Dic pol	-
3	ombro-minero	new ditch vegetated	Sph-Carex	Sph rip	Car can	Sph rip	-	Sph ang	Rub cha Vac uli Eri vac
4	ombro-trophic	old ditch vegetated	Sph	Sph rip	Eri vag Vac uli Vac oxy	Sph fus Sph rip Pol str	Rub cha Vac uli	Sph ang	Rub cha Vac uli Eri vac Emp nig
5	ombro-(minero-trophic sripe)	old ditch vegetated	Sph	Sph rip	Car ros	Sph ang Pol str	Vac uli Emp nig	Sph fus Ple sch	Eri vac Vac uli And pol Emp nig
6	minero-trophic	old ditch not veget.	Carex	-	-	Sph ang	-	Sph ang	-

Results and discussion

The emissions of CH_4 from ditches, sides and strips ranged between 0 - 595 mg m^{-2} d^{-1}, -7 - 78 mg m^{-2} d^{-1} and -3 - 33 mg m^{-2} d^{-1}, respectively. Temporal variation was great on the minerotrophic sites but on the ombrotrophic sites emissions were rather constant during the measuring period. The fluxes were always highest from the ditch bottoms and decreased remarkably to the ditch sides and strips (Table 2). The release of bubbles was rather small ranging from 3 to 37 mg m^{-2} d^{-1} on sites 1 and 2 (Table 2). However, no ebullition was observed at sites 4 and 5, which were dominated by vascular plants such as *Eriophorum vaginatum* and *Carex rostrata*. These plants are probably functioning as gas conduits transporting CH_4 from the peat to the atmosphere and thus hindering the bubble release (Whiting and Chanton 1992, Schütz et al. 1989).

Table 2. Mean (± s.d.) fluxes of CH_4 from ditches and peat surfaces (strips) and the mean depth of water table below the chamber on the drained experimental sites. The emissions were measured by chambers (plant-mediated transport and diffusion) 7 times and by inverted funnels (ebullition) 5 times during the study period, 15.8 - 27.9 1995.

Site	Ditch bottom			Ditch side		Peat surface (strip)	
	chambers (mg CH_4 m^{-2} d^{-1})	funnels	wt (cm)	chambers (mg m^{-2} d^{-1})	wt (cm)	chambers (mg m^{-2} d^{-1})	wt (cm)
1	205 ±183	25 ±8	0 ±0	27 ±19	18 ±3	0 ±1	50 ±16
2	248 ±165	13 ±8	0 ±0	29 ±16	26 ±2	-1 ±2	59 ±10
3	40 ±8	n.d.	0 ±2	11 ±5	16 ±2	4 ±2	29 ±18
4	64 ±18	0 ±0	6 ±6	5 ±4	22 ±7	13 ±4	39 ±16
5	67 ±17	0 ±0	0 ±2	18 ±15	18 ±3	25 ±5	27 ±16
6	96 ±93	n.d.	0 ±14	13 ±32	24 ±15	0 ±0,5	42 ±23

The CH_4 fluxes from the ditch bottoms are similar as those measured in Ontario, Canada (Roulet & Moore 1995). The source of CH_4 from the ditches was not identified. The sediment of the ditches can produce CH_4 from organic matter leached from peat profile or from the fresh material produced in photosynthesis by plants and algae active in the ditches. It is also probable that the soluble CH_4 in drainage water from strips is released in ditches. In fact, there were evidences that especially after rainfall events this mechanism had importance in the CH_4 emissions from ditches (fig. 2, sites 1 and 2).

The results indicate that the ditches were a considerable source for CH_4. The mean emissions from ditches and strips (Table 2) were similar to the mean emissions measured on the same mire for August - September 1991 (Martikainen et.al 1992).

There the fluxes on undrained minerotrophic and ombrotrophic sites were 215 mg m^{-2} d^{-1} and 60 mg m^{-2} d^{-1}, respectively. The corresponding drained sites had fluxes of -0,4 mg m^{-2} d^{-1} and 28 mg m^{-2} d^{-1}.

The role of ditches on the methane emissions from drained mires can be calculated roughly using the mean values from this study for sites 1-2 and 4-5. If the width of the ditch bottom is 0.5 m, the width of ditch sides is 0.5 m (*2) and the space between ditches is 35 meters, the emissions from the ditches calculated for the whole drained area would be following:

Sites 1 and 2:

(0.5 m * 246 mg m^{-2} d^{-1} + 1.0 m * 28 mg m^{-2} d^{-1}) / 35 m

= 4.3 mg m^{-2} d^{-1} .

Sites 4 and 5:

(0.5 m * 66 mg m^{-2} d^{-1} + 1.0 m * 12 mg m^{-2} d^{-1}) / 35 m

=1.3 mg m^{-2} d^{-1}

In the total areal fluxes the importance of the CH_4 released from ditches was much less than the importance of the CH_4 emitted from the undrained peat surface. If the CH_4 fluxes from peat surface of undrained mires was low, as in some drained mires in Canada (Roulet & Moore 1995), the significance of ditches to change the CH_4 balance after drainage would be greater.

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Trace gas fluxes from northern peatlands

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Introduction

Peatlands cover large areas in northern environments: 1.1, 0.1 and 1.7 x 10^4 km^2 in Canada, Finland and the former Soviet Union, respectively (Botch et al. 1995; National Wetlands Working Group 1988). Interest has been generated into the role these extensive areas of peatlands play in controlling the chemistry of the atmosphere. In particular, it has become established that peatlands can be a source of methane (CH_4) and nitrous oxide (N_2O), and a sink of carbon dioxide (CO_2), the latter through the rates of plant production exceeding the rate of decomposition of plant material and peat.

In the past decade, considerable progress has been made in developing techniques to measure the flux of these gases from peatlands, to quantify the fluxes and to identify the important controls on these fluxes. Now, there is a wealth of data available, compared to five or ten years ago. In evaluating these data, however, one must be aware of their limitations. Gas flux measurements from peatlands are often made over short periods of time (less than 1 hour per week in most cases using chamber techniques) and over small areas (commonly < 0.5 m^2 for chambers, < 100000 m^2 for tower-based measurements). Thus, there is a great need to be able to extrapolate measurements of trace gas flux from points in time and space into estimates over longer times and whole landscapes.

Furthermore, it must be realised that this interest in trace gas fluxes has been fed by funds from government agencies which are concerned not only with the present role of peatlands, but what happens to these fluxes under different management scenarios, such as shallow flooding or drainage, as well as how the fluxes are likely to change under scenarios of climatic change. Thus, models of trace gas flux need to be able to accommodate the needs of agencies and be amenable to policy decision options.

The need for extrapolation spatially and temporally as well as to a variety of change scenarios means that we must, in our studies of trace gas flux, look for simple variables which appear to control the magnitude of the flux, which can be applied at the scales of the landscape and different management scenarios. In this paper, I examine recent advances in trace gas flux measurements in northern peatlands and the extent to which we can relate variations in these fluxes to simple environmental parameters that can be used in extrapolation and management scenarios.

Nitrous oxide

Nitrous oxide release from soils is tightly tied to the nitrogen cycle, and particularly to the rates of nitrification and denitrification (Davidson 1991). Relatively few measurements of N_2O flux from peatlands have been made in northern peatlands, and generally show small fluxes when the water table is very close to the surface, and when the peat is of an ombrotrophic origin, in which rates of nitrification are slow. However, when the peatland water table is lowered, especially associated with drainage, then substantial increases in N_2O flux have been observed in both field (e.g. Martikainen et al. 1993; Nykänen et al. 1995; Duxbury et al. 1982) and laboratory (e.g. Freeman et al. 1993) conditions, though fluxes are generally small, a few mg m^{-2} d^{-1}. In terms of developing a simple predictive model of N_2O flux from peatland soils and response to drainage, including position of the water table and N status of the peat may be effective, as suggested in Fig. 1, though other factors, such as the presence of vegetation (e.g. Nykänen et al. 1995) may also be important in controlling fluxes.
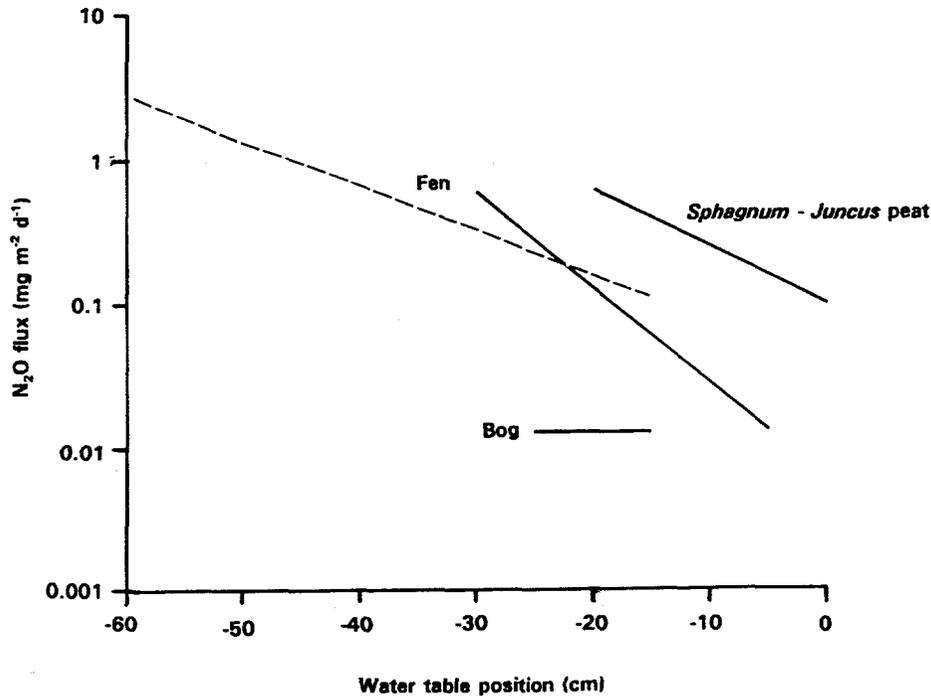
Fig. 1 Relationship of N_2O flux from peatland soils to position of the water table and characteristics of the peat. Derived from Martikainen et al. (1995), solid line, and Nykänen et al. (1995), dotted line, for virgin and drained fen sites in Finland and Freeman et al. (1993), solid line, for laboratory incubations of *Sphagnum - Juncus* peat cores.

Carbon dioxide

The CO_2 balance of a peatland will be derived from the rates of photosynthetic uptake of CO_2 and the emission of CO_2 from the combination of root respiration and peat and plant tissue decomposition. Although field measurements of CO_2 flux are bring made with clear static chambers, much remains to be done in generating general relationships between CO_2 uptake and photosynthetically active radiation (PAR), temperature and plant community. Synthesis of short-term measurements of CO_2 exchange (NEE) into annual estimates of the C budget of peatlands is beginning to reach values close to the long-term C accumulation rates, but is hindered by the difficulties in making flux meaurements year-round and adequately incorporating measurements during the winter and in spring and fall, when there may be substantial gains and losses, respectively, of CO_2.

More is known about the flux of CO_2 from peatland soils, of which about half may be derived from root respiration (Silvola et al. submitted ms.). Rates of CO_2 production from a wide range of peat samples in laboratory incubations cover one order of magnitude, and aerobic rates are greater than anaerobic rates, though the ratio varies with peat type and with duration of incubation (Moore & Dalva submitted ms.). The relationship of CO_2 to peat temperature is well established, with Q_{10} values ranging from 2 to 4 (Silvola et al. submitted ms.; Moore & Dalva 1993). Lowering the water table increases CO_2 emission rates, as shown by both laboratory (e.g. Moore & Dalva 1993) and field (Silvola et al. submitted ms.) measurements (Fig. 2), in a consistent fashion, suggesting that water table position may be a useful variable for predicting CO_2 emission rates and for predicting the effect of drainage.
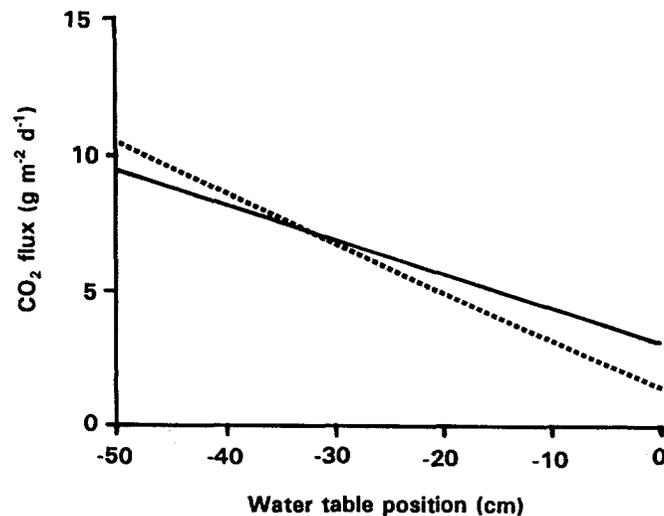
Fig. 2 Relationship between CO_2 emission and water table position for field peatland soils, dotted line, (Silvola et al. submitted ms.) and for a column of fen peat, solid line, incubated in the laboratory (Moore & Dalva 1993). Note the similarity in response to water table. The warmer laboratory conditions and absence of roots partially account for the observed differences in flux.

Methane

The emission of CH_4 from peatland soils is a function of the balance between rates of methanogenesis and methane consumption in the peat profile and the mechanisms for CH_4 movement to the surface. Vascular plants can promote CH_4 flux, either through the production of root exudates used as substrates for methanogenesis, or by venting CH_4 to the atmosphere, by-passing oxidation pathways. Rates of CH_4 production and CH_4 consumption by peat samples under anaerobic and aerobic laboratory incubations cover four orders of magnitude, indicating the extreme variability in these microbially-mediated processes in peatlands (Moore & Dalva submitted ms.). Not surprisingly, annual emissions of CH_4 range from 0 to 100 g m^{-2} yr^{-1} (Bubier & Moore 1994), with drier peatlands exhibiting net CH_4 consumption of up to 0.1 g m^{-2} yr^{-1}. There is often great variability in CH_4 flux within ecological units as well as high spatial variability within an apparently uniform site. Bubier et al. (1995) have shown that plant species, especially bryophytes, can be used a good predictors of CH_4 flux, because their distribution is controlled primarily by water table depth; plants which also distribute according to trophic status are less valuable indicators because differences in peat water chemistry play little role in controlling CH_4 emissions in field conditions.

Where the water table remains close to the peat surface, the seasonal emission of CH_4 often follows the cycle of temperature in the peat profile, but where the water table fluctuates, a hysteresis effect prevents a clear relationship between daily CH_4 flux and water table position (Moore & Dalva 1993). Comparison of CH_4 flux data collected from various regions in Canada suggests that there is a consistent relationship between mean position of the water table during the summer and the mean CH_4 flux (Moore & Roulet 1993). The slope of the regressions are very consistent, but the intercept varies among regions. Comparison of CH_4 data collected from peatlands in Finland suggests that this relationship may be generally valid for northern peatlands (Fig. 3).

Attempts have been made to estimate the contribution of peatlands to the global CH_4 budget, most notably the Hudson Bay Lowland, which covers 0.3 x 10^6 km^2, almost entirely peatland. Using field measurements and remote sensing, an estimate of 0.3 Tg yr^{-1} has been derived (Roulet et al. 1994). Using field measurements from several locations and maps of peatland area and peatland type in Canada, Moore & Roulet (pers. comm.) estimate the Canadian peatland contribution at 3.5 Tg CH_4 yr^{-1}. The recent estimate of peatland area in the former Soviet Union (Botch et al. 1995), with average water table depths in each category, suggests an annual emission of between 4 and 6 Tg CH_4 yr^{-1} from the vast (1.65 x 10^6 km^2) peatlands of Russia and Siberia.

The importance of water table position, through its control on anaerobic methanogenesis and aerobic CH_4 consumption, means that a lowering of the water table, either through climatic change or drainage, has a profound effect on CH_4 fluxes. CH_4 fluxes have decreased to almost zero after drainage of mires for horticultural crops (Glenn et al. 1993) and forestry (Martikainen et al. 1995; Roulet et al. 1993). In the drainage exercise, however, it must be recalled that the drainage ditches may contain water and may flux CH_4 at relatively high rates. Thus, the net effect of drainage on CH_4 emissions from a peatland will depend on the size and spacing of the drainage ditches and their CH_4 flux, compared to the drained section.

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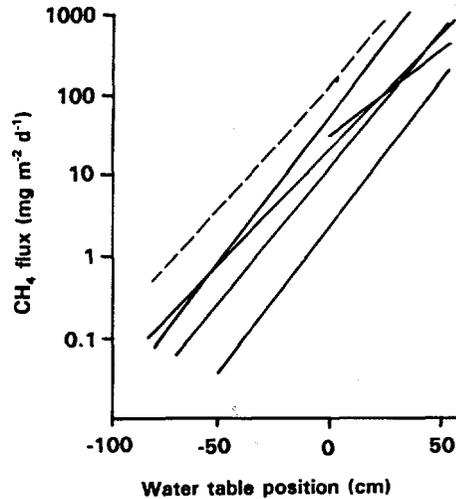
Fig. 3 Relationship between the mean CH$_4$ flux and mean water table position during the summer from various regions in Canada, solid lines, (Moore & Roulet 1993; Bubier et al., in press), with data from Finland, dotted line (Martikainen pers. comm.).
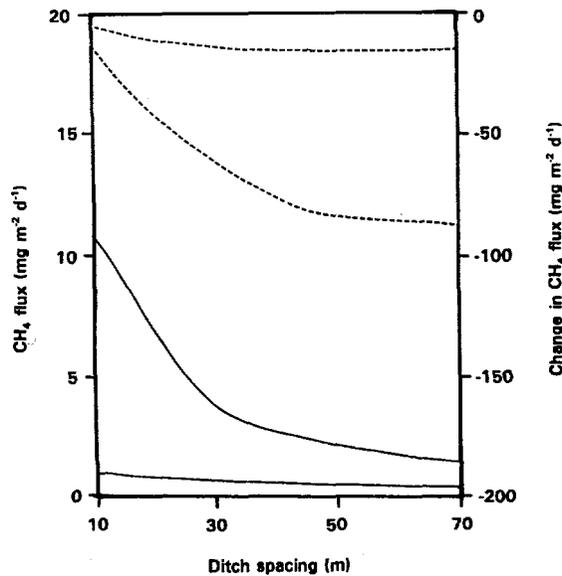
Fig. 4 Predicted overall CH$_4$ emission and change in CH$_4$ emission from the Lukkasuo peatland complex associated with ditch draining. Assumptions are a range of CH$_4$ fluxes from the ombrotrophic (dotted line) to minerotrophic (solid line) sections of the mire before (25 and 200 mg CH$_4$ m^{-2} d^{-1}) and after (0 and 10 mg CH$_4$ m^{-2} d^{-1}) drainage and an estimated ditch emission of 70 mg CH$_4$ m^{-2} d^{-1} with a ditch width of 1.5 m (Nykänen pers. comm.). Based on Roulet & Moore (1995) and Martikainen et al. (1995).

For the Lukkasuo drained mire, for example, the effect of drainage on the reduction in CH_4 emission varies from the minerotrophic to ombrotrophic sections (Fig. 4). A ditch spacing of 30 m on the minerotrophic section has reduced its overall CH_4 flux from an undrained value of about 200 mg m^{-2} d^{-1} to about 4 mg m^{-2} d^{-1}; in the ombrotrophic section, the reduction has been from about 25 mg m^{-2} d^{-1} to about 13 mg m^{-2} d^{-1}.

Conclusion

Over the past five years, much has been learned about the flux of N_2O, CO_2 and CH_4 from peatlands, their role in the global atmospheric budgets of these gases and the processes and factors controlling these emissions. Comparisons of data from different regions, but with similar peatland types, suggests that there are common relationships between simply observed variables, such as water table position, and the flux of these gases. In the next decade, the challenge will lie, not so much in making further field or laboratory measurements, but rather developing predictive models of broad applicability to extrapolate measurements spatially and temporally and to incorporate the effect of anthropogenic disturbances, such as drainage or flooding, or climatic change on these fluxes.

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Changes in fluxes of N_2O and CH_4 after clearfelling of drained spruce mires in southern Finland

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Introduction

Nitrous oxide (N_2O) emissions are generally low on undrained peatlands with high water table (Martikainen et al. 1993; Regina et al., submitted, Regina et al., this volume) whereas fluxes of methane (CH_4) can be high (Bartlett & Harris 1993). Draining of boreal peatlands for forestry substantially changes the hydrological conditions and the processes responsible for N_2O and CH_4 fluxes. In general, the emissions of CH_4 are highly reduced (Martikainen et al. 1995) whereas the emissions of N_2O can rise after lowering of water table (Martikainen et al. 1993; Regina et al., submitted). Reduction in CH_4 release after drainage is due to better aeration of the uppermost peat profiles. The reduction in the CH_4 release in aerobic peat profile is a sum of two factors: 1) aerobic environment inhibits the production of CH_4, 2) there is increase in the oxidation of CH_4. Increase in N_2O production in drained mires results from an increase in nitrification and denitrification activities (Regina et al., submitted).

Clearfelling of drained and forested mires changes hydrological conditions. Availability of mineral nitrogen to micro-organisms increases, as well. These changes may affect the microbial processes responsible for the trace gas fluxes. We studied the effects of clearfelling on the fluxes of N_2O and CH_4 in two experimental areas in southern Finland.

Materials and methods

The experimental areas were located in southern Finland in Ruotsinkylä (60°21'N, 25°03'E) and Vesijako (61°23'N, 25°03'E). Both experiments had a control site (drained and forest-covered) and a site which was clearfelled in the winter of 1994. The site characteristics are shown in Table 1.

In the summer of 1994 the fluxes of N_2O and CH_4 were measured during one day in June, July, August and October. In 1995 the sites were measured in June, July and August. A modification of the method used in Finland for the fluxes on peatlands was applied (Martikainen et al. 1993; Martikainen et al. 1995; Nykänen et al., 1995). In the application metal cylinders (volume 30 l, height 30 cm, diameter 30 cm) were directly installed to soil for the measurement period of 35 min. The use of collars could be omitted because the water tables of the sites were generally low (no problems with bubbling). Each site had two parallel chambers. Four samples (5, 15, 25, 35 min) were collected from the chambers with plastic syringes and sent to Kuopio (National Public Health Institute) for the gas analyses (see Nykänen et al. 1995).

Table 1. Characteristics of the study sites

	Ruotsinkylä		Vesijako	
	Control	Clear-cut	Control	Clear-cut
Peatland area (ha)	1.4	1.9	4.9	2.5
Site type[1]	MTkg	RhTkg	MTkg	MTkg
Peat depth	0.5	1.0	1.0	0.5
Drainage year	1932	1927	1914	1914
Stand volume (m^3 ha^{-1})	250	234	249	259
Picea abies (%)	100	100	85	90
Pinus sylvestris (%)	0	0	5	6
Betula spp. (%)	0	0	5	6
Peat layer (0–20 cm)				
N (%)	1.4	1.9	2.2	2.0
P (mg kg^{-1})	620	950	1000	800
K (mg kg^{-1})	400	350	400	390
Ca (mg kg^{-1})	4300	4800	6300	8900
Mg (mg kg^{-1})	440	510	530	870

[1]Types according to Heikurainen and Pakarinen (1982).
RhTkg: Original peatland of this type has high fertility. Spruce dominated. Ground vegetation consists of ferns, shrubs and herbs (e.g *Rubus idaeus, Oxalis acetosella, Athyrium filix-femina*).
MTKg: moderate fertile. Spruce generally the dominant tree species. Typical ground and bottom layer vegetation: *Vaccinium myrtillus, V. vitis-idaea, Hylocomium splendens, Melampyrum spp., Trientalis europaea, Orthilia secunda.*

The studies on N_2O and CH_4 were part of a wider nitrogen cycling study. Data on the movement of NO_3^- and NH_4^+ in percolation water and run–off water is also available (Nieminen, unpubl.). These results can be used as background data for the trace gas studies.

Results and discussion

N_2O fluxes

In Ruotsinkylä the emissions of N_2O were considerably higher in the clear–cut site than in the control area both in 1994 and 1995. The lower N_2O emission from the control site might be partly due to low nitrogen content in peat (Table 1). However, the emissions of N_2O were higher in the clear–cut area than in the control area in Vesijako, as well. There was also an increasing trend in the fluxes of N_2O in both clear–cut sites from the year 1994 to 1995. This trend was less evident in the control sites (Fig. 1). Thus, the results indicate an increase in the fluxes of N_2O after clearfelling. Leaching of NO_3^- considerably increased in both clear–cut areas after the first growing season since performing the cuttings (Nieminen, unpubl.). This may be an evidence of increased nitrate production after the cuttings. According to Regina et al. (submitted) nitrate production is an important factor controlling the emissions of N_2O from boreal mires. There was also a substantial increase in the NH_4^+ contents of percolation water in both clear–cut areas. The increase in availability of ammonium in peat can be expected to favor nitrification activity.

The fluxes of N_2O on the control sites in 1994 ranged from –40 to 350 ug N_2O m^{-2} d^{-1}. On the clear–cut sites the respective fluxes in 1994 were from 2 to 3800 ug N_2O m^{-2} d^{-1}. In 1995 the control sites had fluxes from –10 to 350 ug N_2O m^{-2} d^{-1}, and on the clear-cut sites they were from 81 to 2600 ug N_2O m^{-2} d^{-1}. The N_2O fluxes for the control sites were in the range generally found for drained and forested minerotrophic mires (Martikainen et al. 1993; Regina et al., submitted). However, the N_2O fluxes on clear–cut site in 1995 in Ruotsinkylä exceeded these flux rates.

CH_4 fluxes

The sites were generally sinks for atmospheric CH_4. However, in 1994 the clear–cut site in Ruotsinkylä was a weak source for CH_4 (Fig. 1b). In Ruotsinkylä the uptake rates of CH_4 on the control site were higher than those on the clear–cut site. The same was also true for Vesijako in 1995 but not in 1994 (Fig. 1b). The effect of clearfelling on microbial CH_4 oxidation is poorly known. After clearfelling water table is risen (e.g. Heikurainen and Päivänen 1970), which can decrease the oxidation of CH_4 thus leading even to a release of it. Mineral nitrogen has been reported to inhibit CH_4 oxidation in peat profile of drained and forested mire (Crill et al. 1994, Nykänen et al., this volume). The higher availability of mineral nitrogen, especially that of ammonium in peat profiles of the clear–cut sites (see above), could be associated with the lower uptake of CH_4 into these profiles
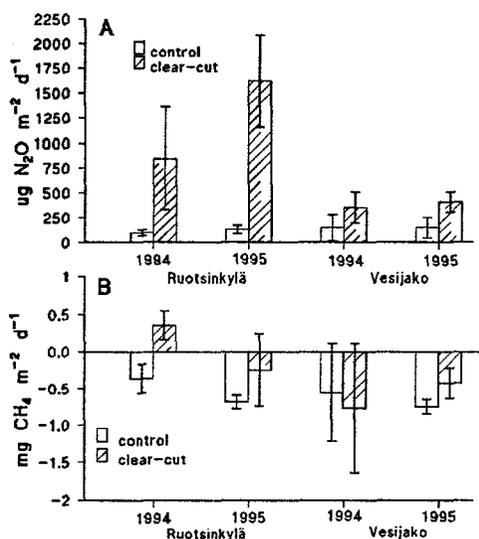
Fig. 1. Fluxes of N_2O (a) and CH_4 (b) in the experimental mires in Ruotsinkylä and Vesijako.

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Fluxes of methane on boreal mires with different hydrology and fertility in Finland

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Introduction

Emissions of CH_4 from peat depends on the amount of CH_4 produced in anaerobic peat and on the CH_4 oxidation in aerobic layers of the uppermost peat profile (Sundh et al. 1994). The oxygen status in peat profile depends on the level of water table. Therefore, water table has a great importance to moderate CH_4 fluxes (e.g. Moore & Knowles, 1989, 1990, Martikainen et al. 1995). Hydrology together with nutrient level determines the vegetation type on mires which also is associated to the CH_4 release (Whiting and Chanton 1993, Bubier et al. 1995). Production and oxidation of CH_4 also are influenced by the prevailing temperature conditions (Crill et al. 1988, Dise et al. 1993).

Most of the data on CH_4 release from northern peatlands is from North America (latitudes from 45°N to 60°N) (Crill et al. 1992). In northern Europe there is data on CH_4 from northern Sweden (Svensson & Rosswall 1984) but not much from the southern and middle boreal zone. Due to the Gulf Stream the mean annual temperature in Finland is higher than the mean temperature in respective latitudes in North America. The differences in climatic characteristics, peatland types and their distribution between Finland and North America probably cause differences also in their CH_4 fluxes.

Anthropogenic activities have changed the biogeochemistry of wetland ecosystems. This is also true for the northern mires, especially those in Finland. In Finland 6.1 – 6.9 Mha of the original 10.4 Mha peatlands have been drained during this century, from drainages 5.4 Mha are for forestry (Arne 1992, Laine & Päivänen 1992, Lappalainen 1995).

We give here a short summary of the studies whose objectives were: (1) To measure fluxes of CH_4 from wide range of natural mires in Finland. (2) To study the long-term changes in CH_4 fluxes on boreal peatlands after lowering their water tables. These

measurements were done on mires drained for forestry 30–40 years ago. The results from the drained mires may reflect the possible changes in the CH_4 fluxes if water table will drop in the northern peatlands as a result of drier climate (Manabe & Wetherald 1986, Mitchell 1989). These results also show some effects of land–use activities (drainage) on biogeochemistry of boreal mires.

Methods

The study sites

The mires are situated on the transition region of the southern and middle boreal peatlands in Finland (Orivesi: 61°47'N, 24°18'E; Ilomantsi: 62°46'N, 29°50'E) and represent typical mire types in the region. Their classification was made according to the scheme used in Finland (Heikurainen & Pakarinen 1982, Ruuhijärvi 1983). The mires are grouped based on their moisture and nutrient level (Fig. 1) as described earlier by Silvola et al. (in press). There were both natural mires and mires drained for forestry 30–40 years ago. Many of the mires has both natural and drained counterparts (Fig. 1).

The flux measurements

All sites had boardwalks to minimize disturbances of the ground when walking on the sites. Fluxes of CH_4 were measured by a closed chamber method (Crill et al. 1988) during years 1991, 1992 and 1993. There were aluminum collars in soil equipped with water grooves for gas tight connection of the chamber during the measurement period. For water table measurements perforated pipes were inserted to soil close to the chambers. Samples (50 ml) from headspace of chambers were withdrawn to plastic syringes stopped with 3–way stop cocks (PD). Gas concentrations were analyzed within 24 hours after collection in Kuopio (Department of Environmental Microbiology) with Hewlett Packard 5890 gas chromatograph having TC– and FI–detectors in series (Nykänen et al. 1995). The samples in Ilomantsi in 1992 were analyzed with a Shimadzu 14A GC with FID. Fluxes were calculated from the linear increase or decrease in gas concentration of the 4–5 samples taken during the measuring period of 30 min. The annual emissions were calculated by summing the monthly mean emissions including the fluxes during winter.

Results

Water table height and the CH_4 fluxes

The drained mires were grouped based on their original hydrological and nutritional characteristics. Fluxes of CH_4 were highly depended on the height of water table and on soil temperature (Martikainen et al. 1995). Here the effect of temperature on the dynamics of CH_4 fluxes was considered to be similar in all mires and was omitted from

g CH$_4$–C m^{-2} a^{-1}
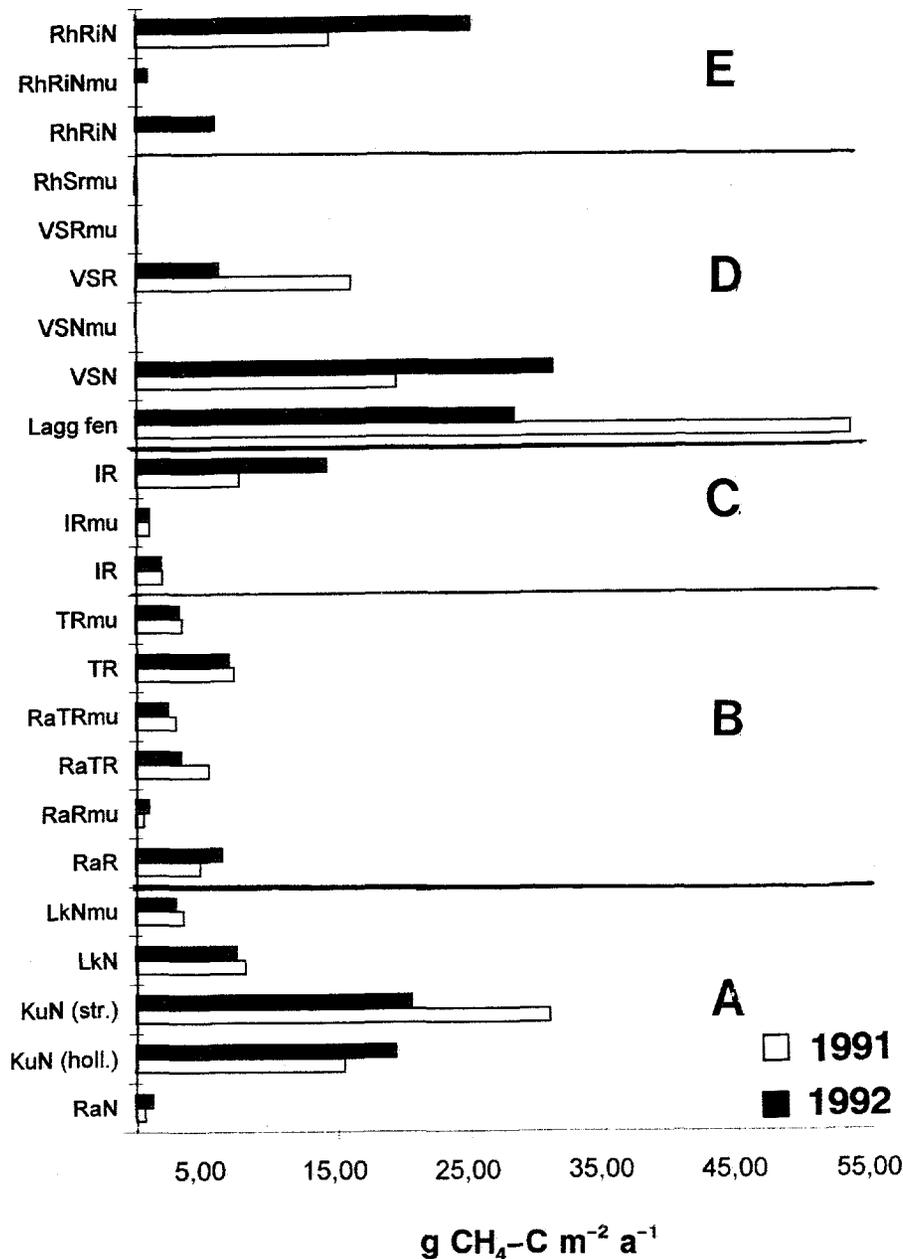
Figure 1. Annual fluxes of CH$_4$ on peatlands in 1991 and 1992.

Group A, Ombrogenous bogs: *Sphagnum fuscum* bog (RaN); Ridge–hollow bog (hollow and string) (KuN); Low–sedge bog, natural (LkN), and drained (LkNmu).

Group B, Ombrogenous pine bogs: *Sp. fuscum* pine bog, natural (RaR), anddrained (RaRmu), Cottongrass pine bog, natural (RaTR), and drained (RaTRmu); Cottongrass pine bog, natural (TR), and drained (TRmu).

Group C, Dwarf shrub pine bogs: Dwarf shrub pine bog, natural (IR), and drained (IRmu).

Group D, Minerogenous oligotrophic and oligo–mesotrophic fens and treed fens:Lagg fen; Tall sedge fen, natural (VSN), and drained (VSNmu); Tall sedge pine fen, natural (VSR), and drained (VSRmu); Herb–rich sedge pirch–pine fen, drained (RhSrmu).

Group E, Minerogenous mesotrophic fens: Herb–rich flark fen, natural (RhRiN), and drained (RhRiNmu); Herb–rich flark fen natural (RhRiN).

the calculations. Fig. 2 shows the regression between the mean methane fluxes and mean water table during summers 1991 and 1992. The regression models were calculated separately for the natural and drained mires:

Natural mires: $y = 0.032 \times WT + 2.26$ $R^2 = 0.49$

Drained mires: $y = 0.025 \times WT + 1.68$ $R^2 = 0.46$

where y is the ^{10}log–transformation of the mean CH_4 flux rates (mg CH_4 m^{-2} d^{-1}), and WT the mean height (cm) of water table (the values below the soil surface are negative).

The regression equations for the natural and drained mires were different. The slope for the natural mires was steeper and the intercept greater than those for the drained mires, i.e the response to the level of water table was greater in the natural mires. For example, at water table level of –10 cm the predicted emissions for natural and drained mires are 84 and 24 mg CH_4 m^{-2} d^{-1} agreeing with the measured values. Thus, the regression model constructed for natural mires would greatly overestimate the CH_4 fluxes from drained mires.
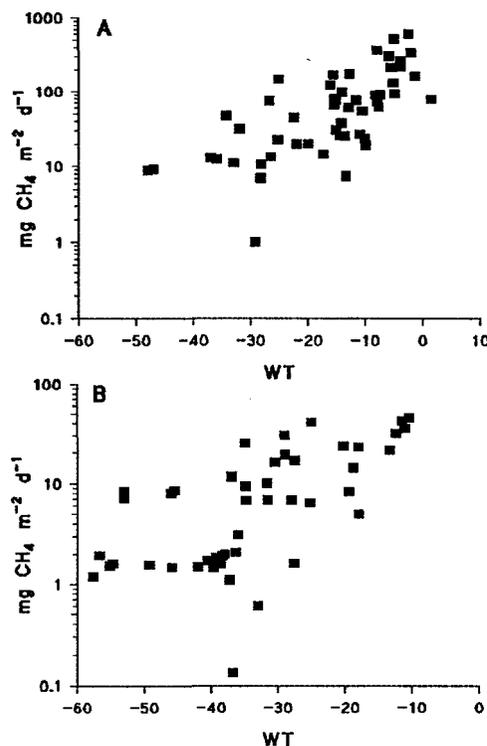
Figure 2. Mean CH_4 fluxes (mg CH_4 m^{-2} d^{-1}) versus mean water table from June to September in 1991 and 1992. Natural mires (A), drained mires (B).

Annual fluxes of CH_4

There were differences in the CH_4 flux rates between years 1991 and 1992. Summer 1992 was warm lowering water table in many mires. This reduced the annual CH_4 flux rates (Fig. 1). However, there were mires retaining high water table and high CH_4 fluxes also in 1992 (KuN hollow, RaR, IR at Korvinsuo, VSN and RhRiN). The early summer (May–June) in 1992 was warm and e.g. the tall sedge fen (VSN) in the group D with high water table had much higher fluxes in 1992 than in 1991 (Fig. 1).

Among the natural mires the annual CH_4 emissions varied greatly in individual mires, also in mires in a group (A–E) (Fig. 1). The annual flux rates from natural mires belonging to the ombrogenous pine bogs (group B) and to dwarf shrub pine bogs (group C) were generally low, generally below 10 g CH_4–C m^{-2} a^{-1}. Among the mires in the group A, ombrogenous bogs, there also were mires having low annual CH_4 emissions but also sites with higher CH_4 emissions (KuN, fluxes up to 31 g CH_4–C m^{-2} a^{-1}, Fig. 1). The highest annual flux rates were measured from the minerogenous oligotrophic and oligo–mesotrophic fens and treed fens (group D), there a wet lagg site produced in 1991 54 g CH_4–C m^{-2}.

Drainage decreased generally the annual CH_4 emissions (Fig. 1). Water tables were lowered more at the minerotrophic sites than at the ombrogenous ones. In drained fens (VSNmu, VSRmu, RhSrmu) the fluxes varied from a weak uptake of 0.06 g CH_4–C m^{-2} a^{-1} to a small emission of 0.2 g CH_4–C m^{-2} a^{-1}. At drained bogs (LkNmu, RaRmu, RaTrmu, TRmu, IRmu) the emissions decreased generally to about half of the emissions from their natural counterparts.

Discussion

The highest methane emissions were found from some wet, nutrient rich fens but also some ombrotrophic mires showed high methane emissions from the wet surfaces they also had (Fig. 1). Methane emissions from the mires correlated rather well with the level of water table irrespective of the fertility of the site (Fig. 1). Correlation between methane fluxes and water table have also been reported for peatlands in other regions of the world as well (see Moore, this volume). However, the effect of water table on the CH_4 release from natural and drained mires was different. There are several possible reasons why the response to water table height was steeper in the natural mires:(1) The mean depth of water tables of the drained mires was lower than that of the natural mires. Therefore, many mires with low water table were included to the regression model for the drained mires. It would well be that drainage decreases the substrate availability for methanogenesis in deeper anaerobic peat layers because in aerobic uppermost peat layers the fresh organic material derived from vegetation is decomposed (Hogg et al. 1992). Furthermore, the uppermost generally aerobic peat profile of drained mires probably have low capacity to produce CH_4 even after a temporal rise in water table. (2) Not only substrate availability for methanogenesis but also changes in microbial populations involving production and oxidation of CH_4 would decrease the net release of CH_4 from drained mires. Anaerobic microbes, including the methanogens, in the

uppermost temporary aerobic peat layers are expected to suffer. (3) The changes in vegetation cover after drainage would change the amount and/or quality of substrates derived from plants. Furthermore, some vascular plants, such as sedges, have internal gas spaces where CH_4 can be transported from peat to atmosphere. This mechanism is more efficient than diffusion of CH_4 from soil (e.g. Torn & Chapin 1993, Schimel 1995). Therefore, the reduction in coverage of such vascular plants after drainage decreases the transportation rate of CH_4 from peat to air. (4) Bulk density of the uppermost peat layers are increased by drainage (especially in fens) as a result of peat compaction lowering the diffusion rate of CH_4 produced in deeper anaerobic peat layers. This may enhance the capacity of the upper peat layers to oxidize CH_4 before it escapes to atmosphere. In fact, the highest potential for CH_4 oxidation in the peat profile of a drained fen occurred in the uppermost 5 cm peat layer (Crill et al. 1994).

Methane emissions from northern peatlands have been studied in North America (Roulet et al. 1992, 1994, Bubier & Moore 1993, Dise et al. 1993), in Sweden (Svensson & Roswall 1984) and some data is also available in Russia (Panikov 1994, Panikov & Dedysh 1995). Annual methane emissions of 3.4 to 70 g CH_4 m^{-2} a^{-1} for peatlands (Dise et al. 1993, Crill et al. 1992) in Minnesota (47°N) are in the range we found for the transition region of the southern and middle boreal zone in Finland. In Canada CH_4 emissions from Hudson Bay lowland varied from 0.2 to 14 g CH_4 m^{-2} a^{-1} (Roulet et al. 1994), and the emissions from peatlands in northern Ontario from 0.1 to 10 g CH_4 m^{-2} a^{-1} (Bubier & Moore 1993). According to Moore (1994) the average annual emission from Canadian peatlands could be 2 g CH_4 m^{-2} a^{-1}. Thus, the annual methane release per m^2 from mires in southern and middle boreal zone in Finland generally is higher than the methane release from Canadian peatlands. There are not much published data on methane fluxes from wetlands in Siberia. There are reports from Russia (Panikov 1994, Panikov and Dedysh 1995) on high methane fluxes (–20 to 2400 mg CH_4 m^{-2} d^{-1}, mean 234 mg CH_4 m^{-2} d^{-1}) from some West Siberian lowlands suggesting that these wetlands may have greater importance in the total release of CH_4 from northern wetlands than generally estimated. There are in Siberia extensive peatlands with wet surfaces (aapa mires, raised string bogs) (Botch et al. 1995) as there are in northern Finland as well. Our measurements in northern Finland indicate (unpublished results) that the aapa mire region continuing east from Finland to Russia has high CH_4 emissions.

Some climatic models predict drier summers following increasing temperature in the boreal region (Manabe & Wetherald 1986, Mitchell 1989). This would lower water tables of the northern peatlands and change their trace gas fluxes. The present results together with results from other peatlands (Sebacher et al. 1986, Bubier et al. 1993, Roulet et al. 1993, Shanon and White 1994) suggest that water table and temperature are among the most important factors regulating release of CH_4 from peatlands. If the northern peatlands are drying with the climate change, the changes in the CH_4 fluxes will greatly differ from the changes in the fluxes of CO_2 and N_2O; the emissions of CH_4 would lower whereas those of CO_2 (Silvola et al., in press) and N_2O (Martikainen et al., 1993, Regina et al., submitted, Regina et al., this volume) would increase. There are several uncertainties, not discussed here in details, associated to the climate change and biogeochemistry of northern peatlands. In any case, the use of data from drained mires to predict gas dynamics in changing climate is problematic. The possible long–term changes in gas fluxes are associated to the temperature, hydrology and development

of mire vegetation. With increasing temperature emissions of CH_4 on wet mires are expected to increase, not to decrease, if mires retain their high water tables. There can be long-term changes in mire vegetation if water tables will lower; sedge-dominated fens would change to *Sphagnum*-dominated bogs (Bubier & Moore 1994) greatly affecting the gas dynamics.

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Fluxes of CH_4, N_2O and CO_2 on virgin and farmed peatlands in Finland

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Introduction

Nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) are important constituents in the chemistry of the atmosphere and in the radiation budget of the earth. N_2O also participates in the reactions causing ozone depletion in stratosphere (Cicerone 1987). Atmospheric concentrations of N_2O, CH_4 and CO_2 have increased during industrial era (Dibb et al. 1994). The annual increase in concentrations of N_2O, CH_4 and CO_2 is about 0.3, 1, and 0.4%, respectively (WMO, 1985).

10 % of the worlds' land area ($1500*10^6$ ha) has been taken to agricultural use (Houghton et al. 1983). The changes from natural soil to managed agricultural soil affect the fluxes of greenhouse gases. Also peatlands have been managed for agriculture. Cultivation activities on peatlands affect the gas balances even more than the cultication practizes on mineral soils. The originally water-saturated, anaerobic peat has to be drained which together with tilling and fertilization have long-lasting effects on the trace gas fluxes and the stores of carbon and nitrogen in peat. Furthermore, drained peatlands are often used to produce fodder for ruminants which are important sources for CH_4 (Grutzen et al. 1986).

During this century more than half of the original Finnish peatlands of $10*10^6$ ha has been drained for forestry or agricultural use. From the peatlands less than half of the $0.7*10^6$ drained for agriculture is still in agriculture. We studied the fluxes of N_2O, CH_4 and CO_2 from a cultivated peatland in eastern Finland and compared these fluxes with those from a corresponding natural peatland site. Also the CH_4-release from cattle consuming the hay grown on the study field is included to the calculations. The atmospheric impacts of the agricultural activity and the natural mire were compared by using the Global Warming Potential (GWP) approach (IPCC 1994).

Materials and methods

Measurements were conducted at a field in Ilomantsi in eastern Finland (62°40'N, 30°50'E) during years 1991 and 1992. The study area has annual precipitation of about 650 mm. Snow free period lasts 190–220 days and the sum of effective temperature is ca. 1150 D.D. The original site type of the peat field was a wet flark fen. The field was drained 60 years ago. The depth of the peat was originally more than 2 m and is still 1.4 m. Soil pH was 5.3 and contents of total N 2.55 % and total C 49 % (Hokkanen & Silvola 1993). There were sub-sites with and without grass, the later was kept free from vegetation with regular tilling. For the reference purposes there was a virgin herb–rich wet flark fen corresponding the site type of the field.

The field was used for hay production and was fertilized with NPK fertilizer (Kemira, 450 kg/ha) and cow manure. Water table was generally more than 50 cm below soil surface. The aluminum chambers used for measurements of N_2O, CH_4 and CO_2 *in situ* were similar to those used at mires (Nykänen et al. (this volume). During measurement period of 30 min 4–5 gas samples were taken from dark chambers with gas tight plastic syringers. In laboratory concentrations of CH_4, CO_2 and N_2O in syringes were analyzed with HP 5890 Series II gas chromatographs equipped with TC–, FI– and EC–detectors. Gas emissions were calculated from the linear change in gas concentrations in chambers during the 30 min incubation time (see the details from Nykänen et al. 1995). Carbon dioxide evolution was measured also with a portable infrared gas analyzer (ADC) using a dynamic chamber technique (Silvola et al. (in press). The long–term rate of carbon accumulation into peat as well as the release of CH_4 from cows were taken from litterature (Ward et al. 1993, Kirchgessner et al. 1991, Tolonen & Vasander 1992).

Results

Nitrous oxide

The emissions of N_2O from the natural peatland site were negligible. The N_2O fluxes from the field site had a maximum in spring (both in 1991 and 1992). These fluxes decreased during summer but increased again in autumn. Some N_2O was emitted also during winter under snow cover. The annual emission calculated from the average monthly flux rates was 9.3 kg N_2O-N ha^{-1} in 1991 and 7.8 kg N_2O-N ha^{-1} in 1992. Winter emissions consisted 4–10% of the annual emissions.

Carbon dioxide

Carbon accumulation during the last 3000 years for the natural mire was estimated to be 200–300 kg C ha^{-1} yr^{-1} (Tolonen & Vasander 1992). Emission of CO_2 from bare sub-area of the field was 1.5 x 10^4 CO_2 kg ha^{-1} yr^{-1} and from site with vegetation 2.2 x 10^4 kg CO_2 ha^{-1} yr^{-1}. The former value was considered as the rate for peat oxidation due to agriculture.

Methane

The release of methane from the natural peat site as CH_4–C was 149 kg ha^{-1} yr^{-1} in 1991 and 259 kg ha^{-1} yr^{-1} in 1992. From the field some release of CH_4 occurred after fertilization in spring and also in winter. During summer there was a net uptake of CH_4 from the atmosphere as a result of oxidation of CH_4 in aerobic peat. The annual release of CH_4 from the field as CH_4–C was 1.97 kg ha^{-1} in 1991 and 1.25 kg ha^{-1} in 1992. The emissions during winter were 20–50% of the annual CH_4 emissions. One lactating cow requires field area of 0.45 ha. Annually the cattle grazing on one hectare would produce 175 kg CH_4–C (see Ward et al. 1993 for CH_4 production capacity of cows). This CH_4 release is rather similar as the CH_4 emissions from the natural mire.

The comparizon of the atmospheric impacts of the ecosystems: GWP approach

The atmospheric impact of the natural mire and the mire used for hay production was estimated by the GWP approach. The annual gas fluxes per hectare were calculated as CO_2 equivalents using a time horizon of 100 years. Then the efficiency of CH_4 is 24.5 and that of N_2O 320 times the efficiency of CO_2 to warm atmosphere (IPCC 1994). The total input of greenhouse gases to atmosphere without taking on the account machinery, CO_2 produced by cows and energy used for fertilizer procuction was 2.2×10^4 kg CO_2–eq. ha^{-1} yr^{-1}. In the GWP calculation carbon dioxide relased from peat oxidation was the most important component.

Discussion

Natural peatland had low N_2O fluxes as found for wet natural mires generally (Martikainen et al. 1993). However, the field had large N_2O emissions indicating high availability of mineral nitrogen. During spring and autumn peat was moist causing depletion for oxygen which favours denitrification (see Klemedtson 1986). The fluxes of N_2O from field were 6 to 10 times the fluxes from drained forested mire (Martikainen et al. 1993). Nitrogen mineralization and nitrification are the processes closely associated with fluxes of N_2O from peat soils (Regina et al., submitted). Availability of mineral nitrogen for nitrification and denitrification in fertilized and tilled peat soil is probably higher than the availability of nitrogen in forested mires.

The loss of carbon from field calculated from the CO_2 fluxes was similar as the loss calculated from the reported subsidence of peat soils located in similar climatic conditions as our sites (Armentano & Menges 1986). On cultivated peatlands having deep water table the carbon originated from vegetation can never compensate the carbon released in oxidation of peat. This is in contrast to drained and forested mires where carbon derived from vegetation compensates sometimes the mineralized carbon (Laine et al., in press), until tree harvesting.

The emission rates of CH_4 from surface of the field were much lower than those from

the natural mire. Release of CH_4 during ditching of the site and the long-term fluxes of CH_4 from ditches were unknown. The results from forested mires and peat harvesting areas indicate that ditches have some importance in the total release of CH_4 from drained peatlands (Minkkinen et al. this volume). The field showed in summer some uptake of CH_4 but the uptake was lower than that found for peat sites drained for forestry (Crill et al. 1994, Martikainen et al. 1994). Nitrogen fertilization, together with high soil moisture, might be reasons for the CH_4 emissions from the field in spring and autumn. Nitrogen is known to inhibit CH_4 oxidation in peat (Crill et al. 1994). Low emissions of CH_4 during winter is possible because of the insulation capacity of snow keeping the soil unfrozen.

The results showed the importance of cattle to produce CH_4 in agriculture: the CH_4 release per hectare from grazed field and natural mire was rather similar. The production of CH_4 by cows thus totally compensated the decrease in CH_4 release induced by drainage. Most important changes in the gas fluxes due to the agricultural activity occurred in the net release of CO_2 and N_2O to atmosphere.

We studied gas balances only in one field used for hay production. We also did not measure the actual CO_2 balance, e.g. by transparent chambers or micrometeorological techiques. There probably are great differences in the gas balances between different sites under various crops. Organic soils are considered to be among the most important biogenic sources for greenhouse gases. The gas balances on organic soils, including peat soils, will be further studied in an European joint project.

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Fluxes of greenhouse gases CH_4, CO_2 and N_2O on some peat mining areas in Finland

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Introduction

The increase in concentration of greenhouse gases (CO_2, CH_4 and N_2O) in atmosphere is associated with burning of fossil fuels and also changes in biogeochemistry due to land use activities.

Virgin peatlands are globally important stores of carbon and sources of CH_4. Peatland drainage changes the processes in carbon and nitrogen cycles responsible for the fluxes of CO_2, CH_4 and N_2O. Preparing of peatlands for peat mining greatly change their biogeochemical processes. Effective drainage decreases water table and allows air to penetrate deep into peat profile. Aerobic conditions inhibit activities of anaerobic microbes, including the methanogens, whereas aerobic processes like methane oxidation are stimulated. Destruction of vegetation cover stops the carbon input to peat.

In Finland the actual peat mining area is 0.05×10^6 hectares and further 0.03×10^6 hectares have been prepared or are under preparation for peat mining. The current total peatland area in the world used for mining is 0.94×10^6 ha and the area already mined is 1.15×10^6 ha (Lappalainen 1995). We report here fluxes of greenhouse gases (CH_4, CO_2 and N_2O) on some mires under peat mining and compare these fluxes with those on natural mires and with the emissions from peat combustion.

When virgin peatland is managed for peat mining an efficient drainage is first needed. Gas fluxes on peatlands drained for forestry (Nykänen et al., this volume, Martikainen et al. 1993, Martikainen et al. 1995) could represent this phase. During the second phase the vegetation is removed and peat mining starts. Gas fluxes of this phase were measured from various mined mires. The fluxes were measured from peat surface and also from ditches.

Materials and methods

The peat mining areas studied are located in Ilomantsi (62°47'N, 24°18E') where fluxes of CH_4 and CO_2 were measured in 1991 and 1992. Fluxes of N_2O were measured in 1991. In Ilomantsi both *Sphagnum fuscum* and *Carex* peat were included to the studies.

In summer 1993 emissions of CH_4 and CO_2 from ditches were studied in three field campaigns. In 1994 gas fluxes were measured in central Finland on *Carex* peat sites (Rastunsuo peat mining area). There were sub-sites representing different phases of peat mining: (1) recently started site, (2) old site, and (3) a cut-away site.

Gas fluxes were measured with a static chamber method. In 1991 and 1992 the method used in natural peatsoils was applied (Nykänen et al., this volume). In 1993 and 1994 galvanized steel cylinders (vol. 30 L) were put to the depth of 5 cm in soil immediately before measurement. At the old and abandoned sites ditches had no water and the standard technique could be applied. For ditches containing water a modified technique using floating chambers were used (for the methodological details of gas sampling and analyses see Nykänen et al., this volume, and Nykänen et al. 1995). At recently started area in Rastunsuo (Fig. 1) bubbles from ditches containing water were collected to bottles with inverted funnels (600 ml). Bubbles were released by disturbing mechanically the sediment of ditch bottoms. Gas concentrations in bubbles were analyzed with a HP 5890 GC equipped with a TC–detector. The standard was 59.97% CH_4 and 1.98% CO_2 in nitrogen (AGA SpecialGas, Lindingö, Sweden).

Annual gas emissions from ditches were calculated by multiplying daily averages by 150 days. The CO_2 emissions from stockpiles were calculated from data published by Ahlholm and Silvola (1990a). The annual amount of CO_2 released m^{-2} in peat combustion were calculated from the average amount of peat mined in Finland during years 1990–1994 and from the mean peatland area under mining during these years (Suo ja Turve 1995). Volumetric amount of peat was converted to peat carbon using published conversion factors (Ahlholm and Silvola 1990b).

Results and discussion

Methane

Emissions of CH_4 on virgin ombrogenous mires in Finland vary generally from 1 to 8 g CH_4-C m^{-2} a^{-1} and the emissions on minerogenous mires from 6 to 30 g CH_4-C m^{-2} d^{-1} (Nykänen et al., this volume). Drainage of mires in the first phase reduces the emissions of CH_4 from peat surface as indicated by results from drained and forested mires. There the emissions from peat surface have ranged on ombrotrophic mires from 1 to 4 g CH_4-C m^{-2} a^{-1}. Peat surfaces on minerotrophic mires have shown even some CH_4 uptake (Martikainen et al. 1995; Nykänen et al., this volume). However, the ditches are always sources for CH_4 (Minkkinen et al., this volume).

On peat mining areas the emissions of CH_4 from peat surface were generally low, there was even some CH_4 uptake. However, some peaks in the CH_4 emissions were found (Fig. 1). The average annual emission on the three sites studied from spring to winter in Ilomantsi was 0.24 g CH_4-C. The ditches emitted 10–1200 mg CH_4 m^{-2} d^{-1}. Annual emissions from ditches having water ranged from 0.2 g CH_4-C m^{-2} (old area at Rastunsuo) to 103 g CH_4-C m^{-2} (Ilomantsi). For calculations the mean of 13.3 g CH_4-C m^{-2} was used (the peak value was excluded). A collecting ditch leading water from peat

mining area in Rastunsuo emitted 60 mg CH_4 m^{-2} d^{-1}. Results from peat mining areas in Sweden indicate that ditches are important in the total areal CH_4 emissions (Svensson 1995). Also the CH_4 emissions from ditches in drained and forested mires increase the areal fluxes (Minkkinen et al., this volume, Roulet et al. 1995). In the present study the bubbles released from ditch bottom contained on average 33 % CH_4. The amount of CH_4 released in bubbles from ditches of the mined area was not determined. The results from mires drained for forestry indicate that the CH_4 emitted in bubbles is less than 10% of the amount of CH_4 released in diffusion (Minkkinen et al., this volume).
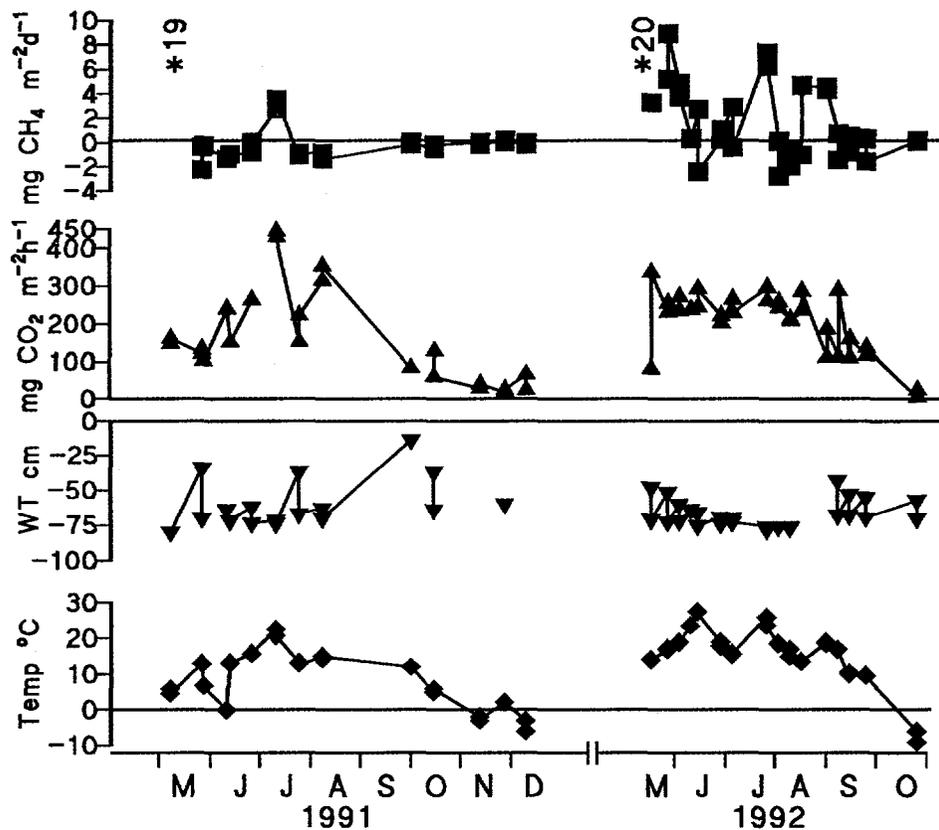
Fig. 1. Mekrijärvi peat harvesting area, *Sphagnum*-peat, measured in year 1991 and 1992. Emissions of CH_4 and CO_2, water table (WT) and air temperature.

Carbon dioxide

Natural peatlands accumulate carbon annually from 2.8 to 88.6 g C m^{-2}. The accumulation rate depends on the peat type, mire location, age and fire history of the site. The average accumulation for bogs is 25 g C m^{-2} a^{-1} and for fens 15.1 g C m^{-2} a^{-1} (Tolonen & Turunen, in press).

The first phase after drainage with developing vegetation could act as a sink for carbon. The carbon balance in drained and forested mires is considered elsewhere in this volume (see also Laine et al., in press). A mire can even act as a sink for carbon if lowering in water table is minor. This is a results of increase in biomass production which compensates the oxidation of peat. However, carbon balance in mires prepared for peat mining can differ from that in forested mires because ditches are deeper in sites drained for peat mining. Silvola et al. (in press) concluded that lowering of water table by 1 cm increases carbon release by 9.5 g CO_2-C m^{-2} a^{-1}. According to this model mires drained for peat mining (WT –50 cm) could release 500 g CO_2-C m^{-2} a^{-1}. The increase in biomass production probably can not compensate the high carbon loss from peat. There are results from peatland forestry that efficiently drained minerotrophic sites are loosing peat carbon (Laine et al., this volume).

Bare *Sphagnum* peat surface released 20 to 450 mg CO_2 m^{-2} h^{-1} (Fig. 1). Average annual emission from the three intensively studied sites in Ilomantsi was 240 g CO_2-C m^{-2} a^{-1}. This corresponds well the values reported by Ahlholm & Silvola (1990b) whose annual estimate was 262 g CO_2-C. The CO_2 emissions from ditches ranged from 62 g to 220 g CO_2-C m^{-2} a^{-1}. For stockpiles, data of Ahlholm and Silvola (1990b) was used. Here the emissions of CO_2 were 3000 mg m^{-2} h^{-1} for 180 days, area for stockpiles 4500 m^2 and their volume 14100 m^3. In calculations 33.4 ha of peatland area is needed for a stockpile (Ahlholm & Silvola 1990a, Suo ja Turve 1995, Ahlholm & Silvola 1990b).

Nitrous oxide

The emissions of N_2O from natural mires are negligible but nutrient rich sites drained for forestry can produce N_2O by rates from 0.051 to 0.143 g N_2O-N m^{-2} yr^{-1} (Martikainen et al. 1993). These values would be valid for the first phase of peat mining areas. However, even higher fluxes are possible because the drainage for peat mining is more efficient than the drainage for forestry.

Emissions of N_2O from the peat mining area in Ilomantsi were close to the detection limit. However, low emissions were regularly measured giving annual emission of 0.022 g N_2O-N m^{-2}. New and old areas at Rastunsuo emitted annually 0.05 g N_2O-N m^{-2} (Figs. 2 c, d). The *Carex* peat in Rastunsuo probably had higher capacity to produce nitrate than the nutrient poor *Sphagnum* site in Ilomantsi as suggested by the results from drained and forested mires (Regina et al., submitted). Nitrification activity and availability of nitrate are important factors to regulate production of N_2O in peat soils (Regina et al., submitted). The emissions of N_2O in Rastunsuo were typical for forested peatlands (Fig 1.) (Martikainen et al., 1993; Regina et al., submitted). Emissions were highest from recently started site and from an old site where peat and clay already were mixed. Old, partly dry ditches also were small sources for N_2O. Abandoned area emitted some N_2O, but less than the younger area still used for peat mining.
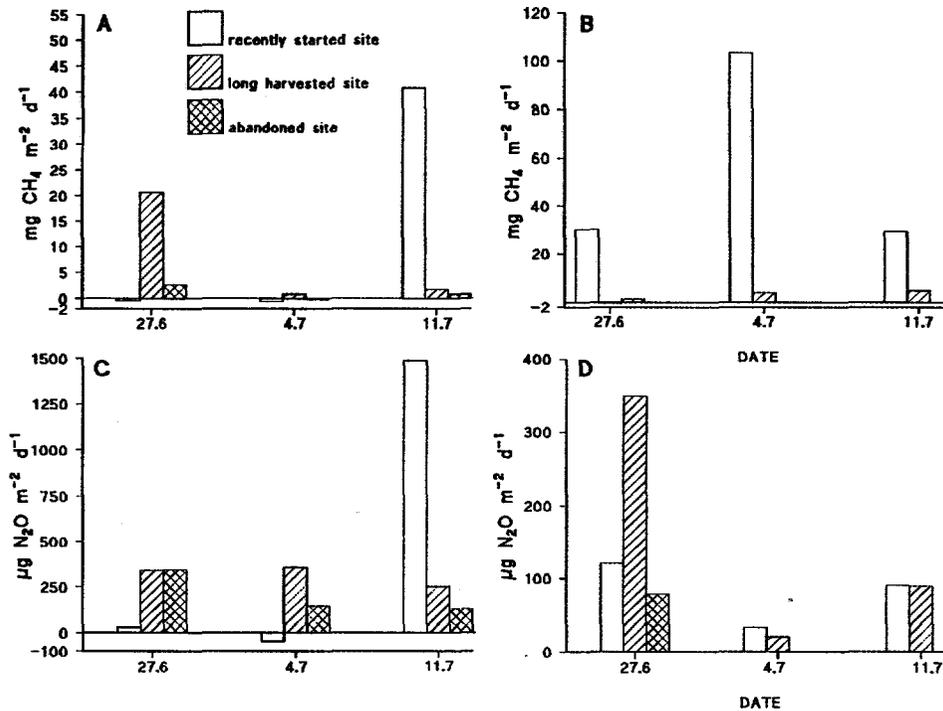
Fig. 2. Rastunsuo peat mining area measured in summer 1994. A) CH_4 emissions from peat surface, B) CH_4 emissions from ditches, C) N_2O emissions from peat surfaces, and D) N_2O emissions from ditches.

GWP calculations

We used in the calculations a time horizon of 100 years. Then the warming potentials of CH_4 and N_2O are 24.5 and 320 times the warming potential of CO_2 (IPCC 1994).

Emissions of CH_4 and CO_2 during ditching as well as the fluxes during the first phase of site preparation (drained, no mining activity) are not included to the calculations because of lack of experimental data for peat mining areas. The data for peatland forestry are discussed shortly in the previous chapters.

The total fluxes of CO_2, CH_4 and N_2O from peat surfaces, ditches and stockpiles of *Sphagnum* and *Carex* peat mining areas corresponded 1200 g of CO_2-eq. m^{-2} a^{-1}. The loss of CO_2 was highest from peat surface (880 g CO_2-eq.), where peat is oxidized and no carbon accumulation occurs. The loss of annual carbon accumulation (20 g m^{-2} a^{-1}) occurring in natural mires was included to the surface flux. Based on the statistics of the recent years, the average release of CO_2 due to peat combustion is 25000 g CO_2 m^{-2} a^{-1}. The annual GWP of the greenhouse gas fluxes from mined area is about 5% of the GWP caused by the peat combustion (Table 1).

Natural mires are cooling atmosphere, except fens having high CH_4 emissions and low carbon accumulation (Laine et al., in press). Therefore, lowering of water table decreases the GWP of the CH_4 release. This reduction associated to the lowered CH_4 emissions is 0.5-3.7 % of the GWP of peat combustion. In a Swedish study the integrated reduction for 100 years of the mining period of 20 years was estimated to be 15 %. Using the same approach for the area and GWP-calculations, the significance of the reduction in CH_4 emissions is rather similar also in the present study (Rodhe & Svensson 1995).

After restoring of peat mining areas peat accumulation is probably higher for the first hundreds of years than the accumulation in old mires. Restored site may accumulate carbon at its maximum 71.6 g C m^{-2} a^{-1} (Tolonen & Turunen, in press). With this maximum carbon accumulation rate about 100 years of peat growth is needed to compensate the annual release of carbon due to combustion of peat. With the average accumulation rate about 300 years is needed.

Table 1. Emissions of CH_4, CO_2 and N_2O and the GWP:s associated with peat mining (mean for various sites).

Source	$g\ m^{-2}\ a^{-1}$		CO_2-eq. $m^{-2}\ a^{-1}$
CH_4 surface	0.24	CH_4-C	7.9
ditch[a]	0.22	CH_4-C	7.2
CO_2 surface	240	CO_2-C	880
ditch[a]	2.43	CO_2-C	8.9
N_2O surface	0.04	N_2O-N	14.0
ditch[a]	$1.7*10^{-4}$	N_2O-N	0.1
CO_2 stockpile[b]	47.7	CO_2-C	175
Total			1100
Peat combustion	6912	CO_2-C	25000

[a] **Emissions from ditches are converted to areal fluxes (The distance between ditches is 30 m, ditch bottom is 0.5 m wide).** [b] **Emissions from stockpile is converted to flux rates per square meter of peat harvesting area (Ahlholm & Silvola 1990a,b).**

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Effects of grassland management on the emission of methane from grassland on peat soils

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Abstract

Net methane (CH_4) emissions from managed grassland on peat soils in the Netherlands have been monitored with vented closed flux chambers in the period January - December 1994. Preliminary results are given here. Net CH_4 emissions from two intensively managed grasslands were low, in general in the range of -0.2 to 0.2 mg CH_4 m^{-2} d^{-1}. On these sites, the effect of management was negligibly small. CH_4 emission from three extensively managed grasslands in a nature preserve ranged from 0 to 185 mg CH_4 m^{-2} d^{-1}. The results presented here indicate that a shift of intensively managed peat grasslands into more natural ecosystems will significantly increase the contribution of Dutch peat soils to the total CH_4 emission.

Introduction

Methane (CH_4) is a greenhouse gas and as such it contributes to the greenhouse effect. Soils can either be a source of CH_4, for example in the case of wetlands and rice paddies, or a sink, for example in the case of well-drained soils (Van Amstel, 1993). Grasslands are generally considered to be a net sink for atmospheric CH_4, especially when they are well-drained. The CH_4 consumption rate depends on grassland management and environmental conditions (Mosier et al., 1991; Bartlett & Harris, 1993).

In the Netherlands (52 °N, 5 °E), about 30% of the grasslands are situated on peat soils. Undrained peat soils often show CH_4 emission, as they have a high organic matter content and are anoxic at some depth (Bartlett & Harris, 1993). The majority of Dutch peat soils are drained by reducing the ground water table, so the oxic top layer is much thicker than in undrained peat soils. As a consequence, the sink-source balance of these soils for CH_4 alters drastically. It is unknown whether Dutch peat soils are net consumers or net producers of CH_4. To assess the effects of grassland management, i.e. drainage, fertilization and mowing versus grazing, on CH_4 emission rates from grassland on peat soils, a monitoring study was conducted. Both intensively managed grasslands and grasslands in a nature preserve were studied.

Methods

Five grassland sites in the major peat soil area of the western part of the Netherlands have been investigated:
* two typical sites on intensively managed, reasonably well-drained grassland at the experimental farm ROC Zegveld:
  * site 8B, with a mean ground water level of 35 cm;
  * site Bos 6, with a mean ground water level of 50 cm.

Ground water levels vary greatly during the year. The vegetation of the sites is dominated by perennial ryegrass (*Lolium perenne* L.). The soil of both sites consists of clayey peat. Soil characteristics are given in Table 1. At both sites there were three different treatments: (i) mowing, no nitrogen (N) application; (ii) mowing, N application; (iii) grazing, N application. Fertilizer N was applied as calcium ammonium nitrate in six or seven dressings. Cumulative application rates were on average about 400 kg N ha^{-1} yr^{-1} for site 8B and about 350 kg N ha^{-1} yr^{-1} for site Bos 6.

Table 1. Characteristics of the 0-20 cm layer of the sites in Zegveld (Segers & Van Dasselaar, 1995; Velthof & Oenema, 1994).

Property	8B	Bos 6
Ground water level, cm, mean	35	50
range	2-70	15-85
Loss on ignition, %	38	45
pH-KCl	5.0	4.7
Clay, %	28	29
Total C content, g kg^{-1}	156	223

* three typical sites on extensively managed grassland in a nature preserve, the Nieuwkoopse Plassen area (see Table 2):
  * Koole, with a mean ground water level of 5 cm;
  * Brampjesgat, with a mean ground water level of 10 cm;
  * Drie Berken Zudde, with a mean ground water level of 15 cm.

Ground water levels vary greatly during the year. The vegetation of the three sites is quite diverse, but consists mainly of grass, moss, sedges, rushes and reed. They are mown once every year in summer.

Table 2. Characteristics of the 0-20 cm layer of the sites in the Nieuwkoopse Plassen area in 1994 (Segers & Van Dasselaar, 1995; Van den Pol - van Dasselaar, not published).

Property	Brampjesgat	Koole	Drie Berken Zudde
Ground water level, cm, mean	10	10	20
range	2-20	0-30	5-40
Loss on ignition, %	50	55	90
pH-H$_2$O	5.3	4.7	3.9

Net CH_4 emissions have been monitored with vented closed flux chambers (Hutchinson & Mosier, 1981) from autumn 1993 onwards. In general measurements took place once every week or every two weeks with six flux chambers at each site. In the Nieuwkoopse Plassen area, boardwalks and steelen frames were installed to prevent artificially induced fluxes due to the very soft topsoil. Gas samples were taken from the headspace of the chambers with glass syringes and analysed for CH_4 by gaschromatography with a flame ionisation detector (relative standard deviation: 0.08%). Monitoring will continue till December 1995. Preliminary results are presented here for the period 1994.

Results

Intensively managed, drained grasslands in Zegveld

In Zegveld, net CH_4 emissions were low, in general in the range of -0.2 to 0.2 mg CH_4 m^{-2} d^{-1}. Only in the relatively warm summer of 1994 (with soil temperatures above 20 °C), consumption of atmospheric CH_4 of about 0.4 mg CH_4 m^{-2} d^{-1} was measured. Effect of mean ground water level in the range of 35 to 50 cm was negligible; site 8B gave equal or only slightly higher net CH_4 emissions than site Bos 6 (Figure 1). There were also no clear effects of N fertilization and grazing versus mowing on net CH_4 emissions from the soil (data not shown).
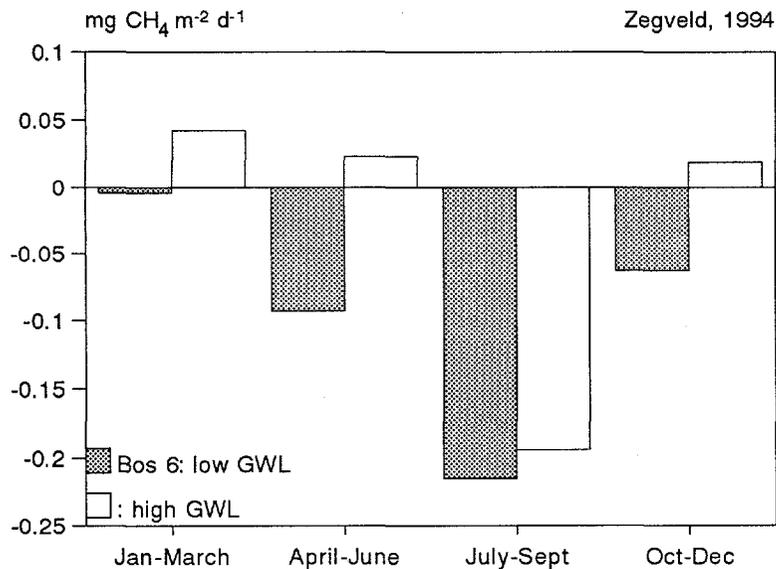
Figure 1. Time course of mean CH_4 emissions (in mg CH_4 m^{-2} d^{-1}) from intensively managed grassland with a mean ground water level (GWL) of 50 cm (site Bos 6) and 35 cm (site 8B) (preliminary results).

Extensively managed grasslands in a nature preserve

Net CH_4 emissions from extensively managed grasslands in the Nieuwkoopse Plassen area ranged from 0 to 185 mg CH_4 m^{-2} d^{-1} (Figure 2). Differences between the different sites were quite large, as were the spatial variations at each of the sites.
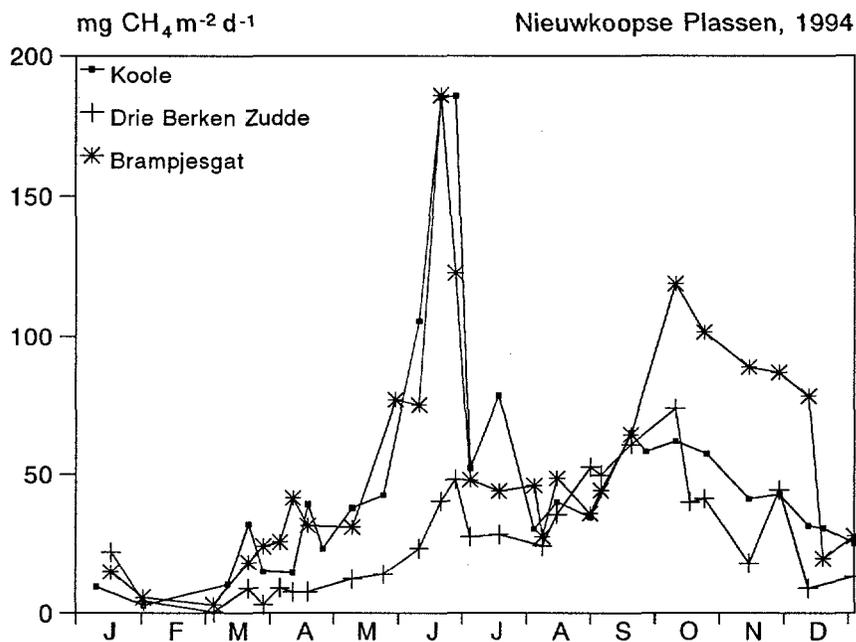
Figure 2. Time course of mean CH_4 emissions (in mg CH_4 m^{-2} d^{-1}) at three different sites in the Nieuwkoopse Plassen area: Koole, Drie Berken Zudde and Brampjesgat (preliminary results).

Discussion

Net CH_4 emissions were low on the intensively managed grasslands (Figure 1). Literature data for comparable sites range from -0.8 for an unfertilized pasture (Mosier et al., 1991) to 0.1 mg CH_4 m^{-2} d^{-1} for a poorly drained grassland soil in winter (Jarvis et al., 1994). Soil analyses in Zegveld showed relatively high nitrate and sulphate concentrations in the soil, especially in the top soil. Both nitrate and sulphate will have blocked CH_4 production. Low soil temperatures in winter and spring will also have contributed to low microbial activities in the soil. In the summer of 1994 with, for Dutch conditions, extreme warm and dry weather, some methane consumption was observed. This could be the result of a lower ground water level in combination with high temperatures.

Grassland soils with a high ground water level generally have a relatively thin aerobic layer. These soils are expected to emit more CH_4 than grassland soils with a relatively low ground water level and a relatively thick aerobic layer. However, site 8B in Zegveld

(relatively high ground water level) gave equal or only slightly higher net CH_4 emission than site Bos 6 (relatively low ground water level) (Figure 1).

Nitrogen fertilization may decrease CH_4 consumption (Mosier et al., 1991; Hütch et al., 1993); mowing or grazing could affect CH_4 emissions by influencing the amount of organic material and nitrogen that is added to the soil annually. However, there were no clear differences between the treatments at the two sites. It has to be emphasized that CH_4 production by cattle is not included in these estimates.

Results presented here indicate that, for intensively managed grasslands, the effect of management on CH_4 emissions is negligibly small.

Compared to the intensively managed grasslands, grasslands in the nature preserve had a high efflux of CH_4. In the period January - December 1994, CH_4 production in the nature preserve ranged from 0 to 185 mg CH_4 m^{-2} d^{-1} (Figure 2). Literature data show great variations in CH_4 emissions from wetlands. In a review, Bartlett & Harris (1993) arrive at a mean estimate of 87 mg CH_4 m^{-2} d^{-1} for boreal wetlands (standard error of mean: 18; range: 0-664 mg CH_4 m^{-2} d^{-1}).

In the Nieuwkoopse Plassen area, several environmental factors have been monitored next to CH_4 emission to be able to, at least partly, explain variations in CH_4 emissions. Both soil temperature and ground water level appeared to be major controlling factors of temporal variation of CH_4 emission (data not shown).

The Dutch government intends to set aside intensively managed grasslands and turn them into more natural ecosystems. The ground water level of these grasslands will then be raised again. As the CH_4 emission from extensively managed, 'natural' grasslands was much higher than from intensively managed grasslands, the contribution of Dutch peat soils to the total CH_4 emission will then increase significantly.

Acknowledgements

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Fluxes of nitrous oxide and nitrification on a drained and forested boreal peatland treated with different nitrogen compounds

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Introduction

Nitrogen rich peat soils are potential sources of nitrous oxide (N_2O). However, in virgin boreal peatlands the emissions of N_2O are low. N_2O is produced both in nitrification and denitrification. High water table inhibits nitrification activity which limits denitrification due to lack of nitrate (Regina et al., submitted; Regina et al., this volume). The better aeration after drainage is known to increase N mineralization in peat (Updegraff 1995) and to enhance the emissions of N_2O (Martikainen et al. 1993; Freeman 1993; Moore 1994, Regina et al., submitted; Regina et al., this volume). The addition of nitrogen compounds to soil generally enhances the fluxes of N_2O by increasing nitrification and denitrification (Eichner 1990). However, in acid forest soils there are differencies in the effects of various nitrogen compounds on nitrification activity (Martikainen 1984, 1985a). We studied the effect of different nitrogen compounds on the net nitrification rate and fluxes of N_2O in situ in a drained forested mire.

Materials and methods

The study site was located in eastern Finland (Ilomantsi) was drained for forestry 40–50 years ago. The annual mean temperature in the area is 1.9°C and average precipitation 648 mm. The experimental site was divided into 12 plots (area 9 m^2) and four treatments: control, KNO_3, urea and NH_4Cl. The plots were treated in September 1992 with nitrogen solutions (100 kg N ha^{-1}) or water (control plots). Description of the study area is in Crill et al. (1994). The accumulation of nitrate was studied in situ in five periods during a year (May 1993–June 1994) using an intact soil core method. Two soil cores (diameter 10 cm, height 15 cm) on each plot were incubated in PVC tubes in soil with ion exchange resins in the bottom. The initial and final concentrations (those after incubation) of nitrate were determined in four soil layers: litter, 0–5 cm, 5–10 cm and 10–15 cm. The amount leached was determined from the resins. The tubes were open at both ends to allow precipitation to flow through. Gas fluxes were measured from the

same PVC tubes after closing them with a cap and also from larger chambers on the same plots in 1993. In 1992, 1994 and 1995 only the larger aluminium chambers were used (method in Nykänen et al. 1995). N_2O fluxes were measured by taking two gas samples from the PVC tube or 4 samples from the larger chamber during half an hour. The samples were analyzed with a gas chromatograph equipped with ECD. Measurements from the period June–September are reported here except in 1992 when there are also results from October after the fertilization.

Results and discussion

Fertilization in September 1992 caused an immediate increase in the N_2O fluxes in all treatments but especially in the KNO_3 treated soil (Fig. 1). In 1993 the effect of fertilization was most clearly seen in the soils treated with urea and NH_4Cl. However, in 1993 the fluxes were higher than in other years also in the control plots. In 1994 and 1995 the effects of fertilization on N_2O fluxes were minor. The addition of KNO_3 increased rapidly the N_2O fluxes (results in autumn 1992) because the surplus NO_3^- can be immediately denitrified. The effects of urea and NH_4Cl are slower because the ammonium liberated from these compounds has first to be nitrified before denitrification can take place. The results from the core experiments showed that all N compounds enhanced net nitrification in June/July 1993 (Fig. 2). Later in 1993 the effects of the N treatments on the net nitrification activity were minor. In June 1993 the highest nitrification rates occurred in soils treated with urea or NH_4Cl (Fig. 2). In July the nitrification activity was highest in soils amended with KNO_3 and urea. During the summer months the mean rates in different treatments varied from 38 to 97 mg N m^{-2} d^{-1} and during the winter from –2.3 to 2.1 mg N m^{-2} d^{-1}. The nitrification rates in general were high compared to some forest soils (Smolander et al. 1995; Tietema and Verstraten 1991; Tietema et al. 1993).
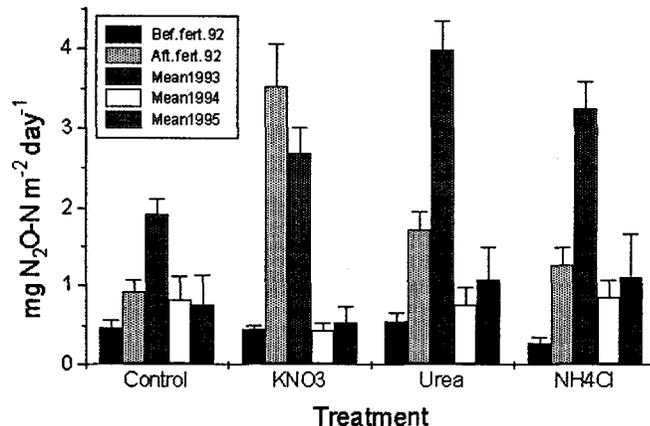
Fig. 1 The effect of KNO_3, urea or NH_4Cl on fluxes of N_2O from a forested mire in 1992–95.
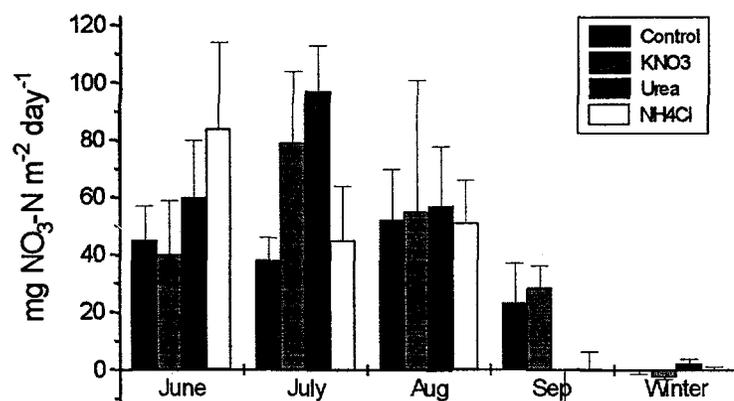
Fig. 2 The effects of KNO₃, urea and NH₄Cl on net nitrification on a forested mire in 1993.

N_2O fluxes from the soil cores correlated with the net nitrate accumulation in the cores and with the nitrate concentration in the litter and in the surface soil (Table 1). During the summer the N_2O flux was 4.5 % of the nitrate accumulation (mean of all treatments). The results show that the production of N_2O is dependent on nitrification. N_2O is either produced in nitrification or nitrate is needed as a substrate in denitrification.

Additions of mineral nitrogen enhanced nitrification. These results differ from those reported for acidic mineral forest soils where merely urea, not mineral nitrogen, has enhanced nitrification (Martikainen 1984, 1985a, 1985b). The reason for the stimulatory effect of KNO₃ on nitrification in July 1993 is unknown. An explanation could be that this treatment has increased mineralization and nitrification of the native N in peat.

Table 1. Pearson correlation coefficients between N_2O flux and soil characteristics

	N_2O flux	Net nitrification	Nitrate in the litter	Nitrate in the 0–5 cm layer
Net nitrification	0.22 p=0.016			
Nitrate in the litter	0.29 p=0.008	0.04 p=0.357		
Nitrate in the 0–5 cm layer	0.44 p=0.000	0.23 p=0.028	0.53 p=0.000	
T°C soil	0.09 p=0.205	0.35 p=0.000	0.11 p=0.055	0.09 p=0.086

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Nitrous oxide production in boreal peatlands of different hydrology and nutrient status

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Introduction

Nitrous oxide (N_2O) is an important component in the atmospheric chemistry both as a greenhouse gas and in the destruction of stratospheric ozone (Cicerone 1987). The concentration of N_2O in the atmosphere is increasing annually by 0.25 % (Houghton 1991). Soils are considered the most important source for N_2O (Bouwman 1990). However, the role of northern peatlands in the global N_2O emissions has not been studied. N_2O is produced in the environment, including soils, mainly in microbial nitrification and denitrification (Davidson 1991; Bouwman 1990). Nitrite/nitrate produced in nitrification or derived from atmospheric deposition or fertilization can be reduced in anoxic conditions in denitrification producing N_2O.

Nitrification is the key process for denitrification in environments with low nitrate input, like mires in the boreal zone. There is evidence that low availability of nitrate limits denitrification in peat soils (Verhoeven 1986). Nitrification activity is limited by low oxygen content (Goreau 1980) and low pH (Focht and Verstraete 1977). In water-logged mire ecosystems the oxygen content as well as pH usually are low. However, nitrification activity (Martikainen and De Boer 1993; Martikainen et al. 1993a) and nitrifying bacteria (De Boer et al. 1991, 1992) adapted to low pH have been found in acid mineral forest soils. An increase in nitrification rate in a mire has been observed after an intensive drainage (Zimenko and Misnik 1970). An increase in nitrification in the aerobic surface peat may enhance denitrification in the anaerobic layer due to the nitrate leached from the aerobic layer of peat profile. Nitrification in soils of different mire ecosystems is poorly known. The dependence of nitrification activity on nutrient content and pH in peat, as well as the organisms responsible for nitrification, are unknown. We examined N_2O fluxes from various boreal mire ecosystems in Finland and correlated the flux rates with mire hydrology, and chemical (nutrients, pH) and microbiological (nitrification, nitrifying microbes) characteristics of peat. Part of the results have been published or submitted for publication and will be only shortly referred to in the text.

Materials and methods

Short-term effects of changes in water table on N_2O fluxes

Intact peat monoliths of 22-35 cm were taken into PVC-tubes (ø 10 cm) from a tall-sedge fen in central Finland (Lakkasuo, site A, virgin and drained subsites) and from hummocks and hollows of a tall-sedge fen in eastern Finland (Ilomantsi, site B) in the end of October in 1994. Site A was a fen with a rather high N content in peat (2 %) whereas the N content in the peat of site B was lower (0.7 %). In half of the replicates from the virgin mires the water table was maintained at the original level and in half of them the water table was lowered. In the cores taken from the drained subsite A, the water table levels were either kept at -25 cm or raised to the surface. The peat cores were kept at 20°C in a greenhouse. After 10 weeks of incubation the cores were kept in dark for two weeks during which time the vegetation died. After this period the cores were kept again in light. During week 22 temperature in the greenhouse rose to 29°C after which the samples were transferred to 15°C (dark). Gas samples were taken after inserting a chamber on the tubes.

Field experiments

The experimental sites were located in central Finland (Lakkasuo) and in eastern Finland (Ilomantsi). The sites represent several types of mires according to their hydrology, nutrient status and management (Table 1). Content of P in the 0-20 cm layer of the sites varied from 360 to 1200 μg g^{-1}, K from 120 to 690 μg g^{-1} and Ca from 1600 to 8400 μg g^{-1}. Some of the virgin mires have drained counterparts which originally (30-50 years ago) have been similar. Details of the sites and methods are found in Regina et al. (submitted).

The N_2O fluxes were measured 6-15 times from the end of April until the middle of December in 1991. In 1992 the fluxes were measured 1-6 times. The samples were analysed with a gas chromatograph equipped with ECD. The method of measurement is described in Nykänen et al. 1995a.

The effect of acetylene on N_2O fluxes was studied on the drained subsite of site 1 (3 chambers in 1992) and on site 3 (2 chambers in 1991) *in situ*. Acetylene is known to block the reduction of N_2O to N_2 in denitrification (Walter et al. 1979) and to inhibit chemolithotrophic ammonium oxidation (Hynes and Knowles 1982). Fluxes were first measured without acetylene and after that with 10 % acetylene added in the gas phase of the chamber.

Nitrification capacity

The peat samples for the enumeration of chemolithotrophic ammonium and nitrite oxidizing bacteria were collected from the sites in September 1990. A soil profile of 8 x 8 x 30 (depth) cm was divided into layers and homogenized manually. The occurrence of bacteria oxidizing ammonium and nitrite was studied in 2-6 layers of the soil profiles with the most probable number (MPN) method using microtiter plates.

Nitrification potentials were determined in the samples from sites 1, 2, 3, 6, 7 and 9. The peat cores for determination of nitrification potential (8 x 8 x 100 cm) were sampled in August 1992. The cores were cut to 5 cm sections. Nitrification potentials with pH adjusted to 4 or 6 were determined by incubating soil samples in a mineral medium (Lång et al. 1994). The samples were shaken in flasks closed with aluminium foil on a rotary shaker at 19–21°C. Nitrate samples (3 ml) were taken after one, two and three week incubation.

Results and discussion

Short-term effects of changes in water table level

There was occasionally some N_2O uptake in the monoliths of the virgin tall–sedge fen (site A) with high water table (mean -0.1 μg N $m^{-2}h^{-1}$) (Fig 1). With the water table lowered to -25 cm N_2O emissions were induced (mean 10 μg N $m^{-2}h^{-1}$). Enhanced production of N_2O was seen on week 12. The plants started to die after a 10–week incubation when the dark period started. Concentration of nitrate had increased in the monoliths with lower water table.

The N_2O flux from the hummock–monoliths of the less fertile fen (site B) with the water table at -21 cm differed only occasionally from the ones with the water table at -15 cm (Fig. 2). Both dark periods caused a short–term peak in the fluxes from the cores with lower water table. In the hollow samples of site B the low initial production of N_2O was not enhanced by lowering the water table to -25 cm during the 22–week incubation (Fig. 3). The transfer of the cores to dark after week 22 induced some enhancement of N_2O flux in the cores with high water table. The nitrate concentration had not increased in the monoliths either from hummock or hollow.
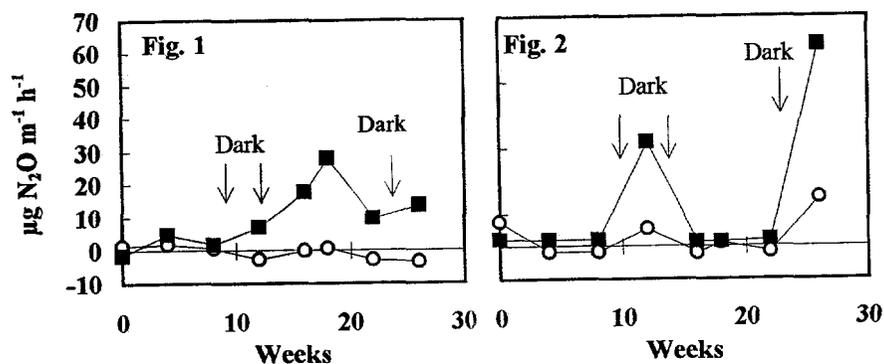
Figure 1 N_2O fluxes from a peat monolith of a virgin tall–sedge fen (subsite A) with water table at -2 cm (-o-) and at -25 cm (-■-).
Figure 2 N_2O fluxes from a peat monolith of a virgin tall–sedge fen hummock (site B) with water table at -15 cm (-o-) and at -21 cm (-■-).

In the samples from the drained tall–sedge fen (subsite A) with water table at -25 cm the mean N_2O flux was 280 μg N $m^{-2}h^{-1}$ (Fig. 4). With the raised water table the

production of N_2O was negligible; similar to that from the virgin part of the same fen with high water table (Fig. 1). Nitrate concentration had diminished remarkably in the samples with the high water table.

Thus, lowering of water table in peat monoliths from these virgin mires increased N_2O fluxes more in a fen with nitrogen content of 2 % than in a fen with nitrogen content of 0.7 %. These results show that N_2O fluxes may rise relatively soon after water table draw-down if the nutrient status of the soil enables nitrification. The increase in the N_2O emissions might be associated to the dying of the vegetation which could have increased the availability of mineral nitrogen. However, nitrification activity is lower in the less fertile fen. Raising water table in a drained fen can cease nitrification and thus cause a rapid decrease in N_2O flux.
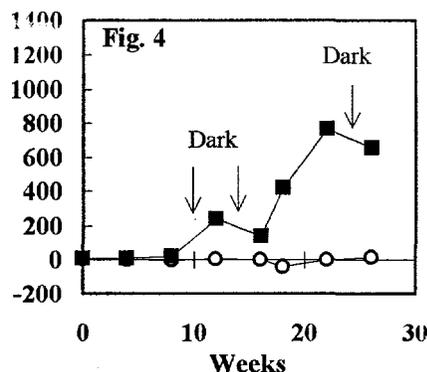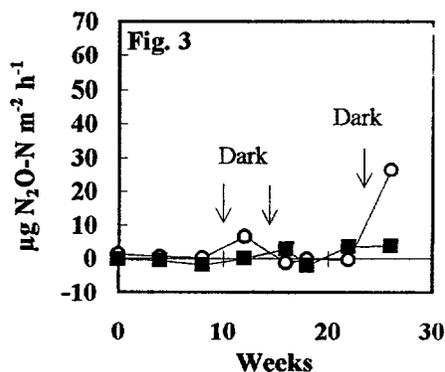
Figure 3 N_2O fluxes from a peat monolith of a virgin tall-sedge fen hollow (site B) with water table at -2 cm (-o-) and at -25 cm (-■-).
Figure 4 N_2O fluxes from a peat monolith of a drained tall-sedge fen (subsite A) with water table at -2 cm (-o-) and at -25 cm (-■-).

Long-term effects of water table draw-down *in situ*

Many virgin sites showed some net consumption of N_2O with fluxes ranging from -30 to 200 μg m^{-2} d^{-1} in 1991. Draining had increased the N_2O fluxes on the sites but, however, much more on the minerotrophic sites (Table 1). Fluxes on the drained sites ranged from -5 to 7900 μg m^{-2} d^{-1}. In 1992 the fluxes were generally higher than in 1991 which results from the drier summer and lower water table in 1992.

The highest N_2O fluxes were observed on the site drained for cultivation of grass (site 11) where also the greatest numbers of nitrifying bacteria were found. This site has the highest nitrogen content and pH and it is the only site that is yearly fertilized and ploughed. Cultivated organic soils are important sources of N_2O (Li et al. 1994; Nykänen et al. 1995). Ploughing and fertilization together with lowered water table ensure the availability of oxygen and nutrients for the microbes. Like the field site also the peat mining area was highly manipulated. There part of the surface peat was peeled off every year. On the peat mining area the emissions of N_2O as well as the numbers of nitrifiers were low which probably results from transferring away the most active bacterial mass with the surface peat every summer.

Table 1. N_2O fluxes and nitrogen content of the experimental mires

Site	Classification	Water table (cm)		pH		N (%)		Mean N_2O in 1991 ($\mu g\ m^{-2}d^{-1}$)		Mean N_2O in 1992 ($\mu g\ m^{-2}d^{-1}$)	
		V	D	V	D	V	D	V	D	V	D
VIRGIN AND FORESTED SITES											
Minerotrophic											
1	Tall–sedge fen	4	31	5.0	4.6	1.9	2.1	2.3 ±16^2	900 ±270	23 ±12	260 ±91
2	Tall–sedge pine fen	21	38	4.4	4.0	1.4	2.3	14 ±19	33 ±18	130 ±47	320 ±137
3	Herb–rich sedge pine–birch fen	–	50	–	4.5	–	1.8	–	960 ±140	–	680 ±160
4	Lagg fen	5	–	4.5	–	0.9	–	−26 ±38	–	200	–
Ombrotrophic											
5	Dwarf–shrub pine bog	28	36	3.8	3.8	0.9	0.9	5.6 ±13	11 ±13	24 ±14	74 ±45
6	Cottongrass pine bog with *Sphagnum fuscum* hummocks	18	24	4.3	4.1	0.5	0.5	−30 ±15	−5.3 ±12	22 ±12	18 ±6.0
7	Cottongrass pine bog	11	13	3.8	3.8	0.9	1.0	−9.6 ±9	22 ±35	19 ±12	48 ±30
8	*Sp. fuscum* pine bog	16	20	4.3	4.3	ND	ND	−1.2 ±25	50 ±25	ND	ND
9	Low–sedge bog	12	27	4.2	4.0	0.6	0.8	−15 ±8.6	13 ±8.8	−3.4 ±9	13 ±16
10	Ombrotrophic hollow bog	13	–	4.1	–	0.6	–	−0.6 ±18	–	2.7 ±14	–
OTHER SITES											
11	Cultivated field, originally flark fen	–	63	–	5.3	–	2.6	–	5000	–	7900
12	Peat mining area, originally tall–sedge *Sp. papillosum* fen	–	73	–	4.6	–	ND	–	67 ±6.0	–	ND

WT= average water table level in 1991 and 1992
V = a virgin subsite, D = a drained subsite
ND = not determined
– = such site was not available

Nitrification potentials of the soil had increased after draining at minerotrophic sites and the numbers of nitrite oxidizers at all sites. More nitrite oxidizers than ammonium oxidizers were found. Nitrite oxidizers may survive better than ammonium oxidizers in peat because they may be acid–tolerant (Hankinson and Schmidt 1988) or able to grow heterotrophically and anaerobically (Freitag et al. 1987; Woldendorp and Laanbroek 1989). Ammonium oxidizers adapted to the acid peat may not be able to grow in the neutral medium used for enumeration (De Boer 1989). Ammonium oxidizers adapted to low pH may be more common in nature than earlier believed because it was recently found that *Nitrosospira* can adapt to low pH if the cell density is high enough (De Boer et al. 1995).

Table 2. Pearson correlation coefficients between biological, physical and chemical characteristics of the peatlands

	N_2O 1991	N_2O 1992	NO_2 ox	Nit pH4	Nit pH6	WT 1991	WT 1992	N	P	K	Ca
N_2O 1991[1]											
N_2O 1992[1]	.83 ***										
NO_2 ox[2]	.82 ***	.83 ***									
Nit pH4[3]	.62 **	.51 *	.79 **								
Nit pH6[3]	.59 **	.32	.43	.87 ***							
WT 1991[4]	−.49 ***	−.53 ***	−.49 **	−.36	−.35						
WT 1992[4]	−.34 *	−.46 **	−.10	−.49 *	−.27	−.70 ***					
N[5]	.64 ***	.70 ***	.80 ***	.55 *	.43 *	−.46 **	−.42 **				
P[5]	.42 **	.39 **	.62 **	.56 **	.36	−.03	−.46 **	.73 **			
K[5]	−.45 **	−.58 ***	−.61 **	−.65 **	−.55 **	.30 *	.30 *	−.83 ***	−.33 *		
Ca[5]	.78 ***	.68 ***	.07	.48 *	.61 **	−.60 ***	−.38 *	.54 ***	.15	−.34 *	
pH[6]	.63 ***	.50 **	.61 ***	.33	.42	−.32 *	.08	.63 ***	.30	−.21	.44 *

Asterisks denote two–tailed significances (* $p<0.05$; ** $p<0.01$; *** $p<0.001$)
[1]Fluxes of N_2O *in situ* in 1991 or 1992
[2]Most probable numbers of nitrite oxidizers in the three upper layers of the soil
[3]Nitrification potentials at pH 4 or 6 in the three upper layers of the soil in one week incubation
[4]Water table in 1991 or 1992
[5]Total N, P, K and Ca content in the 0–20 cm layer of the peat
[6]pH measured in soil–water suspension

The log-transformed fluxes of N_2O increased when the average water tables lowered (Table 2). Soil N, P, Ca and pH correlated positively and K negatively with the N_2O fluxes. K correlates negatively with N_2O fluxes because the higher the bulk density of the soil is the higher is the content of K in the soil (Westman 1981). The bulk densities were highest on the drained sites which produce most N_2O. N_2O fluxes correlated with the level of water table, nitrification potentials and numbers of nitrite oxidizers. However, we can not conclude from this that the observed N_2O was mainly produced in nitrification because denitrification may also take part in N_2O production on sites where nitrate is formed. The nitrate produced in the aerobic surface of the peat may leach downwards and be denitrified in the deeper layers. It must also be taken into account that other processes than chemolithotrophic nitrification and denitrification may be involved in N_2O production. Nitrification actvity is thought to be very low in peat soils (Verhoeven 1986). These acid soils where organic matter is readily available may favour heterotrophic nitrification. Methanotrophic bacteria are known to oxidize ammonium in addition to methane and to produce N_2O (Yoshinari 1985; Bender and Conrad 1994).

The experiments with acetylene on the two sites indicated high spatial variation in the proportions of nitrification and denitrification in N_2O production. In three chambers the emission of N_2O was lowered, in one chamber increased and in one chamber there was no change in the emission with acetylene. According to these results nitrification accounted for 0–97 % of the N_2O production.

This study gives sight to the complex pattern of environmental factors regulating the N_2O fluxes from peat soils. Many of the virgin mires act as a sink of N_2O. Lowering of water table and cultivation practises increase the microbial activities related to the N_2O emissions. Especially draining of nutrient rich mires enhances N_2O fluxes. According to some climate models the summers would be drier and warmer at high latitudes in the future (Manabe and Wetherald 1986; Mitchell 1989). As a result, the northern peatlands would be drier and their N_2O emissions would increase (Martikainen et al. 1993b). Thus, not only changes in land use but also climate change could affect the N_2O fluxes from the northern peatlands.

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Nitric oxide fluxes from natural, drained, and nitrogen-treated peat soils

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Introduction

A significant amount of nitric oxide (NO) is produced in the microbiological processes of soils in addition to the anthropogenic sources which are considered the main source for NO. All biogenic sources are not known and global estimates are uncertain (Logan, 1983; Davidson, 1991). Peat soils containing high amounts of nitrogen are potential sources for nitrogen oxides. It has recently been shown that drainage of peat soils increases their possibility to produce nitrous oxide (N_2O) (Martikainen et al., 1993; Li et al., 1994). However, the emissions of NO from peat soils are poorly known (Davidson, 1991). Nitric oxide is produced both in nitrification and denitrification, the same processes which produce N_2O. Although acidity and lack of oxygen may limit nitrification in peat soils, peat soils have shown to have capacity to nitrify (Lång et al., 1994; Regina et al., submitted; Regina et al., this volume). We studied the emissions of NO from different peatland ecosystems and the effect of nitrogen amendments on the fluxes from a forested site.

Materials and methods

The NO emissions were measured from three different peatland sites located in Ilomantsi in eastern Finland (62°40'N,30°50'E) in 1992-94. One of the sites was an undrained tall-sedge fen where water table was 1-2 cm from the soil surface. One site was a forested mire which has been drained for forestry 40-50 years ago and has two subsites about 200 meters apart (subsites 1 and 2). The third site was a fertilized, cultivated grass field drained 60 years ago. To see the effect of vegetation on the NO fluxes there also was a plot where vegetation was constantly removed by tilling. The measurements of NO were done with a dynamic chamber technique using Environnement SA AC30M chemiluminescent nitrogen oxide analyser. Methodological details are found in Lång et al. (1995).

In September 1992 subsite 2 was divided into 12 plots and 4 treatments. Nitrogen was added on the plots as KNO_3, urea or NH_4Cl (100 kg N ha^{-1} in water). Net accumulation

of ammonium and nitrate were studied with an *in situ* method on this site by incubating peat cores in PVC tubes. Tubes were open on the upper end to allow precipitation to flow through. Fluxes of NO were measured from the same tubes.

Results and discussion

NO fluxes from natural and drained peat soils

We did not find any NO emission from the virgin fen with a constantly high water table (Table 2). Nitrification activity is found to be negligible on this site (unpublished results). Draining for forestry has increased the NO emission. On the forested site we have measured high nitrification rates (Regina et al., this issue). The highest emissions of NO were found on the cultivated field. The higher concentration of nutrients (N, P, K, Ca, Mg) and higher pH in the cultivated, fertilized soil than in the forest soil might be associated to its higher NO emission. Also nitrification activity could be higher in the cultivated soil because it had a high nitrate but low ammonium content, in forest soil the nitrate to ammonium ratio was the opposite. In the peat of the field the numbers of nitrifying bacteria were high (Regina et al., submitted).

Table 2. Average emissions of NO from unfertilized peat soil sites in 1992–1994

Peat soil site	n	NO flux (μg N m^{-2} h^{-1})	
		Average±SE	Range
Virgin fen	6	BDL	–
Forested site			
subsite 1	48	3.8±0.9	0–33.3
subsite 2	496	7.0±1.3	0–138
Field			
with grass	32	64±25	4.8–820
without grass	24	300±110	9.5–2600

BDL = below detection limit

The mean emission was higher from the unvegetated plot than on the cultivated part of the field (Table 1). This could be associated with the uptake of mineral nitrogen by grass which reduces the possibilities for nitrification and denitrification. In May 1993 when the grass growth on the field was negligible the fluxes from the grass covered and tilled plots were equal. However, in June when the grass growth was fast there was a great difference between the gas fluxes. In July 1992 when the fluxes from these plots differed most, the soil with grass had lower nitrate content than the soil without grass. Possible reasons for the lowering of NO emission by vegetation are lower soil temperature and higher soil moisture content under the plant cover. Vegetation also consumes soil mineral nitrogen and can even take NO (Slemr and Seiler 1984). Organic matter from vegetation may also enhance microbial immobilization of mineral nitrogen. The lower availability of mineral nitrogen for nitrification and denitrification in soil with vegetation reduces the NO fluxes.

At present we do not know the relative importance of nitrification and denitrification in the production of NO in peat. In addition to the microbiological formation NO can be produced in acidic, organic soils also in chemical reactions (Wijler and Delwiche, 1954; Remde et al., 1989). However, these reactions require the presence of nitrite which has to be produced either in nitrification or nitrate reduction. Increase in nitrification activity after draining might have a great importance for the NO production. Ammonification and nitrification in peat has been reported to increase when water table lowers (Zimenko and Misnik, 1969; Williams and Wheatley, 1988; Regina et al., submitted). Furthermore, nitrifying bacteria with a low pH optimum have been found in the depth of 5–25 cm in drained peat soil (Lång et al., 1994). In the nitrifying forested peat soil the accumulation of nitrate *in situ* did not correlate well with the emission rates of NO indicating that there were also other important factors controlling the NO emissions (Lång et al., 1995). Only in the litter layer there was some evidence that the availability of nitrate could regulate the NO emission rates in the forested site.

Davidson (1991) reported the average NO emissions from temperate cultivated lands to be 64 μg N m^{-2} h^{-1}, equal as observed here from a boreal cultivated peat soil. The emissions from temperate forests have been lower than from our forested site, from 1 to 4 μg N m^{-2} h^{-1} (Davidson, 1991; Papen et al., 1993). Emissions of NO from forested boreal peat soils can thus be higher than those from temperate forests. The forested site studied here is minerotrophic. It may be that nutrient poor, ombrotrophic, peatlands have very low NO emissions even after draining as has been reported for their N_2O emission (Martikainen et al., 1993). Considering the large area of organic soils in the northern latitudes and the relatively high emissions after drainage these soils seem to be a potential source of NO if they are widely used for agriculture or forestry.

Effect of nitrogen amendment on NO fluxes

Nitrogen added in September 1992 had increased slightly the NO emissions in the urea and NH_4Cl amended soil measured in June and July 1993 (Fig.1). Increasing effect of urea and NH_4Cl on NO fluxes indicate the importance of nitrification on NO production on this site. Nitrification activity in this peat soil is higher than the activity found generally in mineral forest soils (Regina et al, this issue).
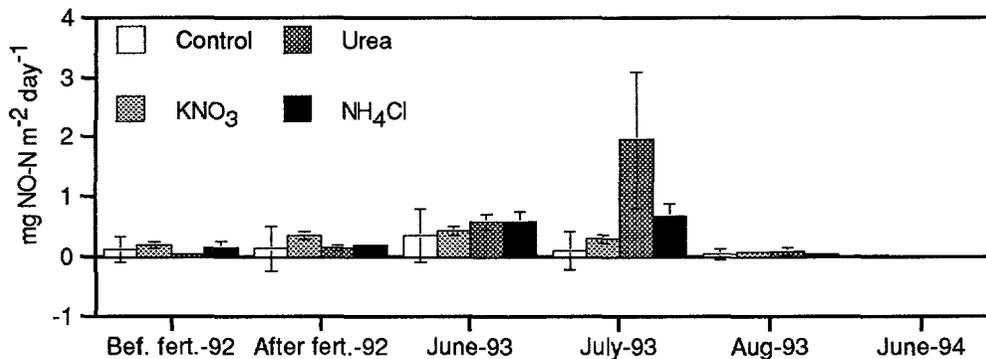
Fig. 1. The effect of KNO_3, urea and NH_4Cl on NO fluxes from a forested mire in1992–1994.

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Seasonal and spatial variation of CH$_4$ emission in an oligotrophic pine fen

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Introduction

Natural wetlands release about 20% (110 Tg) of annual global methane (CH$_4$) emissions (Galchenco 1989), while northern wetlands produce 18-60% (20-62 Tg) of that burden (Matthews & Fung 1987, Crill et al. 1992, Harriss et al. 1993). One reason for the large range in the emission estimates is a huge spatial and temporal variation in the CH$_4$ emissions. Regional differences between study sites might be significant even if much of general climate, vegetation and topography were similar (Crill et al. 1992). Also intrasite spatial variation in CH$_4$ emission rates is high (e.g. Sebacher et al. 1986, Yavitt et al. 1988, Windsor et al. 1992, Bubier et al. 1993a, b). Methane emission rate depends on the difference between CH$_4$ production and oxidation, which are influenced by redox potential (water table depth), temperature, substrate quality etc. (Conrad 1989). More information about the variation of such environmental factors over the mire is needed for reliable regional estimates of CH$_4$ emission in recent and changing climate.

The aim of our study was to determine spatial variation in CH$_4$ emission between different microsites of an oligotrophic pine fen, and the seasonal variation during the growing season 1993. A response fuction was defined between CH$_4$ emission, peat temperature and water table. It was used to reconstruct the CH$_4$ fluxes from different microsites according to time series of the environmental variables.

Material and methods

Methane emissions were measured in Salmisuo mire complex (62°47'N, 30°56'E) in eastern Finland. Study area consisted of minerogenic, oligotrophic low-sedge *S. papillosum* pine fen with slightly more nutrient rich lagg (tall-sedge fen). Three distinctive forms of microsites, hummocks, lawns and flarks could be recognized.

The most typical hummock species was *Sphagnum fuscum*, but in lower parts of hummocks or in low hummocks, also *S. angustifolium* and *Eriophorum vaginatum* were often found. Small, stunted *Pinus sylvestris* -trees were growing on the highest hummocks. Lawn moss layer was dominated by *S. angustifolium* and *S. balticum* accompanied by some *S.*

magellanicum and *S. papillosum*. Field layer in lawns consisted mainly of *E. vaginatum* with *Andromeda polifolia*, *Vaccinium oxycoccos*, *Carex pauciflora* and *C. lasiocarpa*. Major mosses in flarks were *S. majus*, *S. balticum* and *S. angustifolium*. The only vascular plant on flarks was *Scheuchzeria palustris*. Between mineral soil and low-sedge *S. papillosum* fen there was a narrow tall-sedge fen -lagg, consisting of lawn vegetation. Major moss species was *S. angustifolium* and the field layer was dominated by *C. rostrata*.

Permanent aluminium collars (60×60 cm) were installed along boardwalks in the study area, four on flark level, eleven on lawn level (four of them on tall-sedge fen lagg) and nine on hummocks. Gas samples were collected manually using dark, static chamber method (Crill 1991). Four 40 ml samples were drawn into 50 ml polypropylene syringes from the chamber headspace during the 20-30 min sampling period. Air temperature (outside and inside the chamber) and peat temperature profile (surface, -3, -5, -10, -15, -20, -25 and -30 cm) and water table were measured in connection with the gas sampling at each collar. Methane concenrations of the samples were analyzed with gas cromatograph using flame ionisation detector (FID). CH_4 fluxes (mg m^{-2} h^{-1}) were calculated from the linear change in chamber headspace gas concentration.

Seasonal CH_4 emission for the different microsites and for the whole mire was estimated and was used also in carbon balance calculations (see Alm et al. this volume). Dependence between methane emission and environmental factors could be expressed as regression model: $lnCH_4 = \alpha + \beta T + \gamma WT$, where α, β and γ are regression parameters, T = peat temperature at 20 cm depth (°C) and WT = groundwater table (cm). Methane emissions for the whole snow-free period in 1993 were reconstructed using the regression equation as a transfer function and daily time series of peat temperature and water table as driving variables. Daily peat temperatures (-20 cm) were obtained from automatic measurements (see Kettunen et al., submitted) and some absent values were completed on the basis of the nearest observed values. Daily water tables were obtained by spline-interpolation between the measurements.

Canonical correspondence analysis (CCA) was used to study how the CH_4 emission rate can be predicted from vegetation assemblages. CCA is a unimodal direct gradient analysis, which is based on regression between plant species and environmental variables (Ter Braak 1987). Only mosses were treated as plant species in the analysis, because vascular plants have proved to affect CH_4 emissions (eg. Sebacher et al. 1985, Schütz et al. 1991, Chanton & Dacey 1991). Average CH_4 emission in July-August, mean depth of groundwater table in July-August (median), coverage sum of the shrubs, grasses and sedges were used as environmental variables. In addition to that, for every plant species appearing more than once, weighted average CH_4 emission and water table was also calculated using:

$$weighted\ mean\ for\ species\ i(\overline{X_i}) = \frac{\sum_{j=1}^{k} P_{i,j} X_j}{\sum_{j=1}^{k} P_{i,j}}.$$

In the equation, $P_{i,j}$ = plant species i coverage (%) in collar j, X_j = CH_4 emission or water table depth and k = number of the collars (24).

Results and discussion

CH_4 emissions differed clearly among the microsites during the measurement period (Fig. 1). The lowest mid-season CH_4 emissions were measured from hummocks and the highest from lawns. The reason of the low flux rates from hummocks is firmly connected to low water tables (Moore & Knowles 1989, Svensson & Sundh 1992, Bubier et al. 1993b).
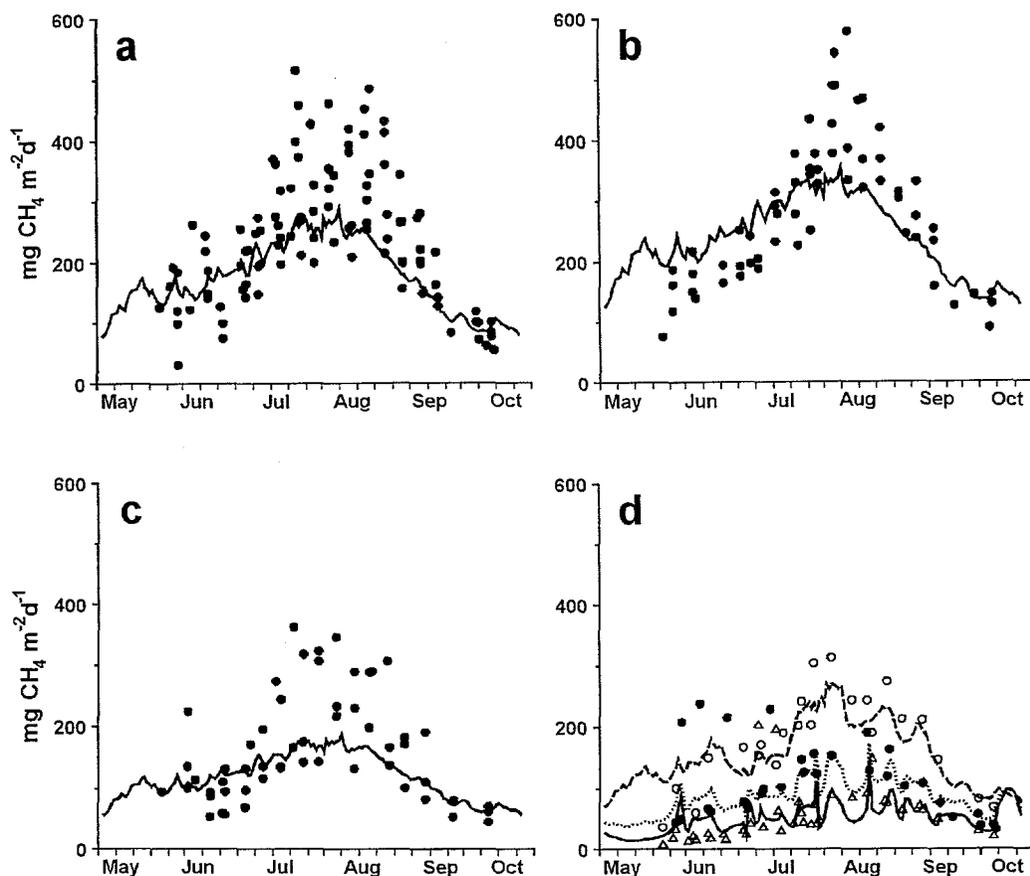
Fig. 1 Measured (markers) and reconstructed (lines) CH_4 emissions in *Sphagnum-Eriophorum* -lawns (a), *C. rostrata* dominated lawns (b), flarks (c) and hummocks (d) in Salmisuo mire in 1993. Three groups of hummocks were distinquished: $\triangle$ = high (water table between 25-35 cm), $\bullet$ = medium (20-25 cm) and $\bigcirc$ = low (10-20 cm). Reconstructions were calculated using peat temperature with "low" (straight line), "high" (broken line) and average (dotted line) water table series.

Seasonal variation in CH_4 emission seems to follow best the general evolution of peat temperature at -30 cm, similarly in all microsites. Because the best predictor was not available for the statistical reconstructions, the second best series (-20 cm) was used. On hummock level, depth of the water table was also a very important control of CH_4 emission (Fig. 1d). In hummocks peat temperature and water table explained 56-72% of the seasonal variation in CH_4 emissions. Average reconstructions, however, did not completely follow the seasonal variation. Spring fluxes were overestimated, midsummer fluxes underestimated but the reconstructions corresponded well with the measured autumn fluxes. Development of vascular plants with changing transport and feeding potential could be one reason for the unexplained seasonal variation. CH_4 emission sums, calculated from daily reconstructions for the whole snow-free period 1993, were highest in lawn surfaces (29-40 g CH_4 m^{-2}) and lowest in hummocks (9-26 g CH_4 m^{-2}, depending on hummock height).

All hummocks were separated from wet microsites in CCA (Fig. 2) and also the weighted mean CH_4 emission and water table correlated negatively (Fig. 3).
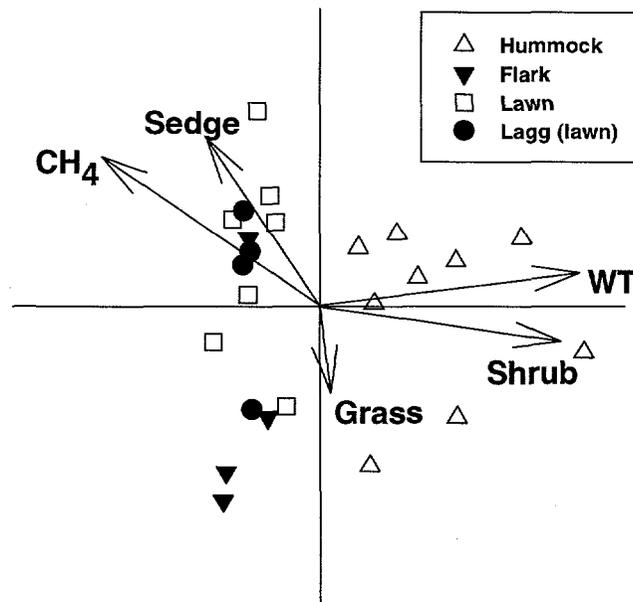
Fig. 2 Distribution of the collars and vectors of the environmental variables in CCA ordination.

On the wet microsites the density of sedge cover seemed to correlate positively with CH_4 emission. Vascular plants promote emissions by transporting CH_4 to atmosphere in aerenchymal tissues and feeding anaerobic microbes with easily dgradable root litter and exudates (Schütz et al. 1991). On the other hand, some O_2 drifts to rhitsosphere in aerenchyme, which may increase CH_4 oxidation (e.g. Conrad 1989, Chanton & Dacey 1991, Schütz et al. 1991), but this O_2 is consumed quickly by plant or aerobic microbe respiration (Bedford et al. 1991).
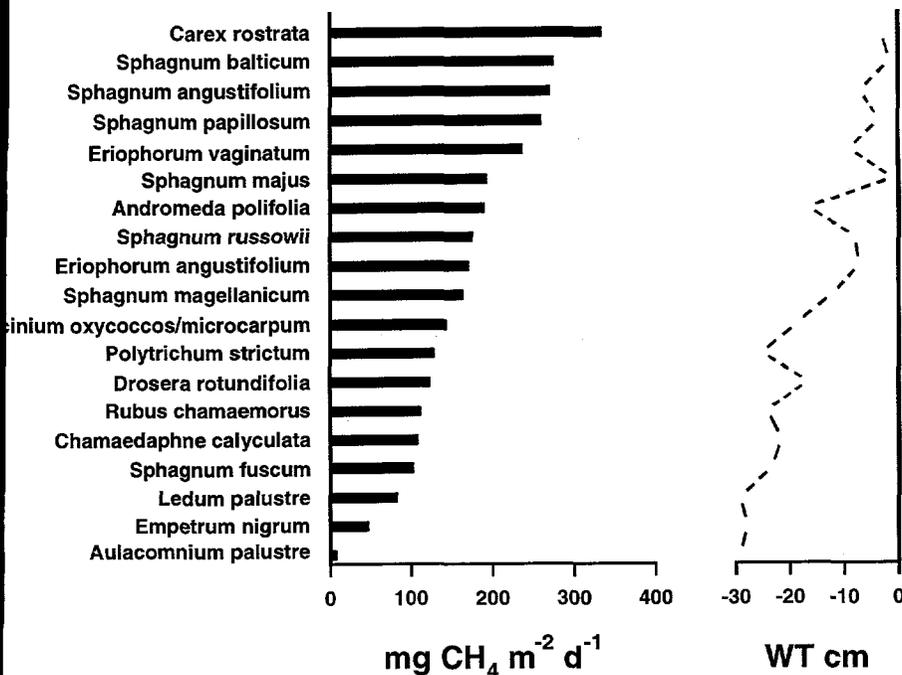
Fig. 3 The average CH$_4$ emissions and water tables weighted for each plant species by its coverage.

There was practically no flarks in Salmisuo, hummocks covered 14% of the mire surface and the rest 86% was lawn surface. With these proportions of the microsites, weighted average CH$_4$ emission for the snow-free period 1993 (140 days) was 27 g CH$_4$ m^{-2}.

Distribution of different vegetation surfaces should be taken into consideration when mire CH$_4$ emission is estimated. Interaction between vegetation and the CH$_4$ emission is obvious, but requires more studies.

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Effects of elevated atmospheric CO_2 concentration on CH_4 emission from peat monoliths of an oligotrophic fen.

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Introduction

Methanogenesis in peatlands depends mainly on water table level, peat temperature and substrate quality (e.g. Svensson & Sundh 1992). The release rate of methane (CH_4) is mostly affected by CH_4 production and oxidation. Also live vascular plants may have remarkable effect on CH_4 emission. Firstly, CH_4 can escape from peat to the atmosphere via plant aerenchyme (Schütz et al. 1991). Secondly, rhitzodeposition offers easily degradable substrates for anaerobic decomposition chain below the water table. Depending on plant species, their physiological stage and environmental conditions, 5-34% of the photosynthetically fixed carbon is released by roots, mostly in respiration (Barber and Martin 1976, Helal and Sauerbeck 1984, Lambers 1987). The main CH_4 forming reactions in soil are carbon dioxide (CO_2) reduction by hydrogen (H_2) or fatty acids or alcohol and transmethylation of asetic acid or methanol (Brasseur & Chatfield 1991). Thirdly, oxygen (O_2) is transported via aerenchyme to anaerobic peat layer (Schütz et al. 1991). This oxygen is quickly used by root respiration and aerobic micro-organisms, because its concentration does not increase in the surrounding soil (Bedford et al. 1991).

Increasing atmospheric CO_2 concentration might increase primary production and carbon accumulation to belowground plant parts and decrease dark respiration (e.g. Drage 1992, Strain and Thomas 1992) and may thus influence to CH_4 emission. We have studied the effect of elevated atmospheric CO_2 concentration on CH_4 emission from peat monoliths with living plants in a greenhouse experiment. In half of the samples the vegetation consisted of both living vascular plants and sphagna and in the other half bare sphagna (some *Eriophorum vaginatum* seedlings were cutted).

The hypothesis was that the elevated atmospheric CO_2 concentration would increase CH_4 emission especially from peat monoliths containing vascular plants. The production of vascular plants and mosses would increase at elevated CO_2 concentration. Thus vascular plants would supply more easily degradable carbon compounds (exudates, fine root litter) to anoxic peat layer and increase CH_4 formation there. On the contrary, most of extra production of mosses would remain in the aerobic peat surface layer and would cause no significant change in the CH_4 emission.

Material and methods

Two sets of 10 cm diameter, 60 cm long peat monoliths with living plants (18 with vascular plants and sphagna and 18 with bare sphagna) was randomly distributed in four thermo-controlled greenhouses (c. 2,67 m^3). In two of the greenhouses, CO_2 concentration was kept at the atmospheric level (360 ppm), and at doubled atmospheric concentration (720 ppm) in the other two. "Sphagnum" samples consisted of Sphagnum angustifolium, S. balticum and S. papillosum and "Sedge" samples of Carex rostrata and E. vaginatum with the above mentioned mosses. The Sphagnum samples were taken from an oligotrophic low-sedge Sphagnum papillosum pine fen and the Sedge samples from a tall-sedge fen.

Each greenhouse contained a refrigerator, cooling the bottom-most 50 cm of nine monoliths of live peat in PVC-tubes. Temperature of the peat monoliths was adjusted for three periods. First 3 weeks the temperature was 1,5 °C, then for 8 weeks it was 9°C, and finally 15°C for 11 weeks.

Greenhouse air temperature (overheating) was controlled by a cooler. Natural lighting was supported by lamps during the mornings and afternoons. The samples were watered with deionised water to keep the water table within c. 2 cm of peat surface.

Gas samples were drawn from the headspace of an opaque, closed, vented chamber (c. 3,5 dm^3) in 1 ml syringes every 5 minutes during 20 minutes sampling period. The samples were analysed within a few hours using gas cromatograph, equipped with a 1,6 m long Haye-Sep Q column and a flame ionization detector (FID).

Results and discussion

Methane emissions were remarkably higher in the Sedge samples than in the Sphagnum samples (Fig. 1). On the other hand, CH_4 emissions between the different CO_2 treatments in corresponding sample types could not be distinguished during the whole measurement period (Fig. 2).

CH$_4$ emissions increased with the temperature of the peat monoliths, but the increase in the emission during the growing season might also be contributed by the development of the plants, because the emissions also increased during periods of stable temperatures. Methane release from the *Sedge* monoliths was c. 50% of the emission measured *in situ* at Salmisuo mire (see Saarnio et al. this volume).
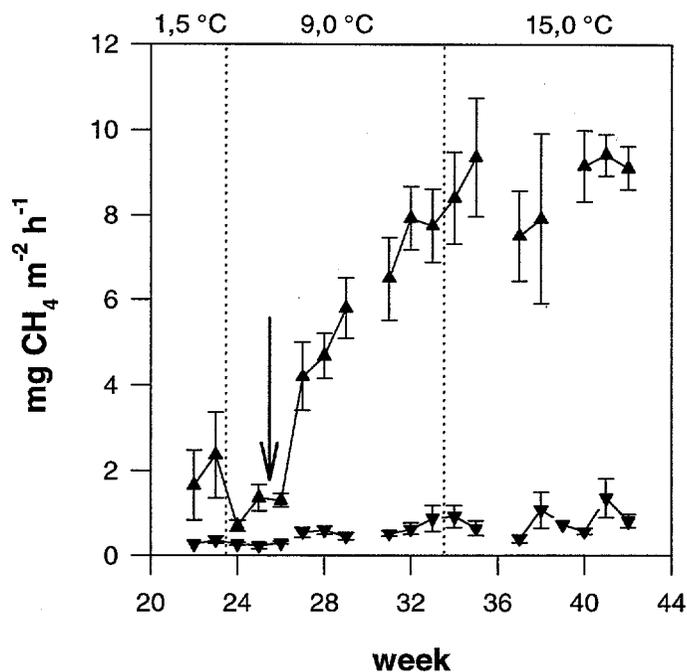
Fig.1 Weekly average CH$_4$ emission (± S.E.) from the *Sedge* samples (▲) were higher than the *Sphagna* samples (▼). Figure includes results from samples grown in atmospheric CO$_2$ concentration (360 ppm). Peat column temperatures were increased twice during the experiment (dotted vertical lines, values inserted). Measurements using 60 cm temperature probe released gas bubbles and probably disturbed the CH$_4$ emissions. The practice was stopped after week 25 (arrow).

In a field experiment (Dacey et al. 1994) and in a laboratory experiment in room temperature (Lee pers. com.) higher CH$_4$ emissions has been found in elevated CO$_2$ concentration than in atmospheric CO$_2$ concentration. It is possible that the lack of response on elevated CO$_2$ in our results is connected to a lower temperature regime of the peat monoliths. One reason could also be that growth of the vascular plants in our experiment was clearly weaker than their growth in natural conditions.
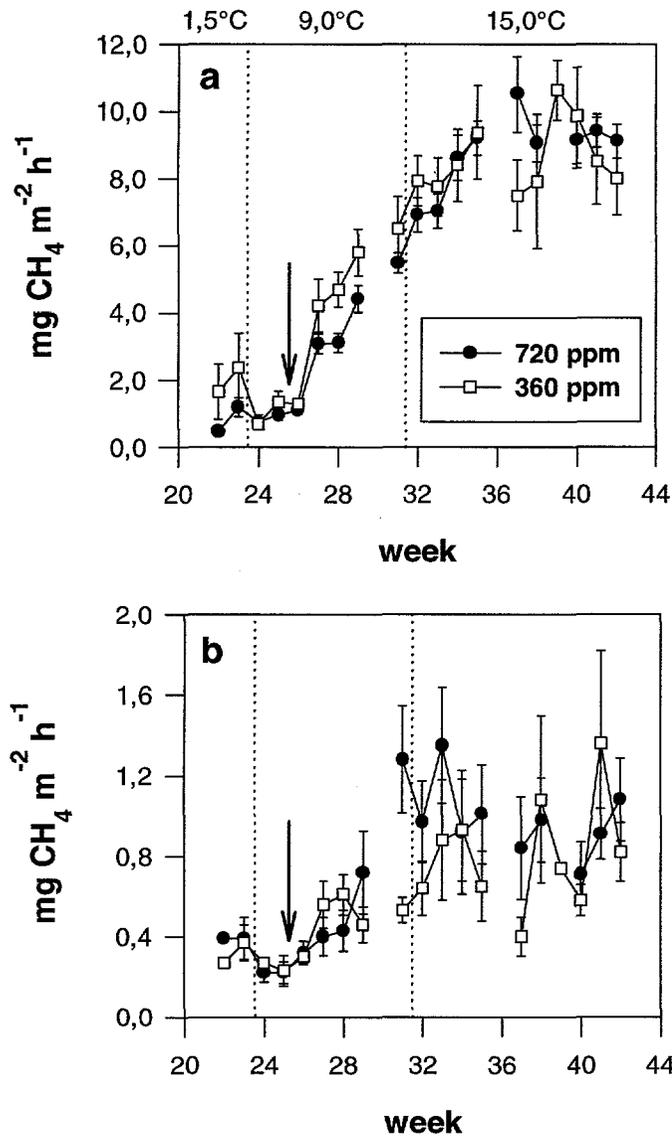
Fig. 2 Weekly average CH$_4$ emission (± S.E.) from the Sedge samples (a) and Sphagna samples (b). Peat column temperatures were increased twice during the experiment (dotted vertical lines, values inserted). Measurements using 60 cm temperature probe released gas bubbles and probably disturbed the CH$_4$ emissions. The practice was stopped after week 25 (arrow).

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Relationship between vegetation and CO_2 balance in an abandoned harvested peatland

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Introduction

The CO_2 balance of an ecosystem is the ratio between the amount of CO_2 fixed through plants in photosynthesis and the amount of CO_2 released by plant respiration and decomposers. When the amount of CO_2 fixed is higher than the amount released, the balance is positive, and the ecosystem acts as a sink for atmospheric carbon. In a barren harvested peatland site the balance is negative. The CO_2 fixing ability of harvested peatland begins with increasing plant colonization and the system develops towards a functional mire ecosystem.

The aim of this study was to examine the vegetation and the CO_2 balance in an abandoned harvested peatland before restoring the water table, and to find out how the vegetation and environmental variables affect the CO_2 balance. Results of this study are published and discussed more detailed in Tuittila & Komulainen (1995). Returning the site to it's original water table level was achieved by blocking the drainage ditches with peat dams in autumn 1994.

Material and methods

The study was carried out at Aitoneva, Kihniö (62°12'N, 23°18'E) during the summer 1994. 11 sample plots, 60 x 60 cm, were established on different kinds of soil surfaces subjectively chosen to represent the variation of vegetation. The daytime CO_2 exchange was measured using two kinds of static chamber techniques. With the first method only the respiration rate could be measured, but with the second method it was possible to measure both the respiration and CO_2 fixation rates (for methods see Komulainen et al 1995).

The relationship between vegetation and the mean rate of CO_2 fixation, respiration, mean water level, species richness and total vegetation cover were analysed by global non-metric multidimensional scaling (GNMDS) (Minchin 1991). Differences in respiration rates between sites were compared using analysis of covariance. In the analysis the soil temperature was used as an independent variable and the respiration rate as a dependent variable.

Results and discussion

The number of plant species ranged from 2 to 13 in a single sample plot: a total of 20 plant species were observed in the present study. The two-dimensional solution of GNMDS was sufficient to describe the variation of vegetation in the data. The main variation in vegetation was connected to the colonizational stage (total cover). The CO_2 fixation and respiration rates increased with increasing total vegetation cover, with the exception of the highest rates which were at sites dominated by a mature *Eriophorum vaginatum* tussock. The variation connected with the water table was not related to the total vegetation cover or CO_2 flow rates (Fig 1).
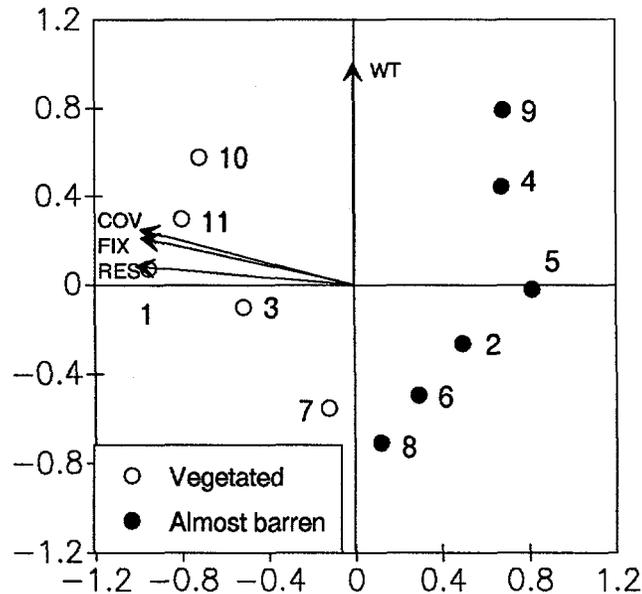
Fig. 1. GNMDS ordination diagram of sample plots and environmental variables in the abandoned harvested peatland site. The variables (arrows) are: FIX = mean CO_2 fixation rate, RES = mean respiration rate, COV = total plant cover, WT = water table level. The correlations and p-values of variables are shown in Table 1.

Table 1. The correlations between environmental factors and the main variation in vegetation of the abandoned harvested peatland.

Variable	n	Max. r	P
Mean respiration rate	11	0.9006	0.000***
Mean CO_2 fixation rate	11	0.8160	0.000***
Mean water table	11	0.7951	0.000***
Species richness	11	0.6472	0.120
Vegetation's total cover	11	0.8972	0.000***

There were more *Pinus sylvestris* and *Eriophorum vaginatum* seedlings in the almost barren sample plots than in the more densely vegetated sample plots. The majority of species had their highest abundances in sample plots dominated by a *Eriophorum vaginatum* tussock (Fig 2).
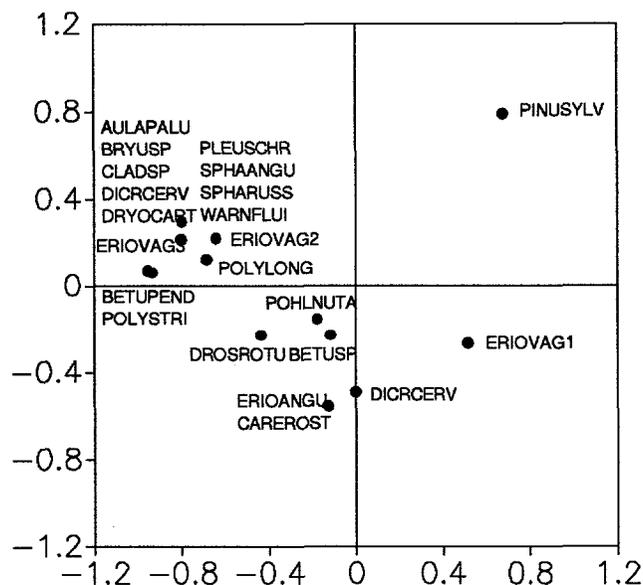
Fig. 2. GNMDS ordination diagram of plant species in the abandoned harvested peatland site. Plant species: **AULAPALU** = *Aulacomnium palustre*, **BETUPEND** = *Betula pendula*, **BETUSP** = *Betula* sp seedling., **BRYUSP** = *Bryum* sp., **CAREROST** = *Carex rostrata*, **CLADSPP** = *Cladonia* spp., **DICRCERV** = *Dicranella cerviculata*, **DICRPOLY** = *Dicranum polysetum*, **DROSROTU** = *Drosera rotundifolia*, **DRYOCART** = *Dryopteris carthusiana*, **ERIOANGU** = *Eriophorum angustifolium*, **ERIOVAG1** = *Eriophorum vaginatum* seedling, **ERIOVAG2** = *Eriophorum vaginatum*, living tussock, **ERIOVAG3** = *Eriophorum vaginatum*, dead tussock, **PINUSYLV** = *Pinus sylvestris* seedling, **PLEUSCHR** = *Pleurozium schreberi*, **POHLNUTA** = *Pohlia nutans*, **POLYLONG** = *Polytrichastrum longisetum*, **POLYSTRI** = *Polytrichum strictum*, **SPAHANGU** = *Sphagnum angustifolium*, **SPHARUSS** = *Sphagnum russowii*, **WARNFLUI** = *Warnstorfia fluitans*.

The respiration rate varied between 34 and 1168 mg CO_2 m^{-2} h^{-1}. The most important abiotic factor affecting the respiration rate was the soil temperature at a depth of 5 cm; the respiration rate rose with increasing soil temperature (Fig 3). The soil temperature influences the activity of soil decomposers and thereby soil respiration. The mean respiration rate of the almost barren sample plots was comparable with those reported by Silvola and Alm (1992) in peat production fields. The respiration rate of the *Eriophorum vaginatum*-dominated sample plots was about three times higher than in almost barren sample plots (Table 2). The difference between the almost barren and

the more densely vegetated sample plots is partly explained by the respiration of plants. Another reason for the higher respiration rate is the increase in heterotrophic microbial respiration due to the larger amounts of easily decomposable organic material exuded by roots.
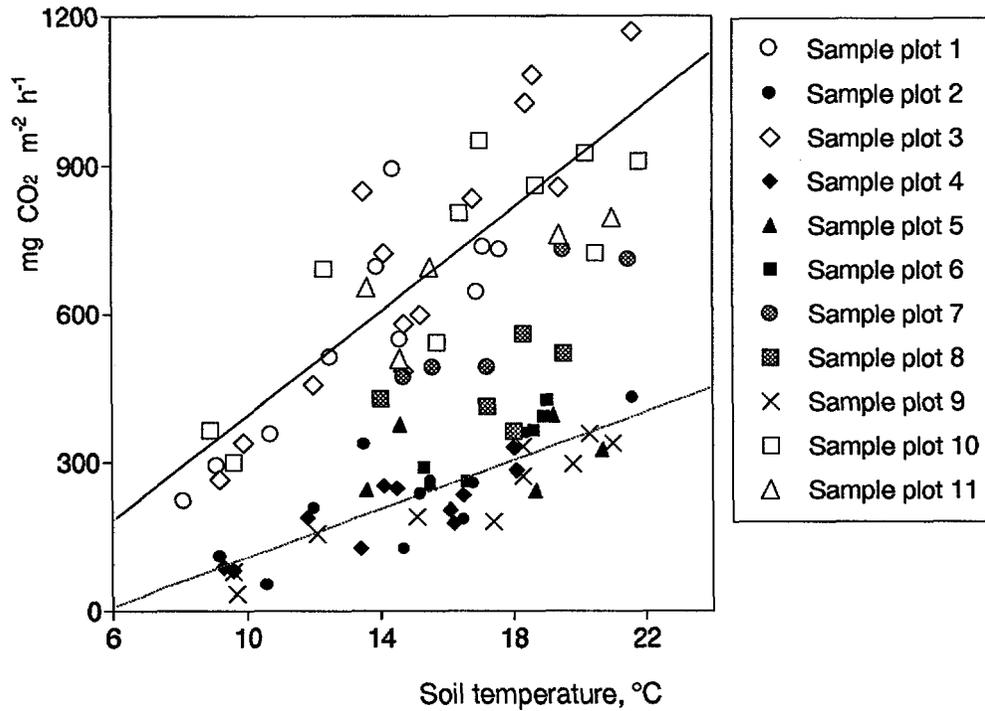
Fig. 3. Respiration rate of different surfaces of an abandoned harvested peatland site in relation to soil temperature at a depth of 5 cm. The dotted line shows the regression for almost barren and the solid line that for more densely vegetated sample plots. Sample plots 7 and 8 which are intermediate according to Tukey's pairwise comparison were not included in the regressions.

The CO_2 fixation rate varied between -56 and 1869 mg CO_2 m^{-2} h^{-1}, and was highest in sample plots dominated by a mature *Eriophorum vaginatum* tussock. Under the environmental conditions prevailing in harvested peatlands, *Eriophorum vaginatum* seems to be the most successful growth form. Despite having the highest respiration rates, the sample plots with a *Eriophorum vaginatum* tussock were the only ones with a positive CO_2 balance.

Table 2. The mean respiration rate of sample plots of an abandoned harvested peatland site, adjusted to a soil temperature (at depth 5 cm) of +15.7 ° C by linear regression.

Sample plot	Mean respiration, mg CO_2 m^{-2} h^{-1}	S.E.	n
1	647.6	35.2	10
2	263.2	32.9	11
3	729.5	30.2	13
4	253.9	33.1	11
5	255.1	44.6	6
6	270.6	49.1	5
7	500.8	49.1	5
8	390.7	49.0	5
9	205.2	34.4	10
10	690.5	34.4	10
11	638.4	48.8	5

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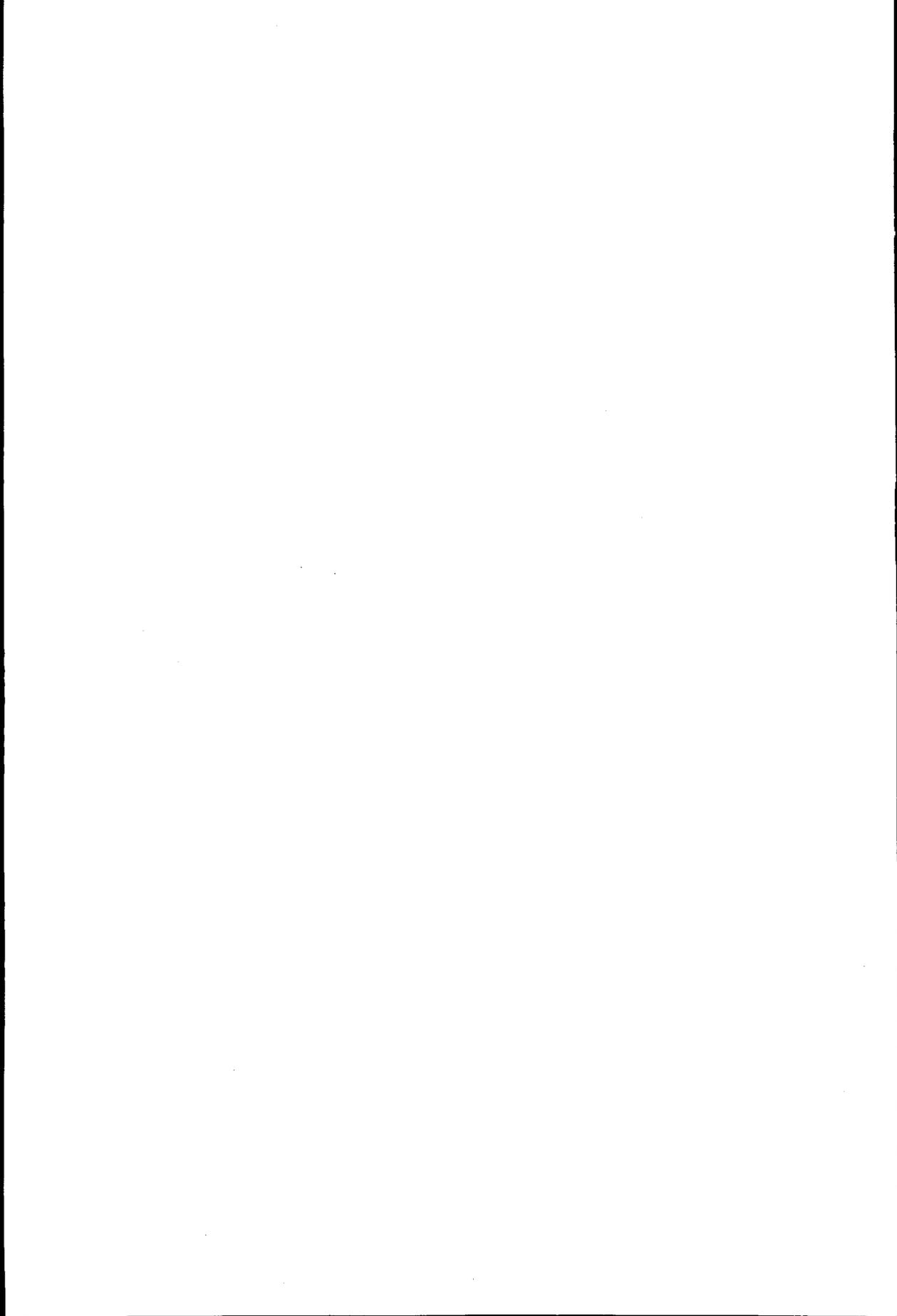
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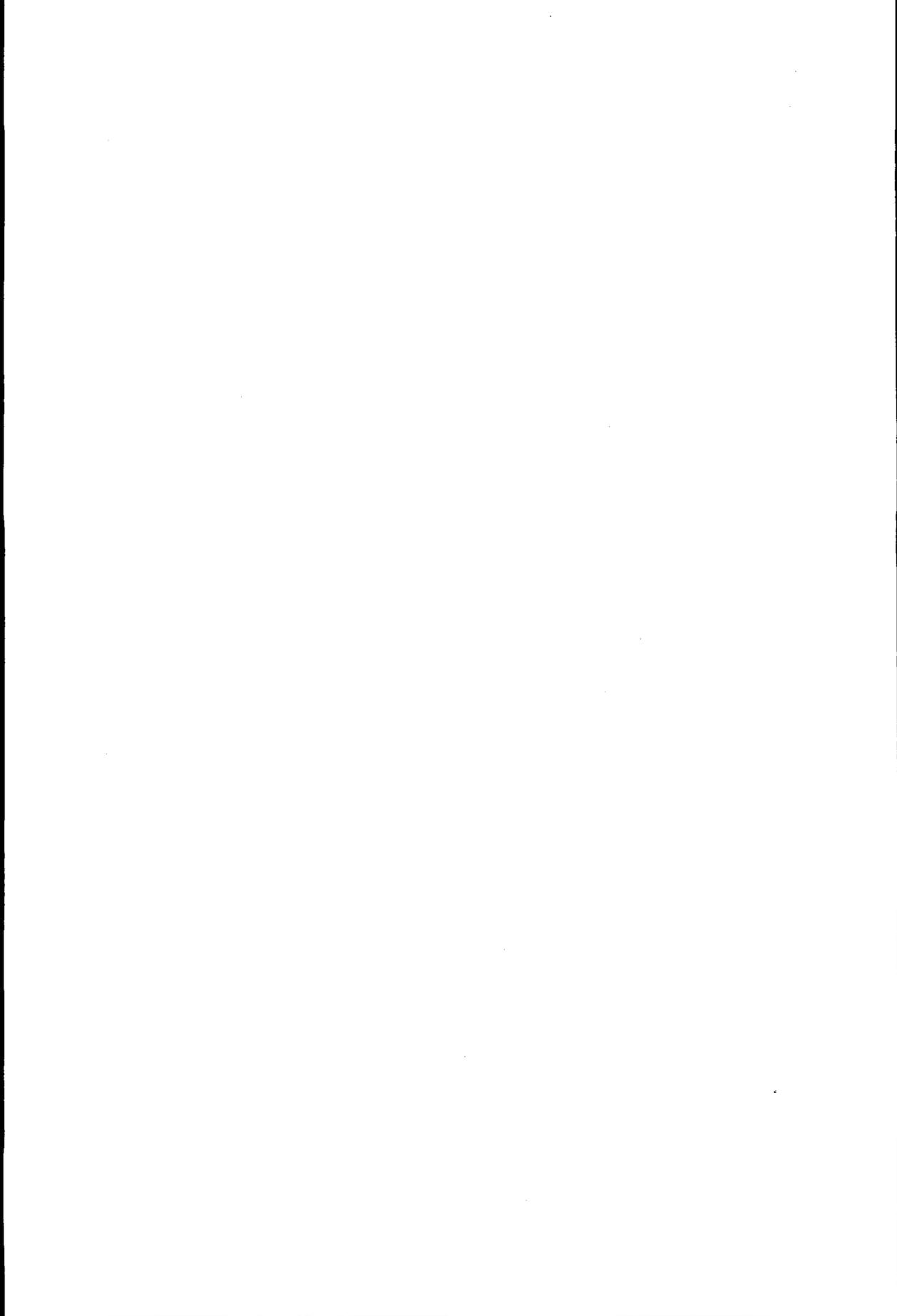
Leaching of organic carbon and nitrogen from Finnish forestry land

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Introduction

One third of the total land area of Finland is covered by peatlands, half of which has been ditched for forestry. 86 % of the total land area is presently classified as forestry land (Aarne 1994). This study provides an assessment of the leaching of total organic carbon (TOC) and nitrogen (N_{tot}) from 22 typical Finnish forested catchments with variable peatland proportion since the 1960's. The long-term monitoring and the regional representativeness of the catchments provide a possibility to consider the impact of changing climatic conditions on the runoff and leaching. The results of the study catchments are used to estimate the total annual organic carbon and nitrogen load from Finnish forestry land.

Material and methods

The 22 study catchments (0.69 to 56 km^2) are located over Finland excluding the northernmost regions. The proportion of the catchments covered by peatlands ranges from 10 to 87 % (Kortelainen and Saukkonen 1994, Table 1). Forestry and atmospheric deposition can be considered the only significant human impacts; less than 6 % of the catchments is covered by agricultural fields.

Daily runoff was recorded in 14 catchments by a V-notch overall weir and a water recorder; in the remaining 8 catchments load calculations were based on runoff data from the catchments nearby. The stream water quality has been monitored since 1962-1976. However, all catchments have not been monitored continuously (Kortelainen and Saukkonen 1994; Table 2). The sampling frequency has been approximately 12 per year. Since the early 1960's samples were taken once a month, in 1981 sampling strategy was changed and sampling was concentrated to spring and autumn high flow periods.

Chemical water analyses were carried out using the methods of the National Board of Waters and the Environment (National Board of Waters 1981). Organic carbon analyses were carried out not until in the 1970's. TOC and chemical oxygen demand (COD_{Mn}) concentrations are closely related in all catchments. Consequently if TOC measurements were missing, site-specific regressions based on COD_{Mn} (r^2=0.60-0.96) were

used to estimate TOC concentrations (Saukkonen and Kortelainen 1995; Table 4).

Trend analysis was carried out using Detect and Exceed -software package which uses non-parametric methods to detect trends in water quality data (Cluis 1988). The tests take into account seasonality and/or persistence. Trend analysis was carried out for COD_{Mn} instead of TOC because COD_{Mn} has been measured since the early 1960's and changes in COD_{Mn} method have been less remarkable than in TOC method.

Annual leaching and spring leaching was calculated separately for each catchment. Spring was not defined by calendar date but using hydrological criteria. The spring was defined to begin when runoff started to increase due to snowmelt. The spring was considered to be over when runoff was under the mean annual runoff or the recession runoff was over. These definitions were made graphically. Using these criteria the average spring period in the study catchments lasted from 38 (Ylijoki) to 56 days (Heinäjoki).

In order to minimize the effect of changes in sampling strategy, the following method (method 2 in Rekolainen et al. 1991) was chozen to calculate the annual leaching of total organic carbon (TOC) and total nitrogen (N_{tot}):

$$L = \sum_{i=1}^{N} c(t_i) \, q[T_i]$$

where N   = number of samples

       $c(t_i)$ = the concentration value at regularly spaced sampling times t_i

       $q[T_i]$ = the discharge for the period $T_i = [\tau_{i-1}, \tau_i]$ with $\tau_i = \frac{1}{2}(t_i + t_{i+1})$, i.e. the runoff period around sampling was used

Results and discussion

The Kruunuoja catchment (No 5) has been in a natural state for the last thirty years. The other catchments represent "normal" Finnish forestry land where a number of forestry practices have been carried out since the 1960's. The ditching intensity of the peatlands ranges from 0 to 100 %. In most catchments clear cutting and scarification has been carried out in less than 20 % and 15 % respectively; in Joutenpuro (No 10) and Kirsioja (No 11) catchment clear cutting has been more extensive, 87 % and 48 %, respectively. Nitrogen fertilization has been carried out in 14 catchments (Saukkonen and Kortelainen 1995, Table 1).

The mean annual leaching of TOC and N_{tot} from the catchments ranged from 2 600 to 8 800 kg km^{-2} and from 100 to 290 kg km^{-2}, respectively (Table 1). Both the concentrations and the leaching of TOC and N_{tot} were higher in the catchments with a high peatland proportion (No 1-13) compared to those with a low peatland proportion (No 21-29). Moreover, the concentrations and the leaching of TOC and N_{tot} were higher in southern Finland compared to northern Finland (Table 1).

Considering the differences in catchment size, location, forest type and peatland type as well as different forest practices the differences in the average long term leaching from the catchments were not large. The regional variation in the mean annual leaching was not higher than the interannual leaching in the catchments. The N_{tot} leaching from

Table 1. Study catchments, median TOC, median N_{tot} and mean annual leaching of TOC and N_{tot}. Upper group: peatland percentage > 35 % ( ). Lower group: peatland percentage < 35 % ( ). Within the groups, the number of the catchments increases from south to north.

	Med TOC $mg\ l^{-1}$	Med N_{tot} $\mu g\ l^{-1}$	TOC $kg\ km^{-2}a^{-1}$	N_{tot} $kg\ km^{-2}a^{-1}$	Runoff $mm\ a^{-1}$
1 Huhtisuonoja	15	600	4300	170	230
2 Katajaluoma	18	950	6500	290	320
3 Heinästönluoma	30	850	8600	250	290
4 Sydänmaanoja	26	880	7400	220	280
5 Kruunuoja	22	460	6300	130	280
6 Töllinoja	11	400	4200	180	280
7 Kesselinpuro	26	700	7300	220	260
8 Vertailualue	27	760	8800	270	320
9 Pahkaoja	18	570	5800	210	320
10 Joutenpuro*	21	460	6600	160	320
11 Kirsioja*	15	380	5900	140	370
12 Kotioja*	15	520	5900	200	370
13 Ylijoki*	13	600	5300	240	400
Mean	20	630	6400	210	310
21 Teeressuonoja	13	850	3800	240	270
22 Paunulanpuro	15	560	4600	180	290
23 Heinäjoki	18	600	6000	200	300
24 Kellojoki*	20	475	6900	140	330
25 Myllypuro*	16	400	6300	180	380
26 Vääräjoki*	8	330	3400	180	420
27 Vähä–Askanjoki*	8	260	4000	150	430
28 Kuusivaaranpuro*	8	340	3500	130	280
29 Myllyoja*	5	200	2600	100	390
Mean	12	450	4600	170	340
Southern Finland	20	680	6100	210	
Northern Finland	13	400	5000	160	
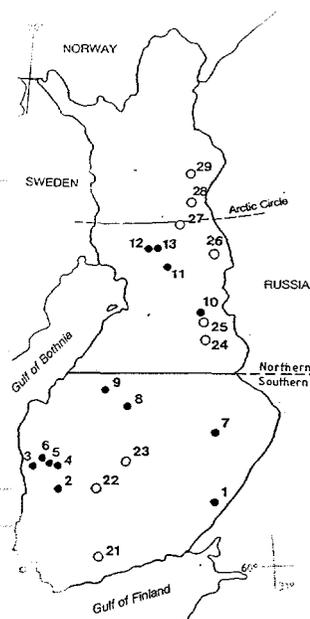
the natural Kruunuoja catchment was among the lowest, it was comparable to N_{tot} leaching from northernmost catchments with a low peatland proportion. The average inorganic nitrogen proportion $((NO_3 + NH_4)/N_{tot})$ was 7.1 % in Kruunuoja. In other catchments it ranged from 9.1 to 53 % (Kortelainen and Saukkonen, Table 2).

There was a large variation in the annual leaching of TOC and N_{tot} in most catchments due to variation in natural hydrological conditions (Kortelainen and Saukkonen1994). The decennial average runoff and leaching values for those catchments with representative water quality data base from all decades were, however, rather close to each other (Table 2). The 1970's was the driest decade (average runoff 300 mm a^{-1}) with lowest TOC and N_{tot} leaching (4 900 and 160 kg km^{-2} a^{-1}, respectively). The highest runoff (350 mm a^{-1}) and TOC and N_{tot} leaching were found in the 1980's (6 000 and 220 kg km^{-2} a^{-1}, respectively).

Half of the annual runoff (on average 50 %) and leaching of N_{tot} (on average 50 %) was concentrated to spring, although the spring period represented only 10-15 % of the

Table 2. Decennial average runoff and leaching for the 7 catchments (No 1, No 7, No 9, No 21-23, No 27) with representative water quality data base from all decades.

	1960's	1970's	1980's	Early 1990's[1]
Runoff (mm a^{-1})	330	300	350	340
TOC leaching (kg km^{-2} a^{-1})	5 300	4 900	6 000	5 800
N_{tot} leaching (kg km^{-2} a^{-1})	200	160	220	190

[1] Early 1990's = 1990-1992

whole year. Nutrient leaching during spring period can be expected to be important to lake ecosystems since increasing temperature and light conditions enable favorable conditions for production. Average TOC leaching during spring period was 44 % in southern Finland and 53 % in northern Finland. In some catchments winter runoff has increased in the early 1990's compared to the previous 30-year study period. Using the temperature and precipitation scenarios presented by Carter et al. (1995) the runoff changes up to 2100 have been modelled for the Siuntio catchment by Kivinen et al. (unpublished). The results suggest increasing winter runoff and decreasing spring runoff in agreement with the results of large catchments by Vehviläinen and Huttunen (1992).

Some statistically significant trends were found in some catchments. N_{tot} concentrations increased in Teeressuonoja and Kotioja and decreased in Huhtisuonoja catchment. In five catchments N_{tot} concentrations dropped to a lower level and in one catchment increased to a higher level in the late 1970's. TOC concentrations increased to a higher level in 5 catchments in the early 1980's (Saukkonen and Kortelainen 1995). The observed trends are probably a combined net effect of forestry practices (e.g. ditching, nitrogen fertilization), nitrogen deposition and changing climatic conditions. Infrequent sampling and missing study years in some catchments cause uncertainty in the true leaching and trends. Rekolainen et al. (1991) have shown that infrequent sampling e.g. during overflow causes underestimation of the true leaching.

Ditching was the largest-scale forestry practice in the study catchments. In earlier studies ditching has been found to result in a short-term increase of organic carbon concentrations (Heikurainen et al. 1978, Moore 1987, Metsä- ja turvetalouden vesien-suojelutoimikunnan mietintö 1987, Ahtiainen 1988). In a long run, ditching lowers the groundwater level and can result in decreased TOC load. The ditching intensity was already high in the early 1960's, whereas most of the catchments with a high peatland proportion were not monitored until the 1970's. This makes it difficult to evaluate the overall impact of ditching. In stepwise multiple regression the proportion of peatlands in the catchments explained 30 % of the mean annual TOC leaching, ditching intensity did not improve the explanation power of the model.

The long term monitoring and the regional representativeness of the catchments enable an assessment of the leaching of TOC and N_{tot} from Finnish forestry land (Table 3). The mean annual runoff from the catchments, 230-430 mm a^{-1} (Table 1) agree with the mean annual runoff from Finland, 301 mm a^{-1}, from 1931 to 1990 (Kuusisto 1992). Moreover, the forestry practices in the study catchments (ditching, clear cutting, scarification and fertilization) have consisted annually about 2.4 % of the catchment area (cf.

Table 3. Leaching of TOC, N_{tot} and NO_3 from Finnish forestry land (86 % of the total land area; Aarne et al. 1994).

	Southern Finland (km^2)		Northern Finland (km^2)		Finland (km^2)
Forestry land	125 440		138 100		263 540
	(kg km^{-2} a^{-1})	(t a^{-1})	(kg km^{-2} a^{-1})	(t a^{-1})	(t a^{-1})
TOC	6 100	770 000	5 000	690 000	1 500 000
N_{tot}	210	26 000	160	22 000	48 000
NO_3	31	3 900	16	2 200	6 100

2,3 % in the entire country in 1980 and 2 % in 1991). The results suggest that the total annual leaching (deposition included) from forestry land in Finland would be about 1,5 million ton of TOC, 48 000 t of N_{tot} and 6 100 t of NO_3. It is likely that only part of this material is transported to large water cources and along rivers to coastal waters (e.g. due to sedimentation and decomposition).

The estimated total annual N_{tot} leaching from Finnish forestry land (48 000 t) is close to the background leaching (including deposition, natural leaching and forestry) to Finnish coastal waters calculated by Pitkänen (1994). In his study the Finnish coastal waters (which receive about 70 % of the annual runoff from Finnish territory) were estimated to receive annually on average 79 000 t of total nitrogen, the contribution of background leaching was 41 000 t. Rekolainen et al. (1995) have recently suggested that nutrient lossess from small agricultural areas mostly enter coastal waters with negligible retention in river channels. This was connected to the fact that most of the nutrient lossess occur in spring, fall or early winter in connection with high water flows and low intensities of biogeochemical processes. The results from agricultural areas are supported by the present study from small forested catchments with average spring leaching about 50 % and the total annual N_{tot} leaching comparable to the riverine N_{tot} input to Finnish coastal waters.

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Water-carried element balances of peatlands

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Introduction

The mire Lakkasuo, Orivesi, southern central Finland, has been a site of intensive carbon and other mass balance studies in the SUOSILMU-project. The hydrology and hydrochemistry of this mire has been monitored since 1991; the results are used here to illustrate the functioning of this mire in the biogeochemical sense and are interpreted from the point of view of the climatic change.

Material and methods

Lakkasuo started to develop some 9000 years ago, initiated by waters from Vatiharju Esker. Today, these waters are still feeding a part of the mire, but large areas have grown beyond the influence of the telluric water inputs from the esker. The variation in the trophic status of this peatland and the possibility to compare same sites in natural state and as drained for forestry 30 years earlier were the main reasons why Lakkasuo was chosen as one of the principal study sites of SUOSILMU in 1991.

The hydrological monitoring in Lakkasuo is presented schematically in Figure 1. There are alltogether 4 water quality monitoring sites for waters leaving the esker (Lä, Rä, Ko and Fe). Runoff water has been monitored in 5 sites in the natural mire and in 4 sites in the drained part of the mire. In addition, deposition has been monitored, too.

Subcatchments B4 and B7 are natural bog catchments; a bog, by definition, receives water inputs from precipitation only. Runoff from these subcatchments is monitored by Weir 4, which is equipped with a water level recorder. Same runoff for both catchments is assumed. DB6 is a corresponding drained bog catchment.

Subcatchments F1, F2 and F3 are natural fen catchments, fed by esker waters from Lä, Rä, Ko, and Fe. Of these, Fe does not enter Lakkasuo today, but, due to a forest drainage, bypasses the mire. Previously these waters were feeding the subcatchment F1. F2 receives water from Ko, Rä and Lä. Rä is the main outlet from the esker entering the mire. F3 receives water from Lä, i.e. Lä waters are diverted partly to F2, partly to F3.

HYDROLOGICAL MONITORING IN LAKKASUO:
A SCHEMATIC PRESENTATION

B = Natural bog   DF = Drained fen
F = Natural fen   W = Weir
DB = Drained bog   R   = Water level recorder
Lä, Rä, Ko, Fe = Springs
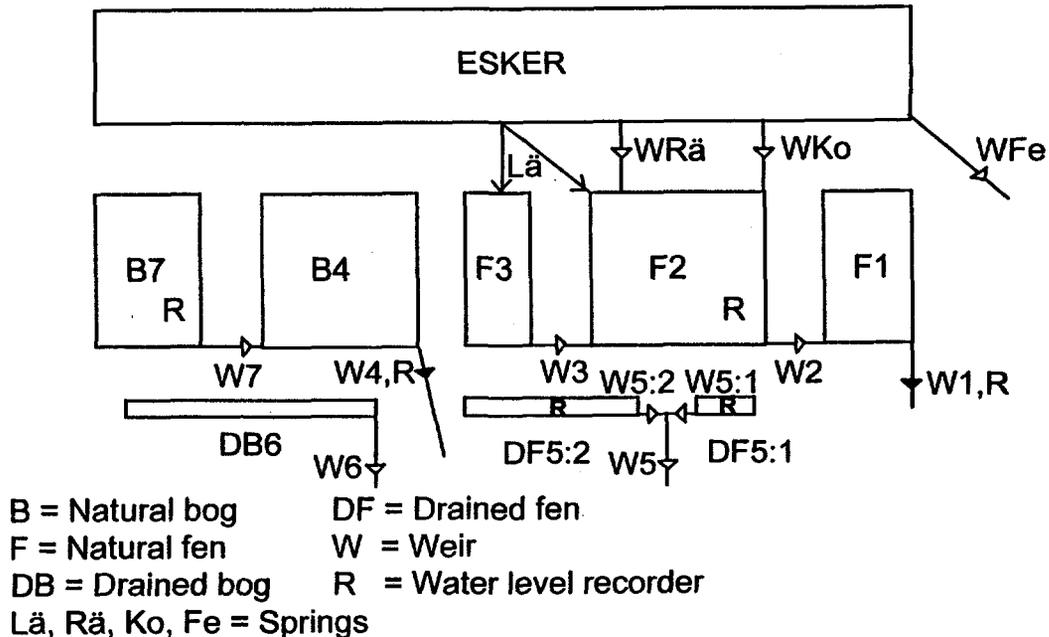
Figure 1. Hydrological monitoring in Lakkasuo.

The waters discharging from F1, F2 and F3 are measured by weir W1, equipped with a recorder, too. Only a part of the inputs from the esker can be measured. The total inputs from the esker are calculated from the water balance, assuming the same evaporation rate in the catchment of weir W4 as in the catchment of weir W1. It follows from this that the excess in weir 1 is identical to the input from esker. In the element balance calculations, 20-day moving average for the input from the esker is used. F3 receives esker inputs approximately in the same proportion as is the area of F3 of the whole area of catchment W1. F1 does not seem to receive esker inputs in any appreciable amounts, so that the remaining inputs of the total are assumed to enter the F2.

There are two drained fen catchments, DF5:1 and DF5:2, which are discharged both through W5. Today, this is equipped with a water level recorder, but no results are yet available.

The element export in runoff waters has been calculated using measured concentrations weighted with the mean runoff of the day of sampling. The runoff data for W7 is the same as in W4: the weirs W3 and W1 have also the same runoff, and for W2, runoff in W1 added by the calculated excess input from the esker is used. The leaching values for each subcatchment (F1, F2, B4) are obtained by substraction.The leaching rates of the drained catchments are not presented in this paper.

The total area of the minerotrophic catchment W1, excluding the additional poorly definable esker area is 35.2 ha, of which F3 is about 8 ha. The ombrotrophic catchment W4 is 30 ha, B4 being 20 ha and B7 10 ha.
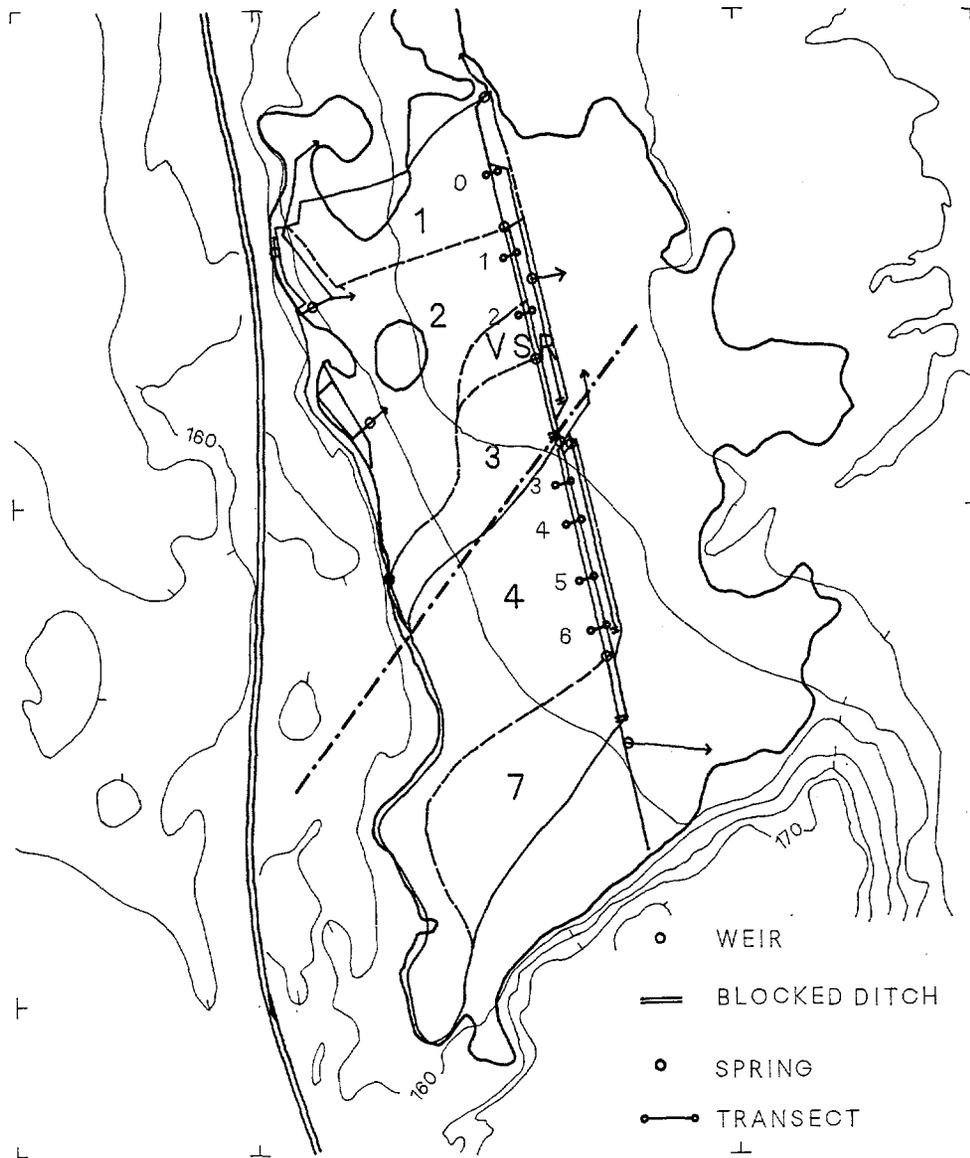
Figure 2. A map of Lakkasuo. The position of the cross section presented in Fig. 3 is marked with a broken line. VSR is a part of F2, but can be chemically joined to F3.

The list of chemical analyses from the waters in Lakkasuo, including deposition, has been very comprehensive, including more than 40 water quality variables. In this context, only carbon, N, P, Cl, S, Na, Mg, K, Ca, Fe and Al are presented (sulphur measured as sulphate-S only). The same constituents, except chloride, have been analysed from the peat layer in the Transects 1-6; these have been dated either by radiocarbon dating or by dated fire horizons (Alm & al. 1992). This allows the calculation of annual accumulation rates of these constituents, which should, in an ideal case, be identical to the difference in inputs and outputs of the catchment concerned.

Results and discussion

The subcatchments B4 and B7 are ombrotrophic by their vegetation, and by definition, should receive their inorganic elements from deposition only. However, when comparing the leaching rates of W4 (=B4+B7) with deposition data (Table 1), it seems evident that an extra source of mineral elements must exist. The leaching rate of iron is nearly four times higher than measured in deposition, and also Na, Mg, K and Ca are slightly higher.

For the subcatchment B7 data exist for a short period only, but this subcatchment gives very much lower leaching rates for many mineral elements than W4 or its other subcatchment B4, and seems to be truly ombrotrophic, i.e. both botanically and chemically. Subcatchment B4 has clearly slight mineral water influence.

At present, no peat chemistry data from the subcatchment B7 is available, but the Transect 4 has been considered as a truly ombrotrophic site and a preliminary element balance for an ombrotrophic bog can be calculated. The dated fire horizons (A.D. 1845 and 970) are assumed to be situated at 46 and 112 cm depth in the site of chemical sampling (L4L25). Approximate peat accumulation in this period has been 46 g m^{-2} a^{-1} and from the peat chemistry data in the layer 50-100 cm the accumulation rates presented in Table 1 can be calculated.

Table 1. Mean annual deposition values (1992-1994), leaching rates in W4 (1992-1994) and in B4 (1.7.1994-30.6.1995) and accumulation rates in peat (L4L25) in Lakkasuo, ombrotrophic part.

	water mm a^{-1}	C	N	P	Cl	S	Na	Mg	K	Ca	Fe	Al	Mn
						mg m^{-2} a^{-1}							
Deposition	740	1900	480	8	180	470	130	35	75	110	23	18	3
W4	340	9000	120	4	120	140	170	51	79	130	82	16	4
B7	430	7000	95	3	110	160	120	20	39	49	18	7	3
Accumulation	-	21000	190	8	n.d.	19	2	12	4	51	26	6	0,1

The results indicate that for an ombrotrophic bog, a reasonable balance for most of the elements measured can be constructed. The bog (B7) receives more nitrogen and especially S than can be found in runoff waters and peat together. This can be interpreted as an indication of the increase in the deposition of these constituents during the last decades. The peat layer used to construct the balance is in the order of 200-800 years old. Today, these elements are well retained and are gradually changing the chemistry of accumulated peat.

There is slightly more phosphorus and iron in runoff waters and peat compared with the measured input. Even a very slight groundwater input from the mineral soil below peat can explain this discrepancy, since these elements are greatly enriched in anoxic conditions prevailing in these groundwaters.

The retention of chloride is in line with the findings of Steinmann (1995), who concludes that chloride cannot be used as a conservative tracer in hydrological studies of peatlands due to the considerable reaction with peat. Organic halogens, either soluble or retained by peat, are formed.

Sodium and manganese are easily leached, as observed e.g. by Damman (1978), whereas Mg, Ca and Al are retained in considerable amount. On the contrary to findings of Damman (op. cit.), Ca is retained more effectively than Mg. In B7, the discrepancy of K, which should be well retained biologically but only poorly by peat, is probably an artefact caused by the short measuring period of runoff water quality in this site.

Similarly, an element balance for a minerotrophic catchment can be constructed (Table 2). The F3 catchment is fairly uncomplicated, since only one water source in addition to deposition has been detected and this site is not contaminated by road salt, as are F2 and F1. The Transect 2 should be the best suited site for constructing the element accumulation rates in peat. The site type in this transect is ordinary sedge pine fen (VSR), perhaps slightly less influenced by esker waters than the main part of this subcatchment.

Table 2. Mean annual deposition values (1992-1994), input values from the esker and leaching rates for a minerotrophic catchment F3 (1993-1994) and corresponding accumulation rates in peat (L2L25).

	water $mm\ a^{-1}$	C	N	P	Cl	S $mg\ m^{-2}\ a^{-1}$	Na	Mg	K	Ca	Fe	Al	Mn
Deposition	740	1900	480	8	180	470	130	35	75	110	23	18	3
Input Lä	270	1300	25	3	270	280	590	140	150	480	10	4	0.3
Input Total		3200	505	11	450	750	720	175	225	590	30	22	3
Leaching F3	700	8400	150	5	420	230	830	190	270	610	250	12	7
Accumulation	-	14300	480	15	n.d.	65	1	6	2	56	76	39	0.2

The radiocarbon datings (Hel3645 and Hel3646) show that the peat layer between 103-146 cm took 1160 years to develop (L2L25) and a peat accumulation rate of 27 g m^{-2} a^{-1} for this site can be calculated. Multiplying this accumulation rate by chemical composition values in 50-100 cm depth gives the accumulation values presented in Table 2.

The same trends of accumulation and leaching of different chemical constituents than in the bog catchment are still visible. However, here it is self-evident that waters, rich in P, Fe and Mn, must enter both the surface peat and runoff waters, since these elements are not balanced at all with the input assumptions presented. Mineral soil groundwater below peat is the likely source. If we look at e.g. iron, there is about 290 mg m^{-2} a^{-1} too much leached and retained; if the iron concentration in the groundwater below peat is 10 mg l^{-1}, 29 mm a^{-1} of groundwater input to the surface layer may explain the discrepancy. The groundwater influence can be well understood when looking at the cross section of Lakkasuo (Figure 3).
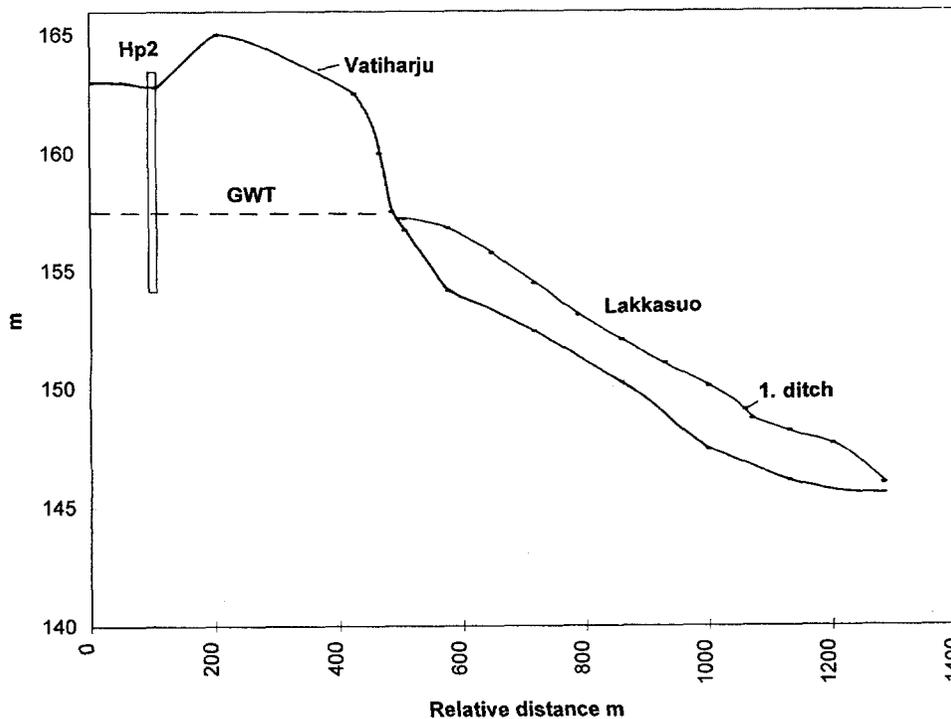
Figure 3. A cross section of Lakkasuo and Vatiharju situated in a site where ombrotrophy is changing to minerotrophy (see Fig 2.).

Siegel & al. (1995) stress the importance of weather conditions on the groundwater influence of mires. Groundwater stores are dependent on long-term infiltration conditions, whereas peatland waters are susceptible to evaporation and the water levels are greatly lowered in dry summer periods. The most recent HBV-model runs for

Längelmävesi watershed, of which Lakkasuo is a part, suggest that soil moisture stores during the growing season are decreased in a predicted climate change situation by 20-30 mm on average (see Vehviläinen & Huttunen 1994 for project description and methodoly). Decreased water levels in the mire are favouring the groundwater input by seepage to the mire surface. Due to wet winters in the climate change situation, groundwater stores are not expected to be reduced.

It is also noteworthy that especially the relative but also the absolute retention rates of e.g. Mg and Ca are poorer in minerotrophic conditions than in the bog.

Conclusions

The accumulating role of peatlands for mineral elements varies a great deal for different elements and the relative retention is usually reduced with increasing load. In addition, peatlands mobilise elements, the mobility of which is sensitive to reduced redox potential, from mineral soil below peat and affect thereby their leaching rates.

The chemistry of peat is still greatly undersaturated of S and N with respect to the present inputs. The biogeochemical functioning of mires is bound to change, once an equilibrium is attained.

Climate change is going to increase the role of groundwater seepage to the mire surface layers. The probable overall impacts of drier and warmer summers, increased groundwater influence, expected changes in vegetation and productivity etc. are to be discussed later.

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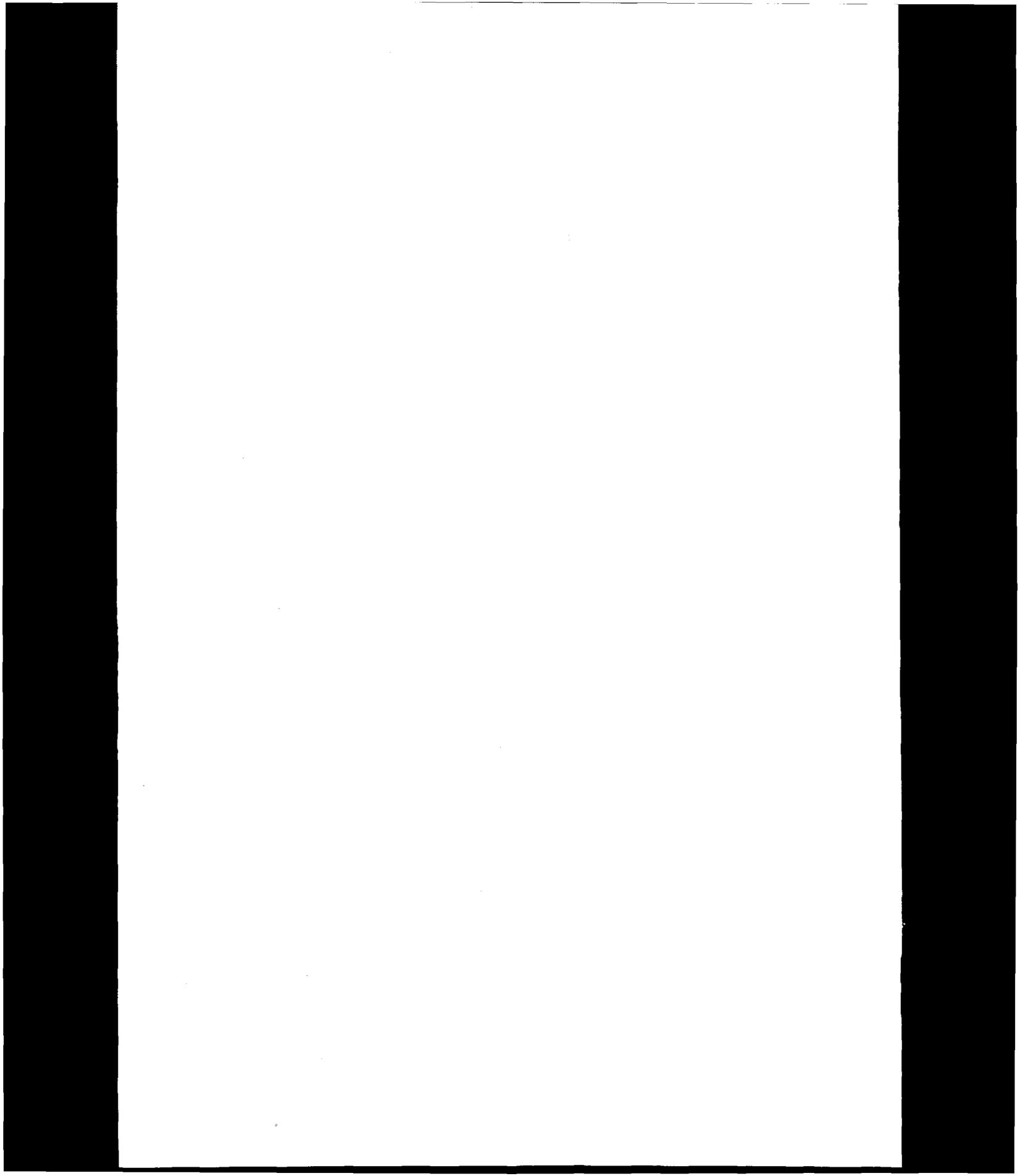
Long-term changes
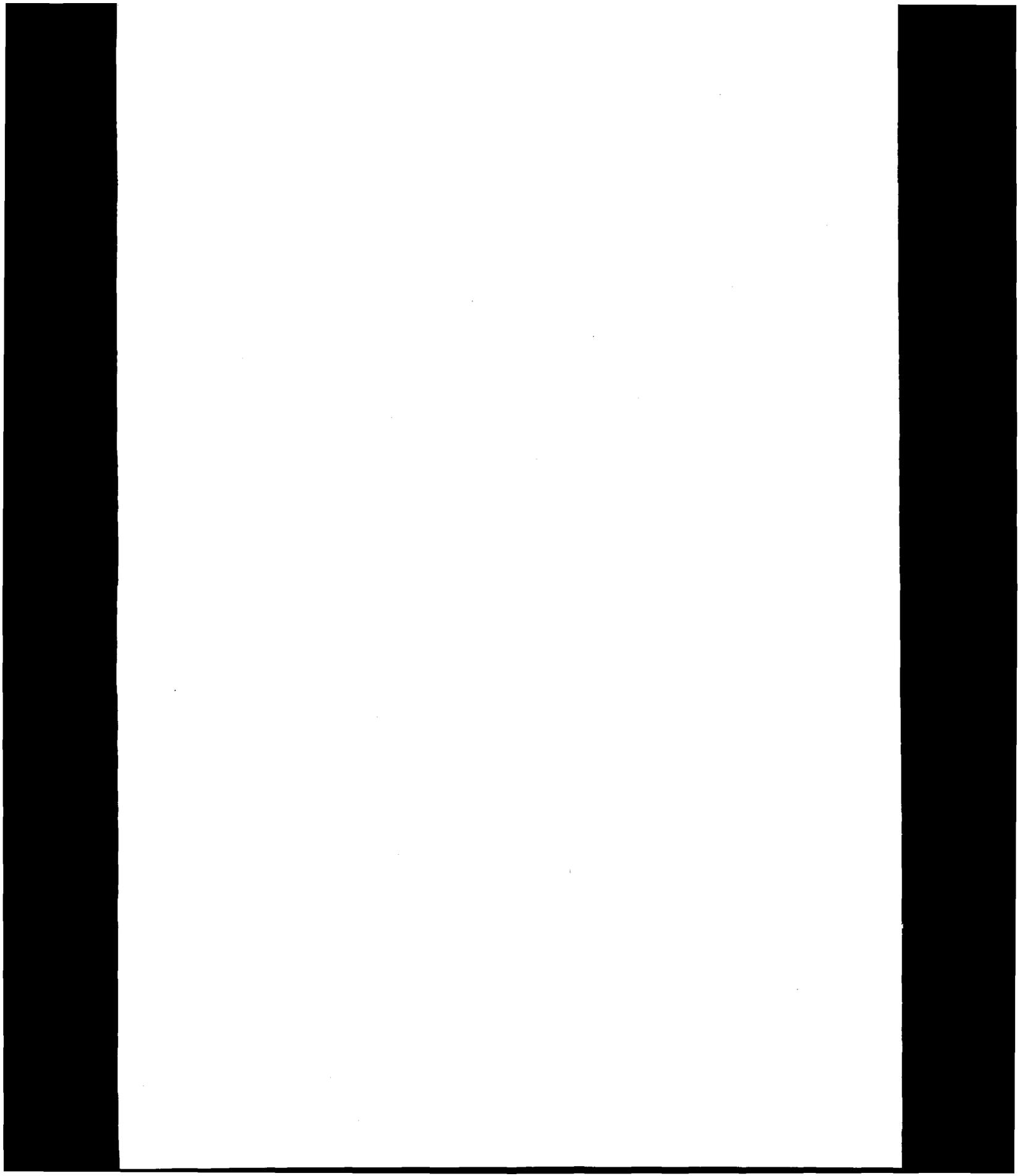
Assessing the accumulation of carbon in peatlands

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Introduction

An assessment of the integrated *amount* of carbon accumulated in peatlands requires two sorts of information: the area of peatland; and profiles of its depth both in linear measure (cm, for example) and as cumulative dry mass (kg m^{-2} for example).

It is difficult to estimate the area of peatlands. They interdigitate with other sorts of ecosystem; many peatlands are difficult to identify from the air, particularly when tree-covered; and many peatlands are remote from roads and inaccessible on the ground. Some of the uncertainties in the estimation of peatland area in the former USSR are discussed by Botch *et al.* (1995).

Area is difficult, but depth and bulk density are more so as they cannot, yet, be estimated reliably from the air and, on the ground, must be measured rather than simply recorded from visual inspection.

The statement that northern peatlands cover about 2.3 % of the Earth's land surface and contain about 450 Pg (Gt) of carbon (Gorham 1991) is, therefore, indicative only. It serves to establish that there is about the same amount of carbon in these peatlands as in the atmosphere: 600 Pg (Houghton, Jenkins & Ephraums 1990).

Improvements in estimates of area, depth and dry bulk density require more work or technical improvements in remote sensing or both, but what is needed is clear enough in principle and I consider it no further here.

An assessment of the *rate* of accumulation may be made in two main ways: using measurements of gas fluxes at the surface and of losses of organic matter in runoff; or by using some model of the accumulation process based on historical performance (coupled with bulk density profiles and knowledge of the age of the peat), or on some belief about the detailed functioning of the peatland ecosystem.

These approaches are complementary: each has strengths and weaknesses. It is with these methods that this article is concerned.

Gas flux measurements

The simplest method of measuring effluxes is the flux-box. This is a closable container placed over a part of the peatland, typically about 50-100 cm across. A collar remains permanently in the peat and the flux-box is put in a shallow trough around the top of the collar. Water in the trough seals the gap between flux-box and collar. When temperature differences alter the pressure inside relatively small amounts of gas bubble into or out from the chamber. This prevents the build up of substantial pressure differences. The earliest such measurements on peatlands are those of Clymo & Reddaway (1971) but there have recently been numerous better and more extensive measurements, with values ranged around the early ones (Bartlett & Harriss 1993 review them). One may follow the accumulation, or loss, of carbon containing gases (CH_4 and CO_2) in the static enclosed volume and calculate flux from the slope of the concentration line, the chamber gas volume, and the area. Portable infrared gas analysers now allow rates to be measured in no more than a few minutes. The instruments are sensitive enough for it to be possible to use a second approach: measure the difference in concentration between inlet and outlet with a gas stream flowing slowly through the chamber. There are many technical problems including heating by the sun and release of gas bubbles by the disturbance caused by a person approaching the site to make measurements. A large number of flux-box estimates have been made, especially of fluxes of CH_4, and relationships have been shown with temperature, watertable, abundance and nature of rooted plants (that provide a low resistance path to the atmosphere), light and so on. A recent review is by Bubier & Moore (1994). Peatland surfaces are usually a mosaic of wet hollows, less wet lawns, and drier hollows. The rates of gas flux with these microhabitats differs by a factor of perhaps 3-30, and they vary much in time as well (Bartlett & Harriss 1993, Bubier & Moore 1994).

Flux-boxes average over an area of about 0.1-1.0 m^2. Scaling measurements to average annual values for a whole peatland is therefore fraught with problems. One needs methods that will do this scaling automatically.

Suitable methods based on micrometeorology are now being applied. In the oldest of these, the flux gradient technique, profiles up to a few metres height of temperature, humidity, windspeed and gas concentration are measured and from these the flux of water vapour, momentum, and carbon-containing gas can be calculated. More recent is the technique of eddy correlation which uses the fact that transport away from the surface is by the vertical component of the wind and there must be similar fluctuations in vertical windspeed and gas concentration. Measurements must be made at a rate of about 10 s^{-1}, so expensive equipment is necessary. A third method also uses the correlated changes in vertical windspeed and gas concentration but it does so by using rapid-acting valves to collect upcurrent air and downcurrent air into two separate bags for later analysis by a cheaper method than is necessary for eddy correlation. Averaging is performed here by the valves.

All these micrometeorological methods integrate automatically though not equally to a distance perhaps 100-500 m upwind from the sampling point and over an area of 10^3-10^5 m^2. But they are technically more expensive and demanding than are flux-boxes, and are difficult to run continuously for long periods. From long runs the effects of temperature can be distilled, and if the peatland to one side of the sampling point is dry and to the other wet then with favourable wind directions some of the effects of watertable may also be inferred (Fowler 1995).

Similar principles may be applied from aircraft, and then the result is an average over 10^8-10^9 m^2.

Assessment of carbon losses in water seeping through the catotelm is only just beginning.

One or more of these methods has been used successfully for CH_4 flux with peatlands in Minnesota (Verma et al. 1992), in North America (Crill, Bartlett & Roulet 1992, Fan et al. 1992), and in Britain (Clymo & Pearce 1995, Fowler et al. 1995). Where comparisons of estimates for whole peatlands have been made using different types of method they have usually agreed within a factor of two and often to within 20 %. There are no obvious reasons why one should not use the micrometeorological methods, at least, for CO_2 as well.

Peatland processes

It is convenient to distinguish a surface layer, the acrotelm (Ingram 1978) which extends down to the depth reached by the watertable in a dry summer, from the underlying catotelm. Processes within the acrotelm are quite complex (Clymo 1992) but the overall effects, with the acrotelm considered as a black box, are these.

- The watertable moves steadily downward during a dry summer to perhaps 50 cm below the surface but its upward movement is limited, as it is in a V-notch weir, by the porous structure of the acrotelm.

- Carbon is fixed from atmospheric CO_2 by photosynthesis. Some is returned to the atmosphere by respiration as CO_2, some is converted to CH_4 and returned to the atmosphere, and some is passed down to the catotelm as peat. Overall the effect is to convert some atmospheric CO_2 to CH_4 and to remove some for a few millennia as peat. Whether the warming potential of the conversion of CO_2 to CH_4 is less than or more than the cooling potential of the removal of CO_2 as peat is not clear.

- Some plant materials (leaves of *Rubus chamaemorus* for example) decay rapidly while *Sphagnum* decays slowly. The slower the decay the greater the increase in the proportion that survives to pass into the catotelm.

- Once established the acrotelm remains of approximately constant thickness. It is not itself a peat accumulator but acts as a selective preprocessor of the plant material before passing it on as peat to the *catotelm: the true site of peat accumulation.*

It is difficult to get accurate dates for layers in the acrotelm, but Malmer & Wallén (1993) show that nitrogen is conserved in the acrotelm and its cumulative total may be used as a surrogate for the passage of time. They show that in hyperoceanic regions the rate of addition of dry mass and the rate of loss by decay, both on an area basis, are both higher than they are in less oceanic peatlands. But the net effect, seen in the rate at which dry mass enters the catotelm (p below) is much the same in both sorts of peatland.

The catotelm is permanently waterlogged - it has a much lower hydraulic conductivity then the acrotelm does - and, because O_2 diffuses through water more slowly than it is used up by micoorganisms, the catotelm is permanently anoxic. Decay is anaerobic and, for reasons that are unclear, is much slower than in the acrotelm. But decay does continue (Clymo & Pearce 1995) and this has consequences for age vs standard dry mass curves.

The sorts of 'rate' of peat accumulation

The accumulated dry mass, [dimension M], in one peatland may be much greater than in another simply because the first peatland is of much greater area. Comparisons of processes almost always require, therefore, that the accumulated dry mass be expressed on a unit area basis. There is no accepted way of making this distinction. Here I will call accumulated dry mass on an area basis the 'standard accumulated dry mass' [$M L^{-2}$]. Neither the accumulated dry mass nor the standard accumulated dry mass are rates. They have no element of time in their definition or value of time in their calculation. Sometimes there is an unspecified assumption that two peatlands whose standard accumulated dry mass is being compared have been growing for the same length of time. If that is true then the one with the greater standard accumulated dry mass must, on average, have had the greater rate of accumulation. But one cannot say what that rate was.

Rates involve time in their definition. There are three sorts of standard rate in common use. They have the same dimensions [$M L^{-2} T^{-1}$] and, therefore, may have the same units (kg m^{-2} a^{-1} for example) but for any one site each of the three has very different values and meanings. They should not, usually, be compared. Suppose we have a graph of the standard accumulated dry mass, M, plotted against time, T, since peat growth began. In general this curve rises rapidly but then turns over and rises ever less steeply. Any slope in this space will be M/T and therefore have dimensions [$M L^{-2} T^{-1}$] i.e slopes on this graph are standard rates.

When peat began to accumulate there was nothing to decay so the rate of accumulation was the same as the standard rate of addition of dry mass to the catotelm i.e. what survives passage through the acrotelm. Call it p, for productivity [$M L^{-2} T^{-1}$]. Implicit in this section is that we are considering timescales of centuries and millennia. On such scales p may be approximately constant. That is not to say that it does not fluctuate greatly during a single day and with the seasons.

Now suppose that several thousand years have passed, but p remains the same. The

peat is decaying very slowly while new plant matter is added at rate p to the top of the catotelm from the base of the acrotelm. To get the true rate of peat accumulation now we must subtract the rate of loss from p. The result, graphically, is the slope of the M vs T graph at the present time, i.e. dM/dT. To evaluate this algebraically requires assumptions about the decay process (Clymo 1992), but all such models show that this true rate is less than p, and that it diminishes still further as time passes. In the context of carbon accumulation this rate has been called TRACA, the true rate of carbon accumulation.

The third sort of rate is used when the standard dry mass and a single basal date are known. This allows the long term average (or apparent) rate of accumulation to be calculated. In the context of carbon accumulation this rate is known as LARCA or LORCA. On the M vs T plot LARCA is the chord between the origin and the present.

In summary, when a peatland starts to grow all three rates are the same. But as time passes, even if p remains the same, the other two decrease. If p is constant then TRACA $<$ LARCA $< p$.

Models of accumulation

The rate of accumulation of dry mass in the catotelm is the standard rate of addition (p) minus the total losses at all depths. In the simplest model p is assumed constant and the rate of decay is assumed to be a constant proportion, a, of what has accumulated so far (Clymo 1984). The relation between standard accumulated dry mass and time is that described above: it rises steeply at first then levels off toward an asymptote at $M = p/a$. Many, but not all, peatlands where there is sufficient evidence to test the fit to this model do have a gently convex curve (concave if plotted as standard dry mass below the surface against age of the peat). The slope at the origin is p and the convexity (or concavity) measures a. But this is not the only possible model. Clymo (1992) shows that models in which the proportional rate of decay of a mass of peat decreases as decay proceeds (and the residue becomes more refractory) fit the data as well as the constant rate model. The value of M in these models does not reach an asymptotic upper limit but continues indefinitely upward, though at an ever decreasing rate. The fitted values of p differ, though the fits are equally good. At present we do not know which value of p to believe. Until we do it is not possible to use this long term approach to make an independent comparison with the flux based snapshots. When that does become possible we will be able to see whether there have been recent changes.

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Peat accumulation in fens and bogs: effects of hydrology and fertility

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Introduction

Most of the solar energy fixed by plants is stored in dead organic matter on the soil surface until decayed by decomposing organisms at rates that vary greatly from place to place. Decay will eventually equal production since the amount of organic matter decayed increases with its mass whereas production remains unchanged, provided all other conditions remain constant (Olson 1963).

In peatlands or mires organic matter has accumulated, often to a considerable depth, and, most of it is now waterlogged and anoxic. In a classic paper that greatly influenced our outlook on peat accumulating ecosystems, Clymo (1984) clearly demonstrated how even in bogs organic matter accumulation is eventually limited by decay. In these ecosystems peat accumulation will cease when input to the catotelm, the permanently anoxic peat, equals decay within it. Although Clymo (1984) established this for raised bogs, it clearly applies to all peatlands.

This leads to three questions:
1) Assuming a stable climate, is input to the catotelm eventually balanced by decay in the catotelm because decay increases with peat depth , or do changes in productivity at the surface and decay in the acrotelm play an equally important role by reducing input to the catotelm? In other words, is maximum peat depth controlled solely by biological processes, or also by hydrological changes during bog development that affect production and decay.
2) Does climate, apart from its obvious effect on productivity and rate of decay, also control the maximum depth of peat accumulation through its effect on hydrology?
3) Are most peatlands still accumulating peat or have many reached a steady state?

These questions cannot be answered for peatlands in general. Peatland type, age, and climatic conditions all affect the answers to these questions. For raised bogs, Clymo (1984) appears to believe that bog growth is primarily limited by biological processes. At least he states: "The usual view that the maximum possible depth is determined by climate operating through hydrology may be incorrect, though hydrology may have an indirect effect on the value of pc, the rate of input to the catotelm at the bog centre."

The purpose of this paper is to:1) consider the effect of climate and hydrology on maximum peat depth, and 2) compare controls on peat accumulation in fens with those in bogs.

Organic matter accumulation

In well-drained habitats with a steady litter production (L) and a decay rate (α^1) organic matter will accumulate at a rate: $dX/dt = L - \alpha X$, , where X is the amount of organic matter present at time t. The amount of organic matter on the soil surface will increase with time following an exponential curve

$$X = (L/\alpha)(1 - e^{-\alpha t})$$

It will reach a maximum mass at steady state, X_{ss}, when $dX/dt = 0$ and $X_{ss} = L/\alpha$ (Olson 1963). Fig. 1 illustrates organic matter accumulation on the forest floor of an *Abies balsamea* forest in western Newfoundland (Damman 1971). For an exponential model, it will take $- \ln(0.01)/\alpha$ or $4.6/\alpha$ years to accumulate 99% of the mass present at steady state, i.e. 96 yrs in this *Abies* forest. The maximum amount of organic matter that accumulates on a site, as well as the time required for it to accumulate, varies greatly with litter fall and decay rate (Olson 1963). The latter will depend among others on climate, habitat conditions (Meentemeyer 1978, Pastor & Post 1986), and litter quality, e.g., litter morphology, nutrient and lignin content (Swift et al. 1979, Melillo et al. 1989).
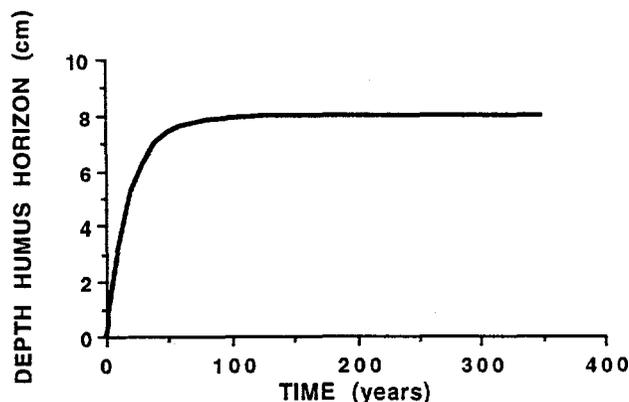
Fig. 1 Organic matter accumulation in a well-drained *Abies balsamea* forest in western Newfoundland (Damman 1971) with combined litter and root fall of 337 g m^{-2} yr-1, decay rate $\alpha = 0.052$, and bulk density of humus horizon = 81.3 g L-1.

Anaerobic decay is about 2 orders of magnitude slower than aerobic decay. Because of this very slow anaerobic decay, lack of oxygen becomes a master factor, and it completely overrules the effect of other factors that affect decay under oxic conditions. As a result, under anoxic conditions decay is so slow that litter input primarily controls organic matter accumulation (Fig.2).

In peatlands, organic matter decays aerobically in the surface peat or acrotelm and anaerobically in the waterlogged catotelm. Peat accumulation in these systems is not so much controlled by production at the surface but rather by the amount of organic matter added annually to the anoxic peat mass, i.e. by the fraction of the organic matter

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[1] α is used for the decay parameter, following Clymo (1984) instead of the more conventional k to avoid confusion with k for hydraulic conductivity, which is used later in this paper.

produced at the surface that is left after decay in the acrotelm (Clymo 1984). At this point it is useful to compare aerobic decay in bogs and fens. Fig. 3 shows this using decay rates typical for these habitats (Clymo 1965, Bartsch and Moore 1985, Johnson and Damman 1991).
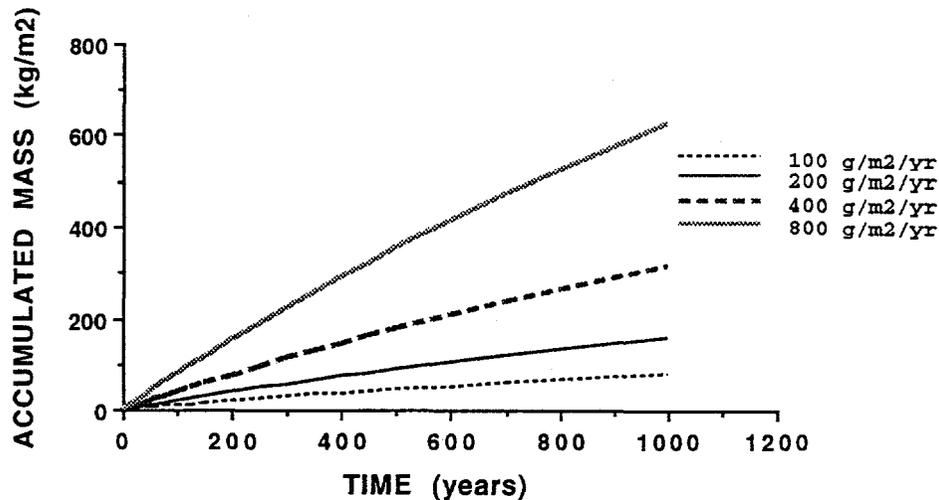
Fig.2 Organic matter accumulation under anoxic conditions for litter inputs of 100, 200, 400, and 800 g m-2 yr-1, assuming $\alpha = 0.0005$ for anaerobic decay (Clymo 1984). At steady state and assuming a bulk density of 100 g L-1 , organic matter will have accumulated to a depth of 2, 4, 8, and 16 m, respectively. It will take about 10 000 yrs to accumulate 99% of the mass at steady state.

In ombrotrophic bogs peat accumulates aerobically for hundreds of years (Fig. 3). During this period the peat structure breaks down, its density increases (Johnson et al. 1990), and its permeability decreases (Romanov 1961, Baden and Eggelsmann 1963, Päivänen 1973). This humified peat becomes waterlogged long before a steady state is reached. This means that under oxic conditions decay in the acrotelm is not complete and some organic matter is passed on to the catotelm. Here it accumulates under anoxic conditions until eventually the input into the catotelm is balanced by decay within it (Clymo 1984). Because of the very low rate of anaerobic decay, peat continues to accumulate for thousands of years (Fig. 2).

Productivity is higher in fens than in bogs but aerobic decay rates are much higher in fens (Reader & Stewart 1972, Bartsch & Moore 1985). Therefore, in spite of the higher productivity, peat accumulates under oxic conditions only to a limited extent[2] before a steady state is reached between production and decay (Fig. 3). Expressed as peat depth this difference is even more striking because of the much higher bulk density of fen peat compared to bog peat (Tolonen 1977, Tolonen et al. 1982, Tolonen & Ijäs 1982). In addition, the balance between production and decay will be reached in a much shorter time in fens, since for an exponential model 99% of the maximum depth will be reached in 4.6 / α years, and the aerobic decay rate (α) is so much higher in fens than bogs.

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[2] In fen hummocks, the acrotelm is often thicker because the surface peat is held up by dwarf shrubs and tree bases and, therefore, not in contact with minerotrophic fen water.
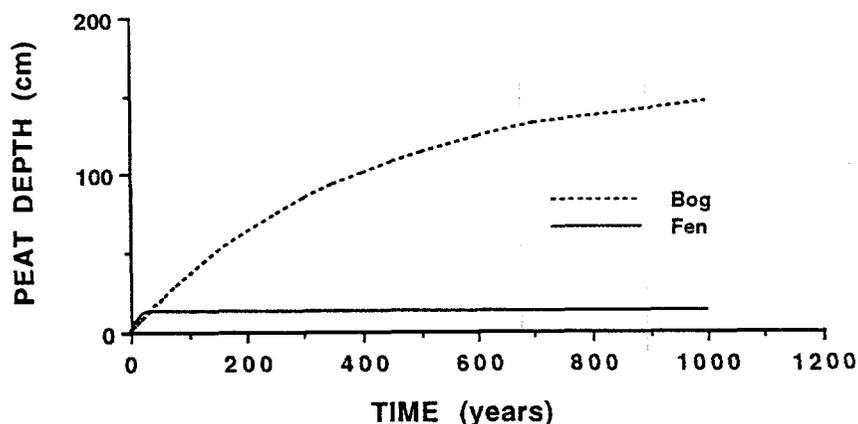
Fig.3 Organic matter accumulation under oxic conditions in a bog and fen. Peat will accumulate above a summer water table to a depth of 143 cm in the bog and only 11 cm in the fen. For the bog: productivity = 100 g m-2 yr-1; decay rate α = 0.0028, and bulk density Db = 25 g L-1; for the fen: productivity = 632 g m-2 yr-1; decay rate α = 0.1109, and bulk density Db = 50 g L-1.

Obviously decay rate varies among fens, but even in nutrient-poor fens, all organic matter produced can decay within a relatively thin oxic horizon (Fig. 3). This means that the water table will not rise into peat that has accumulated under oxic conditions, as it does in peat bogs, because all peat will decay before it becomes anoxic. Peat can become anoxic before it is decayed only if the water table rises because of external causes, i.e. if drainage water backs up or if water seeps in at a higher level. Under these conditions, peat accumulation can continue until decay in the catotelm balances the organic matter input to the catotelm, as it does in bogs.

Therefore, water level rise controls peat accumulation in both bogs and fens, however, whereas water level rise in bogs is an autogenic process resulting from peat accumulation in the acrotelm, in fens water level rise is an allogenic process, i.e. controlled by external factors.

Ground-water mounds in peat deposits

In fine textured deposits with low hydraulic conductivity, such as clays, a ground water table can rise above that in surrounding deposits. These so-called ground-water mounds are maintained by a balance between recharge by influent seepage and discharge by effluent seepage (Tolman 1937, Marino 1974). If the influent seepage is derived from rainfall, then for any given size and permeability of the deposit the dimensions of the ground-water mound will depend on the climatic conditions, specifically the amount of precipitation that is not lost by evapotranspiration. For each combination of hydraulic conductivity (k), long-term supply of surplus moisture (w) and distance to the discharge site (L), the ground-water mound will have a critical profile (Ivanov 1981), which indicates the level to which the water level can rise. Its maximum height can be expressed by the equation (Childs 1963, Ingram 1982):

$$H_{max} = L \sqrt{\frac{w}{k}}.$$

In peatlands, hydraulic conductivity is high in the surface horizons and decreases with depth and humification (Romanov 1961, Baden and Eggelsmann 1963, Päivänen 1973). In the catotelm the hydraulic conductivity is low enough to maintain a ground-water mound in humid climates (Ingram 1982).

However, peat has to accumulate first in peatlands before a ground-water mound can develop. Thus, there exists a dynamic equilibrium between peat accumulation and growth of the ground-water mound (Damman 1986). As long as the bog surface is below the critical profile of the ground-water mound, the water level will rise as humified peat accumulates. However, once the ground-water mound has reached its critical profile, it cannot rise any further and the catotelm cannot become thicker. Therefore, from this point on peat can accumulate only aerobically.

If this happens when production is not yet balanced by decay in the underlying peat, the acrotelm will become thicker. A deeper acrotelm will increase decay because organic matter will remain longer in the oxic peat, and it will also reduce *Sphagnum* productivity by raising the surface higher above the summer water table.

Ultimately, decay will limit the depth of the bog in this case too. However, there is a fundamental difference with the previous case. Here the critical profile of the ground-water mound first limits the depth of the catotelm and decay within it, and then the bog surface continues to grow upward until production equals decay in both catotelm and acrotelm.

Peat accumulation in bogs

Peat will continue to accumulate in raised bogs until either production is balanced by decay (Clymo 1984) or until the acrotelm reaches its maximum depth above the ground-water mound that can be maintained under the existing climatic conditions. Whichever occurs first is the primary factor limiting peat accumulation.

Peat stratigraphy can show the changes in peat accumulation over time. Therefore, a comparison of peat stratigraphy under both scenarios will be useful. To simplify the comparison, I will assume that the climate did not change and that fire or local hydrological disturbances have not affected peat stratigraphy.

Maximum peat depth controlled by production and decay alone

As long as the catotelm surface remains below the critical profile of the ground-water mound the water table will rise as humified peat accumulates at the base of the acrotelm. This will continue until the bog reaches its maximum height and peat additions to the catotelm will balance decay within it (Clymo 1984). The amount incorporated into the catotelm will slightly increase as peat depth increases to compensate for decay within the catotelm. Therefore, the peat added to the top of the catotelm, and thus decayed within the acrotelm, will show roughly the same degree of humification over time. This means that, considering the last 1000 yrs or so, peat within the acrotelm will gradually increase in humification with depth, and catotelm peat will be at least as decayed as that in the lower part of the acrotelm.

A peat profile in the bog center should show this pattern of decay if peat accumulation is limited by the balance between production and decay, or also in immature bogs still below the critical profile of the ground-water mound. This appears to be the case in bogs in the British Isles (Ingram 1982, Clymo 1984).

Maximum peat depth controlled by climate and its effect on the height of the ground-water mound

If peat accumulation is limited by the climatic moisture surplus, the critical profile of the ground-water mound will coincide with the upper level of the catotelm. When the ground-water mound reaches this level, and thus its maximum height, organic matter additions to the catotelm will still exceed decay within it. Since the catotelm cannot increase in depth beyond this, organic matter will decay longer in the acrotelm, and productivity at the surface may decrease, until organic matter added to the catotelm will equal decay within it. These changes in degree of decay of the peat will be reflected in the peat stratigraphy.

Until the catotelm reaches its maximum depth, peat accumulation will proceed as in the previous scenario, where it was controlled only by production and decay. However, after the catotelm reaches the critical profile of the ground-water mound, less peat will accumulate, and it will be more decayed. Therefore, a clear change should be visible within the catotelm from more decayed peat in the upper catotelm to less decayed peat below it. The peat in the lowermost acrotelm will be similar to or only slightly less decayed than that in the upper catotelm, but much more decayed than that below it. In a peat profile, a horizon of decayed peat will extend from the lower acrotelm well into the catotelm. The lower boundary of this humified horizon originates from the time the catotelm reached its maximum depth; its thickness will depend on how long ago this happened.

Bogs that have grown up above the critical profile of the ground-water mound will be further distinguished by a deep acrotelm and low summer water level. In contrast, bogs whose development is not limited by the maximum dimension of the ground-water mound will be wet and have a shallow acrotelm.

The maximum height of many raised bogs in eastern North America appears to be limited primarily by climate and its effect on their ground-water mound rather than solely by the balance of production and decay. This is suggested by the following:
1) In almost all raised bogs outside the extremely oceanic zone, a well-developed humified horizon occurs in the upper catotelm (Damman 1988). This horizon overlies less decayed peat.
2) The convexity of raised bogs decreases with increasing continentality of the climate (Damman 1979). It is difficult to explain this geographic pattern by differences in age or productivity. With the exception of some oceanic raised bogs, they are also considerably lower than those with comparable precipitation in Sweden (Granlund 1932), where summer temperatures, and thus evapotranspiration, are much lower..
3) Peat accumulation rates for recent peat are much lower than those for peat accumulated during earlier stages of bog development (Damman 1988). This is also shown in a peat profile from western Newfoundland (Fig.4) in which accumulation has been much lower during the last 2000 yrs than before.

There are also indications that the maximum height of raised peat bogs in other regions is limited by climate rather than solely by the balance between production and decay. Granlund (1932) established a clear relationship between the elevation of the bog surface and the annual precipitation and dimension of the bog. A reduced peat accumulation rate in the upper 2 m of peat, similar to that mentioned above, was also observed in the finnish raised bog Häädetkeidas (Damman et al. 1992).

ACCUMULATION RATE (g/m2/yr)
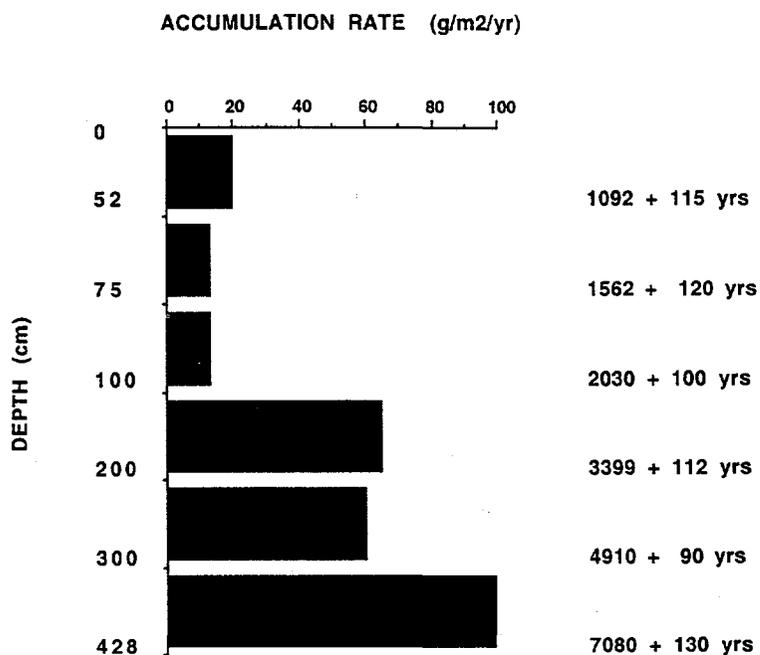
Fig.4   Changes in peat accumulation during development of a raised bog in western Newfoundland. Note the low accumulation rates during the last 2000 yrs. Presumably this represents peat that has accumulated after the critical profile of the ground-water mound was reached. It results from increased aerobic decay and probably also reduced productivity at the surface. The summer water level at the core site was 37 cm below the surface. Accumulation rates are average annual accumulations based on oven-dry weight of peat present between radiocarbon dated levels. Dates are calendar years determined from calibrated ^{14}C dates.

Peat accumulation in fens

Aerobic decay is much faster in fens than bogs. Consequently in fens, peat does not accumulate above the water table to the extent that a ground-water mound can develop. Therefore continued accumulation above an existing water table will depend on a rise in the water table controlled by external conditions.

Peat accumulation, measured as increases in peat depth, will cease when the amount of peat left after decay in the oxic zone balances the decay in the underlying peat. As in peat bogs, the organic matter additions to the catotelm will have to increase with peat depth to keep up with decay in the catotelm. This means that, assuming constant productivity, the acrotelm will become shallower as the fen peat increases in thickness, so that peat will be less decayed when it is incorporated in the catotelm. As a result, apparent accumulation rates (Tolonen et al. 1992) based on peat in the upper catotelm will increase slightly as peat depth increases.

As long as fen litter accumulates under anoxic conditions below a water table, the peat will be poorly decayed and peat accumulation rates will depend mostly on organic matter production, and thus on fertility of the fen water. However, once peat starts to accumulate above a water table most or all newly produced peat will decay in the

acrotelm, and peat depth will increase only slightly or not at all. Therefore, peat accumulation rates in fens are generally low compared to bogs (Tolonen & Turunen 1995), and fen peat is usually highly decayed near the surface.

Fen peat will continue to accumulate if the water level rises. Consequently, variation in peat accumulation among fens in the same area depends mostly on local hydrological conditions rather than on differences in production and decay. This obscures geographic patterns in peat accumulation in fens. In addition, peat accumulation rates will be poorly correlated with habitat type and nutrient regime.

Conclusions

The maximum depth of bogs can be limited by decay within the accumulated peat as well as by the maximum elevation of the ground-water mound that can be maintained in the peat deposit. Apparently, the former can control bog development in cool, humid climates, but the latter seems to become the limiting factor as the climatic moisture surplus decreases.

If decay limits the maximum peat depth, the production of *Sphagnum* , the major component of bog peat, will not decrease as a bog matures. In contrast if climate limits peat depth, *Sphagnum* production decreases as the bog matures and the acrotelm reaches its maximum elevation above the critical level of the ground-water mound. Consequently, present productivity will underestimate productivity during earlier stages of bog development under similar climatic conditions. This needs to be considered in regional comparisons of productivity and in calculating decay losses in dated peat profiles.

Large amounts of carbon have been sequestered in temperate peatlands. However, peatlands are carbon sinks only until they are mature and decay balances production. Many peat bogs may have reached an equilibrium between production and decay, especially in those where climate limits maximum peat depth. Fens reach a balance between production and decay in much shorter time than bogs. Therefore, most fens will no longer actively accumulate peat, although they may appear to do so when only the surface peat is considered. An exception are those fens in which the water level rises as peat accumulates.

Since not all peatlands can be considerd carbon sinks, attention needs to be focused on the developmental stage of peatlands. Carbon balance studies, such as those of the Suo-SILMU project, are critical in this respect. Peat stratigraphy can be a useful tool in some cases. Studies on decay in the acrotelm are potentially valuable, but their accuracy is presently hampered by the difficulty of reliably dating modern peat.

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Spatial reconstruction of C accumulation and CH_4 emission during Holocene in an ombrotrophic mire complex

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Introduction

In northern mires atmospheric carbon dioxide (CO_2) has been sequestered in peat and some has been converted to methane (CH_4) and returned to the atmosphere. According to the ice-core records, atmospheric concentrations of these gases have varied in accordance with changing temperature (Lorius et al., 1990). With the development of peatlands contributing either as a mediator or as a positive feedback of such atmospheric changes (Prinn, 1994). Most previous estimates of the long-term C fluxes in peatland ecosystems rest on analyses of single peat-cores. However, mire C accumulates as peat both in time and in space, and at rates dependent on dynamic interactions between landscape, substrate, vegetation, and climate. Thus, traditional one-dimensional (vertical) modelling of the carbon cycle cannot give a full account of peatland C dynamics.

In this report, we examine the temporal and spatial dynamics of peat C accumulation using a number of peat cores of known age and mass from a single representative mire. We link carbon stocks, rates of accumulation, and CH_4 emissions to the landscape history and incorporate bog dynamics to model C sequestration in the developing mire during the last 9,000 years.

Material and methods

Reksuo bog, some 577 ha in area, is located on the south-west coast of Finland (60°30' N, 23°16' E), 130 km west of Helsinki, at 93 m above sea level (a.s.l.) (Fig. 1). The region lies in the southern boreal zone in the phytogeographical system of Europe. The mire is concentrically domed, largely natural and supports ombrotrophic vegetation; the central bog plain is nearly flat with steep slopes only at the margins of the peat body. The mean peat depth is 4.3 m and the maximum depth 7.3 m. The substrate throughout is clay, and the mire is surrounded by flat clay lands, currently under cultivation. A more detailed decription of the site and its surroundings is published elsewhere (Korhola, 1992). In the years 1989-1993, morphometric measurements and peat analyses were made at 120 points along a network of transects (Fig. 1). The samples for radiocarbon dating and bulk density analyses were taken from 16 peat columns using a box sampler (80 x 80 x 1000 mm) for the upper (~1 m) and a Russian peat sampler (85 x 500 mm) for the lower peat section. A total of 44 ^{14}C dates was obtained for the reconstruction at the Helsinki University Dating Laboratory.
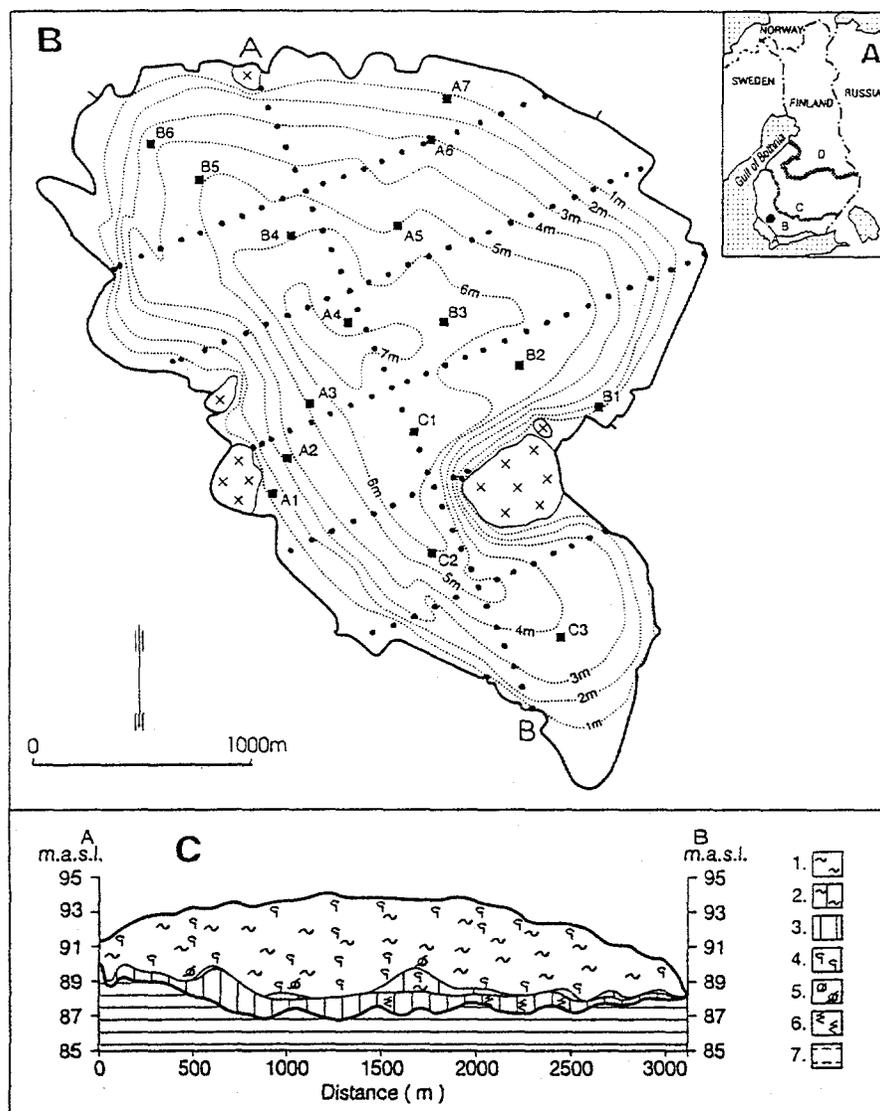
Fig. 1. (A), The location of Reksuo in relation to the regional distribution of mire complex types in Finland: (A) plateau bogs, (B) concentric bogs, (C) eccentric bogs, (D) aapa fens. (B) Depths of the peat deposit in Reksuo (dotted lines), based on measurements at 120 points. The map is derived from unpublished material supplied by the Geological Survey of Finland (by Dr. C.G. Stén), supplemented by the authors' own data. The coring sites, at which bulk density measurements and radiocarbon datings were made, are shown with black squares. Areas marked with crosses refer to rock exposures. (C) Cross-section along the transect A-B as indicated in Fig. B, showing the generalized peat stratigraphy: (1) *Sphagnum*, (2) *Sphagnum-Carex*, (3) *Carex*, (4) *Eriophorum*, (5) *Scheuchzeria*, (6) *Equisetum*, (7) mineral substrate (clay).

Parameters used in modelling temporal change

We divided the mire into three functional entities: plateau, slope and lagg. Each entity is considered separately in our approach, because of the inherent differences in nutrient supply, apparent accumulation and mineralization rate. Lateral expansion is driven by water flowing out of the sides of the peat body and submerging a strip of land just beyond its boundary causing peat to begin to form there. In Reksuo, this process seems largely to be determined by the topography of the mire bottom and its surroundings (Korhola 1994). The lateral growth rate, G_L (m/yr), of the rimming fen lagg, and thus also the mire expansion, was calculated as a function of upslope gradient, S. The values of S were derived from the mire bottom radiocarbon dates and topography of the mire basin, as elevation (cm) per distance (m), data presented in Korhola (1994):

$$G_L = 773.6 \ S^{-1.17845} \tag{1}$$

At slopes with S close zero, the lagg expansion was simply constrained to a maximum of 6 m yr^{-1}, based on prior experience (Korhola, 1992, 1994). The width of the fen-lagg (margin effect) is also constrained to be 50 m at maximum.

Grid layer techniques of ARC/INFO (Geographic Information System, GIS) are utilized in the modelling of mire growth. Spatial algorithms of costdistance and eucdistance functions (ESRI 1992) were utilized in the otherwise tedious grid processing. We, therefore, introduce term "cost", matching the logic of those functions, to describe the amount of mire expansion in new cells, with initial cost 100%, during one model time step (dt). The cost (per metre) that the fen-lagg has to "pay" due to the previously mentioned constrains for spreading into the area of the neighbouring upslope grid cells (width W, m), is defined by Eq. 2, and the remaining cost of accumulation over the entire layer by Eq. 3.

$$Cost_t = \frac{U_{t-1}}{dt \ G_L}. \tag{2}$$

In the equation U_{t-1} is the free space in the cell at the previous step. At each time step (dt=100 yr is applied here) the lagg (currently occupying cell j) may spread into new cells within a distance [Dist(j)<Dist(i)] in its neighbourhood reducing the free space partly or totally in the cells. Cumulative cost for each cell ($CCost_i$) then manages the potential mire expansion into cell (i) from many nearby cells (j) at time t:

$$CCost_{i,t} = \sum_{j}^{Dist(j)<Dist(i)} CCost_j + Cost_{i,t} \frac{W}{2}. \tag{3}$$

If CCost gets a value larger than 0 ($\leq$ 100%) for any of the available grid cells, the lagg area is expanded and, finally, U_t of those cells is updated.

Vertical peat growth was modelled using the actual or true net rate of carbon accumulation (ARCA). It is lower than the long-term (apparent) rate carbon

accumulation (LORCA), because of the slow plant decay in the anoxic deeper peat layers.The net rate can only be estimated by means of peat accumulation models, in which a concave relation is assumed for age *vs.* depth plots in peat profiles (Clymo, 1984; Tolonen *et al.*, 1992). We used the following equation to calculate ARCA:

$$A = pe^{-\alpha t} \tag{4}$$

where A is net rate of dry mass accumulation (g $m^{-2}yr^{-1}$), p is rate of dry matter addition (input) (g $m^{-2}yr^{-1}$), α is decay cofficient as a proportion (yr^{-1}), and t is time (yr). Since the peat accumulation model used here (Clymo, 1984) is mainly applied for *Sphagnum* peats, we considered the bottom *Carex* section and the upper *Sphagnum* layer as separate units. We estimated the production and decay coefficients by fitting Eq. 5 to dated peat bulk density profiles from three cored peat columns, one in the area of each above-determined morpho-units in the present bog. The parameter values for *Carex* peats were taken from Tolonen *et al.* 1994.

The course of ARCA for the entire bog system according to the empirically determined α and p was spatially integrated from the simulated morpho-units at each time step in the GIS model. The result of such simulation over 8600 years, however, greatly underestimated the accumulation for the total peat mass relative to the present bulk (net) volume of the bog. This strongly suggests that the rate of accumulation has not been constant at different parts of the bog over time. It also seriously questions the representativeness of a single core analysis in estimating the carbon fluxes in peatland ecosystems.

Model tuning

To fit the theoretically predicted data to the observed one, we repeated the above procedures after optimising the parameters α and p on the basis of the end product, i.e. the present bog After modification, the calculated loss by decay in the central plateau was 18.5 and 13.7 g C m^{-2} a^{-1} for the margin lagg. In the marginal slope it was clearly higher, 37.5 g C m^{-2} a^{-1}. We found a mean 24.6 ± 6.2 for LORCA (range 8.7 - 33.7). Parameter values are revealed in more detail in Korhola *et al.* 199x.

Results and discussion

The mire started to form from a single central locus at the lowest elevation value around 8,600 years ago (Fig. 2). Lateral extension was fastest between 4500 and 3000 BP, after which there was a decline in the rate of expansion (Fig. 3A). The rate of volumetric extension of Reksuo was very slow in the early Holocene until 3000 BP although by this time it had reached over 80% of its present mire area. As late as 4500 BP the mire was a thin minerotrophic fen, and after that about 5.5 m peat accumulated in the deepest point of the central plateau. The results clearly suggest that about 50% of the current landscape C stock was accumulated during the last 3000 BP during the ombrotrophication (Fig. 3B). Between 8600 BP and 3000 BP, C accumulation was primarily controlled by variations in horizontal growth; *Carex* fens dominated, and only about 25% of the total peatland C was sequestered over that period.

We adopted the annual CH_4 emissions given by Martikainen *et al.* (1995) for our simulated bog and fen areas. Under the simulated mire development, 239x10^6 kg organic matter decayed, corresponding to 120x10^6 kg C. Area-based estimate of CH_4
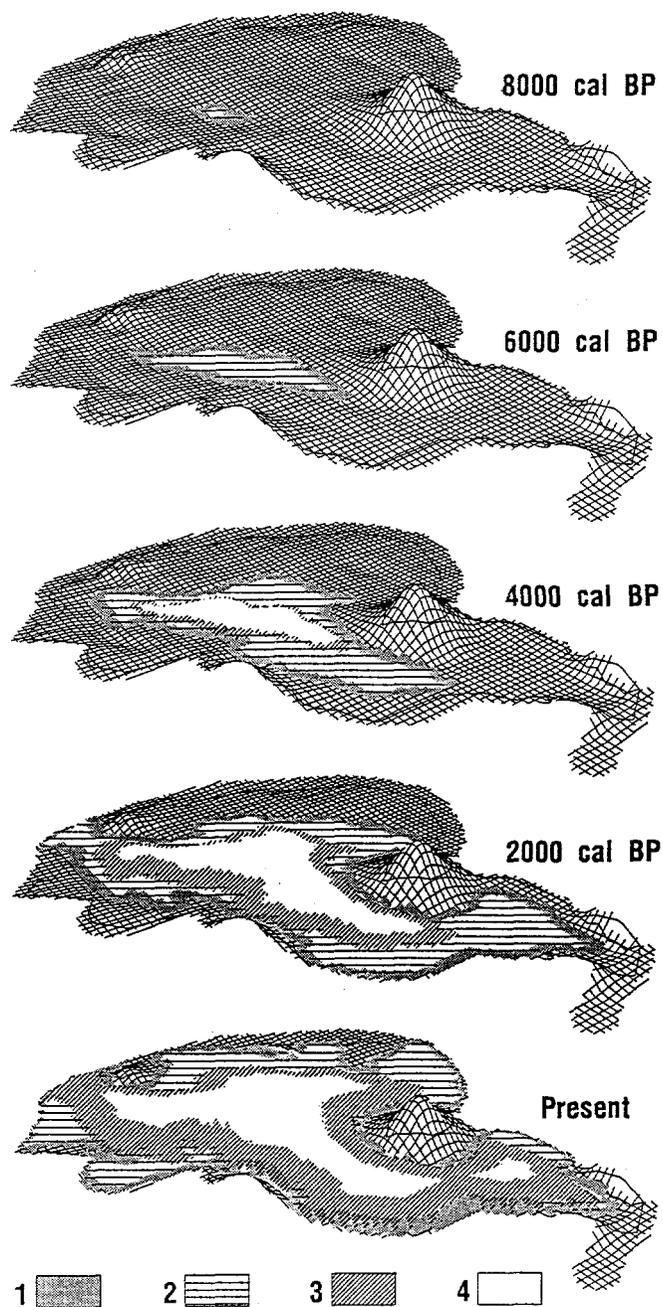
Fig. 2. 2000 year stages of mire growth reconstruction over the clay plateau digital elevation model (DEM), pixel size 10x10 m. The 50x50 m mesh, wrapped over the DEM, covers the full extent of Reksuo; present natural mire edge is indicated with a thin line. Shadowed areas of the functional entities: 1 = lagg, 2 = *Carex* fen, 3 = slope, and 4 = *Sphagnum* plateau.

emission from the mire complex over the same period was 334×10^6 kg CH_4-C, i.e., about three times more than the peat loss (Figs 3B and 3C). This apparent excess of methane can originate in the anaerobic part of the vascular plant rooting zone due to the production of easily soluble root exudates and fine root litter. The fact that observed CH_4 fluxes in earlier studies are strongly seasonal and 3-5 times greater than could be inferred from the decay rates (α) of the anoxic peat (e.g. Warner et al., 1993), gives credence to our model.
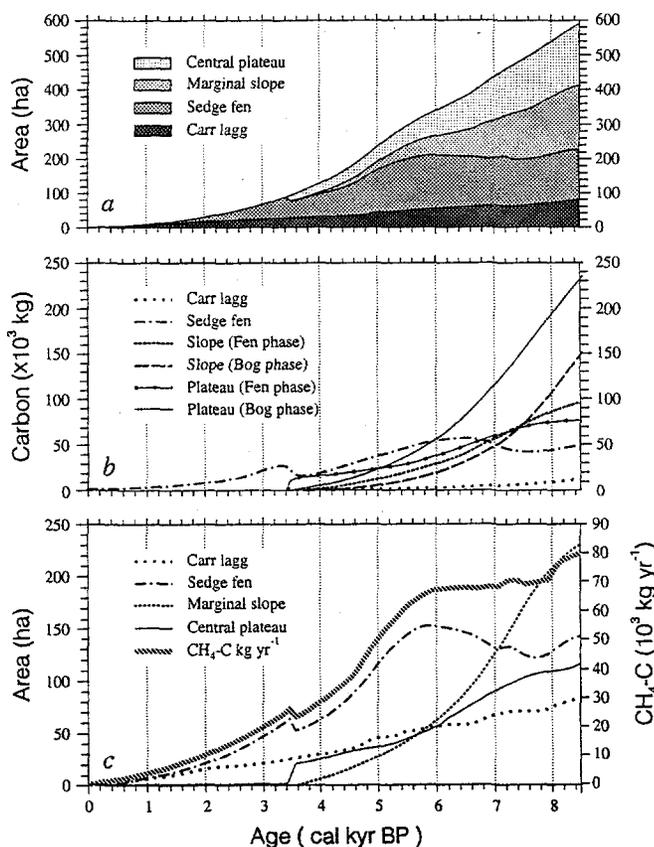
Fig. 3. (A) Areal evolution of Reksuo bog cumulated from areas of the different functional entities. (B) three-dimensional ARCA simulated separately for each entity. (C) specific areas of the functional entities with area-based estimate (Martikainen et al. 1995) of carbon loss in CH_4.

Conclusions

Climatic change due to ongoing shifts in the concentration of atmospheric gases could disturb the natural development of the northern mires, and directly affect the net export of gaseous carbon compounds from the peatlands (Siegel et al., 1995). Annual CH_4

emissions may increase twofold during an exceptionally warm summer with persistent high water tables (Martikainen et al., 1995). On the other hand, if water tables would decline in the northern hemisphere due to climate, and the bog area expands northwards over the present aapa-mires (Boer et al., 1990). Then the increased landscape carbon sequestration and reduced CH_4 emission in bogs would tend to counteract the greenhouse effect.

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Greenhouse impact of Finnish peatlands 1900 - 2100

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Introduction

Northern peatlands are significant in regulating the global climate. While sequestering carbon dioxide (CO_2, ca. 100 Tg C a^{-1}), these peatlands release ca. 24-39 Tg methane (CH_4) annually to the atmosphere (Laine et al. 199x). This is 5-15 % of the annual anthropogenic and 10-35% of the annual natural CH_4 emissions to the atmosphere. The greenhouse gas balance of peatlands may change as a consequence of water level drawdown after land use change, or if summers become warmer and drier, as has been predicted for high latitudes after climatic warming.

Currently, some 15 million hectares of northern peatlands and other wetlands have been drained for forestry. More than 90% of this area is found in Scandinavia and the former Soviet Union (Paavilainen and Päivänen 1995). The area drained annually has, however, been declining during the last two decades and, in Finland for instance the annual drained area of nearly 300 000 hectares in the late 1960's has decreased to ca. 35 000 hectares in the early 1990's (Aarne 1993).

Radiative forcing is the change in the radiative energy balance at the tropopause and it is the driving force behind the greenhouse effect. It is a common quantity for most greenhouse gases and takes into account the dynamics of the greenhouse impact. Radiative forcing model (Savolainen and Sinisalo 1994) was used to compute the greenhouse impact of the drainage of the peatlands, combining the effects of CO_2 and CH_4 balances; N_2O was not included in the calculations because its contribution is minor (Martikainen et al. 1993).

Greenhouse gas emissions and sinks

The calculations were made for 10 site type groups and 5 areas: A1. - Lapland, A2. - Nortern Ostrobothnia and Kainuu, A3. - Western Middle Finland, A4. - Eastern Middle

Finland, A5. - Southern Finland. The change in the undrained peatland area (Fig. 1) was calculated from the data of the third National Forest Inventory (Ilvessalo 1957) and that of Keltikangas et al. (1986).

CO_2 balance was calculated as a net accumulation value; see Tolonen and Turunen (1996, this volume) for undrained peatlands, and Minkkinen and Laine (1996, this volume) for drained peatlands. CH_4 fluxes are from Martikainen et al. (unpublished). Tree stand development was simulated using material from Keltikangas et al. (1986) and growth and yield tables for corresponding sites and areas (Vuokila and Väliaho 1980). The total emissions and sinks due to the whole peatland area of Finland are shown in Fig. 2. The methane emissions are estimated to decrase by approximately one third. The carbon sink due to forest growth reaches its maximum around the turn of the century, and starts to diminish thereafter. The estimated carbon sequestration to the peat layer has increased strongly between 1960-1990, and is assumed to remain at the current level. In the long run this last component has the greatest impact on the greenhouse impact of the Finnish peatlands.

Radiative forcing calculations

The radiative forcing resulting from the emissions is calculated in two stages. First the atmospheric concentration change caused by the emissions is calculated. For the concentration calculations constant adjustment times for CH_4 (14 a) and N_2O (120 a) have been assumed. For CO_2 the pulse response function corresponding to present day consentrations presented by (Maier-Reimer and Hasselmann 1987) is used. The concentration changes are then converted into radiative forcing using the gas-specific functions given in (IPCC 1990). Only the direct radiative forcing is taken into account for CO_2, but for CH_4 the indirect impacts have been taken into account by multiplying the results by 1.3. This factor presents an estimate for forcing due to methane induced increase in the tropospheric ozone and stratospheric water vapour (IPCC 1995) The radiative forcing results in this article have been calculated using the REFUGE model (Korhonen et al. 1993, Savolainen and Sinisalo 1994).

Results

Drainage of Finnish peatlands for forestry appears to decrease the greenhouse impact. The estimated changes in all three emission components (decrease in CH_4 emissions and an increase in the CO_2 sink both in peat soil and CO_2 sequestered in trees) reduce the radiative forcing by approximately similar amounts (Fig. 3).

In the best estimate case (centerline in Fig. 4) the emissions from Finnish peatlands in the past have caused a slight positive radiative forcing of climate which can be regarded as a natural background level. The Finnish radiative forcing due to the emissions from fossil fuel use, waste management and agriculture is at the moment of the order of 3 mW m^{-2}, while according to our calculations the draining of peatlands decreases the radiative forcing presently by ~1.5mW m^{-2} (Fig. 4). In the future the anthropogenic radiative forcing is estimated to increase (Savolainen and Sinisalo 1994), but
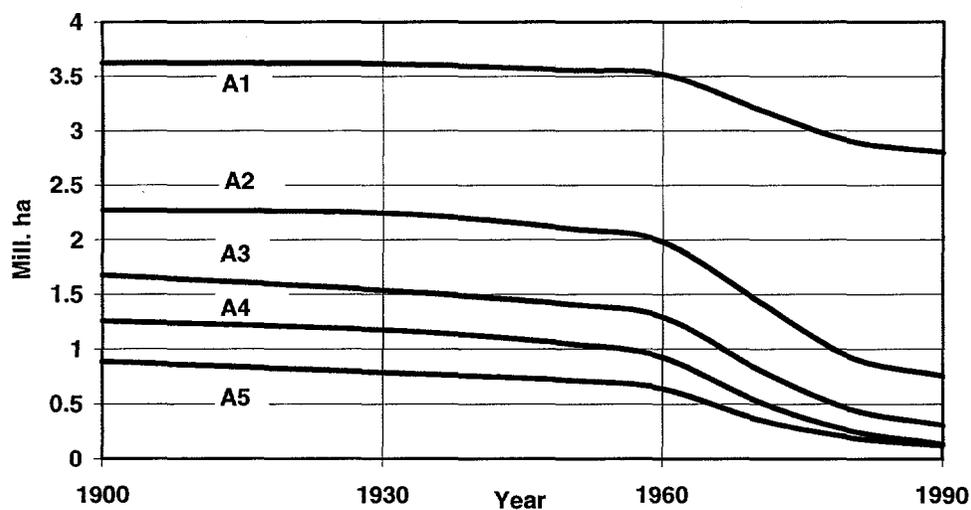
Fig. 1. Undrained Finnish peatland area from 1900 to 1990 by regions.
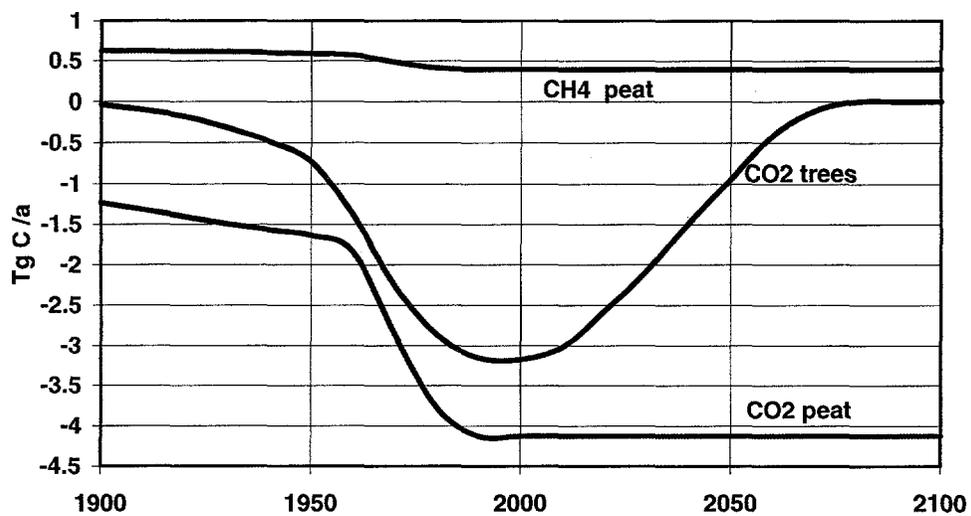

Fig. 2. Estimated methane and carbon dioxide fluxes of Finnish peatlands during 1900 - 2100.
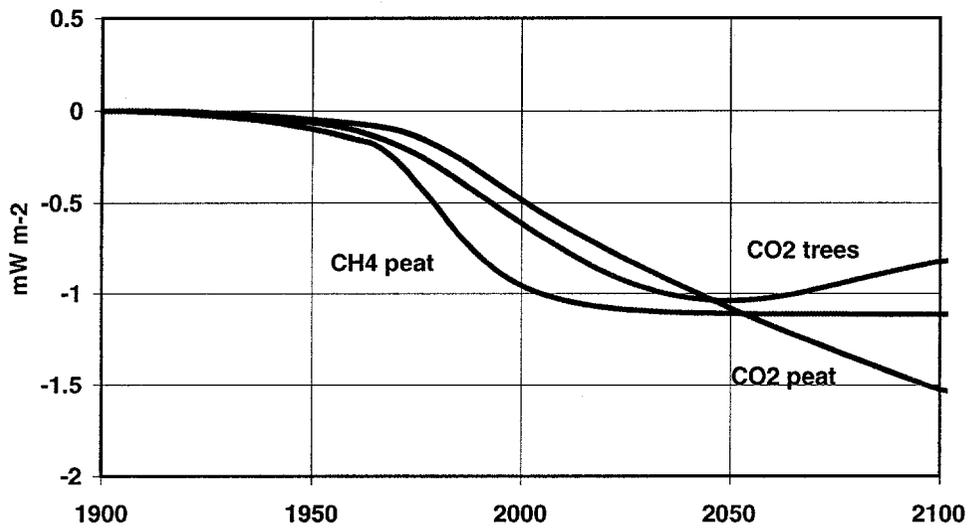
Fig. 3. Estimated change in the radiative forcing of Finnish peatlands caused by draining.
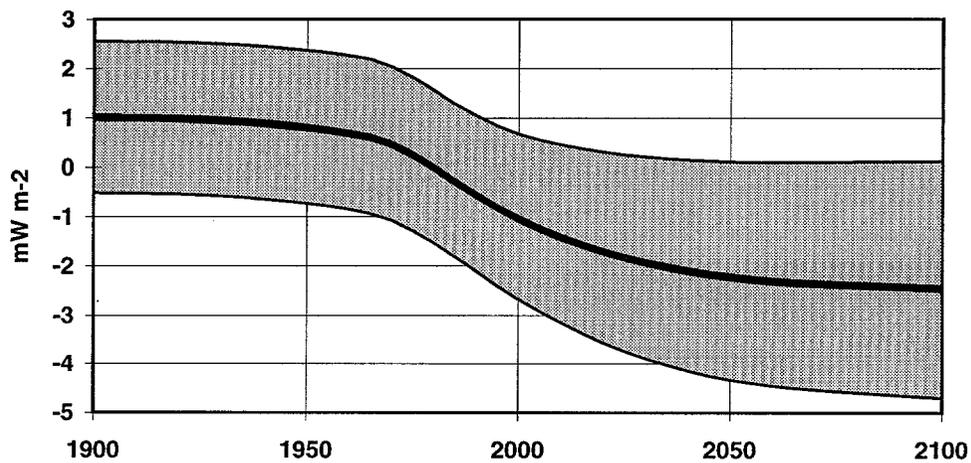
Fig. 4. Radiative forcing caused by Finnish peatlands. Shaded area represents the uncertainty of the results due to inaccurately known emissions of both natural and drained peatlands.

also the reducing impact of peatlands will grow at least for some decades. In the long run the impact of the methane component will saturate, and the forest sink will cease, leaving only the carbon sink in peat soil to exist for an unknown period of time.

Discussion and conclusions

The emission estimates include considerable uncertainty. The upper limit of the uncertainty range shown in Fig. 4 has been generated by deliberately overestimating the methane emissions by a factor of 1.5 and multiplying the reference values of the carbon sequestration of peatland by a factor of 0.3 and the tree growth by a factor of 0.5. The lower limit has been generated by assuming that the methane emissions are only 0.5 of original values and the other two compontents are multiplied by a factor of 1.5. The factors have been selected in a way that maximizes and minimizes the radiative forcing caused by the peatlands. Uncertainty due to radiative forcing calculations is about ± 30 %, which is not included in the figures. The measurements are from sites drained 30 - 60 years ago; for a longer period of time the emissions estimates are based on literature, and linear extrapolations.

Based on the results of the calculations in this paper it can be concluded that the greenhouse impact of CH_4 emissions due to Finnish peatlands in their natural state is on average roughly compensated by carbon sink in peat. This has resulted in a small net impact which may have been positive or negative.

Drainage appears to decrease CH_4 emissions and enhance the carbon sink in the peat layer and in increasing forest cover. All three factors mentioned lead to a decreasing radiative forcing. The increasing soil carbon sink after drainage is contradictory to most previous findings. The carbon balance development appears to be site-specific, i.e. the more nutrient-rich sites experience a net loss and nutrient-poor sites a net gain of carbon (Minkkinen and Laine 1996, this volume). The processes behind these phenomena require further studies before more accurate carbon balance budgets can be produced. The uncertainties involved also in the other emission and sink components causes considerable uncertainty to the absolute numbers arrived at in this study.

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Effect of forest drainage on the peat bulk density and carbon stores of Finnish mires

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Introduction

Drainage for forestry directs biomass production away from the field and bottom layer plant communities to the tree layer. In most cases, both primary production and biomass are increased (eg. Reinikainen 1981). Drainage also enhances microbial activity in the surface peat, resulting in an acceleration in the decomposition and mineralisation of the peat. Whether drained peatlands act as sources or sinks for carbon to the atmosphere, depends on the balance between the rate of organic matter accumulation (mainly from the litter production of the tree layer) and the rate of mineralisation of the peat. A disturbance, such as forest drainage, may cause a shift in the carbon cycle such that the carbon storage in the peat may be reduced but not always resulting in a net increase in the release of carbon dioxide to the atmosphere.

The aim of this study is to determine the changes in the stores of carbon in peat that result from forest drainage.

Material and methods

The material was collected from peatlands drained in 1930's. The drainage plans for these areas were rather detailed and include measurements of peat depth (max 200 cm) along the ditch lines at 40 m intervals. It was thus possible to return to the same measuring points. At each measuring point, the present peat depth was measured and volumetric peat samples taken from the 0 - 80 cm layer. The peat samples were used to derive the present bulk density profile of the peat layer.

The bulk density at the time of drainage was estimated by using material collected from virgin mires from the same regions and site types. The peat samples were collected with similar equipment as from the drained sites. Two peat samples were taken from each site reaching from the surface to the bottom of the peat layer (max 200 cm).
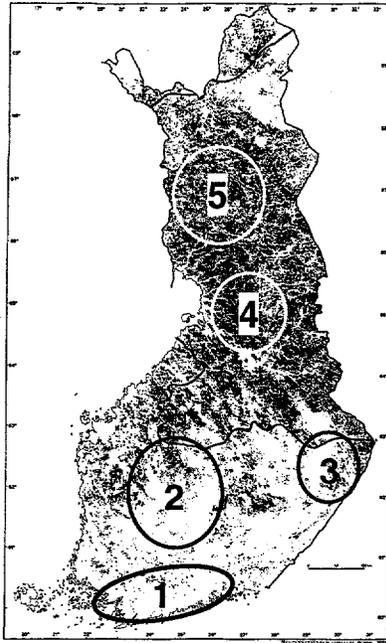
In order to study the effect of macroclimate and nutrient level, the data was collected from 5 regions around Finland (Fig.1) and from three pine fen site types representing different trophic levels (1) mesotrophic (RhSR), 2) oligotrophic (VSR) and 3) oligo-ombrotrophic (TSR and LkR) (see Laine and Vasander 1990 for descriptions of site types).

Fig.1. Location of study regions in Finland. Shaded areas in the map are mires. Border line between raised bogs and aapa mires is also shown in the map.

Approximately 900 peat cores was collected from drained and 400 from undrained sites. On the drained sites the sampling points on each ditch line representing the same site type and peat depth on a mire were pooled into clusters. On undrained sites every site represents one cluster. The number of samping points and clusters in different regions and site types is shown in table 1.

Table 1. Number of sample clusters and sample points (in brackets) in the material.

Region	RhSR		VSR		TSR-LkR	
	virgin	drained	virgin	drained	virgin	drained
1	18 (36)	19 (61)	29 (58)	12 (36)	8 (16)	-
2	5 (10)	27 (76)	33 (66)	82 (249)	17 (34)	22 (62)
3	5 (10)	13 (42)	21 (42)	22 (70)	7 (14)	4 (13)
4	8 (16)	5 (16)	22 (44)	42 (141)	9 (18)	9 (29)
5	4 (8)	24 (68)	12 (24)	13 (41)	-	-
all regions	40 (80)	88 (263)	117 (234)	171 (537)	41 (84)	35 (104)

The calculations were first made for each cluster and the averages for each region and site type were calculated from these values. The subsidence of peat surface was calculated as a difference between peat depths before and after drainage and represents the net change in the surface level. The mean bulk density of the peat layer was calculated from peat samples which include ash. Only the bottom layer near

mineral soil was corrected to an average ash content of the upper peat layer, if the content was exceptionally high (over twice the content of the upper layer).

From every cluster (drained and undrained) one peat profile was selected for further analysis. The carbon and nitrogen contents were analysed by a LECO CSN-1000 - analyser. The carbon contents of these peat samples were used to calculate the carbon densities and carbon stores of the peat profile.

Results

The results are still preliminary, but some trends can be seen. The average subsidence of peat layers after drainage varies from 14 to 33 cm increasing from nutrient-poor to nutrient-rich site types (Fig. 2), but the differences between regions are small. Instead, the mean bulk densities in both virgin and drained sites are higher in Northern than in Southern Finland. The bulk density is also higher in nutrient-rich than nutrient-poor sites, the mean values vary in virgin sites from 74 to 123 kg m^{-3} and in drained sites from 106 to 157 kg . Due to drainage, the bulk density has increased by 22-72 kg m^{-3}, increment being highest in Southern Finland and in the nutrient rich sites (Fig 2). The difference between drained and undrained peat bulk densities is greatest on the peat surface decreasing into deeper layers (Fig. 3). However, a difference of 20 to 25 kg m^{-3} still exists at the depth of 80 cm, which is the maximum depth of the samples from drained peatlands.

The average change in peat carbon stores approximately 60 years after drainage varies greatly: the mean values range from -7 to 18 kg C m^{-2} (Fig. 4). The trend between site types is clear: loss of carbon occurs only in nutrient-rich RhSR-sites - on other types the change is positive and increases towards the two southernmost regions.
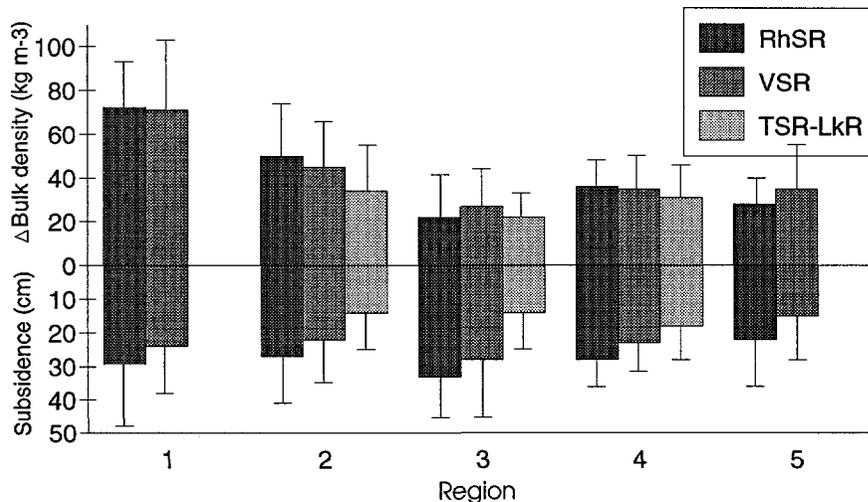
Fig. 2. The increase of peat bulk density and subsidence of mire surface approximately 60 years after drainage in different regions and mire site types (mean and s.d.).
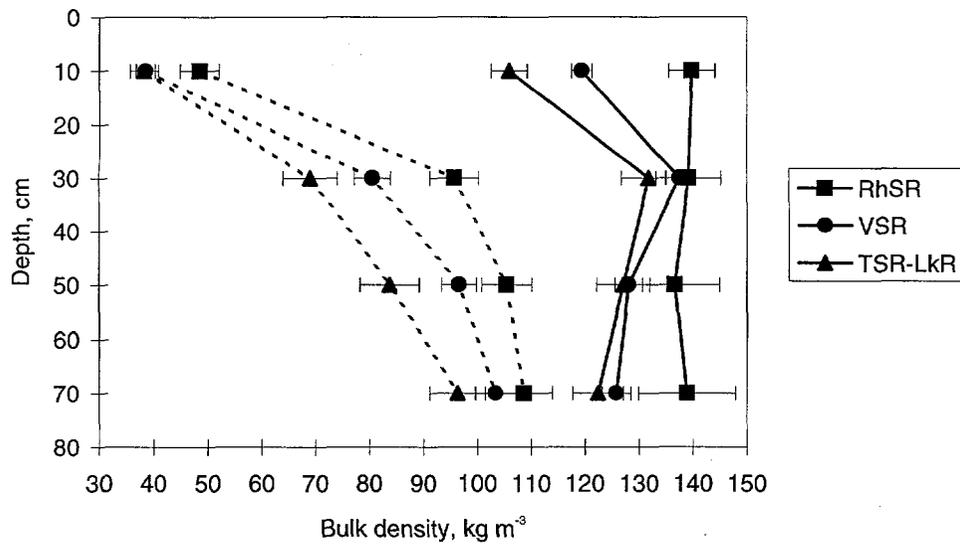
Fig 3. Peat bulk density versus depth in over 70 cm thick peats of undrained (broken lines) and drained mires (solid lines) - all regions together (mean and s.e.m.).
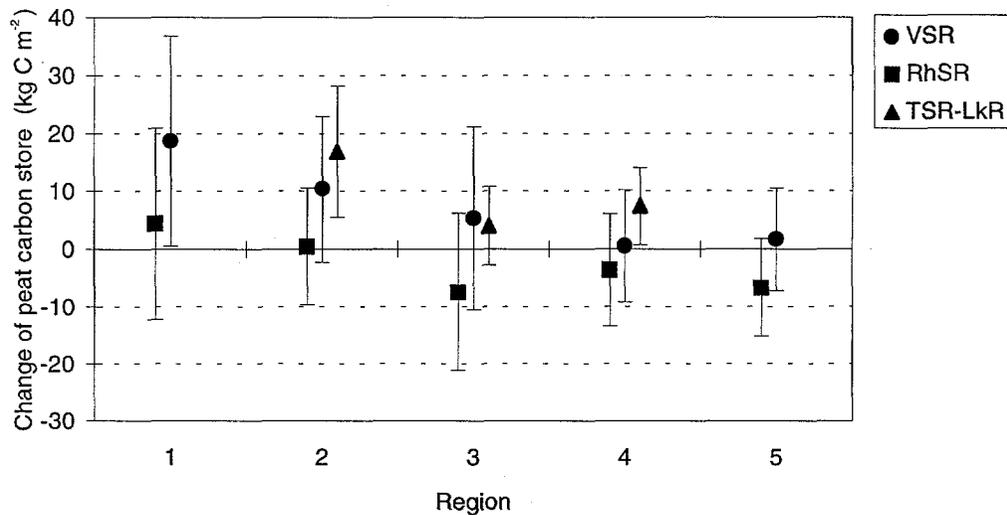
**Fig. 4. The change of peat carbon stores 60 years after drainage in different regions and mire site types (mean and s.d.).**

Discussion

The study shows that the subsidence of mire surfaces due to drainage has been relatively small, on average about 20 cm. The subsidence values presented by Lukkala (1949) for peatlands drained 14 to 36 years previously, varied between 17 to 41 cm. This suggests that the subsidence is largely caused by the physical compression that takes place soon after the drainage. This partly explains why the nutrient-rich types (which in this case are also wetter) have subsided more than the nutrient poor types, where the accumulation of organic matter appears not to have ended after drainage.

The observed increase in bulk density after drainage is caused by the physical compression of peat and the post-drainage input of organic material in the form of litter production from the above and below ground parts of the tree layer. Oxidative decay of organic matter may have further increased the compaction of peat, especially in fertile sites.

Regarding the change of carbon stores this study shows a similar trend with a study by Minkkinen et al. (1996) at Lakkasuo mire, with respect to nutrient level of sites. The decrease in carbon store in nutrient-rich sites may be attributed to higher rates of microbial activity in these sites. Microbial activity is affected mainly by the temperature and the oxic (aerobic-anaerobic) conditions of the peat (Nilsson and Berg 1986), which can be seen as increasing CO_2-emissions with increasing depth of water table (Silvola et al. 1994). Another factor contributing to this result is the relatively smaller fine root production in comparison to poorer sites (Finer and Laine 1994). This factor may also contribute to the observed difference with respect to geographical location from south to north.

The sampling depth on the drained peatlands was inadequate. The change in peat bulk density appears to have reached deeper than 80 cm below the mire surface; this was found also in the study by Minkkinen et al. (1996). Therefore, phycical compression of the peat must reach the deep peat layers or/and there must be a movement of carbon downwards in the peat profile (Domisch et.al 1996). Sampling from deeper layers would have resulted in greater carbon stores in drained peats, as compared to undrained ones.

The changes in the carbon stores after drainage are very large comparing to other estimates from the literature (eg. Cannel and Dewar 1995, Hogg et.al 1992, Braekke and Finer 1991, Silvola 1986), which have shown a loss of carbon from the peat due to drainage. However, the drained forested peatland ecosystems on boreal zone are of special character and their ecology is still poorly known. Although the oxidative layer in drained peatlans is adequate for efficient tree growth, the layer is superficial and the oxidative potential is far smaller than in the upland mineral soils. Considering the decay processes, drained peatlands may actually be very similar to the undrained ones, where the carbon input in the form of litter is largely being accumulated as peat.

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Effect of forest drainage on peat carbon balance at Lakkasuo mire, Central Finland.

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Introduction

Natural mires act as sinks for carbon dioxide (CO_2) and are sources of methane (CH_4) for the atmosphere. Drainage enhances the aerobic microbial decomposition of the surface peat, which may transform mires into net sources of CO_2. However, oxic conditions in drained peatlands are still superficial (Lähde 1969, Burke 1978), which limits the oxidation of organic matter . Also, the increase in growth of the tree stand and consequent fixation of carbon after drainage may be expected to have a compensating effect (Reinikainen 1981, Laiho and Laine 1994). Whether drained peatlands act as sources or sinks for carbon to the atmosphere, depends on the balance between the rate of organic matter accumulation and the rate of the mineralisation of the peat. We studied the changes in peat carbon stores of a mire 30 years after drainage for forestry.

Material and methods

Study site

Carbon balance calculations were carried out at a partly drained (drainage year 1961) Lakkasuo mire, Central Finland (61°48'N, 24°19'E, ca. 150 m. a.s.l.) Lakkasuo is an eccentric raised bog complex where a large variation of Finnish mire site types can be found. For a more detailed description of the mire see Laine et al (1986).

Sampling and analyses

Four transects were placed on the mire, each on different site type, starting from the drained area and extending on to the undrained, virgin part of the mire. The transect sites are described in Table 1. Sampling points were placed on each transect at 10 meter intervals, starting 5 meters from the border ditch (Fig 1.). Each transect was levelled and the sampling points were marked by poles extending from the underlying mineral soil to the peat surface. Three continuous (two from transect 1) volumetric peat cores were collected from each sampling point; one next to the pole and two replicants

5 meters on both sides of the pole. The cores were cut into 20-25 cm slices and the dry bulk density (105 °C) was determined for each sample. The carbon contents were determined from one peat core on each sampling point by LECO CHN-600 analyser.

Table 1. Description of the transect sites. The field measurements were made in summer 1991.

#	Mire site type	Average peat depth (cm)		Average depth of WT (cm)		Peat [**] constit.	Stand vol. (m³ ha⁻¹)		Stand origin	Fertili-sation
		ud.[*	dr.	ud.	dr.		ud.	dr.	dr.	dr.
1	Oligo-mesotr. (VSN)	163	140	3	33	C (L,S,Er)	0	111	Planted (1966)	NPK (-66) PK (-84)
2	Oligotrohpic (VSR)	187	152	8	33	C (L,S,Er)	52	121	Natural	-
3	Ombrotrophic (IR)	259	292	19	31	S (Er, L)	50	79	Natural	-
4	Ombrotrophic (RaTR)	266	244	12	24	S (L,Er,C)	6	16	Natural	-

[* ud=undrained, dr=drained
[** C=Carex, S=Sphagnum, L=Lignum, Er=Eriophorum

Later, another set of continuous peat cores were collected from each sampling point next to the pole. The zero-level (peat surface) of these peat cores was carefully fixed to the top of the levelled poles. The samples were microscopically analysed in 1-2 cm slices and pollen (and charcoal) diagrammes were prepared for a visual determination of synchronous peat layers. The synchronous layers were used to connect the peat cores together for determining the subsidence and calculating the carbon stores of a certain peat layer (Fig 2.)
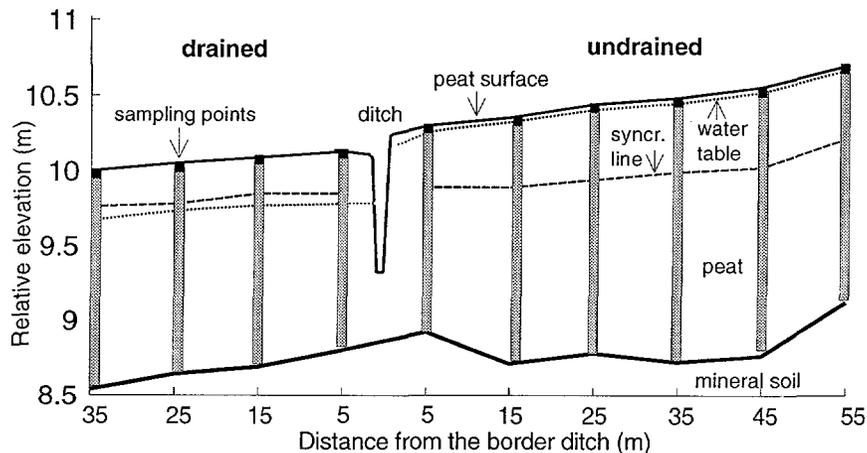
Fig. 1. A vertical section of the peat layer and the layout of sampling points on transect 1. The point next to the ditch on the undrained site (5 m) was left out of the final calculations.

Calculations

The carbon density, carbon store and carbon balance were calculated in 10 cm intervals starting from the top of the peat cores using equations shown in Fig 2. Carbon density and carbon stores were first calculated for each sample and then averaged for drained and undrained sites. Carbon balance was calculated as a difference of the average carbon stores on drained and undrained sites on a certain depth of peat. The calculations are based on the assumption that the peat of drained and undrained sides along the transect have developed similarly before the time of drainage and the observed differences in peat properties between the sides are caused by drainage only.

Carbon density $(D_{ci}) = D_{bi} \times C\%_i$

Carbon store $(S_{cn}) = \sum_{i=1}^{n} (D_{ci} \times t_i)$

Carbon balance $(B_{cn}) = S_{cn(dr)} - S_{cn(ud)}$

where

D_b = bulk density (mean of the three replicants), kg m^{-3}
$C\%$ = gravimetric carbon content of the peat
i=calculation layer (1=0-10, 2=10-20 etc.)
n=number of layers under calculation
t=thickness of the layer under calculation (differs from 10 cm only in the top of the peat core)
dr = drained side
ud = undrained side
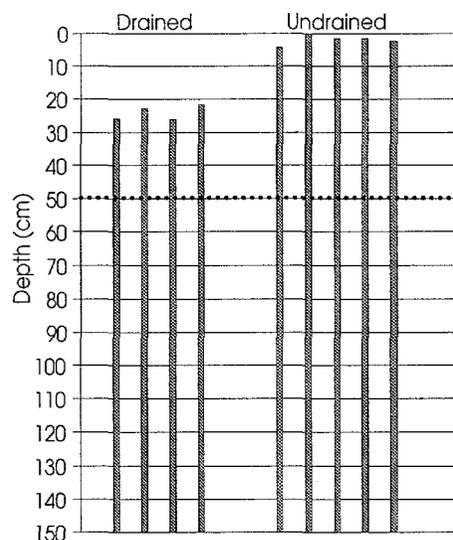
Fig. 2. The principle behind the calculations (example of transect 1). Peat cores (gray bars) were first connected by the synchronous layer (dotted line) and the calculations were made in 10 cm intervals starting from the top of the peat cores using equations shown beside the picture.

Results

Interpretation of the figures

Drainage causes an increase in peat bulk densities and carbon densities. The increase is consentrated on the top of the peat decreasing towards deeper layers. In the carbon balance calculations the carbon stores must be calculated using the whole depth of the changed peat layer. This depth can be determined visually from the Figures 3 and 4: at the changed layers the carbon density is higher on drained side (top graph) and the carbon-balance-line (bottom graph) is rising. The effect of drainage ends on the layer where carbon densities are about the same and the carbon-balance-line is staying at a more or less constant level. Any differing trends below this layer are a result of a different peat development before the time of drainage and therefore those layers must not be included into calculations (Fig 4.).

Minerotrophic sites

On transect site 1 (VSN) the subsidence of the drained side after drainage was on average 22 cm. A uniform synchronous layer was found at the average depth of 50 cm (30 cm on the drained side, Fig 3) below which no subsidence of the peat was observed. However, the carbon density values of the drained side exceed those of undrained side down to 110 cm. From Figure 3 it can be seen that at between 110 and 150 cm the carbon-balance-line stays at the level of -2 kg m^{-2}, meaning an average loss of 67 g C m^{-2} a^{-1} compared to the undrained site (the age of drainage was 30 years). Statistical significance depends on the depth of the calculation layer: variation grows towards deeper layers (Fig 3).

On transect site 2 (VSR) the peat has subsided on average 25 cm after drainage. No subsidence was observed below 35 cm in the drained peat profile (=60 cm on the undrained side). Carbon density was greater on the drained side through the whole peat profile causing a continuous rise in the carbon-balance-line towards the deeper layers. However, variation is large due to great differences in the depth of the syncronous layer on both sides of the ditch. At the depth of 170 cm (undr. side) there is more carbon at the drained side (1,2 ± 1.8 kg C m^{-2}, mean and s.e.) but the difference is not significant (two way t-test: p=0.520).

Ombrotrophic sites

The results for ombrotrophic sites are still preliminary since the preparation of pollen diagrammes still continues. From transect 3 only two peat cores have been analysed microscopically so far. The preliminary results show that the drained side has not subsided at all but has grown slightly faster (4 cm) than the undrained one. This was found also by Alm et al. (1992). There are, however rapid changes in the peat chemistry between different layers and sampling points (unpublished data), which are probably a result of at least three forest fires at the site (Alm et al. 1992). Therefore, the assumption of similar peat developments before the time of drainage may be incorrect, which makes the method unreliable. However, Alm et al. (1992) reported a difference of
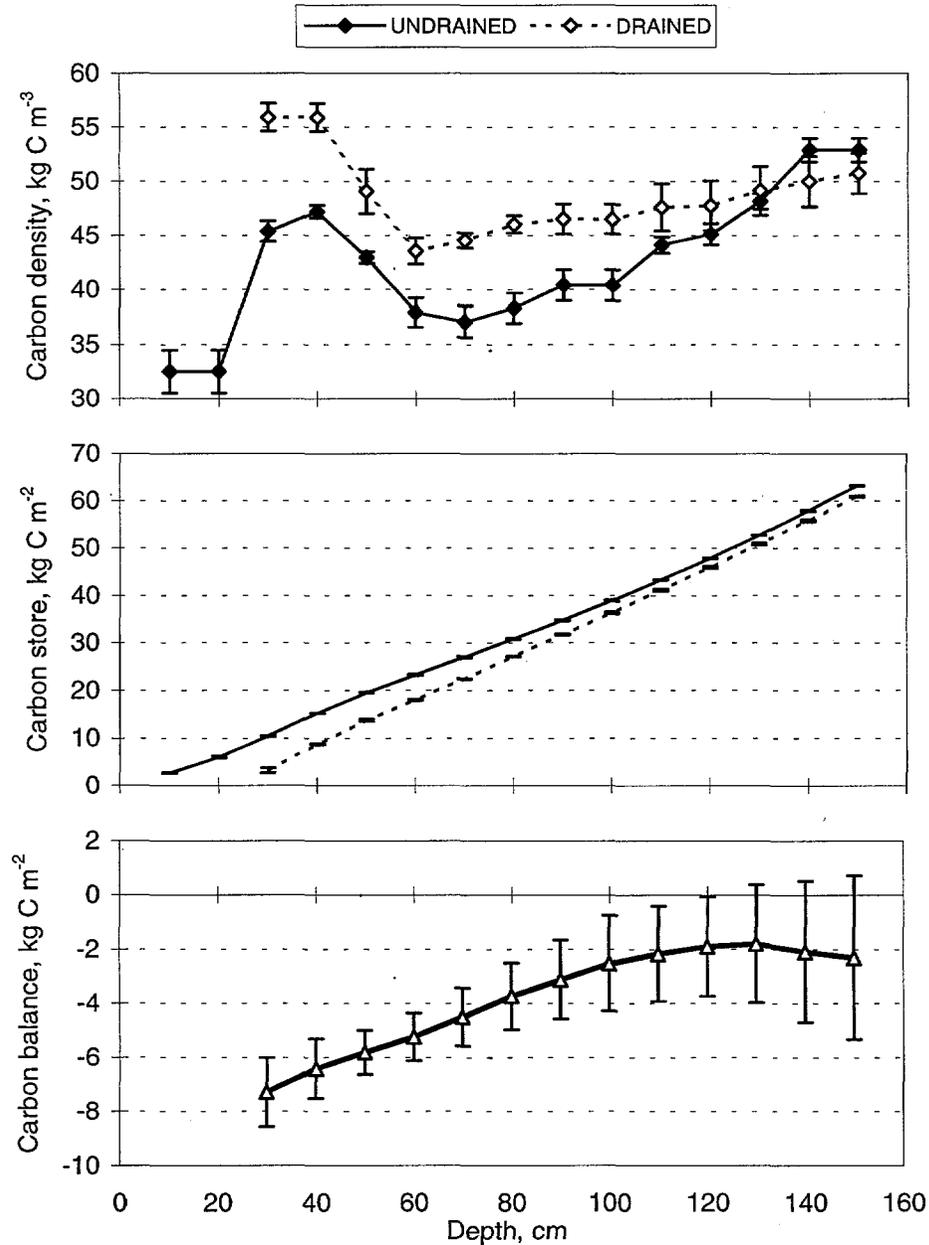
Fig. 3. Carbon density, carbon store (mean ± s.e.) and carbon balance (mean ± 95% conf. limits) versus peat depth on transect site 1 (VSN). Carbon density refers to a certain 10 cm layer in the peat profile whereas carbon store is a cumulative sum of carbon from the top of the peat to the referred peat depth. Carbon balance is a difference between the carbon store lines in the middle graph; if the error bars do not include the value zero, the difference is statistically significant (p<0.05). See also Fig. 2. for the principle of calculations.
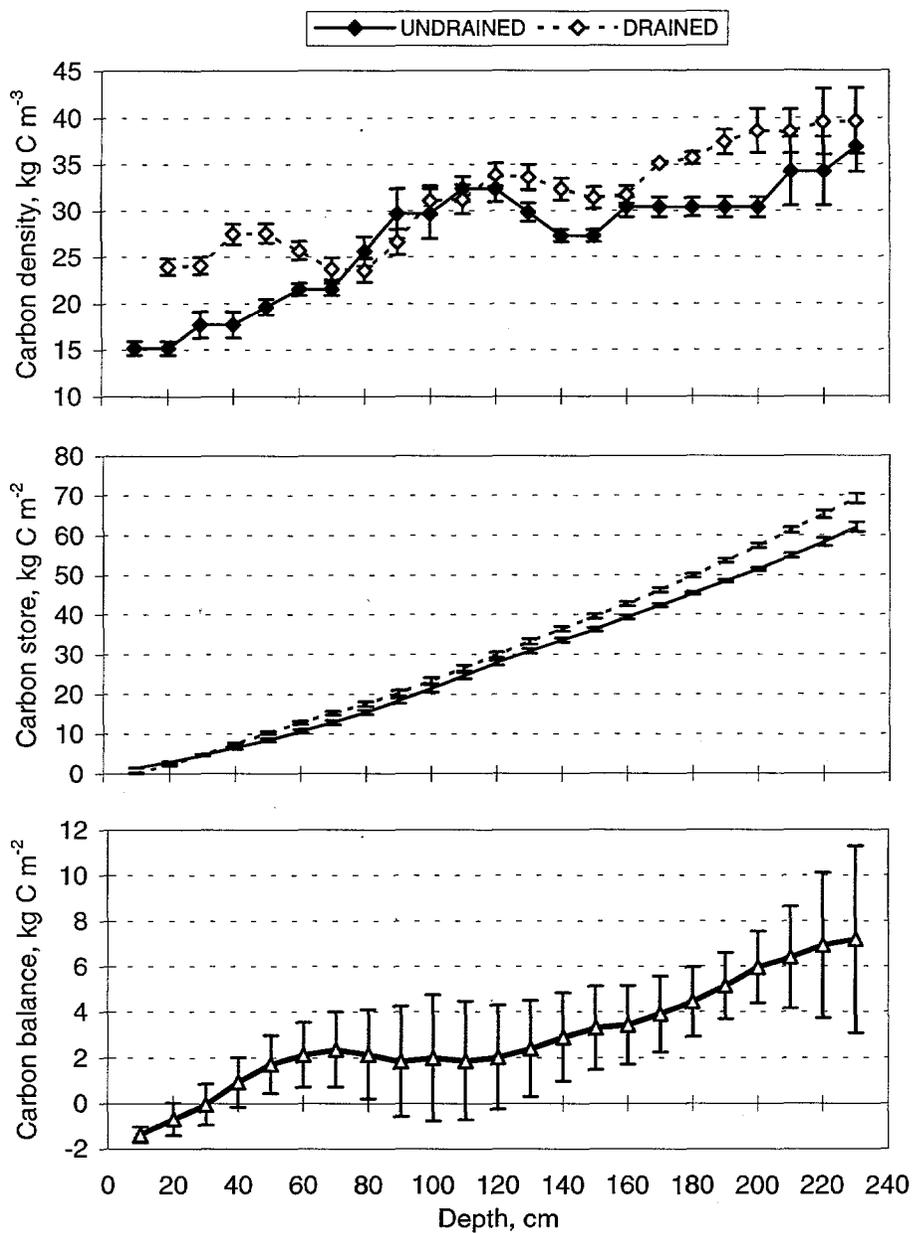
Fig. 4. Carbon density, carbon store (mean ± s.e.) and carbon balance (mean ± 95% conf. limits) versus peat depth on transect site 4 (RaTR). Carbon density refers to a certain 10 cm layer in the peat profile whereas carbon store is a cumulative sum of carbon from the top of the peat to the referred peat depth. Carbon balance is a difference between the carbon store lines in the middle graph; if the error bars do not include the value zero, the difference is statistically significant (p<0.05). See also Fig. 2. for the principle of calculations.

+0,9 ±0.9 kg C m^{-2} (mean and s.e., calculated from table 1 on p. 193) on the drained side compared to the undrained one, using a fire horizon at the average depth of 107 cm as a base line for calculations.

On the transect site 4 (RaTR) the synchronous layer has been determined from five points. The mean subsidence of the drained side was 10 cm. The carbon density is higher on drained side on the top 70 cm after which it is similar on both sides down to 130 cm. Naturally, also the carbon-balance-line stays at the same level between 70-130 cm, after which it starts rising again. The reason for this another rise is most probably a difference in pre-drainage peat accumulation rates between the sides. Between 70 and 130 cm the carbon balance is about +2 kg C m^{-2} and the statistical significance is near to the 95 % level, but depends greatly on the depth under calculation (Fig 4). On a yearly basis, the drained site has accumulated carbon 67 g m^{-2} a^{-1} faster than the undrained one.

Discussion and conclusions

The method introduced here can be used for calculating long term changes in peat carbon balance. However, the accuracy of the method depends greatly on the determination of the synchronous layers, Here, the accuracy was about ±1 cm, which equals a difference of 0,35 - 0,55 kg C m^{-2}. At this study 200 - 400 pollens were calculated from each sample, which was a minimum amount for the purpose. Due to uneven peat surface the layers are more difficult to distinguish than on more homogenous stratified deposits, for example lake sediments. More pollens should therefore be calculated for more reliable results. This is, however, a limited possibility, since the microscopical analyses are very time consuming.

The results were quite clear on transect 1 and transect 4, while transects 2 and 3 were rather difficult to interprete. On the minerotrophic site 1 (VSN) the accumulation of carbon has decreased on the drained site resulting in a difference of -2 kg C m$^{-2}$ compared to undrained side, while at the ombrotrophic site 4 (RaTR) the accumulation has been as much faster on the drained side. Similar results, regarding the site types, are also reported in a larger study by Minkkinen and Laine (1996). Transect sites 1 and 2 were affected by drainage into quite deep layers. If compaction of the peat did not occur on those layers, there would have to be a movement of carbon downwards in the peat profile (Domisch et al. 1996).

Oxidation of organic matter depends greatly on the decomposing material, temperature and oxygen availability (Nilsson and Berg 1986) or in other words the depth of water table (Silvola et al. 1994). The greatest drop of the water table level has happened in transect 1 (Table 1), where the loss of carbon has also been largest. Also, the net production of fine roots is greater on poor sites (Finer and Laine 1996). It has been noticed that, at least on the nutrient poor sites, the height growth of mire surface may continue at a similar speed after drainage (unpublished data). It seems thus possible that the net primary production and litter production may increase more than the oxidation of organic matter resulting into a net increase of the peat carbon accumulation after drainage.

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Carbon accumulation in mires in Finland

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Introduction

The long-term (apparent) rate of carbon accumulation (**LORCA**) can be estimated at a given mire site from peat columns of known (dry) bulk density, carbon content and age (for the deepest peat). Analogously, the recent (apparent) rate of carbon accumulation (**RERCA**) is based on the column section between the surface and a given dated horizon in a surface core. The actual net rate of carbon accumulation (**ARCA**), however, is lower because the slow decay that takes place in the anoxic deeper layers is ignored in this approach. The actual rate can only be estimated by means of "peat accumulation models" like that of Clymo (1984). Results based on vertical growth in single cores but not on the lateral expansion of mires can be misleading. Measuring the rate of the latter in mass units is laborious and no published studies exist. Accumulation of carbon in mineral subsoil underneath the peat strata appears to be an additional sink that has been, thus far, very little studied.

Material and methods

In Finland (between 60°-70° N Lat) the long-term (apparent) rate of carbon accumulation, LORCA was studied in 1121 peat profiles dated by ^{14}C (321 cores), by pollen chronology (258 cores) or by land-uplift chronology (542 cores). The data comprise of peat columns studied by three "methods" as described in Tolonen and Turunen (1996). The RERCA was studied in 222 short cores with known bulk density by means of the so called pine method, where the age of pine root collars, buried in surface peat was counted from the tree rings microscopically (cf. Ohlson and Dahlberg 1992). Synchronous fire horizons provided additional material for 287 short cores. They were dated by dendrochronology using fire scars of mire pines (Alm *et al.* 1993) and the AMS radiocarbon "wiggle matching" technique (Tolonen *et al.* 1993). If not measured by means of a LECO CHN-analyzer, the carbon content was estimated using the dry mass estimate of 0.5. A correction for ash was made in the density values when possible (in a small number of cases). In most peat columns the ash content was low.

In this paper we present the updated results based on "empirical" data from Finland, where both bulk density and bottom age were measured. Numerous further cores are still in laboratory analyses, as replicates for the subsoil carbon accumulation (see Turunen *et al.*, this volume).

Results

Within the boreal area studied, great variation was found in the average LORCA (unit g m^{-2} a^{-1} of carbon). In the large data set for Finland we found a **mean 22.6 ± 0.5 (SE)**, and a range of 2.3-102.8. Significant differences in LORCA were evident between peat columns of different age classes, different areas and different vegetation types (Figs.1 and 2). The highest values in the empirical data set were found in very young mires along the eastern shore of the Gulf of Bothnia. The lowest figures are from a geologically old and fairly continental area in Eastern Finland (Patvinsuo National Park), mires that are 9500-5000 years old and have experienced numerous (at least 31) visible mire fires.

The average LORCA was higher in bogs (**25.5 ±** 0.5, range 6.6-85.8, n =667) than in fens (**17.2 ±** 0.3, range 2.3-102.8, n = 454). In Finnish peatlands younger than approx. 9000 years, LORCA was much higher and almost double in the southern compared to the northern mires (Tolonen et al 1994, Tolonen and Turunen 1996).
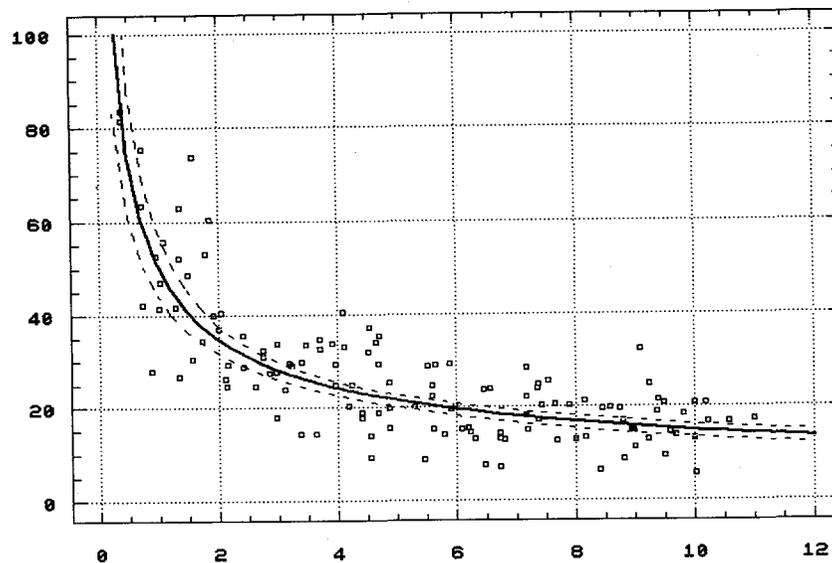
Fig. 1. Long-term rate of (apparent) carbon accumuluation, LORCA (in grams of carbon m^2/a) in bogs in Finland plotted against the calendar age (in kilo years) of deepest peat of each peat column (n=135). Both bulk density and age of bottom peat are measured. The multiplicative model Y = ax^b (solid line and its 95 % confidence limits). R^2 of regression was 57.3 %.
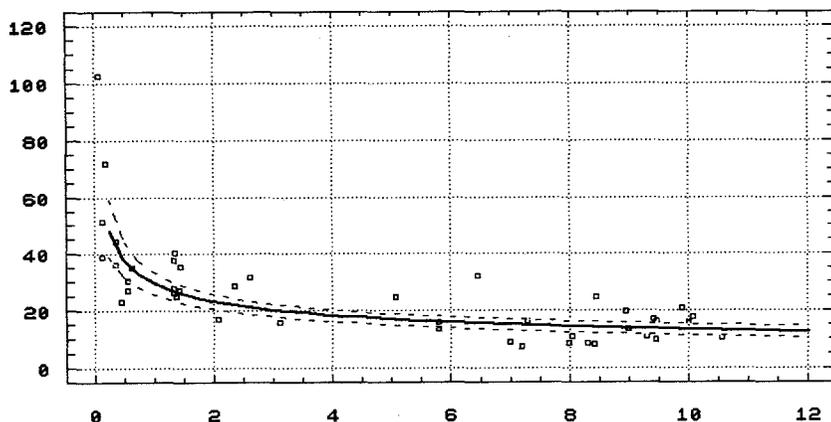
Fig. 2. LORCA (in grams of carbon m$_2$/a) in fens in Finland plotted against the calendar age (in kilo years) of deepest peat of each peat column (n=46). Both bulk density and age of bottom peat are measured. The multiplicative model Y = ax^b (solid line and its 95 % confidence limits). R^2 of regression was 67.4 %.
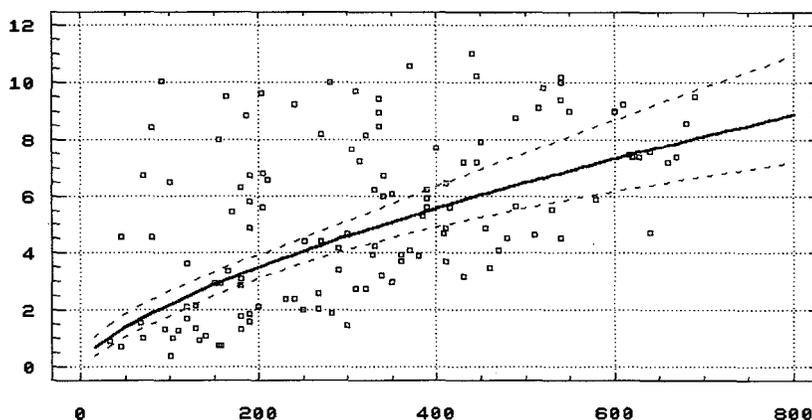
Fig. 3. Calendar age (in kilo years) of deepest peat plotted against depth in bogs in Finland of each peat column (n= 135). Both depth and age of bottom peat are measured. The multiplicative model Y = ax^b (solid line and its 95 % confidence limits). R^2 of regression was 33.0 %.

The pine method comprises peat layers dated from 9 to 102 years old (mean 34.6 ± 1.0 (SE) yr, n = 270). The apparent carbon accumulation ranged from 11.8 to 290.3 g C m^{-2}a^{-1} (mean 100.2 ± 3.1 g). The RERCA was much higher in virgin ombrotrophic mires, at 125.6 ± 4.1 g (age 37.4 ± 1.6 yrs) than in virgin fens, at 76.8 ± 4.3 g (age 31.5 ± 1.4 yrs). The average RERCA was much lower in 100 - 200 years older peat layers that were studied by means of other stratigraphical datings. In the peat columns above the dendrchronologically dated fire horizons the RERCA varied during the last 212-147 years from 39.8 to 80.7 g C m^{-2} a^{-1} (Alm et al.1992). In the same study sites the RERCA was lower for the last 1040 (± 90) years: 28.5 - 42.8 g C m^{-2} a^{-1}, indicating a clear loss between those dates. Clearly, the

Table 1. Long-term rate of (apparent) carbon accumulation, LORCA (in grams of carbon m²/a) for two main hydrolocigal mire groups, bogs and fens in five mire regions in Finland (see the index). The values are weighted averages based on the mire ares for different mire site groups and their average peat thickness given by Ilvessalo (1957). More details in text.

Mire region	BOGS		FENS	
	LORCA, g / m² / a	% of mire area	LORCA, g / m² / a	% of mire area
A	32,9	42,5	23,0	8,7
B	33,9	44,0	23,1	12,5
C	34,8	45,5	23,1	16,2
D	35,2	38,7	22,6	39,4
E-H	27,5	30,4	22,1	49,0

carbon accumulation was highest at sites where *Sphagnum fuscum* was the main component in the bottom layer. The ARCA ranged between 8.1 and 23.0 g (Tolonen et al 1994, Tolonen and Turunen 1996) being from 29 to 85 % (mean 63 %) of the LORCA within the individual peat columns.

Discussion

The role of mire fire in carbon accumulation was clear in some raised bog profiles from western Finland with numerous visible charcoal horizons (Tolonen *et al* 1992). The same is true for an extensive aapa mire complex Patvinsuo, in eastern most North Karelia, Finland. The obvious loss of material due to the fires could be estimated by comparing individual peat sequences with a very large number of other peat sequences with basal dates. Probably more than half of the carbon storage was lost in repeated (up to 31) fires between some 8000-4000 B.P. at the sites where the average LORCA figures remained as low as 2.3 g C m⁻²yr⁻¹.

Our material is non-representative. Most of the profiles are from the deepest points of the mire basins, as they are most often collected for purposes other than the general history of mires. Therefore, the shallower mires are under-represented. In the so called geological mires (over 20 ha in area and more than 0.3 m deep) of Finland, the mean depth is 1.52 m (Lappalainen and Hänninen 1993), but in the total mire area of Finland the mean depth probably is slightly less (Ilvessalo 1957). There was a significant correlation between the depth and age of peat columns ($R^2 = 65.6$ %) and between the age and the LORCA ($R^2 = 52.4$ %) (cf. Table 1) in our "empirical" data, where both bulk density and the bottom age of the peat columns were measured providing a more representative average for the peatlands in Finland to be calculated. For 1.5 m deep peatland, this still tentative value was **26.1 g C m⁻²yr⁻¹** (mean age 2414 yr).

The surface cores revealed apparent rate figures that were clearly higher for the past 30 years (mean 100 g C m-2 yr-1) than those in the youngest LORCA columns for the whole peat strata (Figs 1-2). This implies the strongest loss during the first years in peat accumulation. It is interesting that the RERCA figures for both the 147-212 and about 1000 years old surface layers were so close to the LORCA figures of whole strata in the young mires. This holds for all vegetational types. Our interpretation is that the young mires (especially those initiated by terrestrialization and primary mire formation *sensu* Huikari, 1951) were effective carbon accumulators owing to their optimal moisture regime for production of wetland species (first reed and sedges, then Bryales and *Sphagnum*), but had suppressed decay owing to the very poorly developed and thin acrotelm (cf. Malmer 1992). The highest RERCA figures in the surface layers of *Sphagnum fuscum* bogs are as expected, because this moss species is most resistant to decay (Johnson and Damman 1991).

The figures for carbon acccumulation by means of single cores can be misleading as it ignores the lateral expansion of mires. Studies by Korhola (1992) in southern Finland indicate that the volumetric lateral growth of mires can be great, even though there may be only minor vertical growth. Figures about the spatial mass accumulation exist only for one concentric raised bog, Reksuo in SW.Finland (Korhola *et al.*, this volume). There the spatial mass accumulation was primarily controlled by variations in horizontal growth for the first 5000 years (between 8.6 and 3.0 cal kyr B.P.); *Carex* fens dominated, and only about 20 % of the current peat was sequestered at that time. The first *Sphagnum* peats began accumulating ca. 5.0 cal kyr B.P. Their contribution to ARCA was low until a dramatic increase in the accumulation rates occurred about three millenia ago with the establishment of the raised bog stage.

Gorham (1991) estimated the total annual acumulation rate of carbon for the boreal and subarctic peatlands at 96 Tg C (LORCA) or 76 Tg C (ARCA). According to Botch *et al.* (1994) the LORCA of the peatlands of the former Soviet Union alone is 52 Tg Cyr^{-1} Owing to the scantiness of real empirical data from Russian mires the estimates for the whole boreal zone remain uncertain.

Conclusions

1. The age and thickness of peat enables a prediction about the rate of carbon accumulation in different mires in Finland. Based on good statistics of the areal extension and depth of various mire site types, we believe it would be possible to find quite a real estimate of the long-term and present carbon accumulation in the mires in Finland in the near future.

2. If the proposed warming will shift northward the existing climatic vegetation zones, then the northern aapa fens will change to *Sphagnum* bogs that are more effective in sequestering carbon, but distinctly less effective in their CH_4 and N_2O emanation (Crill *et al.* 1993). Most climatic scenarios predict a decrease in the evaporation surplus during the summer at northern regions. Presumably, the consequent lowering of the water table would improve the growth of forest on mires and simultanously decrease the methane fluxes from peat. The combined net effect could be a clear restraining of the radiative forcing, as

predicted by Laine *et al.* (1995). What will be the role of mire fires in more remote northern areas may then become another important question. The answer may depend on the future precipitation and its seasonal distribution pattern.

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Carbon accumulation in mineral subsoil of mires

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Introduction

Peatlands in their natural state are net accumulators of organic matter and thus an important component of the global carbon cycle with ca. 192 - 455 Pg C ($=10^{15}$ g) stored in peats of Boreal wetlands alone (Gorham 1991).
The amount of carbon stored in the forest soil of the Boreal zone is some 12-13 % of the total carbon storage in the soils of the earth (Post *et al.* 1982). Prevailing soils of the Boreal coniferous forests are podsols, where the accumulation and loss of carbon by decay are in steady state within a few thousands years (Birkeland 1984). According to an unpublished study by Jari Liski, Hannu Ilvesniemi, Anneli Mäkelä and Michael Starr, the corresponding time in oligotrophic sandy soils of Finland is about 2000 years.

Since about one half of the present peatlands was formed by means of paludification of former forests (Huikari 1956), it is of interest to compare the carbon amounts in mineral subsoils beneath peat deposits to those in adjacent forest soil with similar texture. One might hypothesize that the long-term accumulation of carbon might be somewhat lqarger beneath peat due to its waterlogging and usual occurence in lower lying basins. We commenced a special study on this subject in 1994.

Material and methods

Our main study area was Lakkasuo mire complex (61°47'N, 24°18'E, about 150 m a.s.l.; Fig. 1) in the eccentric raiseds bogs area of Finland. We also collected data from

the Mustapuro area in Karstula (63° 01 'N, 24° 25' E' ,170-180 m a.s.l.) and from Patvinsuo National Park (63°05'N, 30°40'E, about 170 m a.s.l.), both situated in the area of southernmost aapa mires of the Middle Boreal zone (Ruuhijärvi 1983).

Subsoils of peatlands that had initiated via paludification and soils of adjacent forests with sorted (sandy) material were investigated in 1994 and 1995. Samples for bulk density, radiocarbon dating and for carbon content analyses were taken using a box sampler (8,5x8,5x100 cm), a Russian peat sampler (5x50 cm) and a special mineral soil sampler (a steel cylinder, 40 x 500 mm, equiped with inner polyvinyl tube) that was hammered into the soil. Extreme care was payed taken when sampling subsamples inside the coring samples for carbon content analyses in order to avoid impurities. Carbon was analyzed using the LECO CHN-600 -analyzer at + 1050 °C. Radiocarbon datings were converted to calendar years (Stuiver and Reimer 1993).

In very marginal areas of all three mires there were clear charcoal layers between mineral soil and peat, and we used the age of the last fire as determined from historically known sources (Lukkala 1933, Potinkara 1993) or from the fire scars in the mire pines to date the age of bottom peat (Alm et al. 1992). .
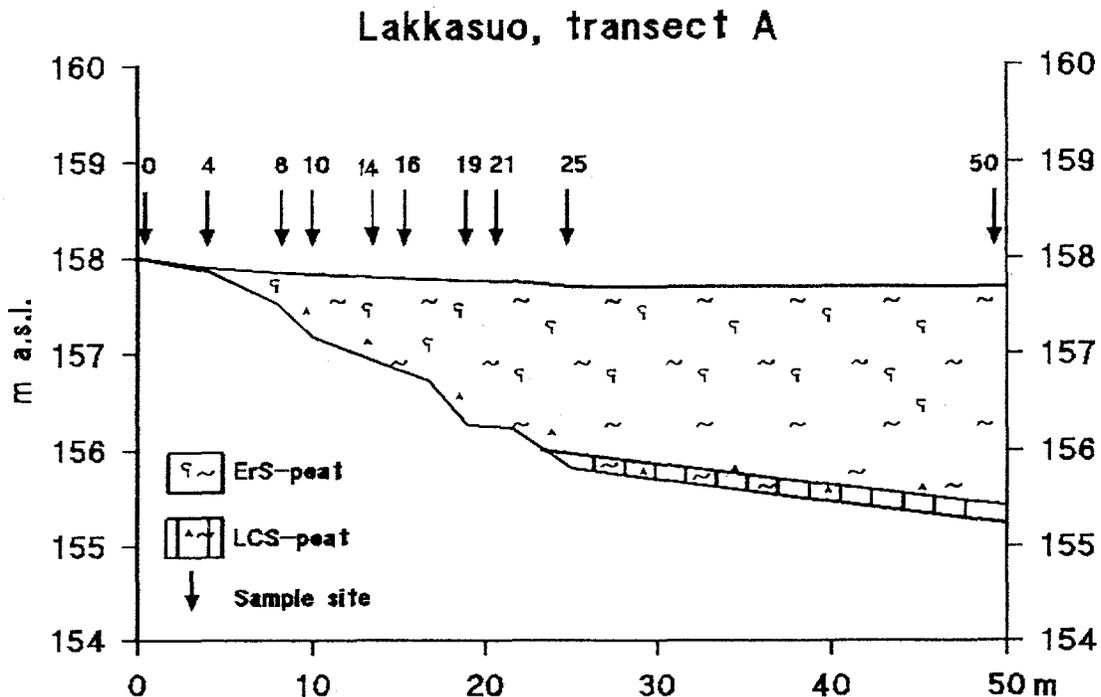
Fig. 1. Profile of the Lakkasuo mire, Orivesi, Finland (survey transect A), showing the peat types (see Table 1) and locations of the sampling sites.

Table 1.Sample description and ^{14}C dates of basal peat from eight coring points of Lakkasuo mire (see Fig.1). Age of the basal peat in site A4m is based on dendrochronological dating (Alm et al.1992). Peat constituents: L=wood, Er=Eriophorum vaginatum, C=Carex, S=Sphagnum. H=degree of decomposition in von Post's 10-grade scale.

Site	Depth, cm	Peat type	Conventional ^{14}C age	Cal. age before 1995
A4m	13	LS, H9	–	148
A8m	32–37	LS, H7–8	660 ± 80	693
A10m	61–65	LS, H7–8	1580 ± 90	1501
A14m	85–91	LCS, H7–8	1820 ± 90	1765
A16m	90–97	LS, H8–9	1790 ± 90	1751
A19m	143–150	LS, H8–9	2860 ± 120	2996
A21m	145–152	LErS, H8–9	2680 ± 90	2812
A25m	182–189	LS, H8–9	2840 ± 90	2988
A50m	239–246	LErCS, H6–7	3990 ± 100	4467

Results

We here concentrate to the mineral parts of podsolic profiles beneath peat deposits and of those in adjacent forest sites. We will compare only the mineral soils from A$_2$ - horizon downwards (that horizon included) in order to avoid errors in the interpretation.

Clear podzolic soils with strongly bleached (eluvial) horizons (A$_2$) up to 15 cm deep and with dark brown humus + sesquioxides rich illuvial horizons (B) up to 45 cm deep were found at all the sites that paludified wihin the last 3000 years. Under Patvinsuo mire, clear podzolic horizons were found in medium-grain sandy soils older than 8000 years, but in Lakkasuo bleached (A$_2$) layers were indistinct beneath peats older than some 4000 years.

In most of the cases, the mineral subsoil of mire sites (when sampled down to 70-100 cm) had much higher carbon density and carbon storage (up to 10-fold) in both A and B horizons than in adjacent forest profiles. When the carbon store in the forest site was subtracted from corresponding figures beneath peat at each paludified·site (cf. Figs 1-2), the addition or loss in the carbon store during the mire stage was obtained
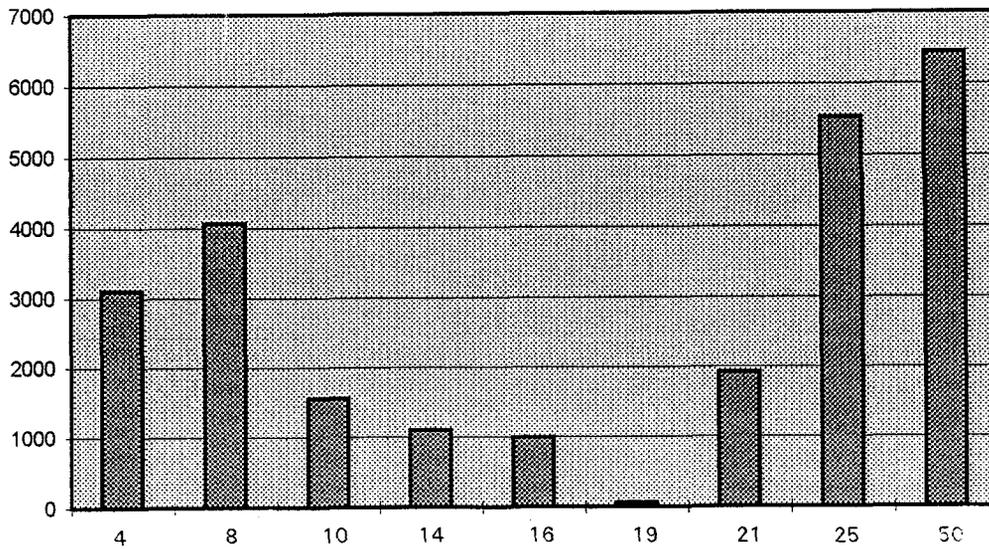
Fig. 2. Accumulation of carbon (in grams of carbon $m^{-2}a^{-1}$) in uppermost 70 cm of mineral subsoil (horizons A_2 through C) at nine sampling sites as shown in Fig. 1, after the sites were paludified. Assumed original carbon storage (2683 g m^{-2}, see text) in corresponding mineral soil during preceeding forested period was subtracted from the total carbon store beneath the basal peat to obtain accumulation figures.
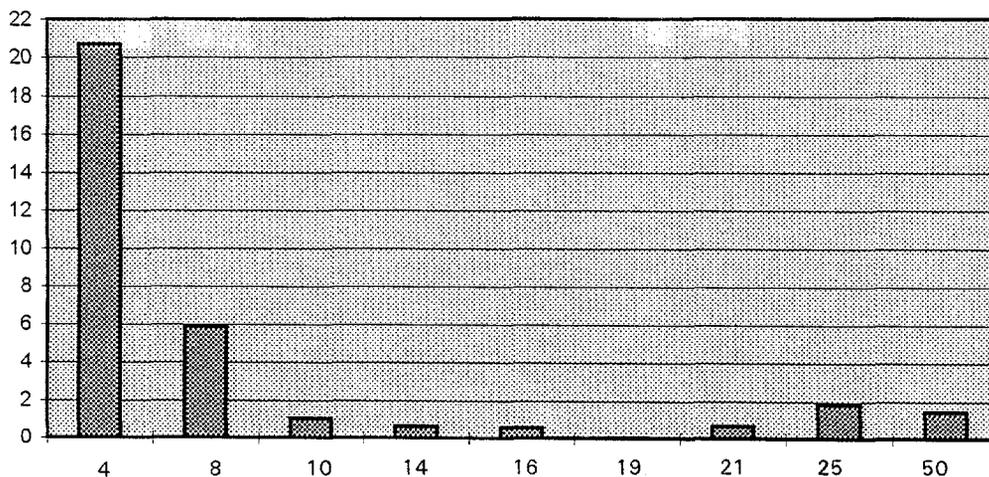
Fig. 3. Long-term (apparent) rate of carbon accumulation (LORCA) during the mire stage in the uppermost 70 cm of mineral subsoil at the sites as shown in Fig. 1. The average rate was estimated by dividing each carbon store of Fig. 2 by the calendar age of the basal peat (Table 1). For A4m, the dating (150 years) was according to the last fire (Alm *et al.* 1992).

according to the principle described in the introduction. In Lakkasuo we used soil carbon store (in the 70 cm deep profiles) of the surrounding pine heath forests at the margin of the bog with a mean value 2686 ± 853 (SD, n=12).

In the southern margin of Lakkasuo mire there was a cumulative increase in the carbon accumulation in the subsoil of the two first and two last sites (sites A4m, A8m, A25m and A50m). Figures were lower at five succesive sites (A10m, A14m, A16m, A19m and A21m) in between. We assume that the steep bottom topograpy is the main factor that explains this discrepancy, since the soil texture and the peat stratigraphy were essentially equal.

The accumulation figures enabled the rate of long-term (apparent) carbon accumulation for the mineral soil of the mire stage (LORCA, see Tolonen *et al.* 1992) to be calculated provided the age of the peat initiation was known (as in Fig. 3). This carbon input ranged from 0.2 - 116.3 g C $m^{-2}a^{-1}$ depending on the age of paludification, but also on vegetation, slope, and texture. The highest figures found thus far were at a shallow spruce fen sit (in Lakkasuo mire) that initiated after forest fire some 230 years ago. Typical input figures in oligotrophic pine bogs (about 150 years old) ranged 10 - 30 g C $m^{-2}a^{-1}$. Very similar values were found in sandy soils of all three mires studied, when the mire sites and the adjacent forest sites with similar soil texture were compared pairwise.

It is thus likely that the LORCA in mineral subsoil with equal texture is lower in steeper sites than in more gently sloping parts of the basins (Fig. 3). More importantly, the LORCA figures decrease with increasing age.

Discussion

We do not know any corresponding study, but Ms. Jaana Ronkainen, in the University of Oulu, is gathering similar data from Kuusamo and Rokua areas in northern Finland. She will have preliminary results in the near future. .

According to the dated transect of Lakkasuo mire, an increase in the carbon store of the mineral subsoil beneath peat seems to continue at least within 4500 years, if five successive sites of the steep slope are excluded. We are anlyzing older (up to 8000 years) parts from Lakkasuo mire clarify, if a steady state in the cumulative carbon accumulation can be found within that time. Clearly, a basin with more even bottom topography and gently sloping mire margins would be ideal. Such an area is under study from Patvinsuo mire complex.

The LORCA figures of mineral subsoil beneath peat resemble those obtained for peat deposits in Finland (e.g. Tolonen and Turunen, this volume) in two respects:

i) the figures are much higher in younger than in older mires. The explanation of this regularity might, however, be different for peat and mineral subsoil, because carbon is much less mobile in peat than in mineral soil,

ii) the dimensions of LORCA in mineral subsoil beneath peat may well be of the same class order as in peat.

Acknowledgements

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RATE OF CARBON ACCUMULATION IN HUMMOCKS AND HOLLOWS

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Background

The overall rate of carbon accumulation in northern peatlands has been estimated to 23 g m^{-2} y^{-1}, or 7.5% of the annual net primary production (approximated to 307 g m^{-2} y^{-1} using several uncertain assumptions, Gorham 1991). The pool of carbon contained in boreal and subarctic peatlands is estimated to c 200-450*10^{-5} g, or between 14% and 33% of the total world pool of carbon (Post et al 1982). Because of that, the understanding of the carbon dynamics in peatlands, and how environmental changes affect the carbon balance in both a short term and a longer term perspective, is important for the understanding of the global carbon balance. The climatic warming will, besides direct effects on the rate of biological processes, i.e. production and decomposition indirectly influence the carbon balance by altering the mire hydrology.

Fig. 1 Mean dry bulk density (g dm^{-3}) at successive 2.5 cm levels in hummocks (solid line) and in lawns (dotted line). Horizontal bars indicate 2SE (N=10).

Most mires are characterised by micro-topographical surface patterns of alternating wet depressions (moss dominated carpets and non-vegetated mud bottoms and pools), and dry, vascular plant vegetated parts (hummocks or hummock strings), and

moss or graminoid dominated lawns in-between these structures (Malmer et al 1994). The distributions and proportions of these structures over the mire surface, vary with climate, and by tradition has been used as the main basis for classification of different mire types (e. g. Ruuhijärvi 1960, Eurola 1962, Ruuhijärvi 1983, Sjörs 1983). These mainly originate from interactions between vascular plants and mosses (Wallén and Malmer 1992, Malmer et al 1994), together these structures form a system of interdependent functional units with different carbon accumulating properties. The hummocks constitute sites with the highest productivity of litter with the highest decomposition rate, and below which highly humified peat with high hydraulic resistance is deposited. Hollows and lawns constitute patches with comparatively lower productivity of a recalcitrant litter, and they thus deposit peat with a comparatively low degree of humification and with low hydraulic resistance (fig. 1). The patchiness of the mire surface vegetation and structure, is thus supposed to be the cause of considerable variation in rate of mass accumulation over the mire surface and thus to an uneven distribution of peat with different bulk density and hydraulic properties. Since the main flow of excess water from the bog is in the surface layers, from the hummocks towards the lawns and hollows through interconnected hollow systems towards the mire margins (Streefkerk and Casparie 1989) the quantitative distribution of these topographical units is of vital importance for the mire hydrology and consequently for the carbon balance of the entire mire. Both model calculations in Clymo (1984) and empirical data (Malmer and Wallén 1992) suggest that even very small changes in water level are decisive for the peat accumulation rate.

An increased proportion of hummocks compared to hollows may be supposed to shift the carbon balance for a mire as a whole towards an increased rate of peat accumulation since the average height of the ground water table will increase due to an increased average peat density. On the other hand, a decrease of the area covered by hummocks will cause a lowered average water table, and thereby a shift towards reduced carbon accumulation rate. A change in climate towards increasing temperatures and increasing CO_2-concentrations will influence both the rate of production and the rate of decomposition, and thus probably bring about a shift in the carbon balance in both hummocks and hollows, in either a negative or positive direction. If this shift is unequal in size between hummocks and hollows, i.e. if the relative proportion of carbon accumulation rates between hummocks and hollows changes, the relative covering of hummocks and hollows will change - towards an expansion of hummocks if lawns exhibit a relatively increasing inclusion of carbon compared to hummocks, and a reduction of hummocks if the opposite is valid.

Production and decomposition

We attempt to compare the current dry mass balance and apparent carbon accumulation rate in ombrotrophic hummocks and lawns, using ^{14}C-inoculations to determine the short term rate of input of litter to these two peat-forming systems, and using the variation of nitrogen in the surface layers to determine the rate of transfer of mass from acrotelm to catotelm (Malmer and Wallén 1993). The study was performed in two ombrotrophic tundra mires around Abisko, N. Sweden.

Methods

Since the mires around Abisko ($68^{\circ}21'N$, $18^{\circ}50'E$), N. Sweden, represent a climatically conditioned transition between eastern and western tundra mires, they may supposedly be sensitive to climatic changes. The sampling was partly done on the Stordalen mire c 10km E Abisko consisting of extensive ombrotrophic mire areas with hummocks and hollows, and partly on a mire close to Abisko, consisting only of hummocks. This last mire is smaller and more sheltered than the Stordalen mire.

The productivity of Sphagnum fuscum hummocks and of S. balticum hollows was measured over 14 years using a method involving labelling of the moss layer with $^{14}CO_2$ to get an innate marker in the moss plants. Each plot was covered with a 35 cm high, transparent Perspex chamber with a basal area of either 20 * 20 or 31.5 * 31.5 cm. Each chamber was supplied with 50 - 500 µCi ^{14}C by acidification of a of $NaH^{14}CO_3$ -solution inside the chamber by injecting 1-M HCl through a rubber membrane in the wall. The treatment lasted for about 8 hours. For the sampling we used a corer, 10 cm in diameter and 50 cm in length. The cores were deep frozen within a few hours after sampling, and later sliced (slices 1 cm thick) by sawing.

Subsamples of moss material (stems, branches, leaves) were extracted from each slice. The samples as well as the remaining part of each slice were dried at 85 $^{\circ}C$ and weighed to obtain the dry bulk density. The ^{14}C-activity in the moss samples was determined by combustion in a Packard Tri Carb Sample Oxidizer, trapping the CO_2 in a Carbo Sorb solution with a scintillation fluid added, and counting the disintigrations of ^{14}C in a Packard Tri Carb Scintillation Counter.

The apparent rate of litter formation in the hummocks and in the hollows was obtained from the ^{14}C -inoculated plots. The total amount of organic matter accumulated since the inoculation was divided by the number of vegetation periods since the inoculation.

Nitrogen was determined by a semi-micro Kjeldahl method. The loss of carbon by decomposition in the acrotelm has been estimated from the concentrations of N based on the assumption that all nitrogen once supplied to the bog surface becomes incorporated in the organic matter and remains there (Björck et al. 1991, Malmer and Wallén 1993). This is a particularly reasonable assumption for ombrotrophic mires with a conservative nitrogen cycling (Rosswall and Granhall 1980) and with a N-limited plant growth (Aerts et al. 1993) like the mires around Abisko.

The decomposition of the peat in the acrotelm implies weight loss from the system at a constant proportional rate, viz.:

(1) $M_t = M_0 e^{-kt}$

where M_0 is the amount of organic matter added to the system during a certain period of time; M_t is the amount of M_0 remaining at time t, and k the proportional decay

constant. As nitrogen is conserved the loss of organic matter due to decomposition will cause an increase in nitrogen concentration, from N_0 in the newly formed plant litter to N_t at time t. In equation (1), M_0 and M_t can then be replaced by the nitrogen free organic matter, W_0 resp. W_t, where

(2) $W_0 = M_0(1 - N_0)$ and: $W_t = M_t(1 - N_t)$

Provided that nitrogen is continuously conserved in the organic matter and S_N = **rate of supply of N**, has remained constant over time t., the accumulated amount of nitrogen can be used as a time scale:

(3) $t = N_{cum}/S_N$

where N_{cum} is the amount of nitrogen accumulated down to a defined depth.

Substituting the time factor t in equation (1) with equation (3), k with k_N as the time scale is changed to N_{cum}, and calculating with W_0 and W_t instead of M_0 and M_t (eq. 2) gives:

(4) $W_t/N_t = (W_0 e^{-kN*Ncum/SN})/N_0 = (W_0/N_0)e^{-kN*Ncum/SN}$

Taking the natural logarithm of (4) gives:

(5) $\ln(W_t/N_t) = \ln(W_0/N_0) - (k_N/S_N)N_{cum}$

which is the equation for a straight line with cumulative nitrogen (N_{cum}) as the X-scale and $\ln(W_t/N_t)$ as the Y-scale. The intercept will indicate the proportion W_0/N_0 in newly formed litter, and from the slope of the line, k_N, it is possible to estimate the decay constant for the peat as:

(6) $k = -k_N*S_N$

All these calculations presuppose that (1) there has been no change in the nitrogen deposition rate during timespan t, (2) the concentration of nitrogen in the litter at litter deposition level (N_0) has been the same over t, viz., there has been no systematic change in productivity or species composition during t, and (3) any decrease in the decay rate (k) with depth as an effect of a differential decay is disregarded. In the acrotelm the decomposition rate is much higher than in the the catotelm (Clymo 1978). The regression of $\ln(W/N)$ on N_{cum} in eq. (5) will therefore have a steeper slope for the samples above the transition from acrotelm to catotelm (here the group of samples below litter deposition level, together giving the highest value on R^2 (Malmer and Wallén 1993).

The proportion, a, of the original organic matter remaining at the transition between acrotelm and catotelm has been calculated as:

(7) $a = (W_t*N_0*100)/(W_0*N_t)$

Results and discussion

In each of the sampled peat cores there was a distinct peak value in the ^{14}C activity at the level corresponding to the moss surface at the time for pulse labelling, in 1980 (Fig. 2). The depth of this level was between 6 and 8 cm below the surface for most of the cores from the more sheltered hummocks at the Abisko-mire, and for the cores from the hollows at the Stordalen mire, while the cores from the hummocks on the Stordalen mire showed greater variability - between 2 and 8 cm.
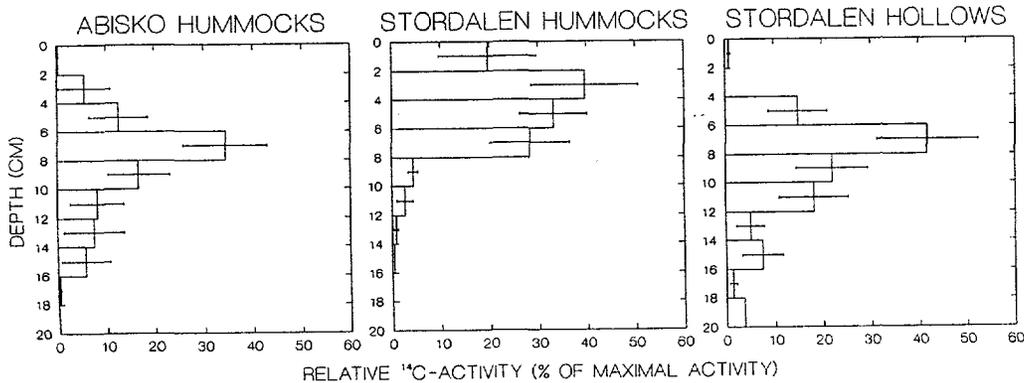
Fig. 2 Distribution of ^{14}C (calculated as % of the highest activity (DPM mg^{-1}) found within each sampled core at successive 2cm levels below surface in S. fuscum dominated hummocks on the mire at Abisko (left, N=11), in S. fuscum dominated hummocks on the mire at Stordalen (centre, N=12), and in S. balticum dominated hollows (left, N=10), 14 years after a pulse labelling. Means ±2SE.

The highest average accumulation rate of organic matter, c 190 g m^{-2} y^{-1}, was found in the S. fuscum hummocks on the Abisko mire (Fig. 3). At the Stordalen mire, the rate of accumulation was nearly half of the above, c 100 g m^{-2} y^{-1}, both in the hollows and in the hummocks. It was evident that exposure to wind and snow drift was a main factor limiting the productivity and thus also the apparent peat accumulation rate. The study also shows that the rate of accumulation in hollows and hummocks on the same mire, did not differ.
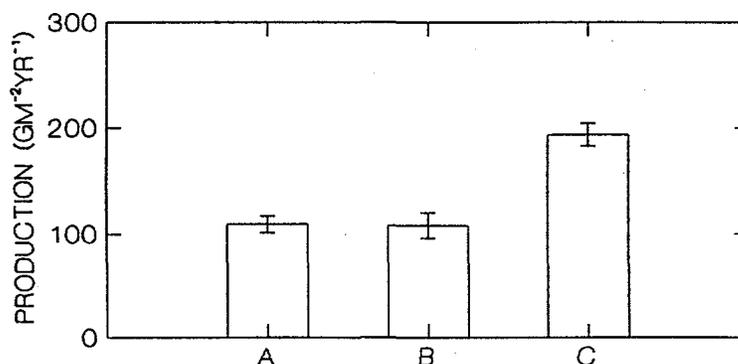
Fig. 3 Mean annual accumulation of dry mass (g m^{-2} year^{-1} ±SE) in S. balticum dominated hollows (A, N=10) and S. fuscum dominated hummocks (B, N=12) on the mire at Stordalen, and S. fuscum dominated hummocks on the mire at Abisko (C, N=11), between the years 1980 and 1994.

Input rates at the Stordalen mire, calculated from the quotients W_0/N_0 and a N deposition rate (S_N) of 0.4 g m^{-2} y^{-1} gives somewhat lower values, c 75 g m^{-2} y^{-1} for the hummocks, and c 80 g m^{-2} y^{-1} for the hollows. Despite similar rates of accumulation of litter at the surface and a considerable difference in the depth of the border between acrotelm and catotelm below hummocks and hollows, the organic matter annually transferred into the catotelm was on average 51% and 52% of the annually produced litter at the surface of the hummocks and the hollows respectively. This study shows that the shorter residence time for the litter in the acrotelm in the hollows is compensated for by a considerably higher decomposition rate of litter compared to the hummocks (Fig. 4).
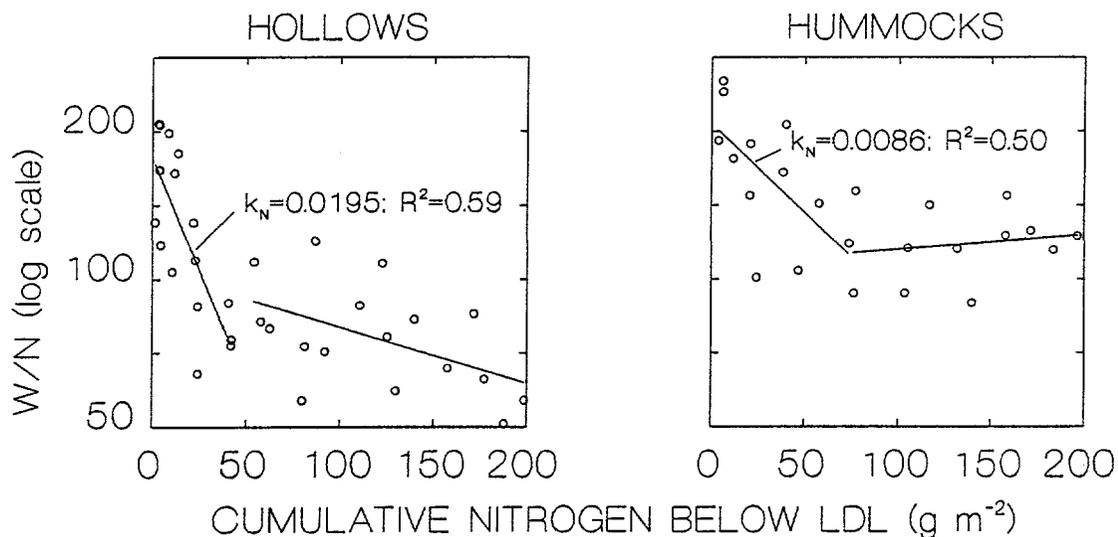
Fig. 4 The variation below the Litter Deposition Level (LDL) in the quotient W_t/N_t (log scale) in relation to depth expressed as cumulative amount of nitrogen (N_{cum}) in cores from S. fuscum hummocks (right) and S. balticum hollows (left). The regression lines are calculated according to eq: $\ln(W_t/N_t)=\ln(W_0/N_0)-(k_N/S_N)N_{cum}$. For the samples above the transition between acrotelm and catotelm, k_N and R^2 are indicated.

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Modelling
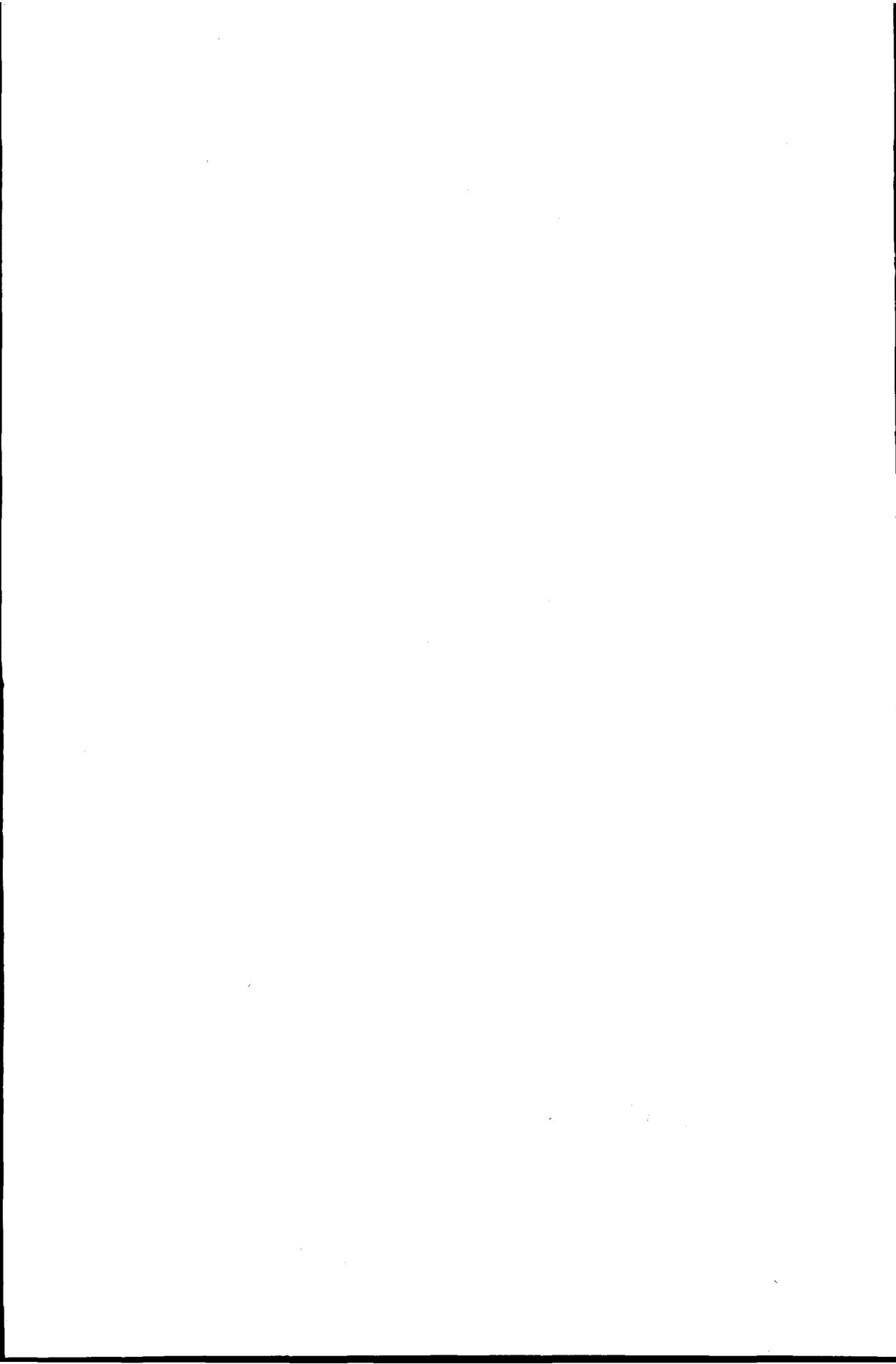
Modelling of peat temperature and moisture profile

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Peat soil is a considerable store of carbon (C) and nitrogen (N), consisting of 50 % (dry weight of organic matter) of C and 0.5-2.5 % of N. In the cool and humid climate of the northern hemisphere, peat deposits currently harbour 300-450 Pg C (Gorham 1991). Carbon is sequestered from atmosphere to peat when annual primary production exceeds ecosystem respiration and litter decomposition. In addition to CO_2 exchange between peat and the atmosphere, other radiatively active trace gases (CH_4, N_2O) are emitted from peat. The processes involved in carbon balance and trace gas emissions depend largely on peat moisture which controls e.g. the oxic/anoxic conditions in peat. Dynamics of these processes depend highly on daily weather variations. Predictions of the effects of climate change must thus be based on simulation of both weather and the microbiological responses to the temperature and moisture dynamics in peat. Models are needed for predicting the response of mire ecosystems to climate change scenarios. Peat properties such as high water retention capacity, gas bubble formation, low hydraulic conductivity and shrinkage and swelling of peat matric following water table fluctuations are crucial elements in peat soil models.

The aim of the current research is to provide a model of physical environment of the microorganisms in peat. This peat soil model will be coupled to modules dealing with forest growth and the processes of CH_4 formation, oxidation and transport (Kettunen et al., this volume). The climate will be simulated according to given scenarios by a weather generator module (Strandman et al. 1993) with regionally interpolated parameters.

Modelling peat water retention

Knowledge of water content of peat profile relative to the level of water table is a prerequisition for modelling both diffusive heat transfer and aerobic status in peat. Such data was available for different peat types and could be used to test the performance of different model approaches (Alm et al. 1995). Three model types for peat water content (θ) were evaluated on basis of data from peat samples, measured over a range of matric potentials (pF 1.0, 1.5, 1.8, 2.0, 3.0 and 4.2). Model shape parameters were estimated using peat characteristics such as bulk density and botanical peat components (remains of Sphagnum, Carex, Eriophorum and wood) determined from the same samples as explaining variables. The objectives of the model identification test were:

(a) to check the performance of the equation on different types of peats,
(b) to test the need of all parameters in describing the data points
(c) to check the sensitivity of the parameters of the chosen models

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Three basic equations tested for describing the moisture retention curve:

1) The equation by Van Genuchten (1980):

$$\theta = \theta_r + \left(\theta_s - \theta_r\right) \cdot \left(1 + (\alpha \cdot h)^n\right)^{-1 + 1/n}$$

2) The bimodal equation by Zhang and Van Genucten (1994):

$$\theta = \theta_r + \left(\theta_s - \theta_r\right) \cdot \left(\frac{1 + c_1\left(\alpha \cdot h\right)}{1 + \left(\alpha \cdot h\right) + c_2\left(\alpha \cdot h\right)^2}\right)$$

3) A semi-empirical equation derived in this study:

$$\theta = e^{\ln(\theta_s) - \left(\ln(\theta_s) - \ln(\theta_{wilt})\right) \cdot \left(\frac{pF}{4.2}\right)^k}$$

Symbols used in equations

θ is the water content ($cm^3 cm^{-3}$) of peat, θ_r is the residual water content ($cm^3 cm^{-3}$), θ_{wilt} is the wilting point water content ($cm^3 cm^{-3}$ at pF =4.2), θ_s is the saturated water content ($cm^3 cm^{-3}$), h is the pressure head in H_2Ocm, h_{pF} is the pressure head in pF ($\log_{10}(H_2Ocm)$, α, n, m are parameters defining the Van Genuchten curve shape; α, c_1 and c_2 are parameters defining the Zhang and Van Genuchten curve shape; k is a single parameter defining the semi-empirical curve. Such models are shown to be most sensitive for errors in saturated water content θ_s. Therefore, we did not estimate θ_s from data, but used a value derived from bulk density ρ and the density of solids g_s as θ_S (%) $= (g_S - \rho)100 / g_S$. A value of 1.5 g cm^{-3} was adopted as g_S.

Table 1. Model families (1-3) derived from equations 1-3 and tested on basis of peat moisture retention data. Symbols are described in the equations box.

Model 1a:	$\theta = \theta_r + (\theta_s - \theta_r)(1 + (\alpha h)^n)^{-m}$
Model 1b:	$\theta = \theta_r + (\theta_s - \theta_r)(1 + (\alpha h)^n)^{-1 + 1/n}$
Model 1c:	$\theta = \theta_r + (\theta_s - \theta_r)(1 + (\alpha h)^n)^{-1}$
Model 1d:	$\theta = \theta_s(1 + (\alpha h)^n)^{-m}$
Model 1e:	$\theta = \theta_s(1 + (\alpha h)^n)^{-1 + 1/n}$
Model 1f:	$\theta = \theta_s(1 + (\alpha h)^n)^{-1}$
Model 2a:	$\theta = \theta_r + (\theta_s - \theta_r)(1 + c_1(\alpha h))/(1 + (\alpha h) + c_2(\alpha h)^2)$
Model 2b:	$\theta = \theta_r + (\theta_s - \theta_r)(1 + (\alpha h))/(1 + (\alpha h) + c_2(\alpha h)^2)$
Model 2c:	$\theta = \theta_s(1 + c_1(\alpha h))/(1 + (\alpha h) + c_2(\alpha h)^2)$
Model 2d:	$\theta = \theta_s(1 + (\alpha h))/(1 + (\alpha h) + c_2(\alpha h)^2)$
Model 3a:	$\theta = \exp(\ln(\theta_s) - (\ln(\theta_s) - \ln(\theta_{wilt}))(h_{pF}/4.2)^k)$
Model 3b:	$\theta = \exp(\ln(\theta_s) - (\ln(\theta_s) - \ln(\theta_{wilt}(\rho,l,c)))(h_{pF}/4.2)^k)$

Six different versions belonging in model family (1), four in model (2) and two in model (3) were prepared for identification of the best model types. A set of 25 series of measured peat water content values representing different peat characteristics at different depths was used in testing the models. Models (1)-(3) were compared to each other and to restricted model variants having fewer parameters with a suitable F-test. The F-test and the holding of the restrictions for m and n in all test cases suggested the models 1e and 3b for further analysis.

The parameters defining the shape of the moisture retention curve by Van Genuchten (1e), n and $\log_{10}$ (α) can be estimated with coefficients of determination (r^2) of 38 % and 53 %, respectively. The most important peat properties controlling both parameters in 1e are the bulk density and, if the sample is situated in the top layer of peat (in the litter layer), the *Sphagnum* content of peat.

The parameter k defining the shape of the empirical water retention curve (3b) can in turn be estimated with r^2 = 68 %. Again the most important peat property controlling the parameter is bulk density and the interaction of *Sphagnum* content with the sample layer depth. Overall coefficient of determination gives 93 % fit for both model 1e and 3b.

The two parameter van Genuchten's model (1e) is slightly more accurate than the semi-empirical model (3b), but the larger number of parameters in van Genuchten leads to lower determination of these parameters. Hence, the semi-empirical should be used to model peat water retention. Peat botanical components clearly determine the shape of the peat water retention curve. In practice, however, determining those parameters from peat samples is very time consuming. A range of peat samples should be inspected with both microscopy and field ranking techniques to obtain distributions of true peat composition and rough field estimations. An approach to establish this connection is under development stage.

Modelling the dynamics of water table and peat temperature

Peat water retention curve (semi-empirical model 3b) is applied in the models of water table and peat temperature (Fig. 1 A,B). Development of these models in still underway, but working prototypes for summer period are already running. Promising results have been obtained when simulating water balance in the well studied Lakkasuo mire complex. Different model approaches can be selected e.g. to assess evapotransipration, including those presented by Laine (1984), Ahti (1987) and Monteith-Penman. The submodels for peat water balance have been coded using object techniques to match the forest growth simulation model FINNFOR (Kellomäki *et al.* 1993) and weather generator (Strandman *et al.* 1993). Thermal submodel is based on conduction theories of Romanov (1968) or on thermal properties of average peat or organic matter. Heat transfer by convection is still under construction.

Fig 1. (Following pages) Schematic presentation of PeatSoil model. Modules for peat temperature (A) and water balance (B) are presented separately in the following pages. Connections to other (*future* and existing) model modules are indicated, as well as to the database containing the parameter layers constraining the weather generator. The model is operating as an Excel-application, programmed using the Excel Visual Basic language.

FINNFOR Weather generator

Hourly values:
- temparature
- humidity
- wind
- radiation

Daily values:
- precipitation

Field measurement timeseries

precipitation

temparature, humidity, wind and/or radiation

selection

selection

Current WATER BALANCE model

FINNFOR Tree stand model (connection not yet estabished)

Net shortwave radiation at soil surface

Infiltration through canopies

Evaporation from vegetation

Vegetation climate parameters:
- rougness length
- reference height

Adjusted potential transpiration

Affecting stomatal conductance etc.

Interception (more detailed model in Finnfor tree stand model)

snow model

Infiltration

Root water uptake

Availability of water to plants, root zone air space

Potential evapotranspiration
Using pure vapour pressure deficit , pure radiation, Priestley-Taylor, Monteith-Penman or other approaches

Actual evapotranspiration
Using Laine(1984), Ahti (1987) or Monteith-Penman approach

Water store change
or
Water balance

Runon/Runoff
Characteristic curve from pond measurements

Water store derivative
or
groundwater coefficient

Previous water table

swelling/shrinking correction

Daily output of water balance model

Water table

input to heat model

and methane model

Peat profile: input parameters

Data from measurements for 1 to more than 20 layers

layer depth and thickness

layer botanical composition, bulk density

moisture retention layer parameters
curve shape parameter, water content at saturation and at wilting point

Peat profile:
layer moisture retention
Corresponding to current water table

moisture retention and average flow profiles
for depth profile of interest

input to methane model

Depth profile of interest
1 to more than 20 depths

274

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Modelling of methane emissions from boreal peatlands

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Introduction

Wetlands, especially boreal peatlands, are an important methane source in the biogeo-chemical methane cycle (Bartlett & Harris 1993). The methane emissions from peat-lands show high temporal and spatial variations (Windsor et al. 1992; Dise 1993; Saar-nio et al. 1996). The connection of environmental factors to methane fluxes is particu-larly important. Due to the complex interactions between environmental factors and methane emissions, one single factor can hardly be used to predict the methane flux satisfactorily. Process based models are needed for getting a deeper insight into the complex processes and for combining the information from measurements and literature to predict pointwise and cumulative methane fluxes from wetlands.

Data and modelling approach

Measurements of methane emissions and environmental factors (peat profile tempera-tures, water tables and precipitation) from a Finnish, boreal low-sedge *Sphagnum papil-losum* pine fen (Salmisuo 62° 47' N, 30° 56' E) during summer 1993 (Silvola et al. 1992) can be used to analyze the interactions in the peat ecosystem and to estimate related model parameters. In the modelling, a system analytical approach is applied.

The effects of environmental factors on the processes

Traditionally, the peat profile temperature has been considered an important control of the methane flux. Methane emissions have been found to increase with increasing tem-peratures (e.g. Shurpali et al. 1993; Kettunen et al. 199xa) when the water table remains high. Methane production rate increases with temperature (Dunfield et al. 1993). On the other hand, methane oxidation is also enhanced by increasing temperature (Dunfield et al. 1993). In sites with deep water table and thick aerobic layer the conditions may fa-vour methane oxidation more than methane production and increasing temperature may have a decreasing effect on the methane emissions. Methane emissions may be en-hanced by the temperature also through the increased diffusion rate of methane in the peat profile. In addition, the solubility of oxygen in peat pore water decreases with in-creasing temperature and oxidation deteriorates thus increasing methane fluxes.

Methane emissions were found to correlate more strongly with peat temperatures at depths of 10-50 cm than with the peat surface temperature (Kettunen et al. 199xa). This suggests that methane is produced in deeper layers than the peat surface layer. The explanatory power of the peat temperatures at various depths differed for the microsites indicating the effect of the depth of the water table and other site spesific factors (Kettunen et al. 199xa).

As the effect of peat moisture on methane emissions is considered, the relationship between water table and methane emissions appears very complicated (Windsor et al. 1992; Moore and Roulet 1993; Frolking and Crill 1994). Spatial variations are apparently related to average hydrological conditions and other site spesific factors such as peat quality, while seasonal and diurnal variations may be related to temporal changes in soil hydrology. Methane production requires anaerobic conditions while methane oxidation takes place in aerobic peat. Consequently, the higher the water table were on average, the greater the flux would be (Roulet et al. 1993).

However, methane emissions measured with high frequency were found to be high when water table was low and vice versa (Kettunen et al. 199xa). Lowering water table and drops in atmospheric pressure can release the pore water methane (Windsor et al. 1992; Moore and Roulet 1993) causing high fluxes with deep water tables. The precipitation has been suggested to suppress the methane emissions for few days (Frolking and Crill 1994) resulting in low emissions with high water tables. Furthermore, the water table time series had high autocorrelative properties and stayed constantly high during the measurement period (Kettunen et al. 199xb). Therefore, the effect of the water table fluctuations were not evident in these data (Kettunen et al. 199xa, 199xb).

Precipitation and the following rises in water table were found to increase methane emissions either with lag or instantly (Kettunen et al. 199xb). The flarks showed a rapid response to rain pulses and rises in water table and the response from the lawns and the hummocks was slower (Kettunen et al. 199xb). The differences in the time lags may be explained by the transport of methane from peat to atmosphere as methane produced in the hummocks and the lawns passes a longer distance before reaching the peat surface than methane produced in the flarks.

Plants contribute to methane flux in several ways. Part of the photosynthetic carbon is secreted as root exudates and fine root litter which act as substrates for methanogenesis (Whiting and Chanton 1993). Plants may have an important role in transporting methane upwards and oxygen downwards between atmosphere and peat profile in aerenchymous cellular tissue (Schutz et al. 1991).

Evaluation of the regression model approach

Methane emissions have been modelled by linear regression models with temperature and depth of water table as independent variables (Kettunen et al. 199xa). The goodness of fits of the regression models were between 60 and 80 % for the flark and lawn microsites. In the hummocks, the fits were less satisfactory as oxidation seems to be important in the hummocks. In addition, the contribution of episodic fluxes is probably

higher in hummock sites than in flark and lawn sites. The regression models produce the general seasonal pattern of methane emissions (Kettunen et al. 199xa). However, the models overestimated the emissions at the beginning of the summer and underesti- mated the high midsummer emissions. The systematic difference in the model predic- tion and measured values may be related to time lag of temperature cycle in the peat profile and the development and role of the vegetation within the microsites during the growing period (Schutz et al. 1991; Whiting & Chanton 1993) which is not included in the regression models. Furthermore, the regression models could not capture the epi- sodic pulses found in the data (Kettunen et al. 199xa). The contribution of episodic fluxes in different microsites range from a few percents up to 20 % depending on the criterion used to distinguish the episodic values. The episodic pulses may be related to the enhancement of methane emissions after rain showers (Kettunen et al. 199xb).

Regression models cannot reliably predict the effect of climate change on the methane flux as the statistical relationships holding in current conditions may not be valid in cli- matic conditions that fall outside the range of observations in model validation. Contra- dictorily, a system analytical model in which the basic processes are correctly identified and structured can be applied to describe the non-linear interactions and functioning of the system even under changing conditions. The effects of climate change on the meth- ane cycle in wetlands could thus be anticipated using the process based model.
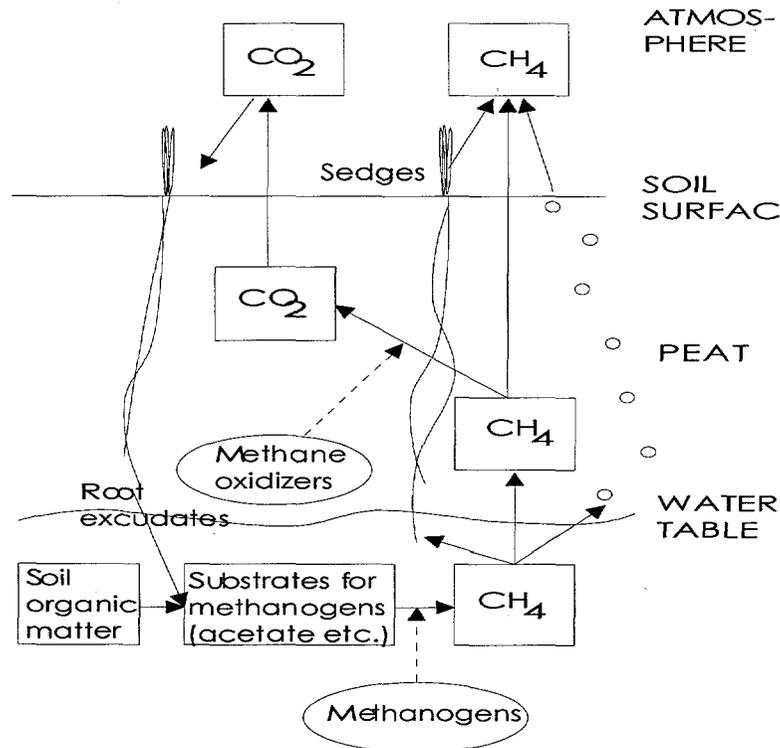

Figure 1: The structure of the methane emission model.

Process based modelling

In the dynamic differential equation system model (see Fig. 1 for the schematic presentation of the system) peat profile is vertically divided to layers and peat microbes and chemical compounds as state variables. Methane emissions are affected by three processes, i.e., methane production, methane oxidation and transport from peat to atmosphere. The methanogens reduce acetate and carbon dioxide to methane in anaerobic conditions. The methanotrophic bacteria consume methane by oxidizing it in microaerophilic conditions. Methane is transported via plants from peat to atmosphere and by diffusion between the layers. In the plant flux, methane is transported into the sedge roots and further to the atmosphere via aerenchymous cellular structures.

Conclusion

Methane emissions are affected by both methane production, methane oxidation and transport from peat to atmosphere, which are all influenced by changes in environmental factors. Thus, very complex relationships exist between the environmental factors and methane emissions. In addition, the environmental factors themselves are often both highly autocorrelated and cross-correlated to each other, leading to difficulties in the interpretation of the statistical relationships.

The modelling work was started by statistical analyses that shed light on the effects of some important environmental factors to the methane emissions and gave information on the basic interactions in the system. These analyses provide a good basis for understanding the complex biogeochemical processes, affecting the methane emissions from wetlands. Based on the knowledge on the system, a process based model is under development.

Overall, the systems analytical modelling is a promising approach for understanding the methane emission dynamics. At later stage, the carbon dioxide balance and dynamics will be studied. By combining the models describing the moisture and temperature balance (Alm and Weiss 1996), the methane model and carbon dioxide model, the peat carbon fluxes can be simulated.

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On explaining methane fluxes from weather, soil and vegetation data via the underlying processes

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Introduction

Methane is responsible for about 15 % of the estimated enhanced greenhouse effect. The uncertainty about the different sources and sinks is large. Wetlands are probably a significant source of methane which magnitude is uncertain. We aim at reducing the uncertainty in estimates of wetland methane fluxes under various climatic and management conditions by investigating the relationship between methane fluxes and weather, soil and vegetation data.

Several statistical models have been developed for explaining methane fluxes. Time averaged water table (Moore and Knowles, 1989; Moore and Roulet 1993), temperature (Hogan, 1993), and Net Ecosystem Production (NEP) (Whiting and Chanton, 1993) were correlated with methane emissions from wetlands. Those models are useful to indicate the influence on methane fluxes of major environmental variables. However, results of such models still contain a large unexplained variation and may not be extrapolated reliably, because the causes of the correlations are not well known.

Methane fluxes from and to soils are a result of methane production, methane consumption and methane transport. Methane production is due to a microbiological process, which can occur when organic matter is degraded anaerobically. Methane consumption is also caused by a microbiological process that generally needs oxygen. Methane transport is a physical process. These three processes depend on each other and on a number of other interacting processes: gas, water and heat transport, and the dynamics of soil carbon, electron acceptor and vegetation. In Figure 1 the major relations between these processes and soil, weather and vegetation data are given.

Here, we investigate the possibility to explain methane fluxes from weather, soil and vegetation data via the underlying processes by developing process based models. We focus on methane production, methane consumption and the role of vegetation, thereby indicating the link to general C–cycling studies in wetlands. We

do not consider the effect of temperature on the different processes, because this can be treated as a correction on the processes. Keynote is the direct or indirect link between process rates and weather, soil and vegetation data.
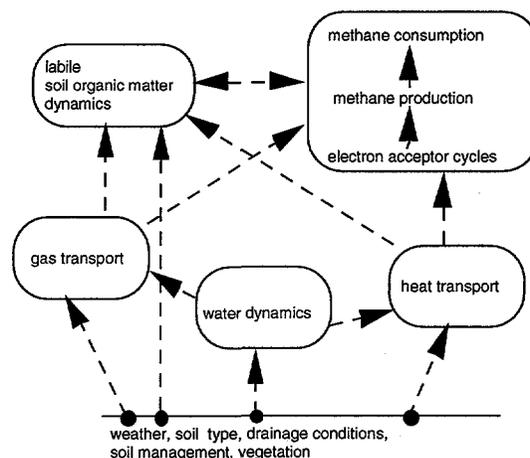

Fig. 1 **The major relations between processes, which affect methane fluxes and the influence of weather, soil and vegetation data.**

Methane Production

Methane production can occur when organic matter is degraded anaerobically. To put this in a formula:

$$MP = I \cdot C \cdot F$$

where MP is the methane production rate. The other symbols are explained below.

The inhibition function, I, is zero under aerobiosis and one under anaerobiosis. Initially one could assume, that soils are aerobic above the water table and anaerobic below the water table. In second instance, partial anaerobiosis above the water table in microsites and partial aerobiosis below the water table in the rhizosphere of vascular plants can be included. Both these heterogeneities are situated in zones with fresh organic matter, which may show a high methane production rate.

The rate of anaerobic organic matter mineralisation, C, can be related to the availability of organic material. A principle question concerning this relation is, whether principles of aerobic organic matter mineralisation can be extended to anaerobic C-mineralisation. Generally, anaerobic C–mineralisation proceeds slower than aerobic C–mineralisation. Furthermore N limitation is less likely to occur under anaerobic conditions compared to aerobic conditions (Gaunt et al., 1995), because the yield of biomass on consumed carbon is relatively low, which causes that less structural N is needed per amount of carbon transformed. Both the relatively low C–mineralisation rate and the low yield of biomass on consumed carbon can be explained by the relatively low energy yield per transformed carbon in anaerobic conditions.

The fraction of anaerobically mineralised carbon that is converted into methane, F, is probably the most difficult factor to determine. If methane production is the major terminal electron acceptor process a ratio of about 0.5 would be expected for stoichiometric reasons: $CH_2O \rightarrow 0.5\ CO_2 + 0.5\ CH_4$. However, reported values of F vary from about 0.5 to 10^{-4} (Valentine et al.; 1994; Amaral and Knowles, 1994; Yavitt et al., 1987; Yavitt and Lang, 1990). When F is below the theoretical maximum of about 0.5, CO_2 is produced which cannot be attributed to methane production. Yavitt and Lang (1990) and Amaral and Knowles (1994) could explain part of the non-methanogenic CO_2 production by SO_4^{2-} reduction, but still a large part remained unclear. There are several other processes, which could explain the non–methanogenic CO_2 production, such as accumulation of H_2 and alcohols, anaerobic S–cycling, reduction of NO_3^-, Fe^{3+} or Mn^{4+} or reduction of large organic molecules. However, not much is known about the importance of these.
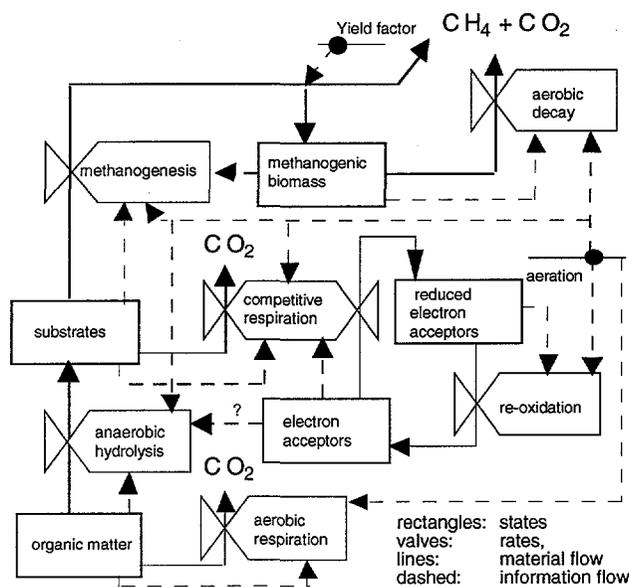
Fig 2. The conceptual methane production model. Methane production is controlled by organic matter, the competition for substrates of methanogenic reactions with electron acceptor reducing reactions and methanogenic biomass.

In Figure 2, the relational diagram of the methane production model is given. It is an extension of the equation $MP = I \cdot C \cdot F$. Methane production is driven by organic matter availability and aeration. Besides, the size of the pool of electron acceptors should be known. Three kinds of anaerobic reactions are distinguished: (i) methanogenesis, (ii) reactions that control the production of substrates: anaerobic hydrolysis, and (iii) reactions that compete with the methanogens for these substrates. Two kinds of aerobic reactions are included: (i) aerobic respiration, which depletes the organic matter pool and (ii) re–oxidation of the electron acceptors.

It is assumed that of these reactions only methanogenesis could be limited by the concentration of biomass. Kengen and Stams (in press) observed acetate

accumulation, indicating methanogenic biomass limitation, in anaerobically incubated soil samples from an originally aerobic soil. However, in wet soils biomass limitation does not seem to occur often, as the addition of substrates mostly enhances methane production (Valentine et al., 1994; Amaral and Knowles, 1994). So, only prolonged aerobic periods seem to poison or starve methanogenic bacteria.

Competition for substrates of methanogenesis with electron acceptor reducing reactions plays a central role in this model . It could explain the large variation in F. Because methanogenesis is energetically unfavorable compared to many other terminal electron acceptor processes, it is a bad competitor. This implies that the methanogens have to live on the left–over substrates. The fraction left–overs could vary strongly, depending on the amount of electron acceptors and on the amount of organic matter available. Consequently, the relation between organic matter availability and methane production could be dynamic and non–linear. This would imply that heterogeneity of organic matter availability could influence methane production. About the cycling of electron acceptors in wetland soils little is known. In conclusion, the predictability of methane production may be limited.

Methane Consumption

In fresh water wetlands it is often assumed that methane consumption is an oxic process (Nedwell and Watson, 1995), performed by methanotrophs. This can be expressed by the following equation:

$$V = V_{max} \cdot \frac{[CH_4]}{[CH_4] + K_{CH4}} \cdot \frac{[O_2]}{[O_2] + K_{O2}}$$

where V is the volumetric methane oxidation rate, K_{CH4} and K_{O2} are the half saturation constants for methane and oxygen. V_{max} is the maximum volumetric methane oxidation rate at a certain concentration of active methanotrophs. These parameters have been measured in laboratory. Typical values of K_{CH4} vary between 1 and 92 µM and values for K_{O2} vary between 0.1 and 37 µM (e.g. King, 1992). V_{max} ranges from zero to several tenths of µmol m^{-3} of soil s^{-1} (King et al., 1990; Sundh et al., 1995; Moore et al., 1994).

It is investigated whether it is possible to explain variation in V_{max} with a submodel for methanotrophic biomass driven by both methane and oxygen availability. In this respect it is encouraging that maxima of V_{max} are found near the interfaces of aerobic and anaerobic zones, namely near water tables (Sundh et al. 1995; Vecherskaya et al., 1993) and in the rhizosphere (Gerard and Chanton, 1993). At these interfaces both the methane and oxygen concentration could be high. There are kinetic data on growth (e.g. Nagai et al. 1973) and decay (Roslev and King, 1994) of methanotrophs. Observed yields are mostly about 0.5 mole C–biomass per mole C–CH$_4$, but values of 0.1 and 0.8 have been observed as well. (Megraw and Knowles, 1987, Vecherskaya, 1993; Nagai et al., 1973; Lidstrom and Somers, 1984).

Vegetation

Vegetation influences methane fluxes in two ways. Firstly, it acts as a source of fresh organic matter by decay and root exudation and, secondly, vascular plants can exchange both methane and oxygen between rhizosphere and atmosphere via the aerenchyma.

Methane production often varies strongly with depth. To explain this the depth distribution of fresh organic matter, a major controlling factor, should be known. Consequently estimates of root distribution and root turnover / root exudate production are necessary.

To explain the gas transport capacity of a certain vegetation, one should relate transport parameters to vegetation parameters. If methane production and consumption are treated depth dependently, also transport should be treated depth dependently. The simplest way to do this is as follows:

$$\begin{matrix} \text{vegetation} \\ \text{mediated} \\ \text{CH}_4 \text{ flux} \end{matrix} = \int_{\text{soil bottom}}^{\text{soil surface}} h_{\text{trans}}(z) \cdot [CH_4](z) \cdot dz,$$

where z is the distance to the soil surface, and $h_{\text{trans}}(z)$ the depth dependent gas transport coefficient. In this relation it is assumed that gas transport is proportional to the concentration difference between soil and atmosphere, where the latter concentration was assumed to be negligible compared to the first one. These parameters may be related to depth dependent root density. Vegetation mediated oxygen transport into the soil may not depend linearly on the concentration difference, because a large part will be consumed within the plant. In some vascular plants gas transport is partially controlled by radiation, humidity, or wind (Armstrong et al., 1992; Morrissey et al., 1993). This implies that $h_{\text{trans}}(z)$, should also be related to weather conditions.

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Miscellaneous

Boreal peatlands and global climate change

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Peatlands occupy in FSU 165 mha. Peatlands of boreal zone (BZ) in Russia are represented by Sphagnum bogs mainly which cover about 68.5 mha (Botch et al.,1995).Their carbon pools are 92.1 Pg and they accumulate 31.4 g C m^{-2} yr^{-1}. It is 41% of C accumulated by russian mires. Bogs of BZ are lagest carbon pools and net sinks for atmospheric CO_2.

Bog area is restricted by boreal zone with average yearly air temperature of 3-4 ^oC. Crictical year temperature for bogs distribution is of 11 ^oC (Schouten et al.,1992). Bogs are the most common peatlands in BZ of the northern hemisphere. The diversity of bog form is connected with variety of environmental factors in BZ. Northern bogs are treeless and flat. They are strongly patterned. Southern bogs covered with pine trees and they are not patterned.

The rise of mean summer temperature at 3 ^oC will not be affected by negative effects for bogs. Climate warming may accelerate growth of pine trees and plant productivity.

Possible effects of climatic change on mire ecosystems have been discussed for last decade by some authors(Shouten et al., 1992; Van Dam and Beltman, 1992; Zoltai, 1994 and others).

Long-term monitoring can provide us tools for prediction of mire changes in the case of rise of mean annual temperature.

The paper is a result of 20-years monitoring on permanent plots of fens and bogs located in Ladoga lake basin (Leningrad region, 60^0 l.n.).Their vegetation belongs to following groups:
1. Pine bogs (2 plots)
2. String bogs with ridges (dwarf-shrubs, cotton-grass, Sphagnum fuscum) and wet hollows (Scheuchzeria, Sphagnum balticum, S. cuspidatum) (8 plots)
3. Sedge- and sedge-Sphagnum fens (53 plots).

They have been described every year since 1976. Cover (in %) of every species including mosses was noted. Weather characteristics such as an average air temperature for every month and precipitation were got from meteorological stations. Variations between years in precipitation and air temperature were correlated with cover of mire plants.

It is seen that cover of plants varied (table 1,2). We explain the cover changes by weather conditions in summer time of every year. In some cases plant cover correlated with sum of precipitation (table 3). It was noted for vegetation of all bog permanent plots. The cover of vegetation was more in rather humid years and it correlated positively with sum of precipitation ($r = 0.80$-0.94).

Fen and forest vegetation has an opposit reaction.It correlated positively with average summer air temperature ($r = 0.71$-0.94).

The results of Principal component analysis (PCA) showed that hollow vegetation is more stable in different years and fen ones is more unstable and varies strongly depending on weather conditions. Fen vegetation has a most cover in warm summer and rich in precipitation autumn of presciding year. Bog vegetation develops well in wet summer.

So impact of climatic change on bog and fen ecosystems in BZ will not change them strongly. They will stay in the frame of their annulal fluctuation.

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Table 1.   Cover dynamics of Chamaedaphne calyculata in permanent plots of BOR bog (%).

Plots   Years

	1976	1983	1989	1992	1993	1994	1995
105	<1	1.2+0.6	1.5+0.5	<1	2.3+1.1	1.8+0.7	1.8+0.5
107	3.0+0.7	0.6+0.2	0	<1	1.2+0.2	<1	1.3+0.6
109	8.3+2.0	<1	1.1+0.4	1.8+0.8	1.2+0.4	1.5+0.3	3.4+1.2
112	3.0+0.8	<1	1.1+0.8	<1	1.2+0.4	<1	<1
114	3.2+0.8	<1	1.1+0.4	1.1+0.4	<1	1.5+0.5	1.2+0.4
209	4.5+1.1	2.5+0.6	---	1.7+0.5	<1	1.5+0.4	2.6+0.9
216	1.7+0.5	1.3+0.5	---	1.7+0.8	1.7+0.7	2.3+0.8	4+1.6
602	5.2+0.9	2.5+0.5	2.9+0.6	3.2+0.6	3.4+0.8	5.5+0.2	4.8+1.0
604	2.2+0.5	2.1+0.7	4.2+0.9	1.4+0.7	4.2+1.1	2.2+0.6	3.2+0.5

Table 2.   Average values of species covers (%) in permanent plot
            No. 115 (BOR bog, Novgorod region)

                                    sedge-herb-Sphagnum bog

species   \   years   1976   1983   1989   1992   1993   1994   1995

Alnus glutinosa   0   0   0   0   0   0   1
Calla palustris   0   0   1   0   1   0   1
Carex canescens   0   1   0   1   1   1   1
Carex chordorrhiza   1.6   1   1   1.2   2.2   2.4   1
Carex lasiocarpa   22.3   3.8   5.9   5.7   4   3.8   1.8
Carex limosa   0   0   0   0   0   0   1
Carex rostrata   1   0   1   1   1   0   1
Comarum palustre   26   23.2   15.7   26.8   15.2   17.7   6.2
Epilobium palustre   1   1   1   1   1   1   0
Equisetum fluviathile   8.7   6.7   7.7   15.6   5.7   7   12.3
Eriophorum polistachion   0   0   1   1   0   1   0
Eriophorum vaginatum   0   0   0   1   0   0   0
Galium palustre   1   1.7   1   1   1   1   1
Iris pseudocorus   0   0   0   0   1   1   1
Menyanthes trifoliata   26.5   19   4.8   18.3   4.6   15.1   9
Naumburgia thyrsiflora   1.5   1.5   2.9   5.6   1.7   1   1
Peucedanum palustre   1.2   1.7   3.5   2   1   1   1
Squtelaria galericulata   0   1   0   0   0   0   0
Utricularia intermedia   1   0   0   0   0   0   0
Calliergon giganteum   4.3   1.4   1.7   0   0   0   0
Drepanocladus exannulatus 1.2   1   1   0   0   0   1
Mnium cinclidioides   15.7   16.5   19.3   3.1   0   1   1
Sphagnum fallax   0   0   0   0   0   10.2   2.3
Sphagnum magellanicum   0   0   4   0   0   0   0
Sphagnum obtusum   10.6   21.5   25.4   60.5   59.3   35.8   81
Sphagnum riparium   0   4.8   0   0   0   0   0
Sphagnum squarrosum   0   1   5   0   0   11.1   0
Sphagnum subsecundum   2.3   3.1   1   0   0   0   0

Table 3. Coverage correlation (all plots)

	shrub-herbs coverage vs.			moss coverage vs.		
plot	temp avg.	prec. sum	water level	temp avg.	prec. sum	water level
Bog:						
105	-0.23	0.88	-0.39	0.36	-0.84	0.28
107(ridge)	-0.59	0.85	0.34	-0.57	0.98	0.73
107(hollow)	-0.32	0.44	-0.20	0.03	0.48	0.55
109	-0.37	0.85	-0.62	0.39	0.07	0.47
112(ridge)	-0.43	0.66	-0.24	0.65	-0.12	-0.06
112(carpet)	0.14	0.29	0.39	0.55	-0.15	-0.28
112(hollow)	-0.39	0.88	0.04	0.62	-0.47	-0.56
114	-0.35	0.80	-0.33	0.62	-0.48	-0.07
209	-0.35	0.91	-0.05	0.91	-0.82	-0.33
212	-0.65	0.77	0.28	0.95	-0.76	-0.60
216	0.10	0.55	-0.72	0.97	-0.71	0.97
601	-0.17	0.43	-0.67	0.70	-0.73	0.99
602	-0.52	0.51	-0.09	0.70	-0.74	0.78
604(ridge)	0.19	0	-0.67	0.66	-0.68	0.75
604(hollow)	-0.42	0.94	0.57	0.59	-0.71	0.18
Fen:						
115	0.12	0.51	-0.67	0.62	-0.48	0.22
6	0.72	-0.70	-0.65	0.56	-0.72	-0.59
8	0.71	-0.75	-0.43	0.33	0.12	0.22
9	-0.41	-0.66	0.38	-0.37	-0.55	0.49
11	-0.06	-0.63	-0.28	-0.40	-0.37	-0.13
12	0.85	-0.56	-0.63	0.87	-0.30	-0.69
14	0.25	-0.42	-0.43	0.09	-0.84	-0.84
15	0.94	-0.23	-0.80	-0.09	0.36	0.70

FTIR analysis of peat water samples

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Introduction

Drainage and fertilization alter the activity of micro-organisms and the decomposition pathways of organic compounds and thereby affect the release of methane and carbon dioxide to the atmosphere. Very little is known about the quality and quantity of organic compounds and their mineralization in peat and peat waters.

Spectroscopic measurements have been shown to provide valuable information about the nature of soil organic matter. Attractive features of these methods are that they are quick and cheap.

Fourier transform infra red (FTIR) spectroscopy

The structure of the biological molecules is often very complex. Molecules may contain many different functional groups, which each make a contribution to the infrared spectra. The net result is a spectrum which is difficult to interpret. Broad absorption bands are overlapping and mixing of different vibration modes makes band assignments confusing (Gendreau, R.M. et al. 1987).

The situation is even more complicated when dealing with decomposed samples of biological origin such as peat. Such samples contain a large number of compounds. In addition to overlapping absorption bands, interaction of compounds makes interpretation of spectrum difficult (Holmgren, A. et al. 1988).

Multivariate methods

Statistical, multivariate methods have been used successfully in near-infrared spectroscopy (NIR) with biological samples for years. Most NIR applications quantitative, based on principal component regression. Principal Component Analysis (PCA) and Projection to Latent Structures (PLS) can effectively extract information from complicated data. Besides regression, PCA can be used for instance to classify samples and to interprete of spectra (Hasenoehrl, E.J. et al. 1993, Dupuy, N. et al. 1995). In this work, PCA is used to classify mid-IR spectra of peat water samples.

Materials and methods

The material for this report was collected from the Jaakkoinsuo water table regulation experimental area at Vilppula (62°3′N, 23°34′E). Sample plots are located in the drained part of dwarf schrub cotton grass pine bog. The peat samples were taken from

five different layers (0-30cm, 30-50cm, 50-70cm, 70-90cm and 90-100cm). Water was pressed out of the peat samples, filtrated and evaporated.

Spectra of the residue were collected with Perkin Elmer System 2000 FTIR spectrometer with DTGS detector using KBr-techniques. The transmittance spectra were recorded in the region 4000 - 450 cm^{-1}.

Statistical analysis were carried out with UNSCRAMBLER II v. 5.5 software package (CAMO AS). Classification were performed by using Multiplicative Scatter Correction (MSC) (Martens, H. et al. 1991) and Principal Component Analysis (PCA) (Martens, H. et al. 1991).

Results

Peat is a very heterogeneous material, and its chemical properties depend on the decay process of different plant material. IR spectrum of peat is complex and quite difficult to interpret.

The spectra of peat water samples are characterized by broad absorption band at 3600 - 2300 cm^{-1}, strong absorption at 1720 cm^{-1}, 1630 cm^{-1} and 1050 cm^{-1} and wavy absorption profile at region 1500 - 1100 cm^{-1}, which is caused by overlapping of several broad absorption bands.

PCA is the only reasonable approach to analyze spectral data from peat water samples. Some preliminary results of applying The PCA model to the spectra of peat water samples are presented in Fig. 1. After removing outliers, the PCA model divided
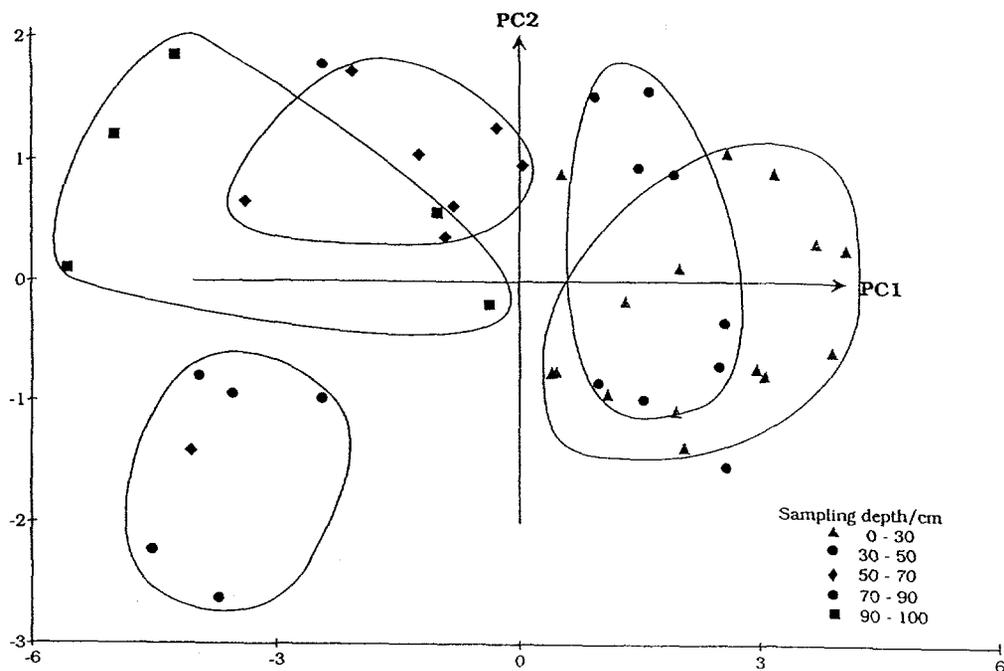

Fig.1. Score plot for the PCA of peat water samples.

the samples to five subgroups according the sampling depth.

The five first principal components (PC) explained 99% of data matrix variance. Only the first two PC's was used in classification of samples. The first PC explains 77% of variance and the information was extracted from 1720 cm^{-1} , 1500-1100 cm^{-1} , 1620 cm^{-1} and 3300-2300 cm^{-1}. The second PC (13%) used information from the bands at 1050 cm^{-1} , 3500-3100 cm^{-1} and 2900-2800 cm^{-1} (Figures 2 a and b). Principal components after the second relate to nonspecific variations, which can be assumed to be caused by noise and sampling variations.
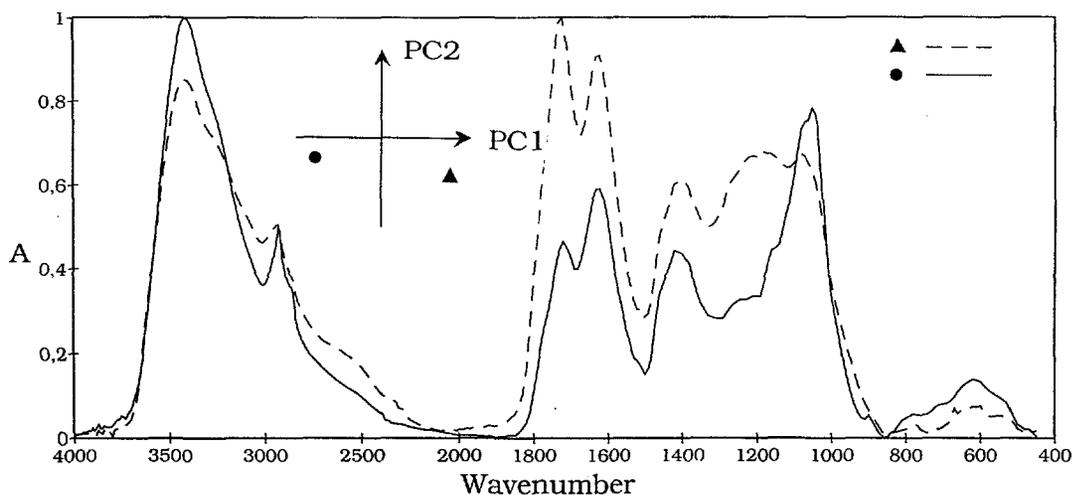

Fig. 2 a. Spectral variation along the first principal component.
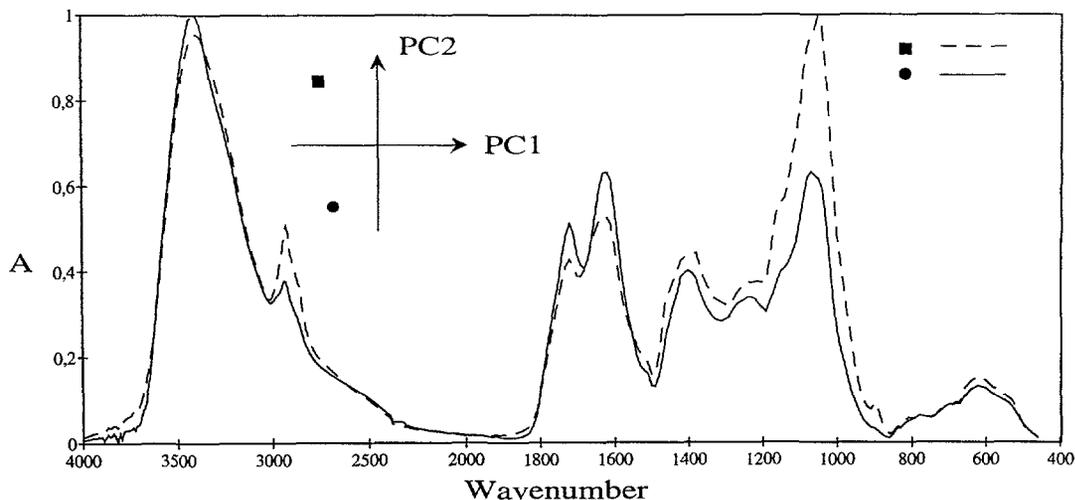

Fig. 2 b. Spectral variation along the second principal component.

The first PC was most important. Information was extracted mainly at carbonyl group range of the spectra. Although the first PC have a strong carbonyl character, it might also have a biological aspect. This part of work is still under research. The second PC has more clearly chemical character, classification is done on infrared absorption of one or several compounds. These compounds have a strong carbohydrate character.

Conclusions

These very preliminary results show that it is possible to characterize peat water samples with FTIR spectroscopy.

Specific information from complicated data can be extract effectively by using statistical, multivariate methods.

There was enough specific information to classify peat water samples according the sampling depth.

The first principal component was most important and classification was done mainly by using carbonyl group range of spectra.

Our other preliminary results show that these methods can be used for more exact determinations of organic components in peat.

We have used many different methods for analyzing peat organic fractions. The data from FTIR-analyses will be compared to specific properties, e.g. the organic compounds found in peat, fractions of organic components (measured according the molecular weight) and microbial biomass carbon.

These variables will be added to the PCA model and the results published later.

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BERI: Bog Ecosystem Research Initiative - objectives, hypotheses and research methods

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Introduction

The BERI project is funded by the European Commission and will start January first of 1996. Nine partners from five EU countries are participating in this project contributing each their unique knowledge and experience to this bog ecosystem research initiative (Table 1). Associate partners from Eastern Europe and North America are also involved in the BERI project. The coordinator is Nico van Breemen.

The concentration of CO_2 in the atmosphere is increasing with 2.8 Gt C y^{-1}. Anthropogenic CO_2, i.e. mostly due to burning of fossil fuel, is released at a rate of 6 Gt C y^{-1}. The difference is mainly due to feedback mechanisms of terrestrial ecosystems (Schimel et al., 1995). The total annual terrestrial CO_2 turnover is estimated at 60 Gt C y^{-1}, which is about 20 times the annual net increase. Therefore it is likely that small changes in net primary production or in decomposition rate of soil organic carbon (SOC) will significantly influence the net increase of atmospheric CO_2. Feedback mechanisms to pools of SOC are expected to play a major role.

Northern peatlands contain 20 to 30% of the world's SOC and, if growing, they constitute a nearly continuous sink of atmospheric CO_2. However, northern peatlands are also a major source of CH_4 with an estimated emission of 24 to 39 Tg y^{-1} (Matthews and Fung, 1987; Bartlett and Harris, 1993).

Oligotrophic peat bogs, on which BERI focusses, have a characteristic microtopography of hummocks, lawns and hollows, each with a specific mixture of *Sphagnum* species and vascular plants (Malmer et al., 1994). Through their anatomical morphological, physiological and organo-chemical properties, *Sphagnum* species, in particular those from hummocks, create water logged, nutrient poor, acidic conditions, thereby outcompeting most vascular plants (Van Breemen, 1995). Plants of hummocks, lawns and hollows differ in their decomposability and therefore differ both in carbon sequestration potential and in the hydraulic properties of the resulting peat. The poor decomposability of *Sphagnum* litter from nutrient poor hummocks causes a large fraction of photosynthetically fixed C to be turned into relatively dense and highly water impermeable peat. By contrast, a larger

fraction of photosynthetically fixed C is decomposed in peat of nutrient richer lawns and hollows, resulting in a somewhat higher hydraulic conductivity of the peat litter in laws and hollows. So, peat formation, and therefore C sequestration, increases if hummocks grow at the expense of hollows and lawns. By contrast, emission of CH_4 is highest from the relatively wet and nutrient rich hollows (Clymo and Pearce, 1995). Increased CO_2 concentrations and N deposition probably affects plants in hummocks differently than plants in hollows (Hogg et al., 1994), indicating a number of potentially important feedbacks between these environmental changes and C sequestration and emission of CH_4 from bogs.

Hypotheses

Even in a nutrient-poor bog environment, elevated CO_2 will stimulate plant growth. We expect growth of *Sphagnum* to be stimulated more than that of vascular plants (a; letters refer to figure 1) because increased growth lowers N availability, which affects vascular plants more than *Sphagnum*. For the same reason, *Sphagnum* of hummocks will expand at the cost of *Sphagnum* at hollows. Increased peat growth (b), positively feeds back to Sphagnum itself (c), but further depresses vascular plants (d). This sequence of events forms a strong negative feed back of increased sequestration of CO_2-C (e). Increased CO_2 would increase CH_4 emissions in the short run where peat expands over non-wetland soils. However, it would decrease CH_4 emissions in the long run if it stimulates formation of hummocks in existing peats.
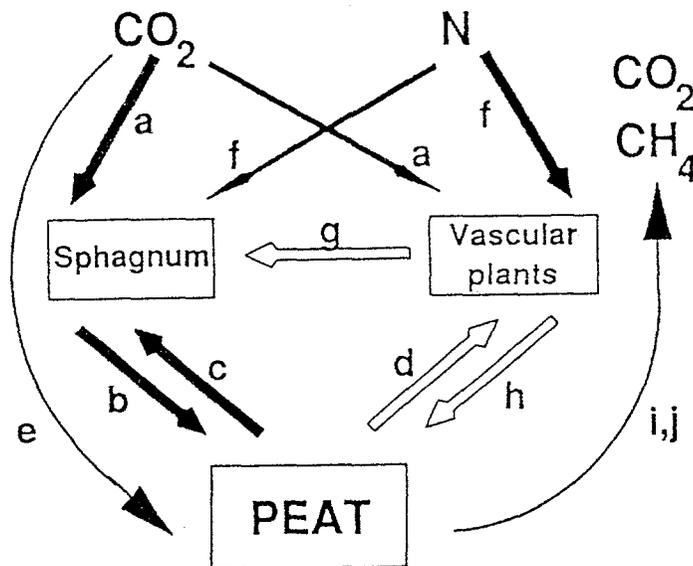

Figure 1. Hypothetical positive (black arrows) and negative (open arrows) effects of elevated atmospheric CO_2 and N deposition on bog processes important for sequestration of CO_2-C and emission of CH_4. Litter of peat moss is more recalcitrant than that of vascular plants, and forms the bulk of bog peat. Letters are explained in the text.

Elevated N-deposition, by contrast, favors (f) vascular plants and *Sphagnum* species of hollows over those of hummocks. This depresses peat growth for two reasons: 1) *Sphagnum* is outshaded by vascular plants (g), and 2) the decomposability of litter increases (h) because of an increasing contribution of vascular plants and of *Sphagnum* species of hollows. So, elevated N deposition depresses C sequestration by peat bogs and may even cause peats to turn into net sources of CO_2 (i). However, as long as a water-logged anoxic subsoil is present at shallow depth, elevated N will increase CH_4 emissions, because of increased substrate availability for methanogenic bacteria and decreased methane oxidation (j).

Objectives

The primary aim of the BERI project is to study, at five climatically different sites across Europe, the effects of elevated CO_2 and N deposition on the net exchange of CO_2 and CH_4 between bogs and the atmosphere. Secondly, to study the effects of elevated CO_2 and N deposition on the plant biodiversity of bog communities.

These objectives further include the quantification of the impact of elevated CO_2 and N, specifically: 1) on the population dynamics of different species of peatmoss and selected vascular plants; 2) on the growth of dominant species in the bog ecosystem; 3) on the organochemical and structural properties of dominant mire plants; 4) on the decomposability of litter from mire plants; and 5) on the emission of CH_4 from the bogs.

Quantification will facilitate the objective to develop process-oriented models that describe responses of botanical composition, C-sequestration and CH_4 emission in bogs in Europe in response to possible scenarios of future changes in atmospheric CO_2, N deposition and climate.

Plan of methods

Field sites

The following five *Sphagnum* bog research sites are considered for BERI:
• Sweden, Roshult mire (56°41' N, 13°13' E). Partners 2 & 3 (See Table 1).
• Finland, Salmisuo in Ilomantsi (62°47' N, 30°56' E). Partners 5 & 6.
• United Kingdom, The Migneint, North Wales (53°N, 3°50' W). Partner 4.
• The Netherlands, Poort Twee, Dwingelo (52°50' N, 6°40' E). Partner 7.
• Switzerland, Les Chaux-des-Breuleux (47°N, 7°E). Partner 9.

The field research will be done in ombrotrophic lawn communities (*S. magellanicum*, *S.balticum*, or *S. papillosum*) and associated vascular plants, (if possible including at least one graminoid, e.g. *Eriophorum vaginatum*, and one ericoid species). A lawn community will be studied because under the influence of changed environment (e.g. in terms of atmospheric CO_2 concentration or N input) it will easily shift towards either a hollow (with low CO_2 sequestration and high CH_4 emission) or a hummock (with high CO_2 sequestration and low CH_4 emission), as indicated by shifts in species composition.

Field manipulation experiments

During two growing seasons, at each site a field experiment will be done that involves four treatments, replicated five times:
• Elevated atmospheric CO_2 (target concentration 560 ppm) in Mini-FACE rings (see next paragraph) that vent air with CO_2 added.
• Ambient CO_2 controls: Mini-FACE rings that vent ambient air.
• Small (2-3 m^2) plots receiving extra Nitrogen, to be added weekly during the growing season as dissolved NH_4NO_3 in a fine spray (5 g N m^{-2} yr^{-1}).
• Control N-plots receiving a similar volume of distilled water.
The treatments will be randomly assigned to five areas on the bog surface.

Mini-FACE technology

Each Mini-FACE ring consists of a circular (1.0 m ⌀) arrangement of about 70 venting pipes that blow CO_2-enriched air. An IR gas analyzer and PC maintain a target CO_2 concentration of 560 ppm after mixing with ambient air inside the ring. The system was developed by Dr. Franco Miglietta, Dr. Marco Bindi and Mr. Mario Lanini and will be supplied by the Foundation for Applied Meteorology, Italy.

Field vegetation studies

In each experimental plot (Mini-FACE ring, N-plot, control) the reactions of the vegetation to the treatments will be studies as follows:
• At the beginning and end of the growing season, (1) abundance of each species is measured by estimation of cover and (2) relative cover is measured with a frame of pins ("point quadrants") combined with a photograph of the plot from a fixed position above the plot.
• In Sweden, Finland and Switserland every six weeks *Sphagnum* dynamics will be studied in more detail with the aid of digitizing techniques.
• Primary production of mosses will be estimated by a modified "cranked-wire" method. In addition, in Sweden, Switzerland, and Finland C^{14} labelling techniques will be used.
• Material collected at the end of each season will be analyzed for nutrient composition, and used for measurements of decay rate as CO_2-emission by respirometry in the laboratory.
• Growth of vascular plants is monitored by measuring shoot length and stem thickness and counting leaf numbers.
• After two years the total moss and vascular plant biomass will be harvested and analyzed.
• In The Netherlands N^{15}-pulse labelling will be applied to 50*50 cm plots (4 plots in 5 blocks). After 1, 6, 12 and 24 months *Sphagnum* and vascular plants will be analyzed to follow the distribution of N^{15} over different ecosystem and plant compartments.
• In Switzerland, partner #9 will do a factorial experiment with two water table depths combined with the usual CO_2 and N treatments, focusing on the moss layer.

Studies on litter composition, decay and gas fluxes

- In the UK the effects of enhanced nitrogen deposition on decomposition will be studied utilizing plots that received NH_4NO_3 over the last 8 years to assess the long-term effects of nitrogen deposition on microbial processes. Other detailed decomposition experiments will be performed as well.
- At all sites, except Switzerland, seasonal and annual CH_4 fluxes will be estimated from manual measurements done 2-4 times per month during the snow-free season.
- In Finland net exchange of CO_2 between plots and atmosphere will be measured by putting a transparent thermostated chamber on the collars in the FACE and control plots.
- Organo-chemical and structural changes due to treatments will be studied by partner #8, using material from *Sphagnum* species, *Eriophorum* species, and *Ericaceous* dwarf shrubs.

Greenhouse experiments

- In Finland, partners 5 and 6 will perform additional growth chamber experiments at ambient and elevated CO_2 in the laboratory.
- In Sweden, partner 3 will carry out additional greenhouse experiments on the effect on nitrogen addition on a selection of *Sphagnum* species.
- Partner 7 will investigate the effects of CO_2 (560 and 350 ppm) and N (0 and 5 g N m^{-2} yr^{-1}) on the competition between *Sphagnum* and vascular plants.

Simulation modeling

Partner 7 will adapt existing models on competition and N- and C-flows in heathlands (NUCOM, HEATHSOIL) to peatbog ecosystems.

Partner 1 will integrate the processes discussed in this proposal in one overall model aimed at testing the proposed hypotheses. The model will consist of several modules and each of these modules will comprise of one or more detailed submodules. The main modules are:
1. Climate and hydrology
2. Photosynthesis and net primary production
3. Competition between plant species
4. Chemical composition of plants
5. Decomposition, methane formation and peat formation

As much as possible, existing models will be adapted, so as to avoid reinventing the wheel for some modules. Individual modules will be tested as much as possible on existing data sets until the BERI data set is available. The use of existing data sets may also improve the robustness of the individual modules. Much emphasis will be on error propagation and sensitivity analysis. Monte Carlo techniques will be used to provide probabilities with the model outcome.

Table 1. BERI Partners

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Estimation of microbial biomass carbon by a fumigation-extraction method: Determination of k_{EC} factor in Finnish peats

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Introduction

Several biochemical methods are now available to estimate the soil microbial biomass carbon. The fumigation extraction (FE) method appears to be quite reliable also in acid soils (Vance et al. 1987; Sparling and West 1988).

The FE method is calibrated in vitro with ^{14}C-labelled glucose (Sparling and West 1988; Sparling et al. 1990; Bremer and Kessel 1990). The conversion factor obtained thereby is marked k_{EC}. It describes the conversion of the extractable organic carbon flush into microbial biomass carbon. This factor is used in the calculation of the total microbial biomass carbon in soil.

The values of k_{EC} factor vary widely between soils; e.g. 0.1-0.9 (Sparling and Zhu 1993), and are strongly influenced by soil water content (Sparling et al. 1990). Therefore, WHC 60 % is recommended to ensure maximum extractable carbon values (Ross 1989). The k_{EC} values of individual soils also vary according to the season of sampling, but the variations of the k_{EC} values among different soils are markedly greater (Ross 1990). More accurate estimates of microbial biomass carbon can be obtained by using the soil-specific factors rather than the average values (Sparling and Zhu 1993).

The k_{EC} factors have been estimated for some New Zealand peats: 0.33 and 0.49, sampling depth 0-20 cm (Sparling et al. 1990) and 0.29, sampling depth 0-75 cm (Sparling et al. 1988).

No k_{EC} factors have been determined for Finnish peats with ^{14}C-labelled glucose method.

Our objectives were:

1. to determine the specific k_{EC} factors for different peats with the help of ^{14}C-labelled glucose,
2. to determine the turnover of ^{14}C-labelled glucose in different peats,
3. to determine microbial biomass in different peats.

Sites

The material for this experiment was collected on June 16th, 1994, from four sample plots on the Lakkasuo mire complex in the central Finland (61°48'N, 24°19'E). Two of the sample plots are located in the virgin part and two in the drained (1961) part of the mire. Site types were mesotrophic flark fen and tall sedge fen. Sampling depths were 0 - 10 cm and 10 - 20 cm.

Experiment

^{14}C-labelled glucose was added to different peats and incubated at 25 °C for 5 days. Measurements were made for $^{14}CO_2$ respired from labelled glucose during incubation. The ^{14}C in peat after incubation was measured. Also the ^{14}C extracted before and after fumigation were analyzed. The k_{EC} factor was calculated on the basis of the above measurements. (k_{EC} = extractable organic carbon flush (i.e. extracted after fumigation - extracted before fumigation) / microbial biomass carbon (i.e. ^{14}C added - respired as $^{14}CO_2$ - ^{14}C extracted before fumigation))

Microbial biomass carbon in the peat was measured with fumigation extraction method (Martikainen and Palojärvi 1990) before and after the experiment.

Results and discussion

The turnover of ^{14}C in peats is shown in Figure 1. The amount of respired $^{14}CO_2$ was lowest and the amount of ^{14}C in peat was highest in virgin peat (0-10 cm) than in other peats. Also the amount of ^{14}C in extracted carbon flush is highest in this peat (Fig. 1). The respired $^{14}CO_2$ values were lower in virgin peat (0-10 cm) than in the drained peat (0-10 cm). (Fig. 2)

The values of the k_{EC} factors were low and varied from 0.08 to 0.18. The values of the k_{EC} factor were highest in order virgin peat (0-10 cm) > drained peat (0-10 cm) > virgin peat (10-20 cm). (Fig. 3)

After incubation microbial biomass carbon was about 3.5 times higher than before incubation in all peats, except for drained peat of the mesotrophic flark fen where it was even 8.4 times higher. Microbial biomass carbon before incubation in the peat of mesotrophic flark fen were lower than in the peat of tall sedge fen. After incubation microbial biomass carbon was higher in the virgin and drained peat (0-10 cm) than in the virgin peat (10-20 cm). (Fig.4)

Conclusion

1. More than half of the ^{14}C recovered in peat.
2. The $^{14}CO_2$ respiration was higher in drained peat than in virgin peat.
3. The values of the k_{EC} factors were in order virgin 0-10 cm > drained 0-10 cm > virgin 10-20 cm.
4. This experiment with glucose showed that there is a lack of available carbon for microbes in peat.
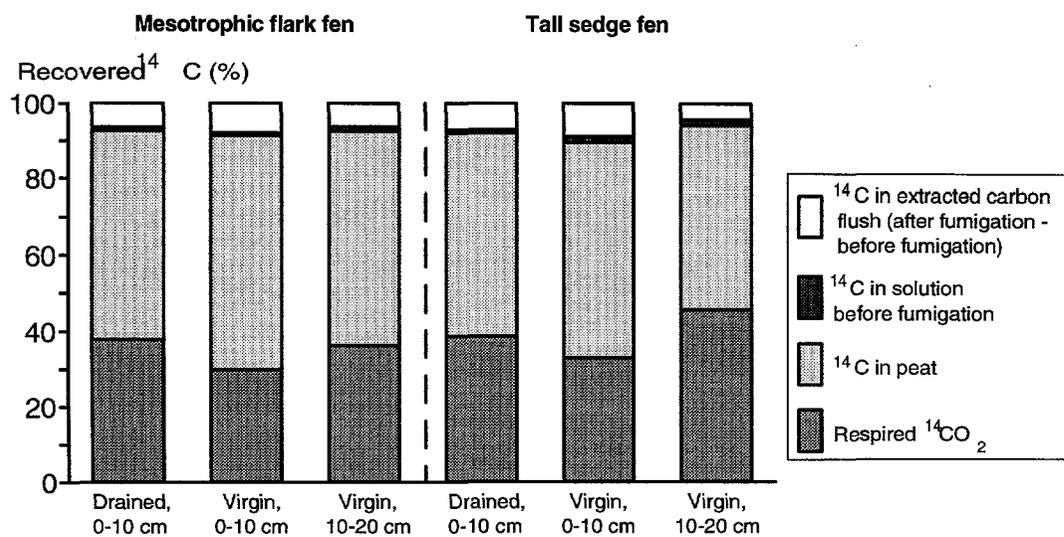
Fig. 1. Turnover of ^{14}C.
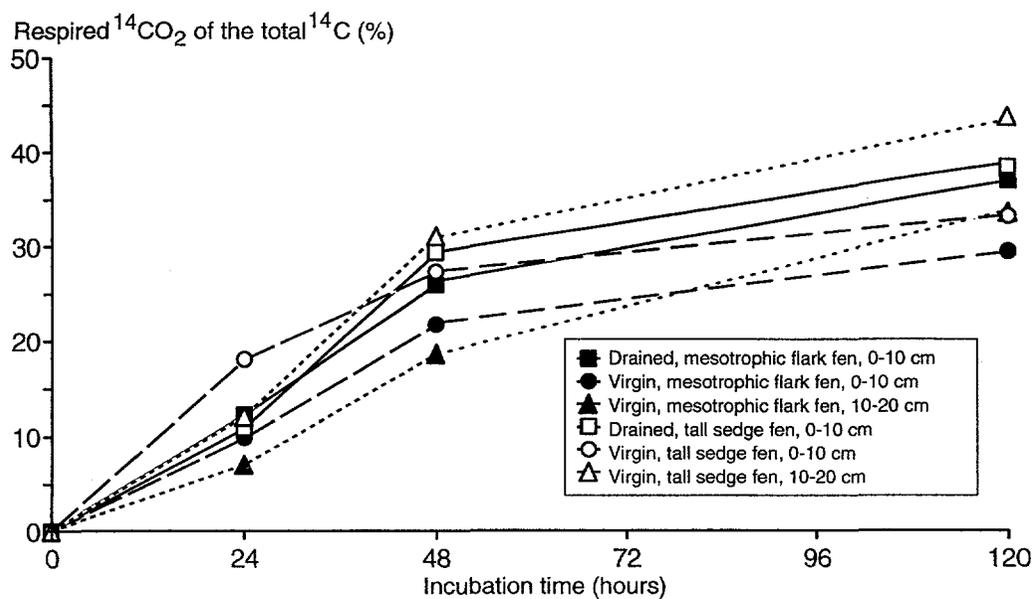
Fig. 2. Respiration of $^{14}CO_2$ in different peats.
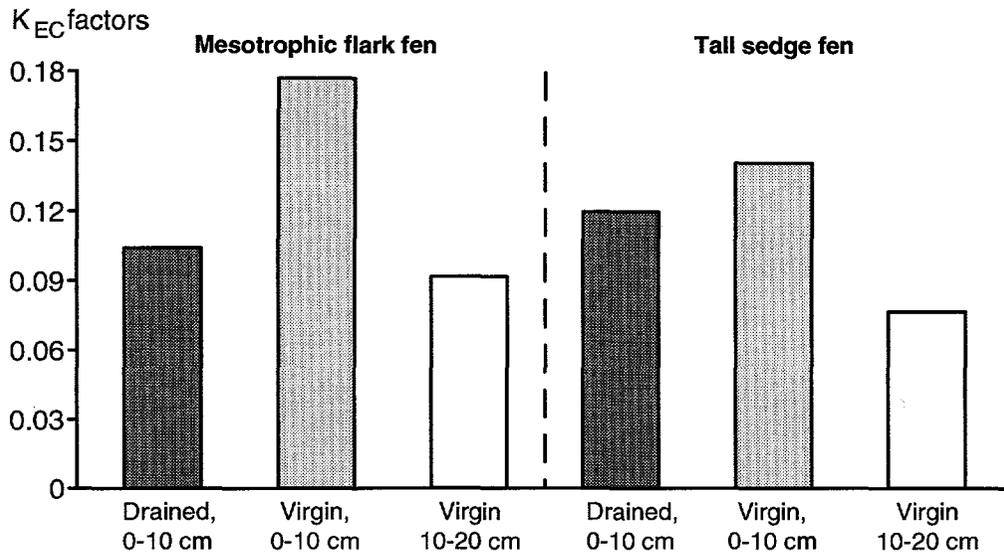
Fig. 3. K_{EC} factors of the estimation of microbial biomass carbon from extractable organic carbon flush in different peats.
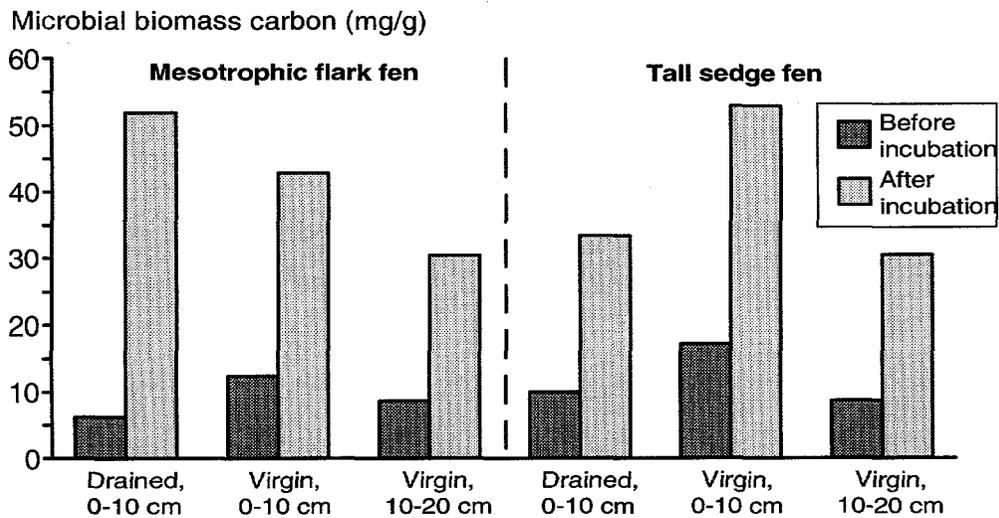

Fig. 4. Microbial biomass carbon in different peats.

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