PEATLANDS

AND

CLIMATE CHANGE

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EDITED BY MARIA STRACK

International Peat Society | IMTG MTO

Cover

Main photo: A flark fen in the northern aapa mire Luovuoma. Flarks and strings are the most essential elements of the mire centre. Photo Markku Mäkilä.

Small photos:

Upper row from left

- Tropical peatland fire. Photo by Marcel Silvius
- Typical flat palsa landscape. Photo by Markku Mäkilä
- Ditching of peatland for forestry. Photo by Juhani Päivänen.

Second row from left

- Peatland used for agriculture. Photo by Hannu Salo
- Peat extraction. Photo by Association of Finnish Peat Industries
- Smoke plume from peatland fire. Photo by M. Turetsky.

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ACKNOWLEDGMENT

The human impact on global climate and the role of peatlands in this process has been widely studied and debated in media, but also within a scientific audience and peatland experts during recent years. Controversial opinions have been put and different parties and experts have emphasised their points of view with the latest research data, historical evidence and statistics.

It seems that there is lack of fundamental cooperation on an international level to coordinate research efforts. There is a need to find solutions for the management of peatlands in the best way from a climate, but at the same, from a human needs point of view.

To deal with this demand, the International Peat Society IPS, a non-profit organisation of about 1,400 individual and corporate stakeholders in peat and peatlands from all over the world, established a joint IPS Working Group on Peatlands and Climate Change in the end of the year 2005.

Specialists from all fields of interest covered by the IPS were asked to join the Working Group to provide knowledge from their special areas of expertise.

The Working Group's task was to compile information into a summary of available knowledge to help the IPS and other actors to understand the role of peatlands and peat within the current context of global climate change.

The work took two years and finally the book "Peatlands and Climate Change" was launched in the IPS International Peat Congress 2008 in Tullamore, Ireland.

Without voluntary efforts of the writers of the eight chapters this book could not have been completed. The IPS warmly thanks all experts who are:

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Once better informed, actors can promote further actions into the topic of peatlands in relation to global climate change. The IPS hopes the book “Peatlands and Climate Change” is one step ahead in the road of wise use of peatlands and peat.

Jyväskylä, Finland, the First of May 2008

The International Peat Society IPS

Jaakko Silpola
Secretary General

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EXECUTIVE SUMMARY FOR POLICYMAKERS "PEATLANDS IN GLOBAL CHANGE"

Extent and importance

1. Peatlands cover an estimated area of 400 million ha, equivalent to 3% of the Earth's land surface. Most (c. 350 million ha) are in the northern hemisphere, covering large areas in North America, Russia and Europe. Tropical peatlands occur in mainland East Asia, Southeast Asia, the Caribbean and Central America, South America and southern Africa where the current estimate of undisturbed peatland is 30-45 million ha or 10-12% of the global peatland resource.

2. Peatlands represent globally significant stores of soil C that have been accumulating for millennia and currently, peatlands globally represent a major store of soil carbon, sink for carbon dioxide and source of atmospheric methane. In general, nitrous oxide (N_2O) emissions are low from natural peatlands but there is evidence that those used for agriculture are releasing significant amounts of this potent greenhouse gas. Losses of peatland C from storage result from changes in the balance between net exchange of CO_2, emission of CH_4, and hydrological losses of carbon (e.g. dissolved organic and inorganic C and particulate organic C). The greenhouse gas (GHG) balance of a peatland depends on relative rates of net CO_2 uptake or efflux and CH_4 and N_2O efflux.

3. In terms of GHG management, the maintenance of large stores of C in undisturbed peatlands should be a priority.

4. Temporal studies of peatlands reveal that they may act as CO_2 sinks in some years and sources in others, depending on climate. Emissions of CH_4 and N_2O are similarly variable in space and time.

5. When considering the role of peatlands in atmospheric GHG balances, it is important to consider that they have taken up and released GHGs continuously since their formation and thus their influence must be modelled over time. When this is considered, the effect of sequestering CO_2 in peat outweighs CH_4 emissions.

6. Contemporary GHG exchange in peatlands exhibits great spatial variability related to regional and local differences in ecology, hydrology, and climate and the impact of climate change is likely to be large. Some peatlands will emit more CO_2 to the atmosphere and change from net C sinks to become sources; other peatlands may exhibit increased CO_2 sequestration owing to elevated water tables and / or increased primary production as a result of changing vegetation.

7. In some parts of the world the peat C store is being reduced because of fire. Major increases in the area of peatland burned have been documented in recent decades and this may continue in the future if peatlands dry out as a result of climate change and anthropogenic activities. Fire will continue to play an important role in the fate of global peatland C stocks.

8. Climate change may threaten C stocks in unmanaged peatlands because of drought leading to peat oxidation, permafrost melting and fire. Owing to the variability in environmental conditions and GHG exchange across peatlands, predicting the overall response is not simple. Research aimed at improving peatland inventories and enhancing our understanding of the links between climate, hydrology, ecology, permafrost degradation, fire regimes and GHG balances will improve our knowledge of the state of current peat resources and predict the fate of this important store of carbon.

Impacts of peatland utilization

9. Agriculture, forestry and peat extraction for fuel and horticultural use are the major causes of peatland disturbance. As these land-use changes require alteration of peatland hydrology, peat oxidation results and the greenhouse gas balance of the peatland is altered.

10. About 14 – 20 % of peatlands in the world are currently used for agriculture and the great majority of these are used as meadows and pastures. For agricultural use, fens and raised bogs have to be drained in order to regulate the air and water conditions in the soil to meet the requirements of cultivated or pasture plants. In many European countries, GHG emissions from agricultural peatlands dominate national emissions of GHGs from peat sources.

11. The loss of water from the upper peat by drainage, followed by oxidation, leads to compaction and subsidence of the surface. Drainage of peat increases the emissions of CO_2 and N_2O but decreases the emission of CH_4. Emission rates depend on peat temperature, groundwater level and moisture content. Appropriate water management is important in order to minimise GHG emissions from agriculture on peatlands. Increasing the water table decreases emissions of CO_2 (by up to 20%) and N_2O, but may increase emissions of CH_4.

12. The utilization of peatlands for forestry is concentrated in Nordic countries (Norway, Sweden, and Finland) and Russia, where over 10 million ha of peatlands have been drained for this purpose. The climatic impacts of the use of peatlands for forestry are smaller than for agriculture because oxidation of organic matter in the surface peat is much less. Biomass and primary production increase during stand development, contributing also to soil carbon store through increased litter production. Simultaneously, however, the organic matter decomposition rate increases because of increased aeration and this leads to increased CO_2 emissions from soil.

13. The combination of these changed fluxes shifts the C balance of the ecosystem with some forested peatlands becoming sources of CO_2 to the atmosphere, while others remain or become even larger C sinks. These differences are related to climatic condition, site type, intensity of drainage and management regime.

14. Finland, Ireland, Russian Federation, Belarus and Sweden account for almost 90% of the world's production and consumption of energy peat. Peat is also used in horticulture, as a growing medium, but the volume used annually is only about half that of fuel peat. Germany and Canada account for over half of horticultural peat extraction.

15. The main greenhouse gas released as a result of peat fuel extraction and

burning is CO_2 but CH_4 and N_2O are also emitted. In the process of peat extraction, the GHG sink function of the peatland is lost. Emissions also arise in the preparation of the surface for cutting (removing vegetation and ditching), extraction of peat and its storage and transportation, combustion and after-treatment of the cutaway area. Combustion accounts for more than 90% of the greenhouse gas emissions.

16. As with the extraction of energy peat, horticultural peat extraction requires drainage of the peatland to accommodate machinery and facilitate drying of peat prior to extraction. This facilitates peat oxidation, increases CO_2 emissions and reduces efflux of CH_4. Although horticultural peat is not consumed instantaneously, it will decompose over time following extraction and result in CO_2 emissions.

Tropical peatlands

17 Carbon storage in SE Asian peatlands is in the order of 58 Gt. In the late 1980s 3.7 million hectares of Indonesian peat swamp forest were taken for agriculture, leading to an 18% decrease in peat swamp forest area with a consequent reduction in the C-fixation capacity of 5-9 Mt yr^{-1}. The development of palm oil and timber plantations, which require intensive drainage and cause the highest CO_2 emissions of all land uses, are major drivers of peatland deforestation and increases in CO_2 emissions.

18. Present and future emissions from natural and drained peatlands in Indonesia have been quantified recently using data on peat extent and depth, present and projected land uses and water management practices, decomposition rates and fire emissions.

19. Current CO_2 emissions (2005) caused by peat decomposition in drained peatlands are estimated to be over 600 million t yr^{-1}, which will increase in coming decades, and will continue well beyond the 21st century, unless land management practices and peatland development plans are changed. In addition, between 1997 and 2006 an estimated average of 1400 Mt yr^{-1} of CO_2 emissions was caused by fires associated with peatland drainage and degradation. The total current CO_2 emissions from tropical peatland of approximately 2000 Mt yr^{-1} equal almost 8% of global emissions from fossil fuel burning. Emissions are likely to increase every year for the first decades after 2000.

20. Overall, methane emissions from tropical peatland are very low irrespective of whether it is natural peat swamp forest or drained and degraded or used for agriculture. N_2O emissions from natural tropical peatlands are low but evidence is emerging that suggests that these increase following land use change and fire.

Restoration of peatlands

21. Peatland restoration is growing in importance in Europe and North America and is likely to remain important over the next half century. It is also gaining recognition in tropical peatland areas where some of the greatest challenges exist following inappropriate and unsuccessful development projects. While peatland restoration is primarily designed for global biodiversity protection, it can also play an important role in reducing GHG emissions.

22. In general, rewetting of peatlands reduces CO_2 emissions by creating anoxic, reducing conditions, although

it may lead to increase in CH_4 efflux at least for a time. Rewetting also inhibits nitrification, resulting in reduced emission of N_2O. Some restored boreal bogs have become net C sinks again following successful re-establishment of Sphagnum-dominated vegetation.

Peatlands and international climate change conventions

23. Peat-based GHG emissions reported under the United Nations Framework Convention on Climate Change (UNFCCC) are divided between several sectors: Energy, Agriculture and Land Use, Land-Use Change and Forestry (LULUCF). Only human-induced GHG emissions are included in reporting, therefore, emissions from undisturbed/virgin peatlands are not included.

24. While industrialized nations listed in Annex I of the UNFCCC submit annual GHG inventories and have emission limitation targets under the Kyoto Protocol, the heterogeneous groups of developing nations that are non-Annex I Parties are only required to provide information about GHG emissions in national communications. However, peatland fires and wetland degradation in many non-Annex I countries contribute significantly to global GHG emissions. The Clean Development Mechanism (CDM) may provide a means for mitigation of these problems.

25. Methodologies and guidance for estimating peat-based emissions in the good practice guidelines for LULUCF and the 2006 IPCC Guidelines are relatively few. There is a deficiency of data that can be applied to country, region or site-specific conditions with data availability varying for different climate regions and countries, while global scale knowledge of peat-derived emissions remains limited. Development of scientifically sound emission factors for peat soils is complicated and resource demanding owing to the variation between sites.

Mitigation of greenhouse gas emissions

26. Since peatland management generally involves lowering the water table, GHG emissions result from decomposition of stored organic matter and, particularly as has been observed in tropical peatlands, an increase in fire susceptibility. The most efficient method for reducing GHG emissions from peatland is to prevent future land use change although this is not always economically, socially or politically possible. If this is the case, land management strategies should focus on preventing degradation of additional peatlands where possible, and adjusting management practices on developed peatlands in order to reduce GHG impacts.

27. Using peat from peatlands that are large greenhouse gas sources, climatic impact of peat utilisation chain can be significantly reduced. Examples of such peat resources are cultivated peatlands and forestry drained peatlands.

28. It is essential that future land use of peatland incorporates the principles and practices of wise use in order to promote sustainable management, especially with respect to hydrology, water and carbon. Inevitably, however, every type of human intervention on peatland leads to impairment or even loss of natural resource functions (ecology, hydrology, biodiversity, carbon storage). Effective peatland management also requires engagement between scientists, policy makers and stakeholders.

SUMMARY FOR POLICYMAKERS
"WISE USE OF PEATLANDS AND CLIMATE CHANGE"

I. Global extent of peat and peatlands

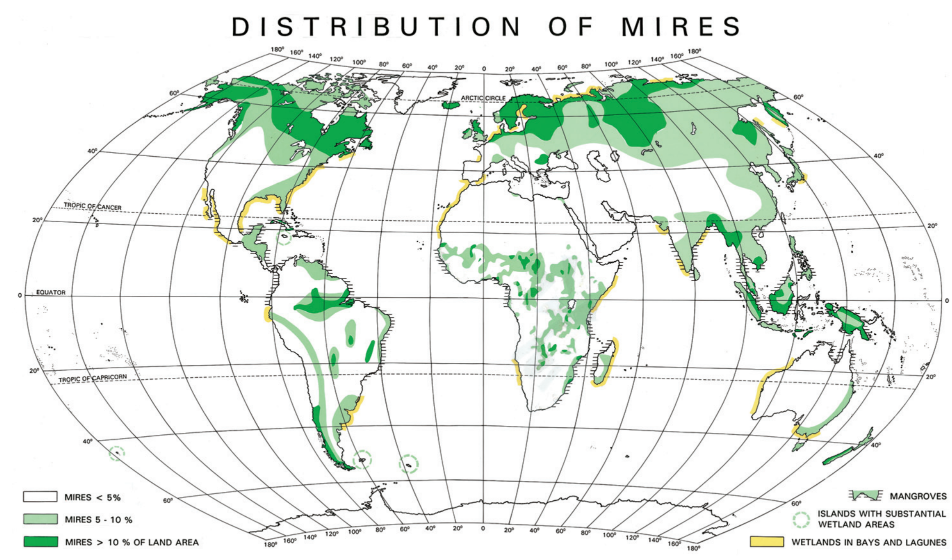
Peatlands cover an estimated area ca. 400 million ha equivalent to 3% of the Earth's land surface (Figure 0.1). Most (about 350 million ha) are in the northern hemisphere, covering large areas in North America, Russia and Europe. Tropical peatlands occur in mainland East Asia, Southeast Asia, the Caribbean and Central America, South America and southern Africa where the current estimate of undisturbed peatland is in the range 30-45 million ha accounting for 10-12% of the global peatland resource.

II. Global peat carbon store

Currently, peatlands globally represent a major store of soil carbon, sink for carbon dioxide and source of atmospheric methane. In general, nitrous oxide (N_2O) emissions are low from natural peatlands but there is evidence that those used for agriculture and receiving nitrogen fertilisers, are releasing significant amounts of this very potent greenhouse gas. The change in peatland C storage results from changes in the balance between net exchange of CO_2, emission of CH_4, and hydrological losses of carbon (e.g. dissolved organic and inorganic C and particulate organic C).

Northern peatlands store around 450 billion metric tons (Bt = 10^{15} g = Pg = Gt) carbon, which is equivalent to approximately one third of global soil C stocks and 75% of the pre-industrial mass of C stored in the atmosphere. In tropical peatlands both the vegetation and underlying peat constitute a large and highly concentrated carbon

Figure 0–1 Distribution of mires
Source: International Peat Society, Available www.peatsociety.fi.



pool amounting to about 60 Bt. The current annual carbon storage rate in the world's peatlands is approximately 100 million tonnes (Mt), which is equivalent to approximately 370 Mt CO_2 yr^{-1} but this has varied greatly throughout millennia depending mainly on climate and sea level. Conversely, however, pristine peatlands (mires), especially those in boreal and temperate zones, emit methane (ca. 20 Mt yr^{-1}), which is a potent greenhouse gas.

In terms of GHG management, the maintenance of large stores of C in undisturbed peatlands should be a priority.

III. Carbon accumulation and greenhouse gas exchange in undisturbed peatlands

Regardless of variability, when considering the role of peatlands in atmospheric GHG balances, it is important to consider that they have taken up and released GHGs continuously since their formation and thus their influence must be modelled over time. When this is considered, the effect of sequestering CO_2 in peat outweighs CH_4 emissions. Thus, peatlands have been net GHG sinks for thousands of years.

Most contemporary peatlands began accumulating peat following the last glacial period and have continued to do so throughout the Holocene, approximately the last 10,000 years. Some peatlands in tropical Southeast Asia, however, started to form towards the end of the Pleistocene more than 20,000 years ago. Carbon accumulation rates have varied over this period in relation to stage of peatland development and climate; average long-term C accumulation rates for northern bogs are 20-30 g m^{-2} yr^{-1}, while a tropical peat core has yielded a long-term average C accumulation rate of over 50 g m^{-2} yr^{-1}.

Contemporary C exchange in peatlands exhibits great spatial variability related to regional and local difference in ecology, hydrology, and climate. Studies of net ecosystem exchange of CO_2 in northern peatlands provide values ranging from uptake of over 220 g CO_2 m^{-2} yr^{-1} to release of 310 g CO_2 m^{-2} yr^{-1}. Accumulation rates in tropical peatlands are also variable, yet they probably accumulate more carbon per unit area than northern peatlands. Present net C uptake may be in the range of 500 g C m^{-2} yr^{-1} (~1800 g CO_2 m^{-2} yr^{-1}). Temporal studies of peatlands reveal that they may act as CO_2 sinks in some years and sources in others, depending on climate. Emissions of CH_4 and N_2O are similarly variable in space and time.

IV. Climate change impact on peatland carbon stocks and greenhouse gas exchange

Owing to the spatial diversity of peatlands, the variability of the response of peatland GHG exchange to climate change is likely to be large. Some peatlands will emit more CO_2 to the atmosphere and change from net C sinks to become sources; other peatlands may exhibit increased CO_2 sequestration owing to elevated water tables. In terms of the stability of peatland C stocks, non-permafrost peatlands will be most at risk because not only are these likely to release CO_2 as a result of peat oxidation under dry conditions, but they will also face increased risk of fire.

Climate variability throughout the Holocene, especially in boreal and temperate zones, has had a major effect on C accumulation rates in peatlands. Increased peat accumulation reflects periods of a more positive precipitation-evaporation balance, which is supported by data on lake level fluctuations from these regions. In contrast, drier periods correspond to lower C accumulation rates. Evidence from the Holocene suggests that climate warming results in permafrost melting and release of GHGs from northern peatlands but this is compensated to some degree by extension of forests northwards. Future climate change may also result in accelerated rates of permafrost degradation, displacing tree communities, and creating water saturated open fens. C accumulation at these sites will be higher than neighbouring permafrost peatlands but CH_4 emissions will be enhanced owing to increased vegetation productivity and waterlogged conditions.

In contrast, climate change scenarios predict that some peatlands will experience lowered water tables, leading to increased dryness and unsaturated, oxic conditions at their surface, resulting in aerobic decomposition (oxidation) and larger releases of CO_2. On the other hand, development of vegetation towards shrub-dominated communities may lead to higher primary production, compensating soil C losses. The larger unsaturated zone will lead to reduced CH_4 emissions and some dry bogs may become CH_4 sinks. Over time, peat subsidence combined with increased ecosystem productivity may keep some peatlands (e.g. fens, bog pools and hollows) wet, maintaining or enhancing C storage although CH_4 emissions may increase under these conditions.

In some parts of the world the peat C store is being reduced because of fire. Major increases in the area of peatland burned have been documented in recent decades and this may continue in the future if peatlands dry out as a result of climate change or anthropogenic activities. Fire will continue to play an important role in the fate of global peatland C stocks.

V. Land-use change impacts on peatland carbon stocks and greenhouse gas exchange

Agriculture, forestry and peat extraction for fuel and horticultural use are the major causes of peatland disturbance. As these types of land-use change require alteration of peatland hydrology, peat oxidation results and the greenhouse gas balance of the peatland is altered (Table 0.1).

V.1 Peatland utilised for agriculture

About 14 – 20 % of peatlands in the world are currently used for agriculture and the great majority of these are used as meadows and pastures. For agricultural use, fens and raised bogs have to be drained in order to regulate the air and water conditions in the soil to meet the requirements of cultivated or pasture plants. In many European countries, GHG emissions from agricultural peatlands dominate national emissions of GHGs from peat sources.

The loss of water from the upper peat by drainage, followed by oxidation, leads to compaction and subsidence of the surface. Drainage of peat increases the emissions of CO_2 and N_2O but decreases the emission of CH_4. The emission rates of the greenhouse gases depend on many factors including peat temperature, groundwater level and peat moisture content. For ploughed temperate fens (arable land), in the central and north part of Europe (Sweden), annual CO_2 mean emissions of 4100 g m^{-2} can be expected, but with a high range of variation. Temperate and boreal fens converted to grassland show mean CO_2 emissions of about 700 g m^{-2} yr^{-1} (Canada) and from 1500 to 1700 g m^{-2} yr^{-1} in Central Europe (Poland, Germany, The Netherlands and Sweden). Finnish studies show average CO_2 emissions of about 2200 g m^{-2} yr^{-1} for boreal fens under grass and barley. Drained peatlands are large sources of nitrous oxide

Table 0.1: *Impact of land-use change on peatland greenhouse gas balance*

	Approximate area (10^3 km^2)	Response of greenhouse gas flux to land-use change CO_2	CH_4	N_2O
Agriculture	300	**Net emission of CO_2** -decreased CO_2 uptake: removal of peatland vegetation, crop residues removed -increased CO_2 emission: lower water table, peat oxidation	**Reduced CH_4 efflux** -emission from ditches remains high	**Enhanced N_2O emission** -N mineralization at nutrient rich sites -fertilizer application
Forestry	150	**Often little change in net ecosystem balance** -increased CO_2 uptake by treestand -increased CO_2 emission from soils: peat oxidation, dependent on extent of drainage	**Reduced CH_4 efflux** -emission from ditches remains high	**Dependent on site type and fertilizer application** -increased N_2O flux on nutrient rich sites -increased N_2O flux if fertilizer applied
Peat extraction	<5 *)	**Net emission of CO_2** -loss of CO_2 via combustion (fuel) or decomposition (horticulture) -decreased CO_2 uptake: removal of peatland vegetation, crop residues removed -increased CO_2 emission: lower water table, peat oxidation	**Reduced CH_4 efflux** -emission from ditches remains high	**Little change** -some increase in N_2O efflux at nutrient rich sites

*) Area of peatland used for energy generation and production of plant growing media. Estimation of the IPS, based on the book "Wise use of Mires and Peatlands", H. Joosten and D. Clarke, 2002 (page 8 and 33) and "Global Peat Resources", edited by E. Lappalainen, 1996.

(N_2O) with fluxes varying between 0.2 and 5.6 g m^{-2} annually. The annual CH_4 fluxes of cultivated peat soils range from a very small sink to low emission.
The position of the water table is one of the most important factors influencing peat conditions and processes in organic soils. Consequently, precise water management for peatlands utilised for agriculture purposes is very important. Increasing the water level in peat decreases emissions of CO_2 (by up to 20%) and N_2O, but increases

emissions of CH_4. German studies showed that for lowland fens in Central Europe maintaining the ground water level at a depth of 30 cm below the surface under grass utilisation will result in 90% of the optimum plant crop yield, the peat mineralization rate will be reduced by 60-70% and the GHG emissions will be only 50-60% of those under lower water table regimes.

V.2 Peatland utilized for forestry

The utilization of peatlands for forestry is concentrated in Nordic countries (Norway, Sweden, and Finland) and Russia, where over 10 million ha of peatlands have been drained for this purpose. In addition, peatland forestry has some importance in the United Kingdom, Ireland, Canada, the United States and Southeast Asia. Forestry in undrained peatlands is currently practiced primarily in Canada, the United States and Indonesia. Forestry in peatlands generally involves the same silvicultural practices (fellings, site preparation, fertilization) as conducted on mineral soils. The fundamental difference is that water management systems (i.e. drainage) are nearly always required when practicing economic forestry on these naturally wet sites.

The climatic impacts of the use of northern peatlands for forestry are smaller than those of agriculture. Subsidence of the peat surface is much smaller than in agricultural sites and the oxidation of organic matter is of less importance. Drainage of peatlands for forestry changes the plant community to one dominated by tree stands and forest flora. The effect is that despite the replacement of common mire-forming plants, perennial plant cover remains. Biomass and primary production increase during stand development, which thereby increases the C input to the soil. Simultaneously the organic matter decomposition rate increases primarily because of increased soil aeration and enhances outflux of C from the system. The combination of these changed fluxes shifts the C balance of the ecosystem with some peatlands becoming sources of C to the atmosphere, while others remain or become even stronger C sinks. According to the few micrometeorological studies, the net ecosystem C exchange in northern organic soil forests varies from a loss of 800 g CO_2 m^{-2} yr^{-1} to a sink of 1000 g CO_2 m^{-2} yr^{-1}. Similar range in variation is reported in soil C balances. This variation is related to climatic conditions, site type, intensity of management and the level of drainage. Despite the possible soil C losses, ecosystem C balance may remain positive because of the increase in tree stand C stock during the first rotation.

Trace greenhouse gas fluxes are also affected by the forest management practices. Methane emissions always decrease after establishment of the drainage network. If the entire site is effectively drained, CH_4 emissions may cease across the site except from ditches. On the other hand drainage increases N_2O emissions. Highest emissions of ca. 1 g N_2O m^{-2} yr^{-1} have been measured on the most fertile sites, while the average emission in Fennoscandian drained peatlands falls between 0.2 and 0.3 g N_2O m^{-2} yr^{-1}.

Clear felling disturbs the GHG balance of the site temporally by decreasing primary production and inducing N_2O emissions owing to liberation of nutrients in the soil. Soil preparation further disturbs the soil C dynamics. Whole-tree harvesting, especially if stumps are removed, greatly reduces the amount of C in the ecosystem compared to conventional harvesting, in which residuals are left on and in the soil. In general, the potential for soil C losses in peatlands increases with intensity of soil disturbance.

Drainage of nutrient-poor peatlands for forestry in the boreal zone typically decreases the radiative forcing of the site in the short term, since CH_4 emissions decrease and the ecosystem (soil and vegetation) usually continue to accumulate C during the first tree-stand rotation (50–100 years). At more nutrient rich sites where the soil often becomes a source of C an increase in radiative forcing is expected in the long term, as the relative impact of CH_4 emissions is decreased in comparison to that of CO_2.

V.3 Greenhouse gas impact of peat extraction for fuel and horticultural use

Peat has been used for domestic energy purposes by local communities in many parts of the world for centuries. Electricity generation, using peat as a fuel, developed in the 20th Century in some European countries and the Soviet Union. Today, Finland, Ireland, Russian Federation, Belarus and Sweden account for almost 90% of the world's production and consumption of energy peat. In terms of greenhouse gas emissions peat combustion which in Finland accounts for 7% of primary energy, is there responsible for 14% of CO_2 emissions from combustion of fossil fuels. Peat is also used in horticulture, as a growing medium, but the volume used annually is only about half that of fuel peat. Germany and Canada account for over half of horticultural peat extraction.

The main greenhouse gas released as a result of peat fuel extraction and burning is CO_2 but CH_4 and N_2O are also emitted. In the process of peat extraction, the GHG sink function of the peatland is lost. Emissions also arise in the preparation of the surface for cutting (removing vegetation and ditching), extraction of peat and its storage and transportation, combustion and after-treatment of the cutaway area. Combustion accounts for more than 90% of the greenhouse gas emissions.

In Finland and Sweden, several studies have been performed to determine the GHG fluxes from different stages of the fuel peat production supply chain and life cycle analyses have been carried out of peat fuel use and its climate impact in terms of radiative forcing. Some studies show that the extraction and combustion of peat from pristine peatland has radiative forcing similar to the combustion of coal. However, by extracting peat from peatlands that are large greenhouse gas sources, radiative forcing of peat utilisation chain can be significantly reduced. Examples of such peat resources are cultivated peatlands and forestry drained peatlands.

As with the extraction of energy peat, horticultural peat extraction requires drainage of the peatland to accommodate machinery and facilitate drying of peat prior to extraction. This facilitates peat oxidation, increases CO_2 emissions but reduces efflux of CH_4. Although horticultural peat is not consumed instantaneously, it will decompose over time following extraction. In Canada the first life cycle analysis of greenhouse gas emissions from horticultural peat extraction has been performed. Decomposition of peat in growing media accounted for over 70% of greenhouse gas emissions. Remaining emissions arise from transportation, processing and land-use change. Although the use of peat in growing media may enhance productivity of those plants it is used to grow, leading to some increase in C storage, this is temporary and short term.

V.4 Greenhouse gas emissions from drained and degraded tropical peatlands

A peat carbon content of 50 kg C m^{-3} is considered to be representative for SE

Asian peatlands in general and, combining this value with peatland area and thickness, indicates that carbon storage in SE Asian peatlands is in the order of 58 Gt. In the late 1980s 3.7 million hectares of Indonesian peat swamp forest were utilized, leading to an 18% decrease in peat swamp forest area with a consequent reduction in the C-fixation capacity of 5-9 million t yr^{-1}. Deforestation, drainage and conversion of peatland in Indonesia and Malaysia continued throughout the 1990s and are still occurring. These changes are converting large areas of peatland from active carbon sinks to carbon sources.

Apart from logging, the development of palm oil and timber plantations, which require intensive drainage and cause the highest CO_2 emissions of all land uses, are major drivers of peatland deforestation and increases in CO_2 emissions. A large proportion (27%) of palm oil concessions (i.e. existing and planned plantations) in Indonesia is on peatlands; a similar percentage is expected to apply in Malaysia. These plantations are expanding at a rapid rate, driven in part by the increasing demand for palm oil as a biofuel in developed countries. Land use change from peat swamp forest to agriculture or plantations affects C sequestration markedly because the tree biomass is removed and replaced with non-peat forming crop plants. Agriculture requires drainage which creates permanent oxic conditions in the surface peat down to the minimum water table required for optimum crop growth; this results in increased CO_2 emissions.

Comparative studies show that CO_2 emissions from drained forest and recovering sites (undergoing succession to secondary forest) are slightly higher than those from undrained forest probably owing to higher autotrophic respiration from tree roots and enhanced peat oxidation as a result of drainage. The highest annual CO_2 emission (4000 g CO_2 m^{-2} yr^{-1}) occurs in drained forest whilst recovering forest has slightly lower emissions than undrained peat swamp forest. The highest CO_2 emission rates in drainage affected sites occur where channels (ditches) are deepest. Annual CO_2 emissions from a drained agricultural site are considerably lower than at all other sites (ca. 500 g CO_2 m^{-2} yr^{-1}) because there are no trees to provide a supply of litter to peat surface decomposers and the replacement vegetation root biomass is small and produces much less respiratory CO_2 than rain forest trees.

Present and future emissions from natural and drained peatlands in Indonesia have been quantified recently using data on peat extent and depth, present and projected land uses and water management practices, decomposition rates and fire emissions. It is difficult to determine accurately the net CO_2 and CH_4 fluxes in natural peat swamp forest because of the uncertainty in measuring gas fluxes into and out of tree leaves in a multi-layered canopy up to 45 metres in height. Most studies of this ecosystem are carried out of gas exchange at the peat surface and measure CO_2 released in autotrophic and heterotrophic respiration of roots and bacteria and CH_4 evolution from anaerobic decomposition. These values tend to be high for CO_2 and misrepresent the true CO_2 balance of the ecosystem.

Current CO_2 emissions (2005) caused by decomposition of drained peatlands are estimated to be ca. 630 million t yr^{-1} (range 350 – 870 million t yr^{-1}), which will increase in coming decades, and will continue well beyond the 21st century, unless land management practices and peatland development plans are changed. In addition, between 1997 and 2006 an estimated average of 1400 Mt yr^{-1} of CO_2 emissions was caused by fires associated with peatland drainage and degradation.

The total current CO_2 emissions from tropical peatland of approximately 2000 Mt yr^{-1} equal almost 8% of global emissions from fossil fuel burning. Emissions are likely to increase every year for the first decades after 2000. As shallow peat deposits become depleted, however, and the drained peatland area diminishes, peat oxidation emissions are predicted to peak sometime between 2015 and 2035 at between 560 and 980 Mt yr^{-1} and will decline steadily thereafter. As the deeper peat deposits will take much longer to be depleted, significant CO_2 emission will continue beyond 2100.

Overall, methane emissions from tropical peatland are very low irrespective of whether it is natural peat swamp forest or drained and degraded or used for agriculture. Annual CH_4 emissions are highest in drainage affected forest and recovering forest sites, both of which are subjected to periodical waterlogging and receive inputs of easily decomposable litter from the canopy (providing substrates for methanogenic bacteria). Peak CH_4 emissions occur when the water table is near to or above the peat surface; oxic conditions increase considerably, however, following deep drainage and the potential for CH_4 oxidation by methanotrophic bacteria is much greater. Cleared but uncultivated peatland has a CH_4 emission of almost zero at all peat water table depths owing to the permanently low water table following drainage to grow crops.

N_2O emissions from natural tropical peatlands are low but evidence is emerging that suggests that these increase following land use change and fire.

Current developments give little cause for optimism because, while deforestation rates on non-peatlands in SE Asia have decreased slightly in recent years, those on peatlands have been stable (on average) for up to 20 years. The current (2000-2005) average deforestation rate is 1.5% yr^{-1}. In 2005, 25% of all deforestation in SE Asia was on peatlands.

V.5 Potential of peatland restoration for mitigation of climate change impacts of peatland management

Peatland restoration is growing in importance in Europe and North America and is likely to remain important over the next half century. It is also gaining recognition in tropical peatland areas where some of the greatest challenges exist following inappropriate and unsuccessful development projects. While peatland restoration is primarily designed for global biodiversity protection, it can also play an important role in reducing GHG emissions.

In general, rewetting of peatlands reduces CO_2 emissions by creating anoxic, reducing conditions, although it may lead to an icrease in CH_4 efflux at least for a time. Rewetting also inhibits nitrification, resulting in reduced emission of N_2O. Some restored boreal bogs have become C sinks again following successful re-establishment of Sphagnum-dominated vegetation. In contrast, it is more difficult to re-establish the C sink function of temperate bogs and fens. In some cases, CH_4 emissions are frequently higher in rewetted peatlands, especially fens, than in pristine peatlands. Emission of CH_4 from restored peatlands can be greatly reduced if the water table is kept below (about 10 cm) the surface so that a high proportion of the CH_4 produced in the lower horizons will be oxidized in the thin, oxic surface layer.

The duration of most field investigations of peatland restoration is too short to evaluate the long-term dynamics of rewetted bogs and fens. After rewetting of peatlands, at least three phases of carbon and nitrogen cycling occur, and only in the third phase, more than 10 years after rewetting, are greenhouse gas fluxes expected to be in the range of natural peatlands. Thus, initially, restoration may result in a pulse of GHG, but in the long-term, the peatland should return to a C and GHG sink with a similar climate impact as an undisturbed peatland. More long-term studies with better spatial coverage are required to better constrain the GHG impact of peatland restoration.

VI. Reporting peat greenhouse gas emissions in international climate conventions

Peat-based GHG emissions reported under the United Nations Framework Convention on Climate Change (UNFCCC) are divided between several sectors: Energy, Agriculture and Land Use, Land-Use Change and Forestry (LULUCF). Only human-induced GHG emissions are included in reporting, therefore, emissions from undisturbed/virgin peatlands are not included.

Emissions of GHGs from peat combustion for energy and heat are reported in the Energy sector and under the 2006 IPCC Guidelines peat is classified to its own class between fossil energy sources and biomass. In the reporting, the emission calculations are, however, based only on the emissions from the combustion. In the Agriculture sector, peatlands are considered only to report N_2O emissions from organic agricultural soils. Emissions of CO_2 from organic agricultural soils are reported under LULUCF. Also under LULUCF are GHG emissions arising from peat extraction areas, biomass burning on peat soils, drained organic forest soils, disturbance and nitrogen fertilization associated with conversion of organic soils to croplands.

Although total national emissions are reported to the UNFCCC both including and excluding LULUCF from the total, the basis for emission reductions under the Kyoto Protocol is total emission excluding LULUCF. Emissions and removals of GHGs are considered only partially when assessing a country's fulfillment of their commitment under the Kyoto Protocol. Emissions or removals from afforestation, deforestation and reforestation since 1990 will be added to or subtracted from a country's assigned amount according to the Protocol, while additional emissions or removals from forest management, cropland management, grazing land management and revegetation may be considered if a country elects for their inclusion. Once a certain LULUCF activity has been added to a country's Kyoto accounting, it must be reported continuously and consistently even if a sink becomes a source.

While industrialized nations listed in Annex I of the UNFCCC submit annual GHG inventories and have emission limitation targets under the Kyoto Protocol, the heterogeneous groups of developing nations that are non-Annex I Parties are only required to provide information about GHG emissions in national communications. However, peatland fires and wetland degradation in many non-Annex I countries contribute significantly to global GHG emissions.

The Kyoto Protocol allows Annex I Parties to fulfill part of their emission reduction commitments by taking actions to reduce emissions in developing countries under the Clean Development Mechanism (CDM). For the first commitment period of the Kyoto Protocol (2008-2012) only afforestation and reforestation activities under LULUCF are eligible for CDM consideration; however, enlarging the scope of LULUCF activities considered under this mechanism could assist in mitigation of fires and degradation in peatlands helping to reduce peat-derived emissions, particularly in developing nations.

Methodologies and guidance for estimating peat-based emissions in the good practice guidelines for LULUCF and the 2006 IPCC Guidelines are relatively scarce. Default methodologies that include all anthropogenic activities likely to alter peatland hydrology, temperature regime and vegetation composition are still lacking. There is still a deficiency of data that can be applied to country, region or site-specific conditions with data availability varying for different climate regions and countries, while global scale knowledge of peat-derived emissions remains limited. Development of scientifically sound emission factors for peat soils is complicated and resource demanding owing to the variation between sites. Still, with more long-term measurements of GHG fluxes on sites with different climatic conditions and land uses, reliable emission factors for inventory purposes can be developed thereby improving the understanding of GHG impacts of different activities under given circumstances.

VII. Wise use recommendations: Peatlands and climate change

Carbon stocks in undisturbed peatlands

Peatlands represent globally significant stores of soil C that have been accumulating for millennia. Thus, these ecosystems have acted as, and continue to act as, important GHG sinks and this function should be considered alongside other functions and values when making management decisions.

Climate change may threaten C stocks in unmanaged peatlands because of drought leading to peat oxidation, permafrost melting and shifting fire regimes. Owing to the variability in environmental conditions and GHG exchange across peatlands, predicting the overall response is not simple. Research aimed at improving peatland inventories and enhancing our understanding of the links between climate, hydrology, ecology, permafrost degradation, fire regimes and GHG balances will improve our knowledge of the state of current peat resources and predict the fate of this important store of carbon.

VII.1 Mitigation of greenhouse gas emissions from managed peatlands

Since peatland management generally involves lowering the water table, GHG emissions result from decomposition of stored organic matter and, particularly as has been observed in tropical peatlands, an increase in fire susceptibility. The most efficient method for reducing GHG emissions from peatland is to prevent future land use change although this is not always economically, socially or politically possible. If this is the case, land management strategies should focus on preventing degradation of additional peatlands where possible, and adjusting management practices on developed peatlands in order to reduce GHG impacts. The incentive to mitigate GHG emissions from peatland management may come from a requirement to include emissions in national GHG inventories (as is the case for most northern peatlands) or from an attachment of an economic value to the C stock (as may soon apply to tropical peatlands).

It is essential that future land use of peatland incorporates the principles and practices of wise use in order to promote sustainable management, especially with respect to hydrology, water and carbon. Inevitably, however, every type of human intervention on peatland leads to impairment or even loss of natural resource functions (ecology, hydrology, biodiversity, carbon storage). Effective peatland management also requires engagement between scientists, policy makers and stakeholders.

Changing the management of peatlands used for agriculture and forestry, for example, reducing the extent and intensity of drainage, converting arable cultivation to grasslands and pasture, and reducing fertiliser application will reduce GHG emissions.

Life cycle analysis of peat GHG emissions from peat extraction indicates that climate impact can be reduced by using already degraded peatland sites, such as those already drained for forestry or agriculture, and reducing the time period during which the peat is extracted, followed by rapid conversion to an appropriate after-use.

New opportunities for protection of the tropical peat carbon store may arise from current negotiations on financial payments for reduced emissions from avoided deforestation and forest degradation (REDD). This could put an economic value on the remaining tropical peat swamp forests and their globally important C stores, and provide an incentive for their protection.

Afforestation following peatland cultivation or peat extraction can greatly reduce radiative forcing as C will be stored in tree biomass; however, the resulting ecosystem will likely be very different than the pre-disturbance peatland. Restoration of the site may assist not only by preventing oxidation and returning the site into a C sink, but can also reinstate other ecosystem functions such as biological diversity. Peatland restoration can be effective for millennia, leaving the work of GHG sequestration to micro-organisms and plants.

SUMMARY FOR POLICYMAKERS
"WISE USE OF PEATLANDS AND CLIMATE CHANGE"

I. Global extent of peat and peatlands

Peatlands cover an estimated area ca. 400 million ha equivalent to 3% of the Earth's land surface (Figure 0.1). Most (about 350 million ha) are in the northern hemisphere, covering large areas in North America, Russia and Europe. Tropical peatlands occur in mainland East Asia, Southeast Asia, the Caribbean and Central America, South America and southern Africa where the current estimate of undisturbed peatland is in the range 30-45 million ha accounting for 10-12% of the global peatland resource.

II. Global peat carbon store

Currently, peatlands globally represent a major store of soil carbon, sink for carbon dioxide and source of atmospheric methane. In general, nitrous oxide (N_2O) emissions are low from natural peatlands but there is evidence that those used for agriculture and receiving nitrogen fertilisers, are releasing significant amounts of this very potent greenhouse gas. The change in peatland C storage results from changes in the balance between net exchange of CO_2, emission of CH_4, and hydrological losses of carbon (e.g. dissolved organic and inorganic C and particulate organic C).

Northern peatlands store around 450 billion metric tons (Bt = 10^{15} g = Pg = Gt) carbon, which is equivalent to approximately one third of global soil C stocks and 75% of the pre-industrial mass of C stored in the atmosphere. In tropical peatlands both the vegetation and underlying peat constitute a large and highly concentrated carbon

Figure 0–1 Distribution of mires
Source: International Peat Society, Available www.peatsociety.fi.

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pool amounting to about 60 Bt. The current annual carbon storage rate in the world's peatlands is approximately 100 million tonnes (Mt), which is equivalent to approximately 370 Mt CO_2 yr^{-1} but this has varied greatly throughout millennia depending mainly on climate and sea level. Conversely, however, pristine peatlands (mires), especially those in boreal and temperate zones, emit methane (ca. 20 Mt yr^{-1}), which is a potent greenhouse gas.

In terms of GHG management, the maintenance of large stores of C in undisturbed peatlands should be a priority.

III. Carbon accumulation and greenhouse gas exchange in undisturbed peatlands

Regardless of variability, when considering the role of peatlands in atmospheric GHG balances, it is important to consider that they have taken up and released GHGs continuously since their formation and thus their influence must be modelled over time. When this is considered, the effect of sequestering CO_2 in peat outweighs CH_4 emissions. Thus**,** peatlands have been net GHG sinks for thousands of years.

Most contemporary peatlands began accumulating peat following the last glacial period and have continued to do so throughout the Holocene, approximately the last 10,000 years. Some peatlands in tropical Southeast Asia, however, started to form towards the end of the Pleistocene more than 20,000 years ago. Carbon accumulation rates have varied over this period in relation to stage of peatland development and climate; average long-term C accumulation rates for northern bogs are 20-30 g m^{-2} yr^{-1}, while a tropical peat core has yielded a long-term average C accumulation rate of over 50 g m^{-2} yr^{-1}.

Contemporary C exchange in peatlands exhibits great spatial variability related to regional and local difference in ecology, hydrology, and climate. Studies of net ecosystem exchange of CO_2 in northern peatlands provide values ranging from uptake of over 220 g CO_2 m^{-2} yr^{-1} to release of 310 g CO_2 m^{-2} yr^{-1}. Accumulation rates in tropical peatlands are also variable, yet they probably accumulate more carbon per unit area than northern peatlands. Present net C uptake may be in the range of 500 g C m^{-2} yr^{-1} (~1800 g CO_2 m^{-2} yr^{-1}). Temporal studies of peatlands reveal that they may act as CO_2 sinks in some years and sources in others, depending on climate. Emissions of CH_4 and N_2O are similarly variable in space and time.

IV. Climate change impact on peatland carbon stocks and greenhouse gas exchange

Owing to the spatial diversity of peatlands, the variability of the response of peatland GHG exchange to climate change is likely to be large. Some peatlands will emit more CO_2 to the atmosphere and change from net C sinks to become sources; other peatlands may exhibit increased CO_2 sequestration owing to elevated water tables. In terms of the stability of peatland C stocks, non-permafrost peatlands will be most at risk because not only are these likely to release CO_2 as a result of peat oxidation under dry conditions, but they will also face increased risk of fire.

Climate variability throughout the Holocene, especially in boreal and temperate zones, has had a major effect on C accumulation rates in peatlands. Increased peat accumulation reflects periods of a more positive precipitation-evaporation balance, which is supported by data on lake level fluctuations from these regions. In contrast, drier periods

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correspond to lower C accumulation rates. Evidence from the Holocene suggests that climate warming results in permafrost melting and release of GHGs from northern peatlands but this is compensated to some degree by extension of forests northwards. Future climate change may also result in accelerated rates of permafrost degradation, displacing tree communities, and creating water saturated open fens. C accumulation at these sites will be higher than neighbouring permafrost peatlands but CH_4 emissions will be enhanced owing to increased vegetation productivity and waterlogged conditions.

In contrast, climate change scenarios predict that some peatlands will experience lowered water tables, leading to increased dryness and unsaturated, oxic conditions at their surface, resulting in aerobic decomposition (oxidation) and larger releases of CO_2. On the other hand, development of vegetation towards shrub-dominated communities may lead to higher primary production, compensating soil C losses. The larger unsaturated zone will lead to reduced CH_4 emissions and some dry bogs may become CH_4 sinks. Over time, peat subsidence combined with increased ecosystem productivity may keep some peatlands (e.g. fens, bog pools and hollows) wet, maintaining or enhancing C storage although CH_4 emissions may increase under these conditions.

In some parts of the world the peat C store is being reduced because of fire. Major increases in the area of peatland burned have been documented in recent decades and this may continue in the future if peatlands dry out as a result of climate change or anthropogenic activities. Fire will continue to play an important role in the fate of global peatland C stocks.

V. Land-use change impacts on peatland carbon stocks and greenhouse gas exchange

Agriculture, forestry and peat extraction for fuel and horticultural use are the major causes of peatland disturbance. As these types of land-use change require alteration of peatland hydrology, peat oxidation results and the greenhouse gas balance of the peatland is altered (Table 0.1).

V.1 Peatland utilised for agriculture

About 14 – 20 % of peatlands in the world are currently used for agriculture and the great majority of these are used as meadows and pastures. For agricultural use, fens and raised bogs have to be drained in order to regulate the air and water conditions in the soil to meet the requirements of cultivated or pasture plants. In many European countries, GHG emissions from agricultural peatlands dominate national emissions of GHGs from peat sources.

The loss of water from the upper peat by drainage, followed by oxidation, leads to compaction and subsidence of the surface. Drainage of peat increases the emissions of CO_2 and N_2O but decreases the emission of CH_4. The emission rates of the greenhouse gases depend on many factors including peat temperature, groundwater level and peat moisture content. For ploughed temperate fens (arable land), in the central and north part of Europe (Sweden), annual CO_2 mean emissions of 4100 g m^{-2} can be expected, but with a high range of variation. Temperate and boreal fens converted to grassland show mean CO_2 emissions of about 700 g m^{-2} yr^{-1} (Canada) and from 1500 to 1700 g m^{-2} yr^{-1} in Central Europe (Poland, Germany, The Netherlands and Sweden). Finnish studies show average CO_2 emissions of about 2200 g m^{-2} yr^{-1} for boreal fens under grass and barley. Drained peatlands are large sources of nitrous oxide

Table 0.1: *Impact of land-use change on peatland greenhouse gas balance*

	Approximate area (10^3 km^2)	Response of greenhouse gas flux to land-use change CO_2	CH_4	N_2O
Agriculture	300	**Net emission of CO_2** -decreased CO_2 uptake: removal of peatland vegetation, crop residues removed -increased CO_2 emission: lower water table, peat oxidation	**Reduced CH_4 efflux** -emission from ditches remains high	**Enhanced N_2O emission** -N mineralization at nutrient rich sites -fertilizer application
Forestry	150	**Often little change in net ecosystem balance** -increased CO_2 uptake by treestand -increased CO_2 emission from soils: peat oxidation, dependent on extent of drainage	**Reduced CH_4 efflux** -emission from ditches remains high	**Dependent on site type and fertilizer application** -increased N_2O flux on nutrient rich sites -increased N_2O flux if fertilizer applied
Peat extraction	>5	**Net emission of CO_2** -loss of CO_2 via combustion (fuel) or decomposition (horticulture) -decreased CO_2 uptake: removal of peatland vegetation, crop residues removed -increased CO_2 emission: lower water table, peat oxidation	**Reduced CH_4 efflux** -emission from ditches remains high	**Little change** -some increase in N_2O efflux at nutrient rich sites

(N_2O) with fluxes varying between 0.2 and 5.6 g m^{-2} annually. The annual CH_4 fluxes of cultivated peat soils range from a very small sink to low emission.

The position of the water table is one of the most important factors influencing peat conditions and processes in organic soils. Consequently, precise water management for peatlands utilised for agriculture purposes is very important. Increasing the water level in peat decreases emissions of CO_2 (by up to 20%) and N_2O, but increases emissions of CH_4. German studies showed that for lowland fens in Central Europe maintaining the ground water level at a depth of 30 cm below the surface under grass utilisation will result in 90% of the optimum plant crop yield, the peat mineralization rate will be reduced by 60-70% and the GHG emissions will be

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only 50-60% of those under lower water table regimes.

V.2 Peatland utilized for forestry

The utilization of peatlands for forestry is concentrated in Nordic countries (Norway, Sweden, and Finland) and Russia, where over 10 million ha of peatlands have been drained for this purpose. In addition, peatland forestry has some importance in the United Kingdom, Ireland, Canada, the United States and Southeast Asia. Forestry in undrained peatlands is currently practiced primarily in Canada, the United States and Indonesia. Forestry in peatlands generally involves the same silvicultural practices (fellings, site preparation, fertilization) as conducted on mineral soils. The fundamental difference is that water management systems (i.e. drainage) are nearly always required when practicing economic forestry on these naturally wet sites.

The climatic impacts of the use of northern peatlands for forestry are smaller than those of agriculture. Subsidence of the peat surface is much smaller than in agricultural sites and the oxidation of organic matter is of less importance. Drainage of peatlands for forestry changes the plant community to one dominated by tree stands and forest flora. The effect is that despite the replacement of common mire-forming plants, perennial plant cover remains. Biomass and primary production increase during stand development, which thereby increases the C input to the soil. Simultaneously the organic matter decomposition rate increases primarily because of increased soil aeration and enhances outflux of C from the system. The combination of these changed fluxes shifts the C balance of the ecosystem with some peatlands becoming sources of C to the atmosphere, while others remain or become even stronger C sinks. According to the few micrometeorological studies, the net ecosystem C exchange in northern organic soil forests varies from a loss of 800 g CO_2 m^{-2} yr^{-1} to a sink of 1000 g CO_2 m^{-2} yr^{-1}. Similar range in variation is reported in soil C balances. This variation is related to climatic conditions, site type, intensity of management and the level of drainage. Despite the possible soil C losses, ecosystem C balance may remain positive because of the increase in tree stand C stock during the first rotation.

Trace greenhouse gas fluxes are also affected by the forest management practices. Methane emissions always decrease after establishment of the drainage network. If the entire site is effectively drained, CH_4 emissions may cease across the site except from ditches. On the other hand drainage increases N_2O emissions. Highest emissions of ca. 1 g N_2O m^{-2} yr^{-1} have been measured on the most fertile sites, while the average emission in Fennoscandian drained peatlands falls between 0.2 and 0.3 g N_2O m^{-2} yr^{-1}.

Clear felling disturbs the GHG balance of the site temporally by decreasing primary production and inducing N_2O emissions owing to liberation of nutrients in the soil. Soil preparation further disturbs the soil C dynamics. Whole-tree harvesting, especially if stumps are removed, greatly reduces the amount of C in the ecosystem compared to conventional harvesting, in which residuals are left on and in the soil. In general, the potential for soil C losses in peatlands increases with intensity of soil disturbance.

Drainage of nutrient-poor peatlands for forestry in the boreal zone typically decreases the radiative forcing of the site in the short term, since CH_4 emissions decrease and the ecosystem (soil and vegetation) usually continue to accumulate C during the first tree-stand rotation

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(50–100 years). At more nutrient rich sites where the soil often becomes a source of C an increase in radiative forcing is expected in the long term, as the relative impact of CH_4 emissions is decreased in comparison to that of CO_2.

V.3 Greenhouse gas impact of peat extraction for fuel and horticultural use

Peat has been used for domestic energy purposes by local communities in many parts of the world for centuries. Electricity generation, using peat as a fuel, developed in the 20th Century in some European countries and the Soviet Union. Today, Finland, Ireland, Russian Federation, Belarus and Sweden account for almost 90% of the world's production and consumption of energy peat. In terms of greenhouse gas emissions peat combustion which in Finland accounts for 7% of primary energy, is there responsible for 14% of CO_2 emissions from combustion of fossil fuels. Peat is also used in horticulture, as a growing medium, but the volume used annually is only about half that of fuel peat. Germany and Canada account for over half of horticultural peat extraction.

The main greenhouse gas released as a result of peat fuel extraction and burning is CO_2 but CH_4 and N_2O are also emitted. In the process of peat extraction, the GHG sink function of the peatland is lost. Emissions also arise in the preparation of the surface for cutting (removing vegetation and ditching), extraction of peat and its storage and transportation, combustion and after-treatment of the cutaway area. Combustion accounts for more than 90% of the greenhouse gas emissions.

In Finland and Sweden, several studies have been performed to determine the GHG fluxes from different stages of the fuel peat production supply chain and life cycle analyses have been carried out of peat fuel use and its climate impact in terms of radiative forcing. Some studies show that the extraction and combustion of peat from pristine peatland has radiative forcing similar to the combustion of coal. However, by extracting peat from peatlands that are large greenhouse gas sources, radiative forcing of peat utilisation chain can be significantly reduced. Examples of such peat resources are cultivated peatlands and forestry drained peatlands.

As with the extraction of energy peat, horticultural peat extraction requires drainage of the peatland to accommodate machinery and facilitate drying of peat prior to extraction. This facilitates peat oxidation, increases CO_2 emissions but reduces efflux of CH_4. Although horticultural peat is not consumed instantaneously, it will decompose over time following extraction. In Canada the first life cycle analysis of greenhouse gas emissions from horticultural peat extraction has been performed. Decomposition of peat in growing media accounted for over 70% of greenhouse gas emissions. Remaining emissions arise from transportation, processing and land-use change. Although the use of peat in growing media may enhance productivity of those plants it is used to grow, leading to some increase in C storage, this is temporary and short term.

V.4 Greenhouse gas emissions from drained and degraded tropical peatlands

A peat carbon content of 50 kg C m^{-3} is considered to be representative for SE Asian peatlands in general and, combining this value with peatland area and thickness, indicates that carbon storage in SE Asian peatlands is in the order of 58 Gt. In the late 1980s 3.7 million hectares of Indonesian peat swamp forest were utilized, leading to an 18% decrease in peat swamp forest

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area with a consequent reduction in the C-fixation capacity of 5-9 million t yr^{-1}. Deforestation, drainage and conversion of peatland in Indonesia and Malaysia continued throughout the 1990s and are still occurring. These changes are converting large areas of peatland from active carbon sinks to carbon sources.

Apart from logging, the development of palm oil and timber plantations, which require intensive drainage and cause the highest CO_2 emissions of all land uses, are major drivers of peatland deforestation and increases in CO_2 emissions. A large proportion (27%) of palm oil concessions (i.e. existing and planned plantations) in Indonesia is on peatlands; a similar percentage is expected to apply in Malaysia. These plantations are expanding at a rapid rate, driven in part by the increasing demand for palm oil as a biofuel in developed countries. Land use change from peat swamp forest to agriculture or plantations affects C sequestration markedly because the tree biomass is removed and replaced with non-peat forming crop plants. Agriculture requires drainage which creates permanent oxic conditions in the surface peat down to the minimum water table required for optimum crop growth; this results in increased CO_2 emissions.

Comparative studies show that CO_2 emissions from drained forest and recovering sites (undergoing succession to secondary forest) are slightly higher than those from undrained forest probably owing to higher autotrophic respiration from tree roots and enhanced peat oxidation as a result of drainage. The highest annual CO_2 emission (4000 g CO_2 m^{-2} yr^{-1}) occurs in drained forest whilst recovering forest has slightly lower emissions than undrained peat swamp forest. The highest CO_2 emission rates in drainage affected sites occur where channels (ditches) are deepest. Annual CO_2 emissions from a drained agricultural site are considerably lower than at all other sites (ca. 500 g CO_2 m^{-2} yr^{-1}) because there are no trees to provide a supply of litter to peat surface decomposers and the replacement vegetation root biomass is small and produces much less respiratory CO_2 than rain forest trees.

Present and future emissions from natural and drained peatlands in Indonesia have been quantified recently using data on peat extent and depth, present and projected land uses and water management practices, decomposition rates and fire emissions. It is difficult to determine accurately the net CO_2 and CH_4 fluxes in natural peat swamp forest because of the uncertainty in measuring gas fluxes into and out of tree leaves in a multi-layered canopy up to 45 metres in height. Most studies of this ecosystem are carried out of gas exchange at the peat surface and measure CO_2 released in autotrophic and heterotrophic respiration of roots and bacteria and CH_4 evolution from anaerobic decomposition. These values tend to be high for CO_2 and misrepresent the true CO_2 balance of the ecosystem.

Current CO_2 emissions (2005) caused by decomposition of drained peatlands are estimated to be ca. 630 million t yr^{-1} (range 350 – 870 million t yr^{-1}), which will increase in coming decades, and will continue well beyond the 21st century, unless land management practices and peatland development plans are changed. In addition, between 1997 and 2006 an estimated average of 1400 Mt yr^{-1} of CO_2 emissions was caused by fires associated with peatland drainage and degradation. The total current CO_2 emissions from tropical peatland of approximately 2000 Mt yr^{-1} equal almost 8% of global emissions from fossil fuel burning. Emissions are likely to increase every year for the first decades after 2000. As shallow peat deposits become depleted, however, and

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the drained peatland area diminishes, peat oxidation emissions are predicted to peak sometime between 2015 and 2035 at between 560 and 980 Mt yr^{-1} and will decline steadily thereafter. As the deeper peat deposits will take much longer to be depleted, significant CO_2 emission will continue beyond 2100.

Overall, methane emissions from tropical peatland are very low irrespective of whether it is natural peat swamp forest or drained and degraded or used for agriculture. Annual CH_4 emissions are highest in drainage affected forest and recovering forest sites, both of which are subjected to periodical waterlogging and receive inputs of easily decomposable litter from the canopy (providing substrates for methanogenic bacteria). Peak CH_4 emissions occur when the water table is near to or above the peat surface; oxic conditions increase considerably, however, following deep drainage and the potential for CH_4 oxidation by methanotrophic bacteria is much greater. Cleared but uncultivated peatland has a CH_4 emission of almost zero at all peat water table depths owing to the permanently low water table following drainage to grow crops.

N_2O emissions from natural tropical peatlands are low but evidence is emerging that suggests that these increase following land use change and fire.

Current developments give little cause for optimism because, while deforestation rates on non-peatlands in SE Asia have decreased slightly in recent years, those on peatlands have been stable (on average) for up to 20 years. The current (2000-2005) average deforestation rate is 1.5% yr^{-1}. In 2005, 25% of all deforestation in SE Asia was on peatlands.

V.5 Potential of peatland restoration for mitigation of climate change impacts of peatland management

Peatland restoration is growing in importance in Europe and North America and is likely to remain important over the next half century. It is also gaining recognition in tropical peatland areas where some of the greatest challenges exist following inappropriate and unsuccessful development projects. While peatland restoration is primarily designed for global biodiversity protection, it can also play an important role in reducing GHG emissions.

In general, rewetting of peatlands reduces CO_2 emissions by creating anoxic, reducing conditions, although it may lead to an icrease in CH_4 efflux at least for a time. Rewetting also inhibits nitrification, resulting in reduced emission of N_2O. Some restored boreal bogs have become C sinks again following successful re-establishment of Sphagnum-dominated vegetation. In contrast, it is more difficult to re-establish the C sink function of temperate bogs and fens. In some cases, CH_4 emissions are frequently higher in rewetted peatlands, especially fens, than in pristine peatlands. Emission of CH_4 from restored peatlands can be greatly reduced if the water table is kept below (about 10 cm) the surface so that a high proportion of the CH_4 produced in the lower horizons will be oxidized in the thin, oxic surface layer.

The duration of most field investigations of peatland restoration is too short to evaluate the long-term dynamics of rewetted bogs and fens. After rewetting of peatlands, at least three phases of carbon and nitrogen cycling occur, and only in the third phase, more than 10 years after rewetting, are greenhouse gas fluxes expected to be in the range of natural peatlands. Thus, initially, restoration may result in a pulse of GHG, but in the long-term, the peatland should

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return to a C and GHG sink with a similar climate impact as an undisturbed peatland. More long-term studies with better spatial coverage are required to better constrain the GHG impact of peatland restoration.

VI. Reporting peat greenhouse gas emissions in international climate conventions

Peat-based GHG emissions reported under the United Nations Framework Convention on Climate Change (UNFCCC) are divided between several sectors: Energy, Agriculture and Land Use, Land-Use Change and Forestry (LULUCF). Only human-induced GHG emissions are included in reporting, therefore, emissions from undisturbed/virgin peatlands are not included.

Emissions of GHGs from peat combustion for energy and heat are reported in the Energy sector and under the 2006 IPCC Guidelines peat is classified to its own class between fossil energy sources and biomass. In the reporting, the emission calculations are, however, based only on the emissions from the combustion. In the Agriculture sector, peatlands are considered only to report N_2O emissions from organic agricultural soils. Emissions of CO_2 from organic agricultural soils are reported under LULUCF. Also under LULUCF are GHG emissions arising from peat extraction areas, biomass burning on peat soils, drained organic forest soils, disturbance and nitrogen fertilization associated with conversion of organic soils to croplands.

Although total national emissions are reported to the UNFCCC both including and excluding LULUCF from the total, the basis for emission reductions under the Kyoto Protocol is total emission excluding LULUCF. Emissions and removals of GHGs are considered only partially when assessing a country's fulfillment of their commitment under the Kyoto Protocol. Emissions or removals from afforestation, deforestation and reforestation since 1990 will be added to or subtracted from a country's assigned amount according to the Protocol, while additional emissions or removals from forest management, cropland management, grazing land management and revegetation may be considered if a country elects for their inclusion. Once a certain LULUCF activity has been added to a country's Kyoto accounting, it must be reported continuously and consistently even if a sink becomes a source.

While industrialized nations listed in Annex I of the UNFCCC submit annual GHG inventories and have emission limitation targets under the Kyoto Protocol, the heterogeneous groups of developing nations that are non-Annex I Parties are only required to provide information about GHG emissions in national communications. However, peatland fires and wetland degradation in many non-Annex I countries contribute significantly to global GHG emissions.

The Kyoto Protocol allows Annex I Parties to fulfill part of their emission reduction commitments by taking actions to reduce emissions in developing countries under the Clean Development Mechanism (CDM). For the first commitment period of the Kyoto Protocol (2008-2012) only afforestation and reforestation activities under LULUCF are eligible for CDM consideration; however, enlarging the scope of LULUCF activities considered under this mechanism could assist in mitigation of fires and degradation in peatlands helping to reduce peat-derived emissions, particularly in developing nations.

Methodologies and guidance for estimating peat-based emissions in the good practice

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guidelines for LULUCF and the 2006 IPCC Guidelines are relatively scarce. Default methodologies that include all anthropogenic activities likely to alter peatland hydrology, temperature regime and vegetation composition are still lacking. There is still a deficiency of data that can be applied to country, region or site-specific conditions with data availability varying for different climate regions and countries, while global scale knowledge of peat-derived emissions remains limited. Development of scientifically sound emission factors for peat soils is complicated and resource demanding owing to the variation between sites. Still, with more long-term measurements of GHG fluxes on sites with different climatic conditions and land uses, reliable emission factors for inventory purposes can be developed thereby improving the understanding of GHG impacts of different activities under given circumstances.

VII. Wise use recommendations: Peatlands and climate change

Carbon stocks in undisturbed peatlands

Peatlands represent globally significant stores of soil C that have been accumulating for millennia. Thus, these ecosystems have acted as, and continue to act as, important GHG sinks and this function should be considered alongside other functions and values when making management decisions.

Climate change may threaten C stocks in unmanaged peatlands because of drought leading to peat oxidation, permafrost melting and shifting fire regimes. Owing to the variability in environmental conditions and GHG exchange across peatlands, predicting the overall response is not simple. Research aimed at improving peatland inventories and enhancing our understanding of the links between climate, hydrology, ecology, permafrost degradation, fire regimes and GHG balances will improve our knowledge of the state of current peat resources and predict the fate of this important store of carbon.

VII.1 Mitigation of greenhouse gas emissions from managed peatlands

Since peatland management generally involves lowering the water table, GHG emissions result from decomposition of stored organic matter and, particularly as has been observed in tropical peatlands, an increase in fire susceptibility. The most efficient method for reducing GHG emissions from peatland is to prevent future land use change although this is not always economically, socially or politically possible. If this is the case, land management strategies should focus on preventing degradation of additional peatlands where possible, and adjusting management practices on developed peatlands in order to reduce GHG impacts. The incentive to mitigate GHG emissions from peatland management may come from a requirement to include emissions in national GHG inventories (as is the case for most northern peatlands) or from an attachment of an economic value to the C stock (as may soon apply to tropical peatlands).

It is essential that future land use of peatland incorporates the principles and practices of wise use in order to promote sustainable management, especially with respect to hydrology, water and carbon. Inevitably, however, every type of human intervention on peatland leads to impairment or even loss of natural resource functions (ecology, hydrology, biodiversity, carbon storage). Effective peatland management also requires engagement between scientists, policy makers and stakeholders.

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Changing the management of peatlands used for agriculture and forestry, for example, reducing the extent and intensity of drainage, converting arable cultivation to grasslands and pasture, and reducing fertiliser application will reduce GHG emissions.

Life cycle analysis of peat GHG emissions from peat extraction indicates that climate impact can be reduced by using already degraded peatland sites, such as those already drained for forestry or agriculture, and reducing the time period during which the peat is extracted, followed by rapid conversion to an appropriate after-use.

New opportunities for protection of the tropical peat carbon store may arise from current negotiations on financial payments for reduced emissions from avoided deforestation and forest degradation (REDD). This could put an economic value on the remaining tropical peat swamp forests and their globally important C stores, and provide an incentive for their protection.

Afforestation following peatland cultivation or peat extraction can greatly reduce radiative forcing as C will be stored in tree biomass; however, the resulting ecosystem will likely be very different than the pre-disturbance peatland. Restoration of the site may assist not only by preventing oxidation and returning the site into a C sink, but can also reinstate other ecosystem functions such as biological diversity. Peatland restoration can be effective for millennia, leaving the work of GHG sequestration to micro-organisms and plants.

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CHAPTER 1:

CARBON ACCUMULATION IN BOREAL PEATLANDS DURING THE HOLOCENE – IMPACTS OF CLIMATE VARIATIONS

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1.1. Introduction

Northern boreal peatlands cover an area of 5.8 million square kilometres according to a recent estimation by Bleuten *et al*. (2006), and thus they form a major biosphere carbon pool as also emphasized by Gorham (1991) (see also Gorham *et al*., 2003). Understanding the rate of carbon accumulation has become increasingly important for estimating the quantity of carbon reserves, especially with reference to their relevance to climate change, CO_2 sequestration and global warming (Tolonen & Turunen, 1996; Moore *et al*., 1998; Mäkilä, 1997; 2001; Mäkilä *et al.*, 2001; Turunen *et al.*, 2002; Yu *et al*., 2003; Belyea & Malmer 2004). Boreal mires can change from carbon sinks to sources in years when the summertime water table is below the long-term average level (Shurpali *et al*., 1995).

Peat accumulation is controlled by the rate of decay rather than productivity (Clymo, 1965; 1978; Damman, 1979). Most decay takes place in the biologically active oxic layer (acrotelm). Decay may also occur in the anoxic layer (catotelm) (Ingram, 1978; 1983). The products of slow decay are mainly removed from the anoxic layer as carbon dioxide (CO_2) and methane (CH_4). The biomass formed at the top of the acrotelm decays selectively and as much as about 10-20% passes into the catotelm (Clymo, 1984). Carbon accumulation into the catotelm is apparent because decomposition continues slowly under anoxic conditions. In boreal mires, about 4-10% of the photosynthetically fixed carbon returns annually to the atmosphere as CH_4 that is formed by the anoxic degradation of organic matter (Alm *et al*., 1997). In addition to losses from decomposition, some carbon and nutrients are also lost from the ecosystem via leaching, herbivory, fires, and erosion.

The homogeneity and age of peat deposits is of primary importance for studying the carbon accumulation dynamics in different intervals of the Holocene. Bogs provide widespread material for palaeoenvironmental analysis covering the Holocene (van Geel, 1978; Barber, 1981; Charman & Mäkilä, 2003; Barber *et al*., 2004). The peat deposits are mainly autochthonous, meaning materials originate within the peatland, and are relatively suitable for dating with radiocarbon, especially when mosses dominate the peat. There are several approaches to

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the reconstruction of past climates using peat deposits, including analysis of the variability in carbon accumulation rates. Increased accumulation rates may reflect periods with a more positive precipitation-evaporation (PE) balance (Mäkilä, 1997; 2001; Mäkilä *et al*., 2001). The surface wetness of all peatlands depends to some extent on PE, but ombrotrophic raised bogs are the only sites where there is no detectable influence of surface runoff or groundwater, and precipitation and evapotranspiration are the only components of the water balance, which is solely regulated by the climate (Charman & Mäkilä, 2003).

The main aims of this article are to examine the changes in rates of carbon accumulation of raised bogs throughout the Holocene, using examples from Finland, where the most detailed regional and site specific data are available, and to compare this information with other climate-sensitive peat properties. These include peat initiation dates, peat growth and humification, variability and variation in the composition of peat-forming plant species. These data are compared with palaeoclimatic records from peatlands and lake sediments in North-West Europe in order to detect the potential role of a regional, external forcing factor, i.e. climate. Comparisons are also drawn to the huge Siberian peatlands and permafrost as well as to North American mires.

1.2. Long-term rate of carbon accumulation in three raised bogs in southern Finland

Three raised bogs studied in southern Finland are used here as an example of carbon accumulation during the Holocene in a boreal environment: Haukkasuo, Kilpisuo and Pesänsuo (Figure 1.1). The initiation of peat accumulation in Finland, excluding Lapland and eastern Finland, depends on the date when the mire locality emerged from the Baltic basin. The result of this continuous isostatic land uplift is the formation of mires of different ages at different altitudes. The oldest raised bog used here as an example is the Haukkasuo bog (Figures 1.2 and 1.3), where peat accumulation started at 10 400 cal BP, followed by the Kilpisuo and Pesänsuo bogs at 10000 and 9200 cal BP, respectively (Figure 1.4). The transition from the minerotrophic, sedge-dominated mire to the ombrotrophic *Sphagnum fuscum* bog phase took place in Kilpisuo already at 9000 cal BP (Mäkilä, 2001), in Pesänsuo at 7300 cal

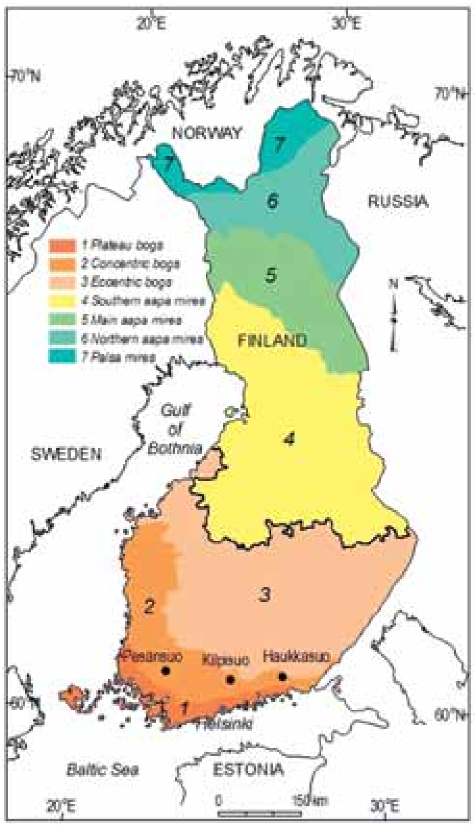

Figure 1.1. *Map showing the location of the study bogs and the regional distribution of the mire complex type regions in Finland according to Ruuhijärvi & Hosiaisluoma (1989). Raised bogs occur to the south of the black line (in regions 1-3) and aapa mires to the north (in regions 4-7).*

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Figure 1.2. *A ridge-hollow pine bog in the raised bog Haukkasuo. All photos Markku Mäkilä.*

BP (Ikonen, 1993) and Haukkasuo at 7000 cal BP (Mäkilä, 1997) (Figure 1.5).

In five peat profiles from the three study bogs, the long-term (apparent) rate of carbon accumulation (LARCA) (Clymo *et al*., 1998) averaged 27.3 g C m^{-2} yr^{-1}. The LARCA was 23.9 g C m^{-2} yr^{-1} for the *Sphagnum* peat section and 35.9 g C m^{-2} yr^{-1} for the *Carex* peat (Figure 1.6). Carbon accumulation is mostly correlated with vertical peat growth rate and to a lesser extent with dry matter content and carbon content of the peat (Ikonen, 1995; Mäkilä, 1997), and this also seems to be the case with our data.

The average carbon accumulation rates decreased from the values of 64 g C m^{-2} yr^{-1} during sedge peat formation to the lowest rates of 15 g C m^{-2} yr^{-1} between the period 5500-5400 cal BP. After this period and especially after 4350 cal BP, an

Figure 1.3. *The stratigraphy of the raised bog Haukkasuo.*

Figure 1.4. *A low-sedge bog in the raised bog Kilpisuo.*

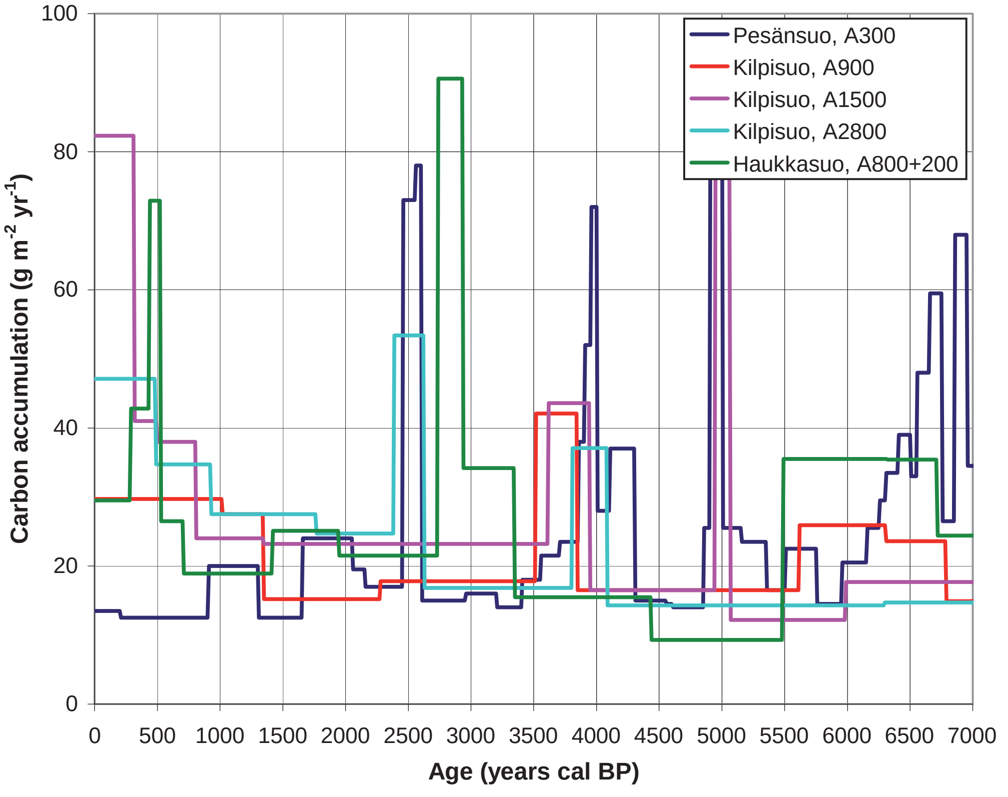

Figure 1.5. *Carbon accumulation rates in five dated profiles from raised bogs in southern Finland over the past 7000 years.*

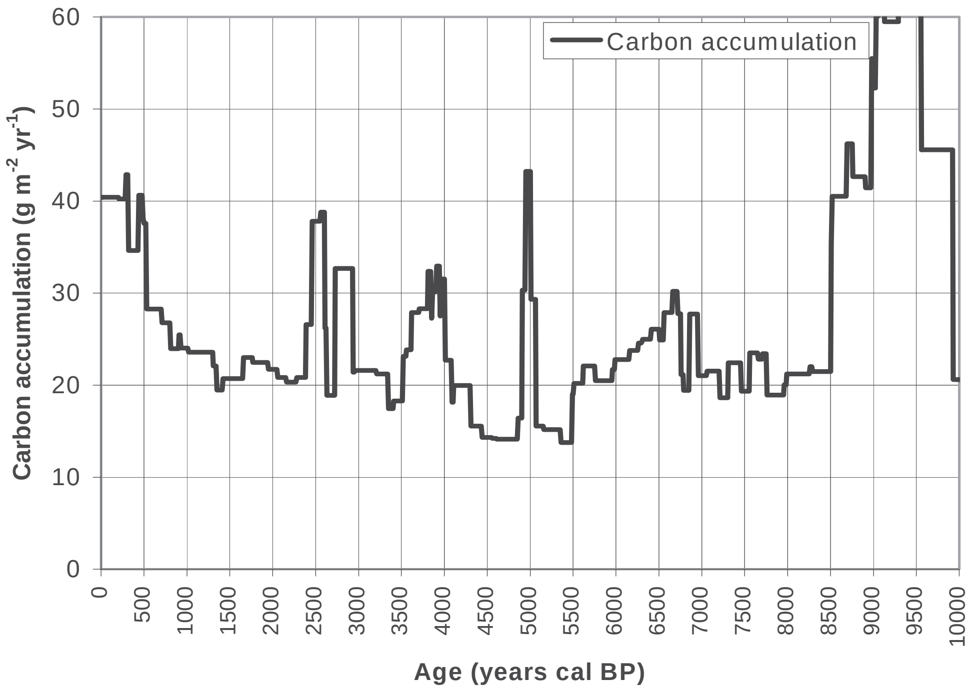

Figure 1.6. *Average rate of vertical carbon accumulation in five peat profiles from three study bogs during the Holocene.*

increasing trend in the carbon accumulation rate was recorded up to the present (Figure 1.6).

The prominent changes to higher carbon accumulation rates in *Sphagnum* peat were dated at 6750-6400, 5100-4950, 4100-3850, 2950-2750, 2650-2500 and 800 cal BP to the present. During these periods the rate of vertical peat growth exceeded the average rate, and the peat was mainly only slightly humified. Periods with low carbon accumulation rates in *Sphagnum* peat were recorded between 5500-5400, 4850-4500, 3500-3400 and 1600-1350 cal BP, when the rate of vertical peat growth was lower and the peat was more humified than the average. The carbon accumulation rates during the period 5500-5400 cal BP were about one half of the present rates. Major variations in carbon accumulation occur at approximately 1000-1500 year intervals.

Variations in peat accumulation rates in the Haukkasuo, Kilpisuo and Pesänsuo bogs do not clearly correlate with the dates of peat initiation in 210 sample series randomly derived from mires in different parts of Finland (Figure 1.7), indicating that the mire initiation is not climatically controlled on a local scale.

1.3. Lateral expansion of peatlands in Finland

Lateral spread of peatlands is also an important control on overall carbon accumulation rates. In southern and central Finland, lateral spread of peatlands occurred rapidly with around half of the total present area paludified by 8000 years BP (Mäkilä, 1997; Mäkilä et al., 2001). In the north, the spread was slower (Mäkilä & Moisanen, 2007). Local differences

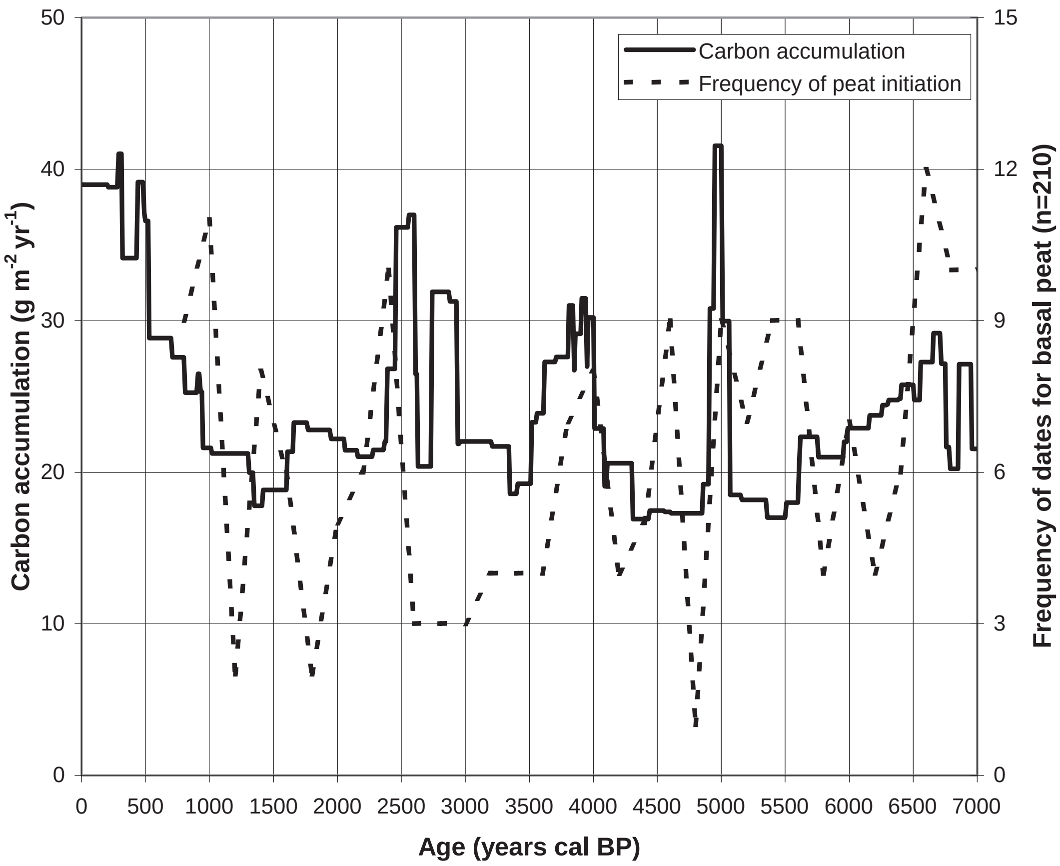

Figure 1.7. *Average rate of vertical carbon accumulation and the frequency of radiocarbon dates (210 in total) for basal peat during a 200-year period in different parts of Finland.*

in rates of mire expansion, and therefore carbon accumulation rates, are also affected by topography, which is a key control on the areas where a peatland can spread (e.g. Korhola, 1992). For Finnish mires in general, by the time they had attained their modern extent, they had accumulated over half of their modern carbon store: 55% of the carbon had formed before 4000 cal BP in a southern aapa mire Ruosuo, and 58% of the carbon before 5000 cal BP in a northern fen, Luovuoma, in Finnish Lapland (Mäkilä, 1997; Mäkilä *et al.,* 2001; Mäkilä & Moisanen, 2007).

1.4. Composition of peat forming plants versus carbon accumulation

The relationships between species composition, humification and carbon accumulation were examined in the data derived from Kilpisuo (Mäkilä *et al*., 2003). No statistically significant relationships were found between the species composition and variation in the peat accumulation rate (p = 0.14) or between species composition and carbon accumulation (p = 0.34). However, species composition explained 41% of

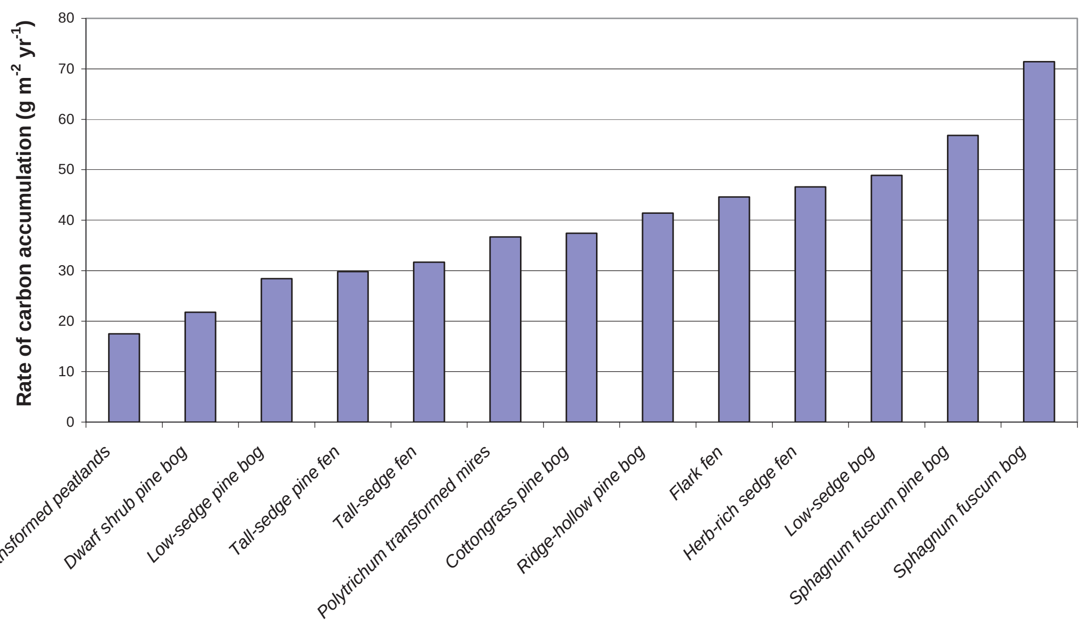

Figure 1.8. *Relation of mire site types to average carbon accumulation rate with layers younger than 300 years.*

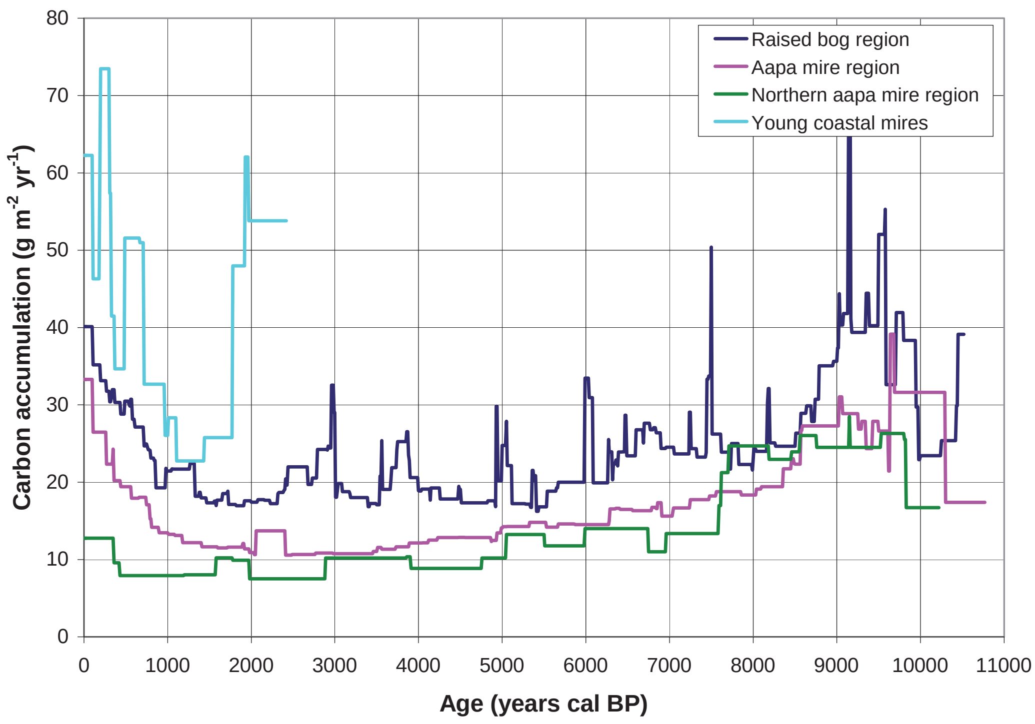

Figure 1.9. *Carbon accumulation rates in raised bog regions, aapa mire regions and coastal mires.*

variation in peat humification, which is statistically significant. Understandably, this relationship is of statistical rather than causal nature, because the same environmental conditions affect both humification and vegetation. There was a clear negative correlation (r_s = -0.49, *p*= <0.001) between carbon accumulation and peat humification. Carbon accumulation is therefore clearly controlled by humification (Mäkilä *et al*., 2003).

The reconstruction of subsoil plant communities provides a reasonably reliable method to identify changes in palaeohydrological conditions in the course of mire development. Based on the Kilpisuo macrofossil and pollen diagram (Mäkilä *et al*., 2003) it is evident that at 5500-5100 cal BP a transition to a moist regime and a change from *Eriophorum-Sphagnum* peat with dwarf shrub remains to *Sphagnum* section *Cuspidata* with *Scheutzeria palustris* are mutually concurrent with the arrival and establishment of *Picea,* an indicator of moister and cooler climate. Aartolahti (1966) suggested that the arrival of *Picea* might have been hindered earlier by dry summers. This period represents the first hummock and hollow phase in the development of Kilpisuo bog. The other transition at around 4500 cal BP to a moist regime, deduced from the pollen diagram at the Haukkasuo site, coincides with a change from *Eriophorum-Sphagnum* (section *Acutifolia*) peat to a hollow-inhabiting species of *Scheutzeria-Sphagnum* (section *Cuspidata*) (Eriksson, 2003**)**. These hollow phases are not directly dependent on climatic conditions. They are a phase in a bog development resulting from growth, which at a sufficient gradient leads to the formation of ridges (dry elongated hummocks).

The correlation between mire site types based on the mire vegetation and the average true or actual rates of carbon accumulation (ARCA) in layers younger than 300 years in Finnish mires can be seen in Figure 1.8. The highest accumulation rates in layers younger than 300 years were measured in the ombrotrophic mire site types (*Sphagnum fuscum* bog and *Sphagnum fuscum* pine bog; Mäkilä & Goslar, 2008). Wet oligotrophic and minerotrophic treeless mire site types come next. The lowest carbon accumulation was found in the most transformed, sparsely forested and forested mire site types. These mires have the lowest water table. The high carbon accumulation in the surface layers is temporary and related mainly to the development of the mire. This is due to the fact that only a small amount of the organic matter in the uppermost layers has had time to decay (Mäkilä & Goslar, 2008).

The highest carbon accumulation rates were found in young *Sphagnum* bogs in coastal mire regions where the uplift rate is the highest in Finland. Moisture, especially its temporal distribution, is the main factor controlling *Sphagnum* production (Backeus, 1988). Thus, both the amount of precipitation and the distance to the groundwater level are important for *Sphagnum* production. However, other climatic factors (e.g. mean annual growing season temperature and growing degree-days) have also been shown to correlate to moss growth (Thormann & Bayley, 1997). The fact that carbon accumulation rates are higher in coastal *Sphagnum* bogs than in older raised bogs is not only due to climate, but also because coastal mires are in the early phase of their development. This kind of young bog produces more moss on the surface and the amount of peat decayed and compacted in the entire bog is lower than in an old bog (Johnson *et al*., 1990).

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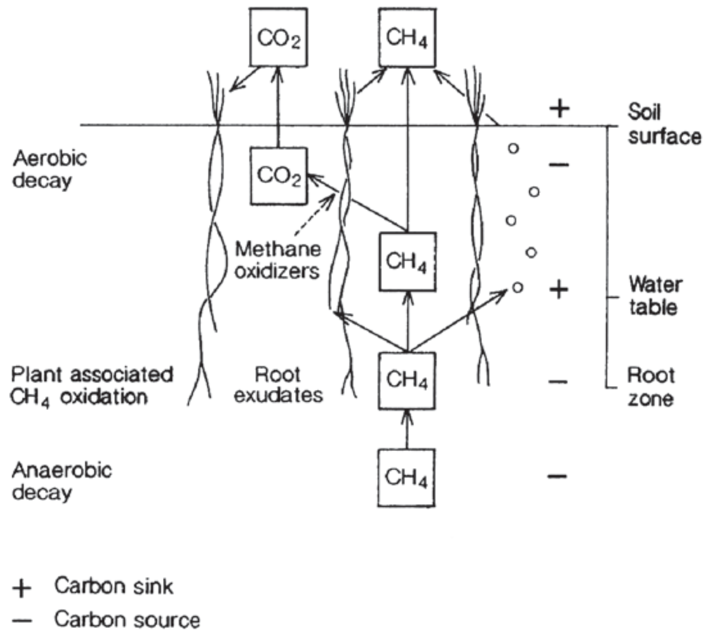

Figure 1.10. *Schematic presentation of the presence of carbon sinks and sources in a wet minerotrophic mire based on profile P20 from Ruosuo (Mäkilä et al., 2001; modified from Kettunen & Kaitala, 1996).*

Figure 1.11. *A flark fen in the southern aapa mire Ruosuo showing the lawn string with wet flark.*

Figure 1.12. *A flark fen in the northern aapa mire Luovuoma. Flarks and strings are the most essential elements of the mire centre.*

1.5. Carbon accumulation in aapa mires

The rate of carbon accumulation is slower in minerotrophic aapa mires than in raised bogs (Figure 1.9). Oxic decay is more efficient in aapa mires, which receive nutrients and oxygenated water from adjacent mineral soils, whereas ombrotrophic raised bogs are fed only by rain water (e.g. Damman, 1996). In minerotrophic aapa mires, oxygen is transported into the peat via the roots of sedges, where it contributes to the decay of peat layers (Figure 1.10). Most old aapa mires are approaching the stage where primary production does not adequately compensate for overall carbon loss (Mäkilä *et al*., 2001; Mäkilä & Moisanen, 2007) (Figures 1.11 & 1.12). Carbon accumulation during the Holocene in Ruosuo, an aapa mire in north-central Finland was studied in detail by Mäkilä *et al*., (2001). It serves as an example of the extensive circumpolar minerotrophic sedge fens. Most of Ruosuo is obviously approaching the stage in its vertical growth where primary production does not adequately compensate for the loss of carbon from the entire peat sequence. The declining net carbon accumulation rates from the base up to the sub-surface may indicate a real delayed trend in net carbon accumulation rates, although it may not appear to do so when the wet surface peat is considered. Although the aapa mires continue to bind carbon, the increase in the decay of the whole peat mass of mires and the fast decay of surface layers have further promoted an increase in gas flux. Therefore, under the present climatic conditions the northern circumpolar wet aapa mires are significant sources of greenhouse gases (Figures 1.11 & 1.12). In addition to high methane production, northern aapa mires also have lower carbon accumulation rates than more southern raised bogs.

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Figure 1.13. *Typical flat palsa landscape (Forest Tundra – SubTundra) close to the Polar circle in Western Siberia. Numerous round thermokarst lakes, flat white lichen covered palsa hummocks with shallow permafrost and fens (green) with permafrost over 70 cm below the surface (or without permafrost) are shown. Photo by Markku Mäkilä; interpretation by Wladimir Bleuten.*

1.6. Siberian and Canadian peatlands

About 60% of the world's peatlands are found in Russia, particularly in Siberia. These peatlands cover an area of more than 300 million ha (Bleuten *et al*., 2006). Most mires in the region started to form about 11,000 years ago, and they have played an important role in the biospheric carbon cycle throughout the Holocene (e.g. Smith *et al*., 2004). "Several mire zones occur in Siberia. From the north to south they are the arctic mineral sedge mires, a zone of flat palsa (a peat bog with an ice core), followed by the domed palsa zone (Figure 1.13). Next is the most abundant peatland type, namely the domed raised bogs with pools and ridges. South of this zone most of the mires are formed by reeds and tall sedges with salt mires occurring in the southernmost part of West Siberia and Kazakhstan. Perhaps, the largest single raised bog in the world, covering some 5.16 million ha is found in West Siberia at Vasuganskoe. Peat accumulation during the Holocene increases from the cold north with frozen bogs, to the south below the permafrost zone by a factor of 3 to 5. In contrast, no significant differences in actual peat accumulation rates could be established between the north and south parts of the West Siberian peatlands" (from The Web site of the Irish Peatland Conservation Council, 2000; available at http://www.ipcc.ie/wpsiberia.html).

Peatlands of Canada cover approximately 113.6 million ha, or 12% of the Canadian land area (Tarnocai, 2006). Most of the peatlands (97%) occur in the Boreal Wetland Region (64%) and Subarctic Wetland Region (33%). These peatlands contain approximately 147 million tonnes of soil carbon, which is about 56% of the organic carbon stored in all Canadian soils. In Canada mean long-term rates of peat accumulation are higher in boreal and temperate peat deposits than in subarctic regions, although peatlands are extensive in the subarctic. Climatic change probably accounts for the development of southern peatlands during the middle to late Holocene and for the late Holocene decline in growth of many subarctic peat deposits (Ovenden, 1990). Tarnocai (2006) suggests that the current climate warming will greatly affect the carbon balance of subarctic and arctic peatlands in Canada as the thawing of permafrost will release great quantities of carbon (see also Zoltai, 1994).

The history of permafrost in Siberia plays a significant role in the history of carbon accumulation in mires, too. The continental ice did not cover the West Siberian plain during the last glaciation, therefore permafrost penetrated deep into the ground

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(Hubberten *et al*., 2004 and references herein). At the end of the glaciation 11,000 years ago, the climate warmed rapidly by more than 10ºC and permafrost started to thaw in millions of square kilometres. This thawing produced countless numbers of thermokarst lakes, which still exist in the Siberian coastal areas. In the warm early and mid postglacial (Holocene) climate, at least 2-4°C warmer that today, trees could penetrate to the Arctic coast of Siberia several hundred kilometres north of their present northern limits (Khotinsky, 1984; MacDonald *et al*., 2000; Kultti, 2004). A similar early Holocene northward forest expansion and later retreat has been worked out in Arctic Canada (e.g. Ritchie, 1987). The warm climate is attributed to strong summer insolation whereas the expansion of forests resulted in reduced surface albedo, which also contributed to climate warming. The cooling of the climate began approximately 6000 years ago. Northern tree lines started to retreat and reached their present position 3000 to 4000 years ago, and permafrost began to form in unfrozen areas.

The formation of permafrost has continued until very recently, and the melting currently observed takes place mainly in those areas where the permafrost is most recent. The fluctuations of permafrost and climate can also be seen in polygenetic ice-wedges, which are exposed along riverbanks all over Siberia in the areas where permafrost is present. Thus, the climate in Siberia has fluctuated between cool and wet and warm and dry during the Holocene, and these fluctuations have been both sudden and of a magnitude at least similar to those observed recently. In the warming climate the northward expanding forests act increasingly as carbon sink, thus in the opposite direction than the melting of permafrost in terms of carbon balance, which is therefore the most difficult to quantify.

Borren (2007) has analysed in detail peat cores from several sites in the southern taiga (boreal region) of Western Siberia. He found that Holocene peat growth and carbon accumulation at different locations were different, especially during the early Holocene time, and caused by variations in vegetation succession. These differences were strongly influenced by regional and local hydrology. Therefore, the effect of climate fluctuation on mire development varied from place to place. The indirect effects of climate change through local hydrology appeared to be more important than direct influences of changes in precipitation and temperature.

At present, western Siberian mires form a significant sink for greenhouse gases (Borren, 2007). The Holocene carbon accumulation rate in Western Siberia shows a spatial variation of 10-85 g m^{-2} yr^{-1}, with an average of 16.2 g m^{-2} yr^{-1}. For comparison, in Finland the Holocene carbon accumulation rate averages 19.8 g m^{-2} yr^{-1} in raised bog region and 14.6 g m^{-2} yr^{-1} in the aapa mire region (Mäkilä & Goslar, 2008), close to the Siberian rates. The mean peat accumulation rate in 32 sites from Alaska to Newfoundland is 50 g m^{-2} yr^{-1} and if we assume that the carbon content of North American peat is approximately 50% as it is in Finland, then the average carbon accumulation rate is 25 g m^{-2} yr^{-1}, with a broad range of 8 to 40 g m^{-2} yr^{-1} (Gorham *et al*., 2003). Long-term net rates of carbon accumulation in Canadian peatlands typically range from 10 to 35 g C/m^2/yr according to Ovenden (1990).

1.7. Carbon accumulation versus climate change

Observations from modern mires show that they are sensitive to changes in the hydrological regime of an area (e.g. Ruuhijärvi, 1983; Charman *et al*., 2004,

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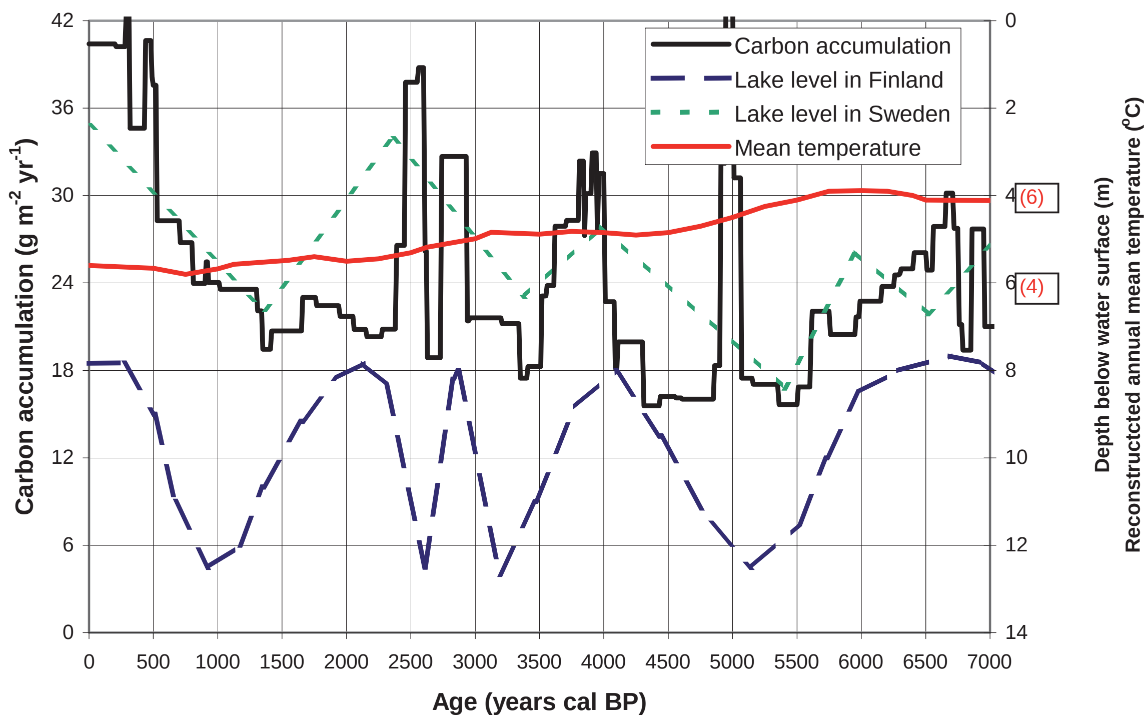

Figure 1.14. *Average rate of carbon accumulation and lake-level fluctuations in Finland and in Sweden. The green line indicates modified Holocene lake-level fluctuations according to Digerfeldt (1988) and blue line the fluctuations according to Sarmaja-Korjonen (2001). Lake-level positions are expressed only as high or low by Sarmaja-Korjonen (2001). The quantitative annual mean temperature reconstruction based on pollen is also shown with the red line and numbers (°C) (Heikkilä & Seppä, 2003).*

Barber *et al*., 2004). Therefore, and based on the data recorded in peat deposits, it is appropriate to assume that this was the case throughout the Holocene period. Summer precipitation is the main factor affecting reconstructed water table variability, but winter precipitation (snow) can also have an effect on reconstructed water tables. Snow melts rapidly in the spring, filling hollows with water and causing floods.

Heikkilä and Seppä (2003) presented a quantitative annual mean temperature reconstruction based on a high-resolution pollen stratigraphy derived about 100 km to the north of our bogs in southern Finland, using a pollen-climate calibration model with a cross-validated prediction error of 0.9°C. According to their model, the annual mean temperature peaked 8000-4500 cal BP, the so-called Holocene thermal maximum period, with particularly high temperatures (2.0-1.5 °C higher than at present) at 8000-5800 cal BP. From 4500 cal BP to the present, the reconstructed annual mean temperature gradually decreased by about 1.5 °C (Figure 1.14). According to Hyvärinen (1976) and Eronen *et al*. (1999), the general cooling in northern Fennoscandia began about 5750 cal BP, as revealed by studies on pollen influx values and subfossil pines in the tree line area. However, it is not appropriate to use temperature alone to describe carbon accumulation, as can be seen in Figure 1.14. Wetness records of peatland surfaces should be interpreted as primarily reflecting summer precipitation variability, with summer temperature increasingly important in more continental bogs (e.g. Haukkasuo) (cf. Charman *et al*., 2004).

A comparison of variations in the carbon accumulation rates in bogs with lake-level fluctuations in southern Sweden (Digerfeldt, 1988) and in southern Finland 50 kilometres to the south-west of Kilpisuo bog (Sarmaja-Korjonen, 2001) reveals quite a good correlation over the past 7000 years, except from 5100-4950 cal BP. A possible explanation for this deviation may be that the resolution of the water level data is crude (Figure 1.14). According to Digerfeldt (1988), a shift to drier conditions in southern Sweden occurred at about 5500 cal BP. The forest composition in south-central Sweden indicates a change from warm and dry conditions to a cooler and wetter climate 4000–3500 cal BP (Jessen, 2006), when carbon accumulation in southern Finnish raised bogs was exceptionally high. Another distinct low in lake levels is recorded between 1600 and 1100 years cal BP when carbon accumulation was also slow in bogs. According to Sarmaja-Korjonen (2001), the period from around 4500 cal BP to modern times is characterized by several short-term fluctuations in water levels.

In the Dümmer basin of NW Germany, the pine forest at Campemoor was replaced by peat when raised bog formation began approximately 5000 cal BP. This indicates the climate became more humid (Leuschner *et al*., 2007), as is also shown in the Finnish raised bogs. In Great Britain, the frequency and severity of floods similarly increased about 5000 cal BP (Macklin, 2005).

Variations in carbon accumulation in the study bogs correlate quite well with lake-level fluctuations, interpreted as representing humid or dry climate (fluctuations in humidity). The climate in both southern Sweden and southern Finland is to a great extent controlled by low pressure belts originating in the North Atlantic (i.e. the North Atlantic Oscillation, NAO) (e.g. Hurrell, 1995; Luterbacher *et al*., 2002), therefore a comparison of climate proxies of these areas is justified.

Variations in peat accumulation rates in the study bogs are not clearly correlated with the dates of peat initiation in the 210 samples randomly derived from mires in different parts of Finland (Figure 1.7). It seems that local environmental factors, particularly topography, have influenced lateral mire growth to a greater extent than the climate (e.g. Korhola, 1992; Mäkilä, 1997).

Quantitative analyses of plant macrofossil remains have been performed on three peat profiles from raised bogs in Ireland and England (Barber *et al*., 2003). The reconstructed vegetation in each level of the profiles was related to changing moisture conditions on the bog surface, and since the bogs are ombrotrophic, these changes were interpreted in terms of changing climate. Thus, prominent changes in the climate coincident with wetter conditions on the bog surfaces have been dated in at least two of the profiles at approximately 4400-4000, 1750, 1400 and 1000 cal BP, and in all three profiles at 3200, 2750-2350, 2250 and around 700 cal BP. Barber *et al*. (2003) also listed shifts to wet conditions from bogs all across Europe, which also cluster around the dates identified in their study. The prominent carbon accumulation peaks at around 4000, 2750 and 800 cal BP are synchronous with the peak identified in Finnish bogs described above. However, the other peaks were not recognized in Finland, which may indicate greater

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variability in the oceanic climate of the British Isles. Barber *et al*. (2004) found periodicities of around 1100, 800, 600 and 200 years, amongst others, from the macrofossil record in peat bogs, and they propose a possible link to solar forcing and oceanic changes.

The results of Yu *et al*. (2003) in continental western Canada suggest that peatland carbon sequestration rates are highly sensitive even to minor climatic fluctuations, which are too small to produce detectable changes in the major plant species compositions of the peatland. Wieder (2000) reported for eight North American *Sphagnum* peatlands that, with increasing temperature and/or increasing precipitation, net primary production, decay in the upper 30 cm of peat, and turnover of photosynthetically fixed carbon increase but net carbon accumulation as peat is unaffected. Wieder further suggests that model predictions indicate that only small decreases in net primary production and/or increases in decay rates (10% over 100 years) could be sufficient to switch boreal peatlands from net sinks of atmospheric carbon to net sources.

A sharp rise in the ^{14}C content of the atmosphere at around 2850 cal BP coincided with indications of an abrupt change in the climate from relatively warm and dry to cooler and wetter conditions, suggesting that climate changes and solar activity are interrelated (van Geel *et al*., 1998; van Geel, 2006**).** Mauquoy *et al*. (2002) suggested, based on evidence from northwest European bogs, that the 'Little Ice Age' climate changes were driven by variations in solar activity. However, this relationship may not be straightforward since the changes in ^{14}C production rate and climate are not exactly synchronous (e. g. Muscheler *et al*., 2004). A rapid increase in carbon accumulation in eastern bogs occurs at around the same time.

The carbon sequestration rates of the raised bogs in our study correlate quite well with Holocene climate records from the North Atlantic (Bond *et al*., 2001). As stated above, the current Finnish climate is largely controlled by the NAO, and this has obviously been the case throughout the Holocene. When the North Atlantic climate was wet and warm, the climate in southern Finland was wet and cool. Bond *et al*. (1997; 2001) found a close correlation between solar proxies (production rates of cosmogenic nuclides ^{14}C and ^{10}Be) and changes in a proxy of drift ice in the North Atlantic (the percentage of hematite-stained grains in North Atlantic cores), which indicate that the Holocene climate has varied with quasi-regularity on a millennial scale. They proposed that atmospheric changes induced by variations in solar irradiance were amplified and transmitted through their impact on sea ice and North Atlantic thermohaline circulation. Our records indicate that these Bond cycles are perhaps important on a hemispheric scale, whatever their possible cause might be.

The stratigraphy of raised bogs suggests that carbon exchange and accumulation in mires have always been sensitive to climatic fluctuations, which have been characteristic of the entire Holocene (Mäkilä & Saarnisto, 2005). A marked decline in carbon accumulation rates may indicate a relatively dry and warm climate. The leveling out and subsequent increase in carbon accumulation rates after 4500 cal BP in raised bog regions indicate not only the development of *Sphagnum*-dominated plant associations, but also a change towards a more humid climate (Mäkilä & Saarnisto, 2005). During humid periods, the decomposition rate was lower and thus net carbon accumulation was higher than usual. Increased accumulation rates reflect periods with a more positive precipitation-evaporation ratio.

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It should be emphasised, however, that climate is not always the major factor affecting the development of mires. For example, the differential rate of isostatic uplift across the Hudson Bay Lowlands in Canada has been the principal parameter of extensive peatland development (Glaser *et al*., 2004), rather than climate. The emergence of Finland from the Baltic basin waters during the Holocene is the result of isostatic land uplift, which has thus led to mire initiation. Otherwise, the differential uplift has played only a minor role in mire development, even in the most rapidly uplifting western coastal regions.

1.8. Conclusions

The natural succession of mires and local environmental factors may obscure the regional relationships between climatic factors and the observed stratigraphical and hydrological changes. Local environmental (i.e. hydrological, topographical and edaphic) factors have been important, especially during the sedge-dominated stage in the early development of mires.

Although the vegetation inferred from each peat profile reflects changing wetness on the ombrotrophic bog surface, humification changes are more sensitive to climatic fluctuations then are changes in plant species composition. Evidence of long-term changes in climate appears in the stratigraphy of ombrotrophic bogs as layers of more or less humified peat. During humid periods, the degree of humification is lower and thus carbon accumulation is higher than usual. Increased accumulation rates reflect periods with a more positive precipitation-evaporation ratio.

When the carbon accumulation rates, dates of peat initiation, lake-level fluctuations in Sweden and in Finland and Holocene climate records from the North Atlantic are compared, they demonstrate that climatic fluctuations, interpreted as representing humid and dry climatic conditions, are quite well correlated with the variations in carbon accumulation. Major variations in carbon accumulation occur in intervals of 1000-1500 years, which correspond to the Bond cycles in the North Atlantic, and are possibly related to changes in solar irradiance. Prominent changes to high carbon accumulation rates in *Sphagnum* peat were dated at 6750-6400, 5100-4950, 4100-3850, 2950-2750, 2650-2500 and 800 cal BP. These periods can be seen in many palaeoclimatic records, and fluctuations in solar irradiance have often been invoked as an explanation for the changes (e.g. Bond *et al*., 2001; Mayewski *et al*., 2004, Solanki *et al*., 2004).

Raised bogs act as large sinks of atmospheric CO_2 during these periods. The natural carbon accumulation rates during the warm and dry periods of the Holocene were about one half of the present rates, indicating the magnitude of climate fluctuations. Thus, potential future climate warming, especially the drying of peatlands, may significantly affect peat carbon sequestration in raised bogs and lead to an increased release of carbon. Although carbon accumulation has varied considerably, these variations are significant for the carbon balance throughout the extensive boreal raised bog region and thus for the carbon balance on a hemispheric scale.

Under the present climate conditions, the boreal raised bogs will act a carbon sink, whereas the northern circumpolar wet aapa mires are significant sources of greenhouse gases. In addition to high methane production, northern aapa mires also have lower carbon accumulation rates than more southern raised bogs. In the calculations of the carbon balance of peatlands, these differences should be taken into account as well as long-term natural variations in their carbon accumulation, which reflect climate

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change. Melting of permafrost in extensive peatlands of Arctic North America and Eurasia will release greenhouse gases, but this will be compensated by the expansion of forests northward into tundra as was the case during the early Holocene warm climate period. The interplay between thawing permafrost and expanding forests and vice versa is inadequately known in terms of carbon balance.

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CHAPTER 2:

NORTHERN PEATLANDS, GREENHOUSE GAS EXCHANGE AND CLIMATE CHANGE

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2.1. Introduction

Peatlands play important roles in the global cycling of carbon (C) as they are net sinks of atmospheric carbon dioxide (CO_2) and a large source of atmospheric methane (CH_4) (Gorham, 1991; 1995). Northern peatlands store between 200 and 450 x 10^{15} g C (Gorham, 1991; Turunen *et al*., 2002). This is equivalent to approximately 10-30% of global soil C stocks and between 30 and 75% of the pre-industrial mass of C stored in the atmosphere.

Conversely, the post glacial development of northern peatlands may have led to an increase in CH_4 emissions as northern peatlands also represent one of the largest natural sources of atmospheric CH_4. Northern peatlands release an estimated 10 to 65 x 10^{12} g CH_4 per year (Walter *et al*., 2001; Mikaloff Fletcher *et al*., 2004) accounting for 4-25% of global wetland CH_4 emissions which make up ~25% of the total source of atmospheric CH_4 (Lelieveld *et al*., 1998).

Peatlands are also important sources of dissolved organic carbon (DOC) to downstream ecosystems with the export of DOC from watersheds found to be related to the proportion of watershed area covered by peatlands and mean annual air temperature (Frey & Smith, 2005). Therefore, warming has the potential to greatly increase the export of DOC to oceans.

Atmospheric C concentrations are predicted to change as peatlands respond to climate change, either through enhanced atmospheric global warming potential due to increased CO_2 and CH_4 losses from peatlands, or through C assimilation resulting in the mitigation of warming due to increased ecosystem productivity (McGuire *et al*., 2000). Many boreal and subarctic regions already are experiencing substantial changes in climate (Serreze *et al.*, 2000) that have resulted in longer and drier growing seasons, increased fire activity, and the degradation of permafrost. The fate of peatland C stocks, in part,

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depends on the response of shallow permafrost and fire regimes to changing climate regimes.

In this chapter we consider the role of peatlands in the natural climate system by examining the controls on greenhouse gas (GHG) exchange (CO_2 and CH_4). Discussion of climate change impacts will focus on seasonally frozen northern peatlands with special consideration of the effects of melting permafrost and fire disturbance in section 2.5.4. We also present a case study of a multi-year investigation of contemporary GHG emissions and conclude with a discussion on how these peatlands may respond to climate change and also present a Canadian case study investigating the effect of water table drawdown on GHG exchange.

2.2. Peatlands and peat

Peatland formation commonly leads to a two-layered, or diplotelmic, structure within the peat profile referred to as the acrotelm and catotelm (Ingram, 1978; 1992). The acrotelm is the upper layer of peat where water table fluctuations occur. This zone is characterized by a high organic matter content that is poorly decomposed with high specific yield and porosity. In natural peatlands, the large pore structure of the acrotelm contributes to a large water storage capacity, particularly specific yield, which aids in limiting water table fluctuations to the near surface zone. Specific yield is the ratio of the volume of water yielded by gravity drainage to the volume of the soil; because of the large pore size in the acrotelm, specific yield is high due to the ability of pores to drain with relative ease by gravity. Similarly, hydraulic conductivity, the rate at which water moves through the peat, in the acrotelm is fast and generally slows with depth (Boelter, 1965). In contrast, the catotelm is the lower and deeper layer of peat that is permanently water saturated and contains highly decomposed organic material. Since catotelmic peat is degraded and consists of smaller pores, less water can be drained by gravity. This results in low specific yield, and therefore greater water retention. As a result, even though total porosity remains high, small pore size leads to much slower hydraulic conductivity (e.g. Boelter, 1965). Consequently, the structural differences of the acrotelm and the catotelm are important in determining the storage of water in bog systems (i.e. Romanov, 1968) and since gas diffusion is largely controlled by air-filled porosity, by extension the exchange of GHGs.

The high water storage capacity of the acrotelm and its ability to shrink and swell, act as regulatory functions that minimizes water table fluctuations, maintaining the water table close to the surface (Ingram, 1983). In a natural peatland, water table position generally remains within the upper metre, maintaining moist surface conditions. Furthermore, it is suggested that the hydrological function of a peatland is controlled by the structure and deformable character of the peat matrix (i.e. subsidence) (Price, 2003) where changes in peat volume can result in changes in hydraulic parameters that govern water retention and flow. More specifically, bulk density, water retention, and hydraulic conductivity may be altered due to seasonal changes in peat surface elevation. Therefore the diplotelmic structure is important in terms of both water transport and water storage where disturbance to this structure can cause significant changes to the hydrological functions.

2.3. Natural peatlands: Greenhouse gas source or sink?

The change in storage of C is determined by the balance between primary production (photosynthesis) and decomposition

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(Clymo, 1984). The contemporary peatland C balance can be represented by:

$$\Delta C = -(NEE + F_{CH4} + F_{DOC} + F_{DIC} + F_{POC})$$

where ΔC represents the net change in C storage in the peatland ecosystem, NEE is the net flux of carbon as CO_2 from the ecosystem to the atmosphere, F_{CH4} is net methane flux (positive if net flux is from the peatland to the atmosphere) and F_{DOC}, F_{DIC} and F_{POC} are the net waterborne exchanges of dissolved organic carbon, dissolved inorganic carbon and particulate organic carbon, respectively (see also Chapin *et al*., 2006). If these waterborne fluxes result in a net loss of carbon from the peatland, then their signs would be positive. Of the waterborne fluxes, only DOC will be discussed in detail in this chapter as it has been observed to be the dominant hydrologic loss of C from peatlands (e.g. Dawson *et al*., 2004).

Table 2.1. *Some Natural Peatland Carbon Fluxes*[a]

Peatland type and location		Reference
Net CO_2 exchange (NEE)	***g CO_2 m^{-2} yr^{-1}***	
Raised bog (Canada)	-7 to -411	Roulet *et al*., 2007
Raised bog (Sweden)	-7 to -37	Waddington & Roulet, 2000
Raised bog (Siberia)	-79 to 132	Arneth *et al*., 2002
Patterned blanket bog (Ireland)	-179 to -223	Sottocornola & Kiely, 2005
Sphagnum-sedge-pine fen (Finland)	-359	Alm *et al*., 1997
Subarctic peatland (Russia)		
Wet hollow	-62	Heikkinen *et al*., 2002
Intermediate hollow	-158	
Wet lawn	-147	
Intermediate lawn	-110	
Hummock	11	
Subarctic peatland (Finland)		
Palsa top	-19 to -53	Nykänen *et al*., 2003
Palsa margin	-62 to -154	
Thermokarst wetland	-71 to -94	
Subarctic peatland (Alaska)		
Palsa top[b]	312	Wickland *et al*., 2006
Palsa margin	191	
Thermokarst wetland	134	

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Net CH_4 exchange (F_{ch4})	***g CH_4 m^{-2} yr^{-1}***	
General peatlands (Canada)	2 to 3	Bridgham *et al*., 2006
Patterned blanket bog (Ireland)	6	Laine *et al*., 2007b
Boreal mires (Sweden)		
Tall-sedge fen	4 to 40	Nilsson *et al*., 2001
Low-sedge fen	4 to 18	
Transitional fen	2 to 3	
Nutrient poor mire (Sphagnum – Andromeda)	3 to 10	
Raised bog (Canada)	4 to 6	Roulet *et al*., 2007
Raised bog (Sweden)	5	Waddington & Roulet, 2000
Subarctic peatland (Russia)		
Wet hollow	16	Heikkinen *et al*., 2002
Intermediate hollow	8	
Wet lawn	6	
Intermediate lawn	2	
Hummock	2	
Subarctic peatland (Finland)		
Palsa top	1	Nykänen *et al*., 2003
Palsa margin	20 to 33	
Thermokarst wetland	12	
Subarctic peatland (Alaska)		
Palsa top	2	Wickland *et al*., 2006
Palsa margin	2	
Thermokarst wetland	31	
Dissolved Organic Carbon (DOC) export (F_{DOC})	***g C m^{-2} yr^{-1}***	
Raised bog (Canada)	13 to 21	Roulet *et al*., 2007
Raised bog (Sweden)	4.2 to 6.7[c]	Waddington & Roulet, 2000
Upland peat complex (UK)	8 to 17	Dawson *et al*., 2002
Temperate poor fen (New Hampshire, U.S.)	3.4	Carroll & Crill, 1997

[a] A positive value represents a flux from the ecosystem to the atmosphere.
[b] Carbon uptake by trees not included and may offset estimated CO_2 efflux.
[c] This estimate includes DIC (dissolved inorganic carbon) and DOC.

Carbon dioxide (CO_2) is fixed by plants via the process of photosynthesis (gross ecosystem photosynthesis, GEP) and then allocated to plant biomass through autotrophic (plant) respiration, returning some CO_2 to the atmosphere. CO_2 is also lost from heterotrophic respiration derived from the decomposition of organic matter mainly by microbes such as bacteria and fungi. Ecosystem respiration (ER) includes soil (heterotrophic), and plant and root (autotrophic) respiration and is dependent on the amount of labile, or easily decomposable material, soil temperature, and soil moisture content. Thus, the total net ecosystem exchange (NEE) is a measure of the difference between CO_2 uptake by GEP and ER. The other components of the C balance include CH_4 and net export of waterborne C (DOC + DIC + POC). Each of the components of the carbon balance are discussed separately below as well as brief mention of the exchange of nitrous oxide (N_2O) another important GHG.

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Gorham (1995) estimated the contribution of the different components of the C balance of northern peatlands (Figure 2.1a). Specifically, this estimate suggests that peatlands emit ~4 g CH_4-C m^{-2} yr^{-1} (5.3 g CH_4 m^{-2} yr^{-1}) and are a net sink of atmospheric CO_2 of ~ 47 g CO_2-C m^{-2} yr^{-1} (172 g CO_2 m^{-2} yr^{-1}). More recently, Saarnio *et al*. (2007) estimate that ombrotrophic peatlands are a net sink of 15 ± 53 g CO_2-C m^{-2} yr^{-1} and source of 5 ± 4 g CH_4-C m^{-2} yr^{-1}, while minerotrophic peatlands emit 15 ± 63 g CO_2-C m^{-2} yr^{-1} and 13 ± 10 g CH_4-C m^{-2} yr^{-1}, respectively. Even though the C balance and relative magnitudes of CO_2 and CH_4 exchange vary within and among peatlands (Table 2.1), a recent multi-year study by Roulet *et al*. (2007) shows very similar results to the Gorham (1995) estimates (Box 2.1, Figure 2.1b). A discussion of the processes that affect the C balance and GHG exchange in peatlands follows.

Box 2.1. The Mer Bleue Case study: contemporary peatland carbon exchange

To determine the contemporary peatland-atmosphere GHG exchange a multi-year study at the Mer Bleue peatland near Ottawa, Canada was undertaken. The Mer Bleue study is one of a few studies that have measured all of the components of the carbon budget of a peatland since May 1, 1998. Mer Bleue is a 28 km^2, raised shrub bog with characteristic hummock–hollow microtopography. A 6-year balance was determined from continuous net ecosystem CO_2 exchange (NEE), regular instantaneous measurements of methane (CH_4) emissions, and export of dissolved organic C (DOC) from the peatland (Roulet *et al*., 2007). The mean exchange of CO_2-C, CH_4-C, and DOC export were -40.2 ± 40.5 (± 1 standard deviation), 3.7 ± 0.5, and 14.9± 3.1 g m^{-2} yr^{-1} which represents a mean carbon uptake of 21.5 ± 39.0 g m^{-2} yr^{-1} (sign convention: negative represents uptake by the peatland, positive represents a source to the atmosphere; Figure 2.1b). The carbon storage is very similar to Gorham's (1991) estimate (Figure 2.1a) but there were considerable inter-seasonal and inter-annual variations leading to the large standard deviations around the means. These variations were in response to varying temperature and moisture conditions in all four seasons over the six years. A simple analysis on uncertainty indicates that the net carbon balance could range from an uptake of 105 g m^{-2} yr^{-1} to a net loss 50 g $m^{-2}.yr^{-1}$. Based on this analysis, Roulet *et al*. (2007) concluded a significantly longer record of measurement would be required to significantly reduce the observed variance and to explain what controls the variance. The Mer Bleue bog has been in its current ecological form – i.e. a shrub bog, for approximately the last 3,000 years. The analysis of the carbon uptake using peat cores indicates that the present-day net carbon balance is not significantly different from the annual carbon accumulation over the last 3,000 years.

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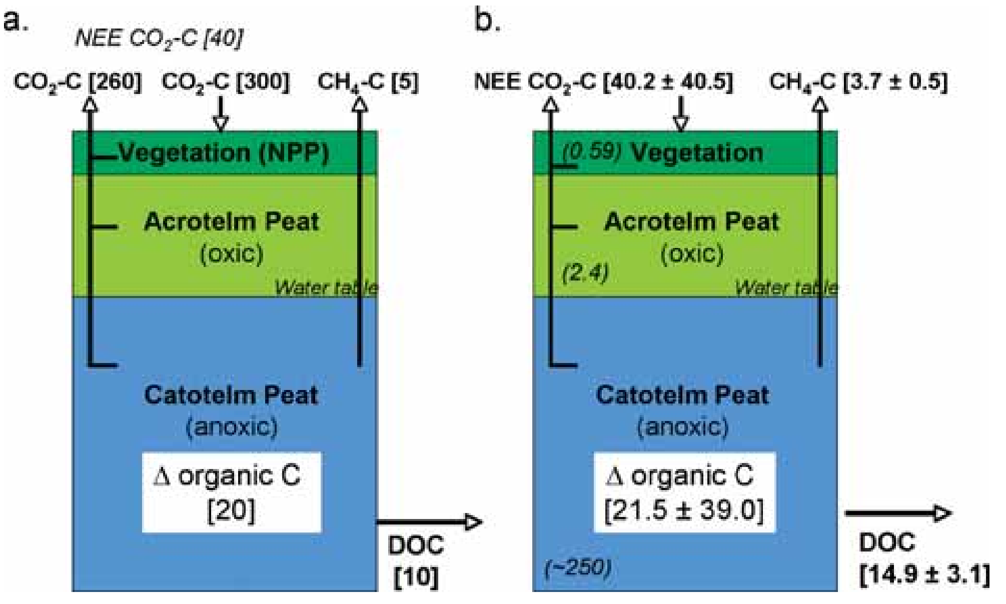

Figure 2.1. *a) The peatland carbon balance deduced by Gorham (1991, 1995) and b) measured by Roulet et al. (2007) for the Mer Bleue peatland. All flux values are given as g C m^{-2} yr^{-1} and values in italics in (b) give carbon stocks in each pool in g C m^{-2}.*

2.3.1. Carbon dioxide

As mentioned earlier, the amount of CO_2 taken up and stored in a peatland results from the difference between CO_2 uptake by gross ecosystem photosynthesis (GEP) and CO_2 release through ecosystem respiration (ER). The productivity of the vegetation is related to the vegetation community present at that location, which is driven by the nutrient status and hydrology of the site (Malmer, 1986). For example, Waddington *et al*. (1998) observed a unimodal relationship between GEP and water table position in a subarctic fen, with the highest productivity occurring when the water table was on average 2 cm below the surface. Frolking *et al*. (1998) considered data from a variety of northern peatlands and determined that at high light levels NEE was greater at rich fens and poor fens than at bogs. Vegetation productivity may also be related to seasonal temperature with warmer temperatures resulting in longer growing seasons and thus higher seasonal GEP (e.g. Griffis *et al*., 2000).

Ecosystem respiration is also related to the vegetation community type, both due to the inclusion of autotrophic respiration in ER and varying decomposability of organic matter of different plant species and peat substrates (Moore & Basiliko, 2006). Since the decomposition rate is faster under aerobic conditions compared to anaerobic conditions, a deeper water table position tends to result in enhanced rates of ER (e.g. Moore & Dalva, 1993). Therefore, a change in the vegetation community may change the substrate quality and hence the rate of ER (e.g. Fisk *et al*., 2003). Also, because respiration is enzymatically controlled, ER is positively related to temperature (e.g. Lafleur *et al*., 2005). Finally, the chemistry or “quality” of the substrate available will affect the rate of decomposition. As new litter is decomposed, the remaining substrate becomes increasingly recalcitrant and more difficult for microbes to degrade, being manifested in lower decomposition rates.

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2.3.2. Methane

Methane is produced under highly reduced conditions by methanogenic bacteria. Thus, it is produced in the saturated zone of peat soil and once produced can be transported to the atmosphere via diffusion, ebullition (bubbling) or as diffusion or mass flow through vascular plants. As CH_4 moves through less reduced zones in the peat, such as the unsaturated soil layer, or the rhizosphere (rooting-zone) of vascular vegetation, it can be oxidized to CO_2 by methanotrophic bacteria. Thus, peatland CH_4 emissions have been found to be related to water table position (e.g. Roulet *et al*., 1992) and peat temperature (e.g. Bubier *et al*., 1995). While both CH_4 production and oxidation, as microbial processes, are positively related to temperature, laboratory studies suggest that production rates are more sensitive than oxidation rates to temperature (Dunfield *et al*., 1993). Also, the quality of the C substrate available affects CH_4 production rates, with CH_4 at depth often produced from newly fixed organic matter leached from the surface, as opposed to the more recalcitrant peat substrate available at depth. Since vascular vegetation can provide fresh substrate for CH_4 and a rapid mechanism for its release to the atmosphere, the presence and productivity of vascular vegetation is correlated to the rate of CH_4 emissions from peatlands (e.g. Bellisario *et al*., 1999). Because fens generally have more sedge vegetation and higher water table position than bogs, CH_4 emissions are generally higher from fens (Bubier *et al*., 1995; Nilsson *et al*., 2001).

2.3.3. DOC dynamics

Although dissolved organic carbon (DOC) is not a greenhouse gas, it is important to include a discussion of DOC dynamics here as it figures prominently in peatland C storage. Also, a significant proportion of DOC exported from peatlands can be broken down to CO_2 during downstream transport and evade into the atmosphere (Billett *et al*., 2004). DOC is operationally defined as a continuum of organic compounds that pass through a filter with pore size of 0.45 µm and it is formed due to the incomplete decomposition of organic matter. Since anaerobic decomposition of organic material is slower and less complete, saturated peats generally have high concentrations of DOC. The concentration of DOC in soil solution also generally increases upon rewetting of the soil after a period of drought suggesting that water table fluctuation is an important control for DOC production. Since the production of DOC is related to microbial decomposition, its concentration in peatland pore water should be related to temperature and several field studies have observed seasonal shifts in DOC concentration that follow soil temperature changes (Bourbonniere, 1989; Waddington & Roulet, 1997; Moore & Dalva, 2001). However, temperature response of DOC production is small in comparison to the temperature response of CO_2 production via soil respiration. Vegetation community composition and productivity is also important for controlling pore water DOC concentration since the production of DOC from different litter types has been observed to vary, while enhanced vegetation productivity is linked to higher concentrations of DOC in soil solution (e.g. Freeman *et al*., 2004).

2.3.4. Nitrous oxide

Nitrous oxide emissions from pristine peatlands are generally low, with many ombrotrophic sites exhibiting net N_2O uptake (Martikainen *et al*., 1993). N_2O emission from minerotrophic peatlands increases when water tables are lowered (Martikainen *et al*., 1993; Regina *et al*., 1996; 1999) and under elevated atmospheric CO_2 concentrations; however, this increase is limited at nutrient poor sites (i.e. ombrotrophic bogs). Therefore, N_2O emissions from pristine northern peatlands play a minimal role in the climate system. Despite the changing environmental conditions expected for northern peatlands under climate change, it has been predicted that increases in N_2O fluxes would account for only 1% of total global emissions (Martikainen *et al*., 1993). Because of this minimal role, the flux of N_2O from natural peatlands will not be discussed further in this chapter. Fluxes of N_2O are important, however, in disturbed peatland systems (see Chapters 3, 4).

The variability in the inter-annual and inter-seasonal C sink/source strength can be very large (Box 2.1).

2.4. Northern peatlands and radiative forcing

The role of peatlands in terrestrial C cycling, and by extension the contribution of these ecosystems to global climate change, is complex. Carbon dioxide uptake contributes to negative radiative forcing (i.e. cooling) while CH_4 emissions contribute to positive radiative forcing (i.e. warming). The most widely used technique to compare the climatic impacts of different GHGs from anthropogenic sources is the Global Warming Potential (GWP) methodology (Ramaswamy, 2001). The GWP index integrates the climatic forcing of a particular gas over time. Over a 20 year time horizon the GWP of CH_4 is 72 times that of CO_2. This falls to 25 and 7.6 over 100 and 500 year horizons, respectively (IPCC, 2007). The reduction of GWP is due primarily to the greater atmospheric lifetime of CO_2 relative to CH_4. A peatland will have a negative GWP when the removal CO_2 results in a negative GWP of greater magnitude than the positive GWP related to the release of CH_4. Therefore, for any given ratio of CH_4 emission to CO_2 uptake there is a time horizon (or a particular GWP value) for which the uptake of CO_2 offsets the CO_2 equivalent emission of CH_4. Over a short time horizon there will be net cooling only when the CH_4/CO_2 exchange ratio is low (i.e. the peatland emits a small amount of CH_4 relatively to its net uptake of CO_2). As the time horizon increases (and the GWP of CH_4 decreases) there will still be net cooling, even with an elevated CH_4/CO_2 ratio. A peatland will be a source when the time horizon is short and the CH_4/CO_2 ratio is elevated.

Applying GWP methodology to assess the climatic effect of peatlands has generally showed that over short periods (i.e. 20 year time horizon) they are net GHG sources (i.e. cause warming), but over longer periods (i.e. > 100 years) they are net GHG sinks. This analysis has also been applied to peatlands experiencing disturbance. Johansson *et al*. (2006) found that over a thirty year period, permafrost thawing and subsequent vegetation change lead to a 47% increase in the radiative forcing impact of a sub-Arctic Swedish peatland when considered on a 100 year time horizon.

There are several assumptions underlying the GWPs that are not generally addressed in ecosystem atmosphere exchanges. First, it is assumed that GHGs are emitted as an isolated pulse and then further emissions are independent. In reality, ecosystem exchanges are continuous, variable, and compound over years. Secondly, when dealing with industrial sources it is clear when the emissions occur and that they had no influence prior to their release, but in the case of an ecosystem the periods of 20, 100 and 500 years are totally arbitrary. That is, there is no biogeochemical justification for the time period considered relative to the turnover time of C within an ecosystem. Finally, using GWPs assumes that the GHG

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source, in this case the peatland, and the atmosphere were in equilibrium prior to the release. But, since northern peatlands continue to take up C and have been for up to 6,000 to 10,000 years, this is not the case – i.e. peatlands are not in steady-state. The consequences of applying the GWP is that one concludes that peatlands are net sources of GHGs on short time scales and that only after several hundreds of years do they become sinks (e.g. Roulet, 2000). However, in reality, natural peatlands are large net sinks for GHGs and have been for thousands of years (Frolking *et al*., 2006). The consequences for GHG management are significant. Using GWPs one may choose strategies to maximize uptake of CO_2 and minimize the flux of CH_4, while when considering GHG exchange continuously it is clear that the maintenance of the large C store is critical.

2.5. Impact of atmospheric and climate change on peatland GHG exchange

The response of peatland GHG exchange to changing atmospheric CO_2 concentrations and resultant climate change depends greatly on the response of the peatland soil-vegetation-hydrology-climate system. In assessing the potential impact of climate change on peatland GHG exchange we begin with a discussion on the expected changes in the hydrologic and thermal regime and then discuss how this may affect GHG exchange in seasonally frozen northern peatlands. The effect of climate change on permafrost persistence and fire frequency, and the subsequent effects on peatland C cycling are then discussed in section 2.5.4.

2.5.1. Hydrologic and thermal regime responses

The Intergovernmental Panel on Climate Change (IPCC, 2007) predicts that boreal and subarctic zones will experience a 1-3 °C warming by 2029 and up to 5-6 °C warming by the end of the century. Predictions of future precipitation patterns are less certain, but tend to predict an increase in precipitation for these latitudes, particularly in the winter months. As a result, soil temperatures in peatlands should increase. Few have estimated the actual magnitude of this increase, but Roulet *et al*. (1992) and Waddington *et al*. (1998) predict increases in surface peat temperatures of 0.8 and 2.3 °C, respectively given an increase in air temperature of 3 °C.

Considering the predicted warmer air temperature under climate change, if the relative humidity in the atmosphere remains constant, evapotranspiration must increase. Roulet *et al*. (1992) applied predicted climatic changes of 3 °C increase in temperature and 1 mm day^{-1} increase in precipitation to a simple peatland hydrologic model and determined that water tables would be lowered by 14-22 cm.

Even at a local scale, the effect of climate change on peat wetness may be highly spatially variable. As the water table is lowered, increased stress is placed on the peat structure and it compresses, reducing the overall peat volume. This results in a lowering of the surface level, or soil subsidence, which may maintain the water table close to the peat surface despite a reduction in the total volume of water stored in the peatland. The potential for peat volume change varies between peatland microforms, with hummocks and ridges tending to be more rigid (Whittington & Price, 2006). Thus, despite increases in evapotranspiration, lawns

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and hollows may maintain relatively wet surface conditions. Also, soil subsidence reduces peat porosity resulting in a reduction in hydraulic conductivity. As a result, water movement through the peat will be reduced, affecting the supply and redistribution of dissolved substrates.

2.5.2. Response of peatland vegetation communities

Peatland vegetation gradients are controlled by nutrient status (minerotrophic vs. ombrotrophic), position within the peatland (mire margin vs. expanse) and microtopography (ridge/hummock vs. hollow/pool) (e.g. Malmer, 1986). The microtopographic gradient is dominantly controlled by water table position relative to the peat surface with both moss and vascular species distributed based on moisture conditions. Variation of vegetation community types between microforms results in corresponding differences in GEP (Laine *et al*., 2007a; Riutta *et al*., 2007) with greatest productivity generally at an intermediate water table/moisture level (Waddington *et al*., 1998).

Aforementioned climate change induced water table drawdown may cause shifts of vegetation communities between microforms in response to shifting moisture conditions. Depending on the extent of drying, peatland vegetation communities may respond with enhanced growth of shrubs and trees. This is well documented in peatlands drained for forestry (Chapter 4). At wetter locations, as pools dry out they may be colonized by S*phagnum* mosses and sedges, commonly present on hollows or lawns, with corresponding shifts in GEP. This has been observed in pools drained naturally by soil pipes (e.g. Foster *et al*., 1988) and in experimentally drained peatlands (Box 2.2). Studies of peat monoliths by Weltzin *et al*. (2000) indicate differential response of bog and fen communities to warming and water table manipulation. Shrub growth was favoured when water tables were lowered whereas sedges and herbaceous species were more productive with water tables at the soil surface. Warming resulted in an increase in belowground biomass relative to aboveground biomass in both bog and fen monoliths, but only increased total biomass at the fen.

2.5.3. Changes in peatland carbon cycling

Carbon dioxide

As described above, NEE results from the difference between vegetation productivity (GEP) and ecosystem respiration (ER). Studies in boreal and temperate peatlands, investigating the effect of higher atmospheric CO_2 concentrations on peatland NEE, have observed minimal response (Kang *et al*., 2001; Saarnio *et al*., 2003; Ruben *et al*., 2006). This occurs because, despite a general increase in productivity and biomass under higher atmospheric CO_2 concentrations, ER also increases. Therefore, it appears that higher atmospheric CO_2 concentrations will have little direct impact on net peatland CO_2 exchange. Studies have shown initial short-term positive increases in photosynthetic uptake in mosses due to elevated CO_2 but after a year there is little, owing to photosynthetic down regulation (Toet *et al*., 2006). In the same experiments there was little change in aboveground biomass in the vascular plant community (Milla *et al*., 2006). Further, in experiments across several peatlands in northern Europe exposed to both elevated CO_2 and increased nitrogen deposition, the elevated CO_2 had no significant effect of moss biomass growth (Berendse *et al*., 2001). With the additional influence of N deposition there was no overall increase in biomass, but the partitioning of biomass between moss and vascular overstory changed.

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Higher temperature may increase the length of the growing season, increase peat temperature, result in lower water table position by enhancing evapotranspiration, and lead to permafrost degradation. Griffis *et al*. (2000) found that the early arrival of spring resulted in enhanced GEP in a poor fen. As well, the annual CO_2 balance at a Finnish subarctic fen was primarily controlled by the timing of snowmelt (Aurela *et al*., 2004). In contrast, in a cool temperate bog, there was little variation in the start of growing season from one year to the next (Moore *et al*., 2006), suggesting that the early arrival of spring may only be important for increasing C storage at more northern latitudes (> 60 °N). Longer growing seasons may enhance CO_2 uptake, but the effect on NEE will likely depend on the resulting hydrological conditions. During a long, dry growing season a peatland may act as a smaller sink, or even a source of CO_2, because increased GEP is more than compensated for by increased ER (e.g. Joiner *et al*., 1999) Increases in ER will also occur because it is positively related to peat temperature, which is expected to rise in response to rising atmospheric temperatures. Waddington *et al*. (1998) predicted that ER would increase by 25% following a 3 °C increase in atmospheric temperature. Thus, longer growing seasons may do little to affect NEE in many northern peatlands.

Temperature will also play an important role in altering peatland hydrology. Increased evapotranspiration will likely lower water table position and reduce soil moisture. These drier conditions should result in higher ER as reduced soil moisture improves peat aeration and increases decomposition rates. The relationship between ER and moisture is complex, as changes in moisture can affect both CO_2 fixation and ER, not necessarily in the same direction (Lafleur *et al*., 2005). Hydrologic changes can also influence GEP as Lafleur *et al*. (2003) showed that moisture availability in July, August and September was a critical factor in determining the size of the annual GEP. Years that had a dry summer had a GEP of ~ 0, while years with frequent rain events through the latter half of summer tended to have a significantly larger GEP. The overall effect on peatland NEE will depend on the initial hydrologic conditions of the peatland. Vegetation productivity at initially wet sites such as fens, and bog hollows and pools can be enhanced by drying (Box 2.2), balancing the increase in ER. Overall, dry peatland areas such as bog hummocks and ridges will act as smaller sinks or sources of CO_2, whereas wet zones will likely become greater sinks.

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Box 2.2. The St. Charles-de-Bellechasse Case study: impact of water table drawdown on peatland carbon cycling

To determine the potential change in carbon cycling caused by climatically induced water table lowering, a controlled ecosystem-scale water table drawdown experiment was carried out in a poor fen in southern Quebec (46°40'N 71°10'W), Canada (see Strack *et al*., 2004; Strack *et al*., 2006; Whittington & Price, 2006; Strack & Waddington, 2007). The site had pool-ridge topography and water table in the pool was drawdown ~20 cm at the experimental site with a ditch connecting the pool to a larger drainage network (Figure 2.2). This was compared to a pristine, control site and a site drained eight years prior to the study.

The resulting water table drawdown was mitigated by peat subsidence which was greater in the hollow/pool area. Hydrological changes resulted in vegetation succession in which *Sphagnum* moss on the ridges was replaced by lichens, sedge cover increased at lawns, and bare peat on pool bottoms was colonized by *Sphagnum*. Overall, this resulted in increased CO_2 release, reduction in CH_4 efflux and lower DOC concentration at ridges, and maintenance of a CO_2 sink, moderate CH_4 emission and higher DOC concentrations at hollows (Figure 2.3). Movement of DOC between microforms was reduced at the drained site because soil subsidence greatly reduced the hydraulic conductivity of the peat; this is also likely to reduce inter-microform water movement and subsurface discharge under climate change. These results reveal that the feedbacks between hydrology, peat soil properties and the vegetation community, as well as peatland microform composition, need to be considered when predicting the response of peatland GHG exchange to climate change.

Methane

Higher concentrations of atmospheric CO_2 have been observed to enhance CH_4 emissions (e.g. Saarnio *et al*., 2000). This increase appears to be linked to higher vascular plant biomass and productivity that likely provides substrate for CH_4 production via root exudates and litter, and an efficient pathway for CH_4 release. However, Kang *et al*. (2001) observed limited increases in CH_4 emission under elevated CO_2 concentrations and suggested that this was linked to better aeration in the root zone resulting in CH_4 oxidation. Therefore, while elevated atmospheric CO_2 will likely result in increased peatland CH_4 emissions, the magnitude of the resulting flux is unclear.

As microbial processes, the rate of both CH_4 production and CH_4 oxidation are positively related to temperature. Laboratory studies suggest that CH_4 production is more sensitive to temperature than CH_4 oxidation (reviewed by Segers, 1998) and thus increased temperature alone should enhance peatland CH_4 emissions.

As discussed above, CH_4 emissions are generally higher from wetter locations. Therefore, in seasonally frozen peatlands, predicted lower water tables will likely reduce CH_4 emissions (Roulet *et al*., 1992). However, reductions in CH_4 emissions are limited, and fluxes may actually increase, in initially wet locations such as hollows and pools. At these areas, soil subsidence maintains water tables close to the surface and drier conditions may result in increased

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vegetation productivity. Both of these effects help to maintain high CH_4 fluxes (Box 2.2).

Recently, it has been suggested that ebullition may account for as much CH_4 release as diffusive fluxes (Glaser *et al*., 2004). Because of the rapid release of CH_4 via ebullition, water table position is not as important for controlling this flux in terms of enabling CH_4 oxidation. These fluxes are more closely linked to large changes in atmospheric pressure, drought resulting in groundwater flow reversals, and sudden shifts in water table position. Therefore, future patterns of weather systems, droughts and extreme precipitation events have the potential to greatly impact peatland CH_4 emissions. Also, Rosenberry *et al*. (2006) suggest that thickness and extent of soil ice, which may be altered by changes in the snowpack presence and depth, could affect springtime release of entrapped gaseous CH_4; further research is needed to quantify the importance of this efflux.

Overall, CH_4 efflux will likely decline from dry locations such as ridges and hummocks and be maintained or increase slightly from wet locations such as fens, pools and hollows. A changing vegetation community can play an important role with more abundant sedge cover enhancing CH_4 emissions at sites that remain relatively wet. Potential changes in CH_4 efflux via ebullition remain unclear.

Dissolved Organic Carbon

Export of DOC from peatlands depends on both production of DOC and discharge of water from the peatland. Increases in DOC export have been observed from European peatlands (e.g. Freeman *et al*., 2001). While sudden decreases in water table position can increase DOC production and concentration, Pastor *et al*. (2003) observed no significant effects of warming or water table position on the export of DOC from peatland mesocosms. Instead it has been suggested that an increase in vegetation productivity resulting from elevated atmospheric CO_2 concentrations has resulted in an increase in DOC production in peatlands (Freeman *et al*., 2004). Net DOC production has been observed to increase under higher atmospheric CO_2 concentrations likely linked to increases in vegetation productivity.

While enhanced net DOC production will increase DOC concentrations within peatlands, export will also depend on total discharge of water. Because climate change will likely increase evapotranspiration, total discharge may decrease and reduce total DOC export, if there is not a corresponding increase in DOC concentration in stream discharge. On the other hand, Worrall *et al*. (2006) suggest that increasing drought severity will result in enhanced DOC export. Also, storms can flush DOC from peatlands suggesting that future frequency and intensity of storms, while uncertain, will be important for determining DOC export.

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Figure 2.2. *The experiment site at St. Charles-de-Bellechasse (SCB) poor fen before (top) and after (bottom) water table drawdown (photos: M. Strack).*

Carbon storage and radiative forcing
Changes in net fluxes of CO_2, CH_4 and DOC will likely result in minimal changes in peat C storage in seasonally frozen areas (Figure 2.3). While dry locations such as ridges and hummocks may release some stored carbon as CO_2, this will be partially balanced by increased carbon storage at pools and hollows. In forested peatlands, loss of soil C may also be balanced by increased C storage in tree-stand biomass. Also, while some authors suggest that

increasing peatland DOC exports indicate a destabilization of C stocks (e.g. Worrall *et al*., 2006), the increases in export may be related to higher vegetation productivity suggesting that stocks may be maintained.

Even if the total C stored in peat is not greatly affected by climate change, shifts in the relative importance of CO_2 and CH_4 in the carbon balance have the potential to change the radiative forcing of natural peatlands in the short term. Strack & Waddington (2007) calculated that dry locations within a poor fen had increased global warming potential following water table drawdown owing to large releases of CO_2, while initially wet zones exhibited a decrease as a result of CO_2 uptake and reduced CH_4 emissions (Box 2.2). Similarly, Laine *et al*. (1996) suggest that drying conditions would slightly reduce peatland radiative forcing. Overall, when short-comings of using GWPs for ecosystems is considered, a maintenance of net C storage will be most important if peatlands are to maintain their function as net sinks of GHGs.

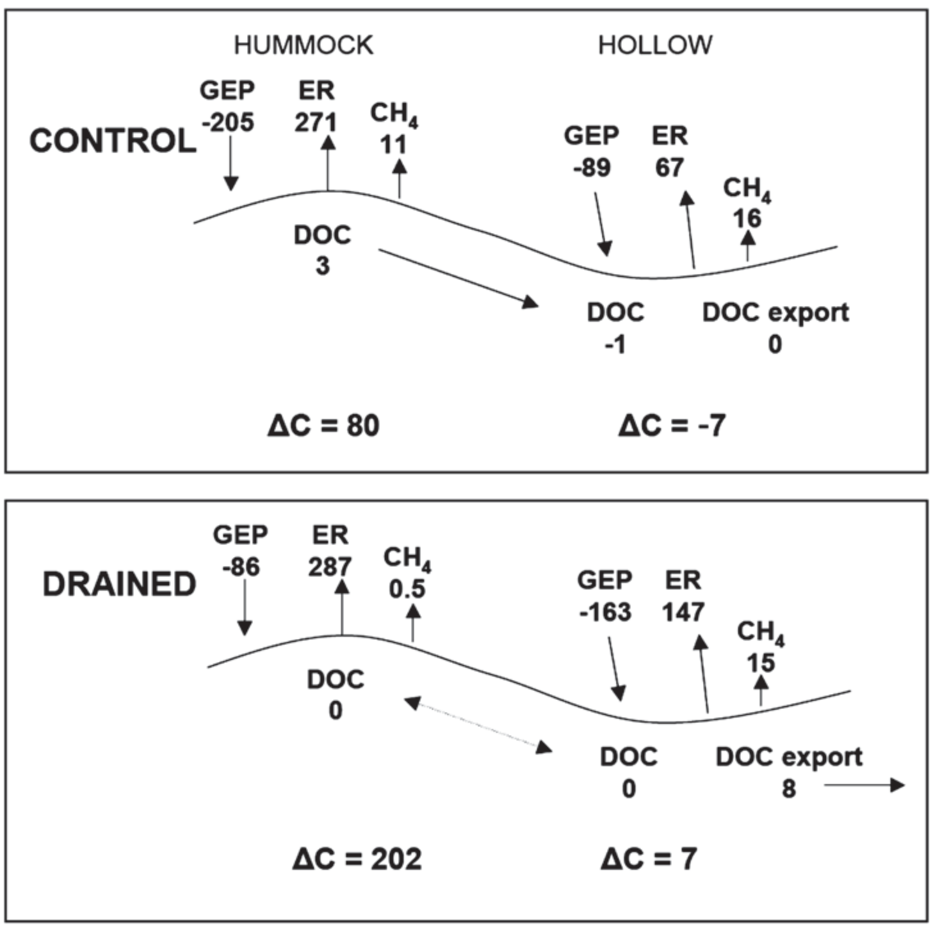

Figure 2.3. *Carbon exchange during the growing season (May-August) at SCB peatland at the control and drained sites. All values are given in g C m^{-2} over the growing season and positive values indicate a flux from the peatland out of the system. Surface export of DOC was enhanced at the drained site because discharge was enhanced via drainage; however, discharge is expected to decline under climate change scenario due to increased evapotranspiration.*

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2.5.4. Climate-mediated disturbance responses

Permafrost degradation

Permafrost is defined as earth materials remaining at or below 0 °C for two or more years. Northern peatlands often are underlain by permafrost, reflecting the thermal insulating qualities of peat. In western Canada, about 30% of peatlands are underlain by permafrost (Vitt *et al*., 2000), which occurs mainly as palsas or peat plateaus. Also, due to the insulating nature of dry surface peat, boreal peatlands near the southern limit of discontinuous permafrost in Canada have harbored relict permafrost since the Little Ice Age, ending ~150 years ago. This relict permafrost creates localized permafrost features that occur as densely treed, elevated landforms situated within unfrozen peatlands (e.g. Halsey *et al*., 1995).

Over the past several decades, increases in near-surface permafrost temperatures and changes in surface energy balance have triggered the degradation of permafrost in northern peatlands (e.g. Jorgenson *et al*., 2001). In particular, permafrost in Canadian peatlands continues to degrade at the southernmost limit of discontinuous permafrost, with no evidence of reaggradation. Over the past ~100 years, the current southern limit of discontinuous permafrost in western Canada has shifted northwards by 39 km on average, and by as much as 200 km in some locations (Beilman *et al*., 2001). Simulations indicate that the area of permafrost landforms in Canada has decreased by up to 22% over the last century, while permafrost landforms that have persisted have thicker active layers (Chen *et al*., 2003). In general, rates of permafrost degradation are expected to accelerate under future climate change. Using a model of palsa distribution in Scandinavia, Fronzek *et al*. (2006) predicted that a 4 °C increase in surface air temperatures would result in the complete loss of palsa landforms from this region.

While lower water tables are likely for many northern peatlands under projected future climate regimes (section 2.5.1), permafrost degradation in peatlands generally results in thermokarst and increased saturation of surface peat, as peat surfaces collapse to levels at or below the water table during thaw. Permafrost degradation in peatlands across continental Canada displaces tree communities and creates saturated, open (non-forested) peatlands called internal lawns that quickly are colonized by mesic species such as *Sphagnum riparium* and *Carex* spp. (Beilman, 2001). In northern Europe, Malmer *et al*. (2005) used vegetation surveys and aerial photographs to document a reduction in dry hummock type communities and an increase in wet sedge and open water areas in mires between 1972 and 2000 due to permafrost degradation.

Several studies have documented increased rates of C storage as peat following surface permafrost degradation in boreal peatlands, likely due to changing vegetation composition and high rates of vegetation productivity in areas of permafrost collapse (e.g. Turetsky *et al*., 2007). Thus, in response to permafrost degradation, peatlands are likely to become larger sinks for CO_2, as GEP will be high while ER is limited by high water tables. However, vegetation change post-degradation is associated with more labile (easily decomposable) organic matter substrates in surface peat, which stimulates heterotrophic respiration and CO_2 production (Turetsky, 2004). Additionally, the saturated conditions following surface permafrost degradation stimulate anaerobic heterotrophic respiration, leading to increases in CH_4 emissions (e.g. Turetsky *et al*., 2007). Thus, permafrost degradation

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in peatlands can serve either as positive or negative feedbacks to net radiative forcing depending on permafrost conditions and differential effects of thaw on GEP, ER and CH_4 efflux (Figure 2.3). Moreover, since streams draining permafrost free, peatland-dominated watersheds have higher DOC concentrations than cold, permafrost influenced watersheds permafrost degradation may greatly enhance DOC export to oceans (Frey & Smith, 2005).

Fire regimes

Generally, while many peatlands historically may have been protected from fire activity due to saturated or near-saturated conditions, lower water tables under ongoing and future climate change (section 2.5.1) will make peatlands increasingly vulnerable to deep soil consumption during burning, with consequences for regional C emissions to the atmosphere and future peatland C balance.

Across the entire North American boreal forest region, annual burn area doubled from the 1960's to the 1990's, primarily due to the area of lightning-initiated fires (Kasischke & Turetsky, 2006). Additionally, the frequency of large fire years increased over this time period due to the increased frequency of large fire events (> 1000 km^2). While trends were consistent across most ecoregions in Canada and Alaska, western ecoregions generally showed greater increases in burn area than eastern ecoregions (Kasischke & Turetsky, 2006). Increases in annual burn areas across Canada since 1920 were correlated to regional warming trends (Gillett *et al*., 2004).

Generally, organic matter consumption during fires depends on the amount and structure of fuels, fuel flammability or moisture, and fire weather conditions. Fire activity in peatlands can occur during the spring, when high water tables or seasonal ice as well as high fuel moisture contents likely limit rates of fuel consumption. However, fires occurring later in the growing season or during drought conditions can result in the consumption of deep peat layers due to deeper water tables and active layers. Zoltai *et al.* (1998) estimated that deep peat fires represent only 15% of all fire events affecting peatlands of Canada, but comprise the majority of regional C emissions due to peat fires.

Estimates of C loss due to organic matter consumption during peat fires average approximately 3 kg C m^{-2}, which does not vary substantially from estimates of organic matter consumption in some upland boreal forest fires. Using these average consumption rates, it has been estimated that ~20% increase in annual burn area and fuel consumption rate would shift continental peatlands in Canada from a net C sink to a net C source (Turetsky *et al*., 2002). In temperate regions, Poulter *et al.* (2006) estimated that temperate peat fires emit up to 0.32 x 10^{12} g of C to the atmosphere annually and that an increase in fire return intervals to 20 years or less would convert these peatlands regionally into a net C source.

It is important to note that while organic matter consumption during upland forest fires can be fuel limited (i.e., burning can consume the majority of ground layer fuels and expose mineral soil horizons), fuel consumption during peat fires is almost always moisture limited rather than fuel limited. Thus, lower water table positions in peatlands under future climate change are likely to exacerbate C emissions from peat fires. However, burning effects on peatland vegetation and soils can show tremendous spatial variability (Figure 2.4). Combustion rates in Canadian bogs appear to be greater in hollows than in hummocks due to the greater water holding

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Figure 2.4. *Smoke plume from peatland fire (top) and spatial variability in fire severity within the peatland (bottom) (photos: M. Turetsky).*

capacity of hummock-forming mosses such as *Sphagnum fuscum*. For example, in a central Alberta fire, fuel consumption rates in hummocks averaged 1.5 ± 0.1 kg C m^{-2}, while consumption rates in hollows at the same site averaged 2.8 ± 0.3 kg C m^{-2} (Benscoter & Wieder, 2003). Given that moss physiology appears to have strong effects on fuel consumption rates in northern peatlands, the expansion or displacement of dominant moss species (particularly hummock species) due to climate change could have important consequences for fire behaviour.

While decomposition rates in boreal forest sites tend to be moisture-limited post-fire, reduced evapotranspiration and increased runoff from surrounding uplands could result in higher water table positions in peatlands following fire activity. Relatively few studies have characterized CH_4 emissions in peatlands with time following fire. In western Siberia, Nakano *et al.* (2006) found that fire converted a forested peatland site from a net CH_4 sink to source for at least several years post-burning. In general, processes affecting CH_4 formation and release in peatlands influenced by fire are likely to be important to the overall GHG budgets of boreal regions.

2.6. Conclusions

Natural peatlands play an important role in the contemporary global C cycle and have played an equally important role in the global C cycle during the Holocene (Frolking & Roulet, 2007; Smith *et al*., 2004; MacDonald *et al*., 2006; Chapter 1). Peatlands are net long-term sinks for atmospheric CO_2 and a large natural source of CH_4. There is tremendous spatial diversity in northern peatlands with local and regional variability in peatland hydrology, climate and ecology. As such, some peatlands are currently net global warmers while others are net coolers according to global warming potentials. However, when the carbon sink function of peatlands is considered over the lifetime of these ecosystems, they represent GHG sinks and maintaining peatland carbon stocks is the appropriate management approach.

The variability of the response of peatland GHG exchange to climate change is likely to be as great as that present within peatland contemporary GHG exchange. Some peatlands will likely emit more CO2 to the atmosphere and potentially switch from a net C sink to a net source while other peatlands may enhance CO_2 sequestration (Figure 2.5). In terms of GHG management, if the goal is to maintain peatland C stock, currently dry, seasonally frozen peatlands are likely most at risk. Not only are these sites most likely to release CO_2 via peat oxidation under drying conditions, but will also face increased risk of fire. Since peatlands will continue to play an important role in the global carbon cycle as an important biospheric feedback to climate change it is critical to continue to improve our understanding of the interactions between climate, peatland ecohydrology, permafrost degradation and fire regimes for controlling northern peatland C stocks.

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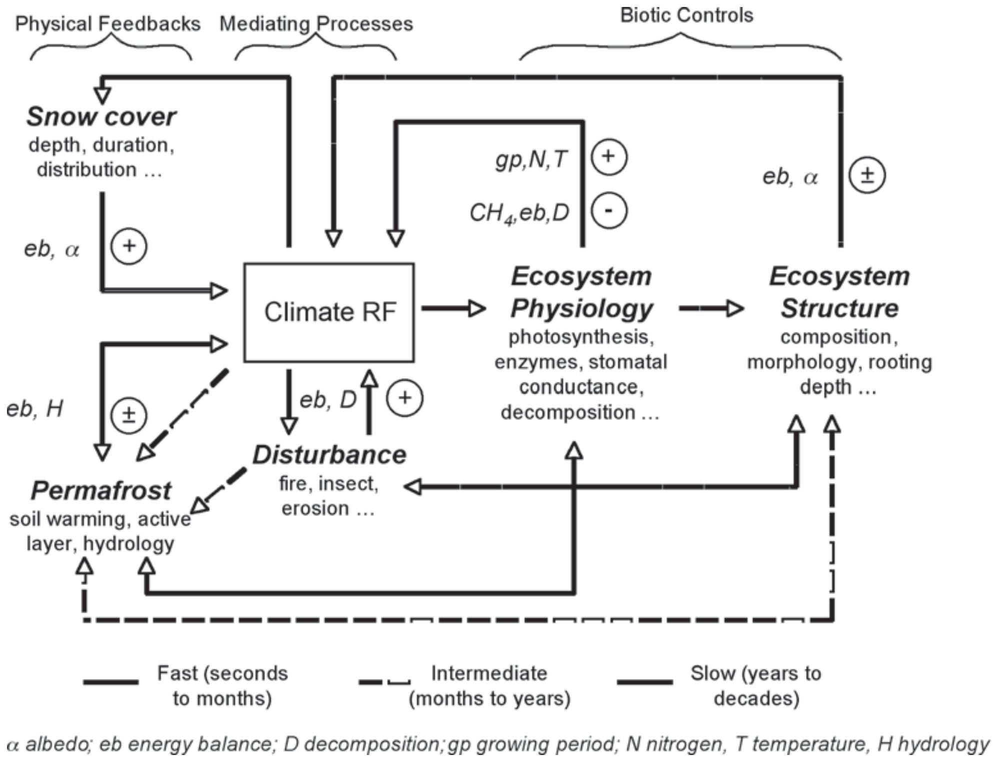

Figure 2.5. *Feedbacks between climate and peatland ecosystems over various spatial and temporal scales. Modified from McGuire et al., 2006.*

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CHAPTER 3:

IMPACTS OF AGRICULTURAL UTILIZATION OF PEAT SOILS ON THE GREENHOUSE GAS BALANCE

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3.1. Introduction

The area and intensity of agricultural production on peatland have increased in many countries during the last two centuries, and agriculture is now the most widespread human use for peatlands globally. Available data are summarised in Table 3.1. However, it is difficult to estimate the fraction of agriculturally used peatland worldwide due to the same difficulties that hamper global estimates of the peatland resource itself, e.g. differing definitions of peatland between countries and uncertainty about the depths of deposits.

Since the Second World War, large areas of peatland in central and Eastern Europe have been converted into pastures, hay meadows, ploughed fields, forestry plantations and fishponds. About 14% of European peatlands are currently used for agriculture, the great majority being used as meadows and pastures (Lappalainen, 1996; Joosten & Clarke, 2002; Ilnicki, 2002, modified). Land privatisation in this part of Europe is now proceeding rapidly, and it is anticipated that this will affect shallow and drained peatlands most strongly (Bragg & Lindsay, 2003). In countries such as Hungary (98%), Greece (90%), The Netherlands, (85%), Germany (85%) and Poland (70%), almost all organic soils are cultivated. Elsewhere, only small areas of peatland are currently under agricultural use (Finland, 2%; United Kingdom, 4%; Sweden, 5%). The great majority of this peatland is used as meadow and pasture, and only a few percent as arable land (Joosten & Clarke, 2002). In the Netherlands, for example, about 43,782 ha are currently used as arable land and 178,226 ha are under grass. In Sweden cereals occupy about 77,288 ha, row crops 4,389 ha and ley and extensive ley 185,263 ha (Van den Akker, 2006; Berglund, personal communication). Canada has one of the largest areas of agriculturally used peatland but this amounts to only 15% of

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the total national resource of peatlands and mires, most of which are undrained and forested. In the United States the largest areas of peatlands are situated in Alaska, Minnesota, Michigan, Florida and Wisconsin. Over 230,000 hectares of fen peatland in the Florida Everglades are cultivated mostly for sugar cane and rice. Only 20% of Indonesia's peatlands are drained and agriculturally used (Ilnicki, 2002; Joosten & Clarke, 2002).

Table 3.1. *The Area of Peatland Used for Agricultural Production in Some Countries of the World (Mutalib et al., 1991; Lappalainen, 1996, modified; Joosten & Clarke, 2002, modified; Ilnicki, 2002, modified)*

Country	Total peat-land area (km^2)	Peatland area used for agriculture (km^2)	(%)
Europe			
Belarus	23 967	9 631	40
Estonia	10 091	1 300	13
Finland	94 000	2 000	2
Germany	14 200	12 000	85
Great Britain	17 549	720	4
Iceland	10 000	1 300	13
Ireland	11 757	896	8
Latvia	6 691	1 000	15
Lithuania	4 826	1 900	39
Netherlands	20 350	2 000	85
Norway	23 700	1 905	8
Poland	10 877	7 620	70
Russia	568 000	70 400	12
Sweden	66 680	3 000	5
Ukraine	10 081	5 000	50
North America			
Canada	1 114 000	170 000	15
U.S.A.	611 000	61 000	10
Asia			
Indonesia	200 728	42 000	20
Malaysia	25 890	8 285	32
China	10 440	2 610	25

3.2. Impact of drainage on the physical and chemical properties of peat soils

Both fens and raised bogs must be drained in order to regulate the air and water conditions in the soil to meet the requirements of cultivated plants. Thus farmed organic soils have the ability to store large amounts of water, and nitrogen and phosphorus are plentifully available to plants due to peat mineralization during the early years after drainage.

Long-term cultivation and agricultural use of peatlands has led to a number of effects including lowering of the water table, increased aeration, and changes in plant

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communities. Grassland on peat requires lowering of the water table from 0.4 to 0.8 m below the soil surface and an air content of at least 6–8% in the surface layer. For arable land, the water table should be 1–1.2 m below the soil surface (Okruszko, 1993; Ilnicki, 2002; Joosten & Clarke, 2002).

The decline in peat soil moisture content resulting from drainage leads to shrinkage of the peat. Volume change due to shrinkage is the result of several forces acting at micro-scale, and its mechanism and magnitude differ from those in mineral (clay) soils (Ilnicki, 1967; Szatyłowicz *et al*., 1996; Oleszczuk, 2006). Peat with a high degree of decomposition and low ash content shrinks more than peat containing *Sphagnum.* The loss of buoyancy of the upper soil horizons due to removal of water also causes mechanical compression of the permanently saturated peat layers below the water table. Thus the drainage of organic soils for agricultural use causes subsidence of the soil surface due to a combination of shrinkage and compaction.

It is generally understood that the rate of subsidence varies strongly with a number of factors such as peat type, rate of decomposition, density and thickness of the peat layer, drainage depth, climate, land use, and period of drainage. Initially, the soil surface may descend by 5–10 cm per year (Schothorst, 1982; Okruszko, 1993; Ilnicki, 1973; 2002; Millete & Broughton, 1984; Wösten *et al*., 1997; Jurczuk, 2000). After a few years the annual subsidence rate decreases to 0.3–1.5 cm for grasslands and to 1.5–3.0 cm for arable peatland (Mundel, 1976). For Swedish peat soils an annual subsidence rate of 0.5–3.0 cm has been reported (Kasimir-Klemedtsson *et al*., 1997), whilst for some arable peatlands in Poland the subsidence rates oscillate around 1.8 cm per year. Values of 0.3–0.6 cm yr^{-1} and 1.0–1.3 cm yr^{-1} were observed for Polish peat grassland with shallow (0.4–0.6 m) and deep (0.8–1.2 m) water table respectively (Okruszko, 1993; Jurczuk, 2000; Ilnicki & Iwaszyniec, 2002; Brandyk *et al*., 2006) (Figure 3.1).

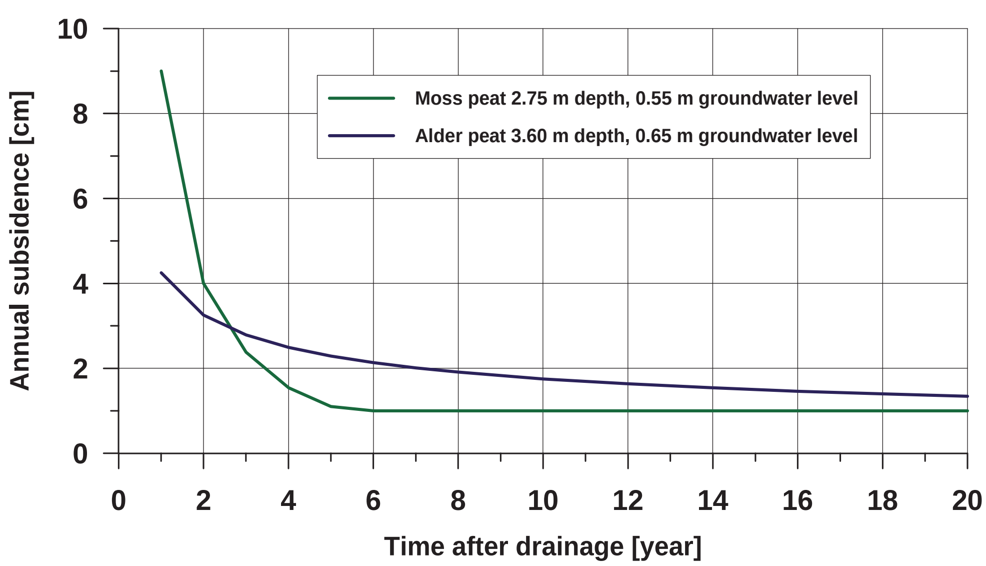

Figure 3.1. *Subsidence rate of the peatlands in North Poland (Jurczuk, 2000, modified)*

Later in the drying process, vertical and horizontal shrinkage cracks are formed (Figure 3.2). The cracks allow oxygen to diffuse into deep layers of the soil profile. Hence oxygen-dependent processes - for example decomposition, mineralization and nitrification - are strongly affected by shrinkage of the soil (Ilnicki, 1967; Szatyłowicz *et al*., 1996; Oleszczuk *et al*., 1997; Van den Akker & Hendriks, 1997; Brandyk *et al*., 2003; Oleszczuk *et al*., 2003; Hendriks, 2004).

Accompanying changes in the physical properties of organic soils are closely related to the depth of drainage and the degree of decomposition (Kasimir-Klemedtsson *et al*., 1997; Brandyk & Szatyłowicz, 2002; Okruszko & Ilnicki, 2003; Schwärzel *et al*., 2002). Drainage and intensive use of peatlands are the main factors causing the moorshing process, which transforms surface peat into a new material called moorsh. This differs from the underlying peat layers in that it has a grainy structure. The structure of undrained peat ranges from fibrous to amorphous depending on the degree of humification of the plant remains. The shrinkage process causes an increase in the density of the solid phase whose value depends on many factors such as botanical composition, degree of peat decomposition and ash content. The changes in soil structure caused by changes in pore space and increasing bulk density have a direct impact on both saturated and unsaturated hydraulic conductivity, which control the flow velocity of water within the soil. Hydraulic conductivity declines with increasing drainage intensity and with time from the onset of peat compaction, due to the reduction of the volume of fast-draining pores (Okruszko, 1993; Zeitz & Velty, 2002; Gnatowski *et al*., 2002; Brandyk *et al*., 2003; Ilnicki & Zeitz, 2003).

Most biochemical and chemical processes in peat require aqueous conditions. The increase in air content that accompanies lowering of the water table arrests the accumulation of organic matter and stimulates humification and mineralization of the peat. The biochemical and chemical character of peat is influenced by the composition of the plant remains from which it was formed, and it is known that high carbohydrate content (e.g. cellulose, pentose and hexose), amino sugars, free amino acids, P_2O and CaO all efficiently accelerate the mineralization process. However, phenols (free and bound), phenolic acids and lignin inhibit this transformation. Also, root exudates and degradation products from the remains of cultivated plants that strongly promote degradation and mineralization of peat have been observed (Grootjans *et al*., 1985; 1986; Lishtvan *et al*., 1989). The mineralization rate in peats with similar botanical composition decreases as peat thickness increases. Bogs can be arranged

Figure 3.2. *Peat covered by cranberry and bare peat in Estonia (photo by L. Szajdak).*

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in the following order according to the accumulation of carbon (taken up as CO_2; C-CO_2) during mineralization: waterlogged *Sphagnum* > *S. fuscum* > Cotton-grass-*Sphagnum* > Scheuchzeria; while for fen peats it is: *Hypnum* > buckbean > woody > woody-sedge peats. The mineralization rate of bog peat depends on its biochemical composition: a high content of waxes and resins, and a high C:N ratio as well as a low content of readily hydrolysable nitrogen forms are factors that result in low mineralization rates. Inisheva & Dement'eva (2000) concluded that mineralization dynamics differ between peat types, taking the form of a simple exponential function typical of classic organic matter decomposition in some fen peats, but otherwise assuming a sinusoidal shape. Thus the extent of mineralization depends on many factors such as degree of decomposition, water and land management, climatic conditions, ash content, air content, and nutrient ratios. In general the mineralization process is faster in peatlands that are used as arable land than in those used as grassland, and greater in fens than in bogs (Eggelsmann, 1990; Okruszko, 1993; Zeitz & Velty, 2002; Okruszko & Ilnicki, 2003).

The development of pastures or meadows that are regularly cut two or three times per year and the creation of new arable agricultural ecosystems both result in peat decomposition and loss of organic matter. Drainage in particular results in a sharp change of biotic and abiotic properties and consequent degradation of peat organic matter. In general, this leads to the progressive differentiation of the hydrophobic and total amino acid contents (Szajdak & Sokolov, 1997; Szajdak, 2002; Sokolowska *et al*., 2005). Peat organic matter regulates long-term C storage and nutrient availability to plants and microbes. The content of dissolved organic matter (DOM) seems to be closely associated with microbial activity, because this fraction of organic carbon is vulnerable to microbial degradation. DOM quantities are sensitive to land management, especially agriculture, which reduces inputs to the soil organic matter through the removal of plant biomass. The mechanism of DOM degradation depends on the aromaticity and complexity of DOC molecules and the process is accelerated by carbohydrates and amino acids. DOM degradation results in relative enrichment of lignin-derived molecules, which affects the thermal behaviour of individual compound classes and increases the thermal stability of residual DOM. Kalbitz *et al*. (2000) showed that the land use of peatlands affects the properties of fulvic acids (FAs) which account for the major fraction of DOM. They suggested that long-term intensive land use (50 to 200 years) resulted in a larger proportion of aromatic structures and a larger degree of polycondensation of FAs (Kablitz *et al*., 1999). However, details of the affected units of FA structure are unknown. Leinweber *et al*., (2001) concluded that in water-soluble FAs, which are the main component (about 60%) of DOM, the proportion of carbohydrates and phenols + lignin monomers increased with increasing intensity of soil tillage, aeration, and peat degradation.

Despite long-term investigations dealing with organic matter in bogs, the transformation sequence of organic matter after drainage is not clear (Vomperskaya, 1982; Bambalov & Belen'kaya, 1997). According to results obtained by Orlov *et al*. (2000), changes in the composition of organic matter in the peat horizons of drained bog peat led to relatively low content of lipid. The chlorophyll content increased from 6.6 to 108.1 $\mu g \cdot g^{-1}$ and decreased slightly in the course of drainage. The humus in most of the undrained peat horizons under study was of the humate-fulvate type and it changed into the fulvate-humate type in drained soils because of an increase in proportion of humic acids in

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the humus. They proposed that drainage decreased the carbohydrates content in the fractions of humic substances. They also suggested that the upper At' peat horizons were characterized by low humification indices (0.7-1.3) because of the low decomposition degree of peat. The lower At'' horizons were noted for the mean values (1.5-2.5) and even high (3.9) humification index of peat. The values of humification index of peats were more closely correlated with the humification degree of peat than the ratio carbon in humic acids to total organic carbon. They suggest that 25 year long period of drainage did not result in significant changes in the group composition of humus.

Methane, carbon dioxide and nitrous oxide are all potent greenhouse gases and thus important in the context of global climate change. Wetland and peatland ecosystems store over 30% of the soil organic carbon in the world and under natural conditions, flooded wetlands are an important source of methane (CH_4) flux to the atmosphere. The sequential changes in physical and hydraulic properties initiated by drainage for agriculture have an important influence on chemical properties and, in turn, on the net fluxes of CH_4, CO_2 and nitrous oxide (N_2O) from agricultural peat soils (Kasimir-Klemedtsson *et al*., 1997). In general, drainage and cultivation of peatlands significantly increases the emissions of CO_2 and N_2O to the atmosphere, whilst CH_4 emissions are reduced. Aeration of the upper peat layers resulting from drainage and agricultural land use triggers the aerobic decomposition process that causes carbon dioxide (CO_2) emissions from the soil. Nitrogen cycling is also intensified in drained peat, the processes of nitrification and denitrification promoting the production of N_2O. Thus peat soils and land use play an important role in the global budgets of these gases, each considered in more detail below.

3.3. Carbon dioxide emissions

CO_2 emissions from soils arise from respiration by plant roots and living organisms, and from the mineralization of organic matter, dead micro-organisms, soil animals and dead plants. They can be explained largely in terms of degradation of DOM (Marscher & Noble, 2000; Marscher & Bredov, 2002). It is well documented that the degradation of peat organic matter results in the emission of CO_2 from peatlands. Here, the emission of CO_2 reflects the state of transformation of peat organic matter whilst the chemical composition and molecular structure of the peat determine the rate and the direction of the pathway. The rate of carbon dioxide emission to the atmosphere depends on many factors including climatic conditions, peat type, degree of decomposition, water table depth and soil temperature (Figure 3.3), in addition to the type and intensity of land use.

Field studies indicate that the rate of peat mineralization and CO_2 emission increases distinctly with increasing groundwater level (Figure 3.3a, Renger *et al*., 2002). Mundel (1976) found a relationship between soil temperature at 10 cm depth and CO_2 emission rate. At soil temperatures in the range 0–6 °C the carbon dioxide flux did not exceed 3 g m^{-2} d^{-1}, and when the soil temperature increased to 14–22 °C the emission rate increased to 5–10 g m^{-2} d^{-1} (Figure 3.3b). The linear relationships between CO_2 emission rate and two parameters: groundwater level and soil temperature at several depths was developed for German fen peatlands by Flessa *et al.*, (1997; Figure 3.3c). A study on drained fen peat soils in the Biebrza river valley (Poland) also showed that the CO_2 emission rate was strongly affected by type of peat. The emission rate was highest for alder peat and lowest for *Sphagnum* peat, and the maximum rates from *Sphagnum*, sedge and alder peat soils

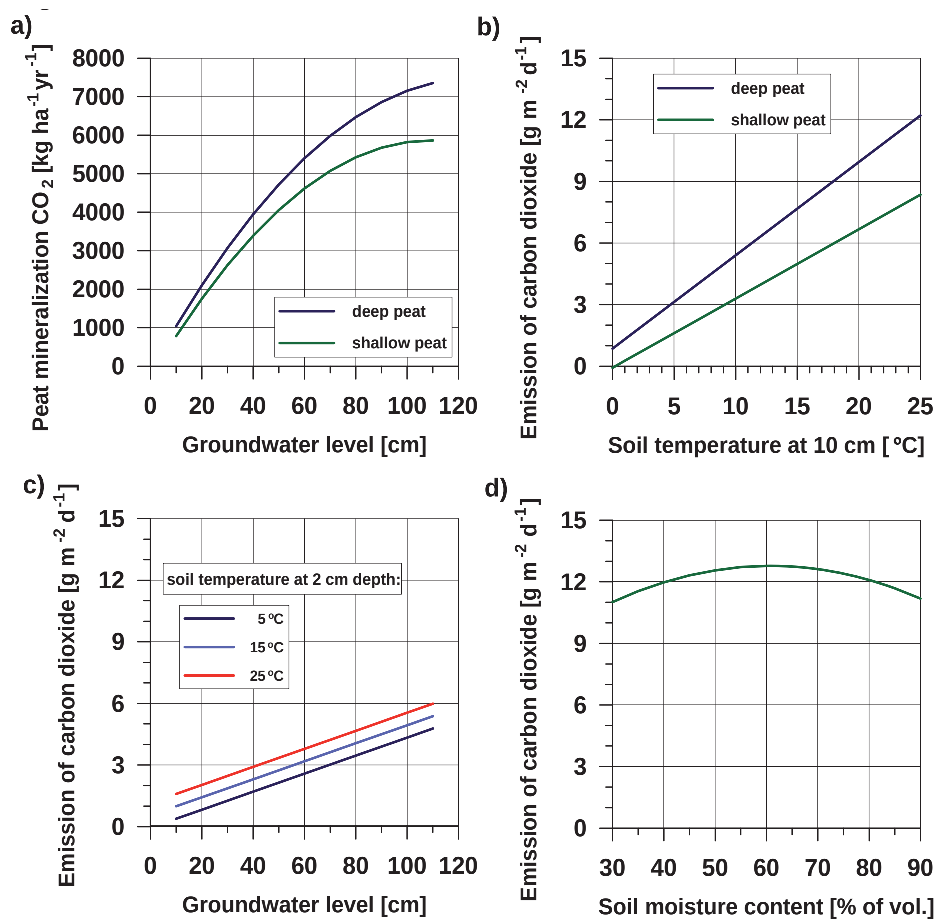

Figure 3.3. *The dependence of carbon dioxide emissions rate a) on groundwater level; b) on soil temperature at 10 cm for shallow and deep peat; c) on groundwater level and soil temperature at 2 cm depth; and d) on soil moisture content (Renger et al., 2002; Mundel,*

occurred at a moisture contents between 58% to 68% by volume (Figure 3.3d, Szanser, 1991).

The relationship between CO_2 emission rate and soil temperature means that the intensity of the peat mineralization process (and thus of CO_2 emission) varies seasonally, being highest between June and September. In the summer months, the highest values are observed between 12 noon and midnight, and the lowest values in the early morning (Ilnicki & Iwaszyniec, 2002). For cultivated peat soils, it is only during the summer months that carbon uptake by plants can compensate sufficiently for peat mineralization to render the system a net sink of CO_2 (Lohila *et al*., 2004).

The most important factors regulating CO_2 emission from drained peat soils are the distributions of organic matter and O_2 in the top layer. The distribution of O_2 in drained peat soils tends to be spatially non-uniform. The degree of aeration depending upon the distance to the next field drain pipe; and this high spatial variability is enhanced by

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the spatial variability of peat properties and processes (Van den Pol-van Dasselaar *et al*., 1998a; Gnatowski *et al*., 2000; Brandyk *et al*., 2003). Thus CO_2 emission is expected to increase with increasing depth of water table and, therefore, increasing aeration. Peat mineralization and CO_2 emission are most intense when the water table is 90 cm below the soil surface. For peat soils in Florida, lowering of the water table from 50 cm to 90 cm depth increased the annual carbon dioxide emission from 40 000 kg ha^{-1} to 75 000 kg ha^{-1} (i.e. from 11 g m^{-2} to 20 g m^{-2} per day). In the Netherlands, the same decline in groundwater level led to an increase of CO_2 emission rate from 10 000 kg ha^{-1} yr^{-1} to 30 000 kg ha^{-1} yr^{-1} (Wösten & Ritzema, 2001). Further lowering of the water table results in drying of the upper peat layers, impeding peat mineralization and reducing CO_2 emission (Mundel, 1976; Szanser, 1991; Augustin, 2001; Renger *et al.*, 2002). According to Okruszko (1993) the largest amount of CO_2 is emitted to the atmosphere from newly drained slightly decomposed peat, in which the moorshing process is in its early stages.

The impact of cultivated mycorrhizal plants on the evolution of CO_2 from soils has also been described (Linderman, 1988; Christensen & Jakobsen, 1993; Larsen *et al*., 1998; Green *et al*., 1999; Ravnskov *et al*., 1999). These authors suggest that mycorrhizae have induced changes in composition of soil micro-organism populations in the mycorrhizosphere and have interacted in the root-free mycosphere with bacteria. Mycorrhizae may modify the effect of elevated CO_2 on the free-living rhizosphere organisms by altering release from roots and by interacting differentially with free-living microorganisms. These direct and indirect interactions may be important for the response of terrestrial ecosystems to elevated CO_2. Diaz *et al*. (1993) proposed that an increase in carbon release into the rhizosphere might lead to an increase in nutrient immobilization by the microbial biomass, resulting in nutrient limitation of plant growth.

About 70-80% of the long term subsidence of drained agricultural peat soils is caused by peat oxidation and mineralization (Eggelsmann, 1976; Okruszko, 1993; Jurczuk, 2000; Ilnicki, 2003). Thus CO_2 emissions can be calculated very roughly on the basis of the annual long-term subsidence rate, the bulk density and the carbon content of the surface peat layers (Kasimir-Klemedtsson *et al*. 1997, Czaplak & Dembek, 2000). The results of such calculations and measured carbon dioxide emissions for bogs and fens that have been drained for agriculture, derived from literature, are presented in Tables 3.2 and 3.3 including the methods of measurements on the top of each table.

There are few data on carbon dioxide emissions for agriculturally used bogs, which emit about 16 000–17 000 kg CO_2 ha^{-1} yr^{-1} (Table 3.2). Thus oligotrophic peatlands, and especially unfertilized bogs, appear to be less subject to peat oxidation than are fertilized peatlands and eutrophic fens, where most studies have been carried out (Table 3.3).

For ploughed fens (arable land), annual CO_2 emissions in the median of 41 100 kg CO_2 ha^{-1} can be expected, with a high range of variation. The gas flux may depend on the degree of disturbance, e.g. soil labour intensity, as indicated by high gas emissions from row crops compared to cereals and field grass. On one hand, the climatic zone seems to have little influence in Europe, as highest emissions were reported for Swedish peatlands. On the other hand, Finnish studies indicate lower CO_2 emissions in the median of about 22 000 kg CO_2 ha^{-1} yr^{-1}. For boreal Canadian peatlands generally low emissions of 7 000 kg CO_2 ha^{-1} yr^{-1} were detected. Fens under grassland show carbon dioxide emissions in the median of 15 000 to 17 000 kg

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Table 3.2. *Carbon Dioxide Emissions from Bogs Drained for Agricultural Use*

Location	Land use	Groundwater level (m below soil surface)	pH-liming fertilization	CO_2 emission ($kg\ ha^{-1}\ yr^{-1}$)	References
		Estimates from field observations on peat subsidence			
NW Germany	arable land	drained	fertilized and limed	16 100	Eggelsmann & Bartel, 1975; Höper & Blankenburg, 2000
NW Germany	grassland	drained	German Bog Culture (fertilized and limed)	17 700	Kuntze, 1992
		Direct measurements in the field– bare soil method			
Sweden	grassland	drained		12 800	Hillebrand, 1993
		Direct measurements in the field – NEE method			
S Germany	fallow, heathland	drained (50 years ago) annual mean: 0.29; amplitude: 0.54	unfertilized, former peat cut area	16 200 ±2 600	Drösler, 2005
S Germany	fallow, heathland	drained (50 years ago)	unfertilized	9 000 ±1 700	Drösler, 2005
Russia	grassland	drained	peat cut	20 000	Krestapova & Maslov, 2004

Table 3.3. *Carbon Dioxide Emissions from Fens Drained for Agricultural Use*

Location	Land use	Groundwater level (m below soil surface)	pH-liming fertilization	CO_2 emissions ($kg\ ha^{-1}\ yr^{-1}$)	References
		Direct measurements in lysimeters– bare soil method			
NE Germany	no	constant: 0.3	unfertilized	10 500-14 300	Mundel, 1976
NE Germany	no	constant: 0.6	unfertilized	14 600-20 600	Mundel, 1976
NE Germany	no	constant: 0.9 / 1.2	unfertilized	13 700-24 500	Mundel, 1976
Temperate			**median**	**14 500**	
		Estimates from field observations on peat subsidence			
Poland, Biebrza	arable land	0.7-0.9	acidic fen, fertilized	41 100	Okruszko, 1989; recalculated by Höper, 2002
NW Germany	arable land	0.8-1.8	calcarous fen, fertilized	38 900-60 500	Eggelsmann & Bartels, 1975; recalculated by Höper, 2002
S Germany (Donaumoos)	arable land	drained	acid to neutral fens, fertilized	24 200-36 300	Schuch, 1977; recalculated by Höper, 2002
Sweden	arable land, cereals	drained		31 000-62 000	Kasimir-Klemedtsson *et al.*, 1997

Sweden	arable land; row crops	drained		62 000-92 000	Kasimir-Klemedtsson *et al.*, 1997
Temperate and boreal	**arable land**		**median**	**41 100**	
Poland, Biebrza	grassland	0.5-0.7	acid fen, fertilized	31 500	Okruszko, 1989;
Poland	grassland			10 000-18 000	recalculated by Höper, 2002
NE Germany	grassland	drained	acid fens	24 200	Czaplak & Dembek, 2000
S Germany	grassland	summer: 1.0-2.0	acid to neutral fens, fertilized	16 900	Lorenz et al., 1992
The Netherlands	grassland	0.7-1.0	acid fen, fertilized	14 100-16 900	Weinzierl, 1997; recalculated by Höper, 2002
The Netherlands	grassland			8 000-30 000	Schothorst, 1976; recalculated by Höper, 2002
Sweden	grassland			15 000-30 000	Kasimir-Klemedtsson et al., 1997
Temperate and boreal	**grassland**		**median**	**16 900**	Kasimir-Klemedtsson et al., 1997
		Direct measurements in the field– bare soil method			
Canada	horticulture	0.7-1.2	cultivated since 10-20 years (sites 1+2) between rows (bare)	7 200-8 300	Glenn *et al.*, 1993
Canada	arable land	0.2-0.9	bare plot (sites 3+6)	5 900-6 400	Glenn *et al.*, 1993
Canada	grass fallow	>0.5 drained	unfertilized (site 7) (vegetated plots)	7 000	Glenn *et al.*, 1993
Boreal, Canada	**arable land, grassland**		**median**	**7 000**	
Finland	grassland	0.2-1.2	limed, fertilised	14 400-14 700	Nykänen *et al.*, 1995
NW Germany	grassland, rewetted	winter: 0.1-0.4 summer: 0.5	unfertilized	14 100-17 600	Meyer *et al.*, 2001
NW Germany	grassland	winter: 0.3-0.5 summer: 0.6	unfertilized	15 100	Meyer *et al.*, 2001
Temperate and boreal	**grassland**		**median**	**14 700**	
		Direct measurements in the field– NEE-method			
Finland	grass, barley		limed, fertilised	14 700-27 500	Meyer *et al.*, 2001
Finland	grass, barley	0-0.5	limed, fertilised	12 100-30 400	Maljanen *et al.*, 2001
Finland	grass, barley	0-0.65	limed, fertilised	16 900-30 400	Maljanen *et al.*, 2004
Boreal	**grass, barley**		**median**	**22 200**	
		Direct measurements in the field – micrometeorology			
Finland	grass, barley	0.8	limed, fertilised	2 900-7700	Lohila *et al*, 2004
Boreal	**grass, barley**		**median**	**5 300**	
The Netherlands	grass	0.3-0.5	fertilised	11 000	Langeveld *et al.*, 1997
Temperate	**grass**		**median**	**11 000**	

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CO_2 ha^{-1} yr^{-1}. No effect of the climate zone on CO_2 emission from fens under grassland can be derived from the present studies. Naturally calcareous fens as well as limed peatlands will mineralize more quickly than acid peatlands.

Finally, a general influence of the different assessment techniques for the carbon dioxide emissions on the result cannot totally be ruled out. Nevertheless, taking only results from the net ecosystem exchange (NEE) method, which takes into consideration all relevant fluxes of carbon dioxide from and into the atmosphere, only few numbers would be available. The results estimated from peat subsidence and from gas emission measurements from bare plots are not totally different from those measured with the NEE method. The micro-meteorological methods tend to lead to lower results than the other methods but there are only few results available using this method.

3.4. Methane fluxes

The observed surface flux of CH_4 is the net balance of microbial production and consumption of this gas in soil. The rate of emission of this gas from wetlands depends on many factors including temperature, depth of water table, vegetation stage and intensity of plant production. Drainage generally turns peat soils from sources to sinks of CH_4 as a result of reduced production of CH_4 in the waterlogged peat and enhanced consumption of CH_4 in the aerated zone of the surface peat. Methanotrophic bacteria oxidise CH_4 in the aerobic layer of the peat and thus convert it to CO_2, which has a lower global warming potential but a longer half-life in the atmosphere. Well-drained cultivated peat soils tend to oxidize CH_4 at rates that are similar to those in cultivated mineral soils. However, poorly drained fields or grasslands may still emit CH_4 when the water table rises, e.g. during periods of snowmelt or rain. Table 3.4 shows published data on annual CH_4 fluxes derived from chamber measurements on agricultural peat soils. The results indicate that most of the sites studied have been net sinks of CH_4, but that emission at low rates occurs at some times during the year. The annual fluxes range from -4.9 to 9.1 kg ha^{-1} yr^{-1}.

Soil temperature, pH, nitrogen status and moisture mainly control methane oxidation in soils. Methane oxidation is less temperature-dependent than CH_4 production (Dunfield *et al*., 1993). This explains the lack of diurnal variation in CH_4 consumption observed in continuous measurement of trace gas fluxes (Maljanen *et al*., 2002). There are observations of rising pH increasing the rate of CH_4 oxidation (Brumme & Borken, 1999) or studies where no response to pH was found (Dörr *et al*. 1993). The population of CH_4 oxidizing bacteria can be adapted to very different pH conditions; in arable soils the populations tend to have higher optimum pH than in forest soils. A small decrease of pH in the range of 7.6–7.1 can cause a strong inhibition of CH_4 consumption in an agricultural soil (Hütsch, 1998) whereas in forest soils the optimum may be found within the range from 4.0 to 7.5 (Saari *et al*., 2004).

An important factor diminishing CH_4 consumption in agricultural soils compared to native soils is nitrogen amendment. Ammonium ions are able to inhibit CH_4 oxidation both by competitive inhibition of the enzyme methane mono-oxygenase and through the resulting decrease in pH when ammonium is applied to soil (Hütsch, 1998). In addition to ammonium, nitrate or nitrite ions reduce the activity of CH_4 oxidising bacteria (Reay & Nedwell, 2004). Decreasing moisture concentration stimulates CH_4 oxidation (Smith *et al.*, 2000) but extreme drought can completely

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Table 3.4. *Methane Flux Rates on Cultivated Peat Soils*

Location	Crop	CH_4 (kg ha^{-1} yr^{-1})	Reference
Canada	Onion	0.2	Glenn *et al.,* 1993
	Celery	-0.6 ± 0.04	
	Grass	-1.6	
Eastern Finland	Grass	1.0 ± 2.0	Nykänen *et al.,* 1995
The Netherlands	Grassland	-0.3 ± 0.1	Langeveld *et al.,* 1997
Southern Germany	Meadow	-1.2 ± 0.8	Flessa *et al.,* 1998
	Rye	-0.2	
	Maize	-0.3	
The Netherlands	Grassland	-0.1 ± 1.1	Van den Pol-van Dasselaar *et al.,* 1999
NW Germany, fen	Grassland drained	-0.4 ± 1.4	Meyer, 1999; Meyer *et al.,* 2001
	partially rewetted	-1.2 ± 0.3	
Eastern Finland	Barley	-4.9 ± 0.1	Maljanen *et al.,* 2003a
	Grass	-1.1	
Western Finland	Barley	-1.8 ± 0.8	Maljanen *et al.,* 2004
	Grass	-2.5 ± 1.0	
Southern Finland	Barley	-0.5 ± 0.1	Regina *et al.,* 2007
	Grass	-0.3	
Northern Finland	Barley	-0.2 ± 5.1	
	Grass	3.6 ± 9.1	

inhibit the activity (van den Pol-van Dasselaar *et al*., 1998b). Physical properties of soils are a key factor determining the soil moisture conditions and the activity of CH_4 oxidisers. Soil structure regulates CH_4 oxidation both by determining the rate of CH_4 and oxygen diffusion into the soil (Dörr *et al*., 1993; Saari *et al*., 1997; Ball *et al*., 1997) and the available surface area suitable for the colonization by methanotrophic bacteria (Bender and Conrad, 1994). Low bulk density in cultivated peat soils compared to mineral soil favors CH_4 oxidation since increasing soil bulk density has been found to limit CH_4 oxidation (Smith *et al*., 2000). On the other hand, fluctuations in the height of water table may change some cultivated peat soils from sinks to sources of CH_4 (Nykänen *et al*., 1995; Regina *et al*., 2007). Tillage breaks down part of the soil microsites of high CH_4 oxidation activity in cultivated soils and it may also lower the diffusion of CH_4 from the atmosphere into the soil as a result of disruption of the soil structure (Ball *et al*., 1999). Other factors reducing CH_4 oxidation in agricultural soils compared to native soils are the use of pesticides (Topp, 1993; Boeckx *et al*., 1998; Prieme & Ekelund, 2001) and irrigation (Kessavalou *et al*., 1998).

3.5. Nitrous oxide emissions

The losses of gaseous nitrogen compounds from organic soils are controlled by the following factors, which limit the microbial processes nitrification and denitrification: quantity of available organic carbon, content of nitrate (NO_3^-), soil moisture, soil aeration, pH and temperature (Tables 3.5 and 3.6). High content of degraded organic matter rich in carbohydrates effectively increases the rate of denitrification and the evolution of N_2O. High concentrations of organic compounds, such as carbohydrates, amines, amides and amino acids in the root exudates of cultivated plants, further favor denitrification. Denitrification activity and the diffusion of its products from soil to the atmosphere are related to the moisture content of the soil. The effect of water is attributed to its function as a regulator of the diffusion of O_2 to the sites of microbiological activity. The rate of the conversion of NO_3^- to gaseous forms is connected with the content of low molecular weight humus substances rather than by concentration of NO_3^-. The content of NO_3^- influences the order of this process. High content of NO_3^- suggests zero order kinetics and low concentrations of NO_3^- show first order kinetics. This substance impacts the N_2/N_2O ratio (Table 3.7). High content of NO_3^- favors the creation of N_2. Behrendt *et al*. (1996) showed that groundwater level and nitrate outputs were closely related. They suggested that strongly humified peat would lose more nitrates than a less humified fen at approximately the same groundwater level. N loss by leaching was greater on shallow fens underlain by sand than on thick fens. Under cultivated land, the N flux was up to 5 times larger than under grassland (Zeitz & Velty, 2002).

The optimum pH for the evolution of N_2O is below 6.5. Denitrification is slow but still significant below pH 5, and absent below pH 4. Soil acidity also impacts on the N_2O/N_2 ratio of the end product of the process (Table 3.6).

Denitrification is very sensitive to changes in temperature. This transformation occurs slowly at 5°C and the increase of the

Table 3.5. *Environmental Consequences of Agriculture with Regard to Nutrient Cycles (Rosswall, 1978)*

Activity	Environmental consequences	Major element involved
Drainage	Decrease of organic matter;	C, N;
	Leaching losses;	N, P;
Cultivation	Decrease of organic matter;	C, N;
	Erosion losses;	C, N, P;
	Mining of nutrients	N, P, K;
Clear cutting	Leaching losses;	N, P;
	Mining of nutrients;	N, P, K;
Irrigation	Leaching losses;	N, P;
	Salinization;	K, Ca, Na;
Fertilization	Decrease of organic matter;	C, N;
	Leaching losses;	N, P;
	Losses of gaseous forms of nitrogen	N;
Grazing	Ammonia volatilization;	N;
	Erosion losses	C, N, P

Table 3.6. *Factors Affecting the Proportion of N_2O to N_2 Produced in Denitrification (Rosswall, 1978)*

Factor	Response on N_2O / (N_2O + N_2)
NO_3^- concentration	Increasing NO_3^- increases ratio
NO_2^- concentration	Increasing NO_2^- increases ratio
O_2 concentration	Increasing O_2 increases ratio in soils and most bacterial cultures
N_2O	Appears to induce synthesis of N_2O reductase which decreases ratio
Length of anoxia	The ratio is initially low (0.2-0.4) then increases (0.4-0.9) and finally drops to 0
pH	Decreasing pH increases ratio when NO_3^- concentrations are moderate or high
Sulfide	Increasing sulfide increases ratio
Carbon source	Increasing carbon increases ratio

temperature accelerates the rate of loss. The optimum for the reaction is at 25^{o}C. Increase in soil temperatures as a result of global warming may increase emissions of N_2O from peat soils by enhancing the decomposition of peat and the nitrogen transformation processes.

The emissions of nitrous oxide from natural peatlands are usually negligible, but lowering the water table enhances mineralization of the organic matter thus triggering the emissions (Martikainen *et al*., 1993; Merbach *et al*., 1996; Augustin *et al*., 1998). The increase of nitrification and the creation of low molecular humic substances after drainage may be the reason for enhanced N_2O evolution (Lång *et al.*, 1994). Nitrate produced in the upper layers of drained soils as a result of mineralization of humic substance moves to the deeper layers of the profile and is reduced to N_2O. Drained peatlands are remarkable sources of nitrous oxide with annual fluxes varying between 2 and 56 kg N_2O ha^{-1} (Table 3.7).

The flux estimates have high uncertainties due to high variability in time and space. The high spatial variability of nitrous oxide emission from drained peat soil was shown by Van den Pol - van Dasselaar *et al*., (1998a) who reported coefficients of variation from 170 to 500%. The variation can be caused by the properties of the peat, fertilization rate, temperature and fluctuations in soil moisture and water table level. Nitrous oxide emission from bogs is low due to low pH and low total nitrogen content whereas in more nutrient rich fens higher N_2O emissions have been observed (Joosten & Clarke, 2002). In the conditions of Dutch pastures on drained soils high fluxes of N_2O (from 14 to 61 kg ha^{-1} yr^{-1}) were found 1-3 weeks after fertiliser application, but in the summer during the dry period the flux was at a lower level (Langeveld *et al.*, 1997). In drained fen peat soil in Germany as well as in Finland the high emissions have been found to be connected with periods of high soil moisture (Nykänen *et al.*, 1995; Augustin *et al.*, 1998). The emissions of nitrous oxide follow the soil temperature and thus the highest emissions in agricultural soils occur in the afternoon (Maljanen *et al.*, 2002). Since the mineralization of peat organic matter itself produces substrate for denitrification, direct fertilization effects are not always seen (Augustin *et al.*, 1998, Regina *et al.*, 2004). With very high application rates (480 kg N ha^{-1}) the

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Table 3.7. *Nitrous Oxide Flux Rates on Cultivated Peat Soils*

Location	Crop	N_2O–N (kg ha^{-1} yr^{-1})	Reference
Eastern Finland	Grass	7.8-9.3	Nykänen *et al.,* 1995
Eastern Finland	Barley	8.3-8.4	Maljanen *et al.,* 2003b
	Grass	11.0	
Western Finland	Barley	5.4-11.3	Maljanen *et al.,* 2004
	Grass	1.7-3.8	
Southern Finland	Barley	6.2-24	Regina *et al.,* 2004
	Grass	5.0-9.9	
Northern Finland	Barley	7.3-18.8	
	Grass	2.6-5.3	
The Netherlands	Grassland	8.8-38.5	Velthof *et al.,* 1996
The Netherlands	Grassland	8.9-39.0	Langeveld *et al.,* 1997
Southern Germany	Meadow	4.2 ± 19.8	Flessa *et al.,* 1998
	Rye	19.8	
	Maize	56.4	
NW Germany	Grassland drained	5.0 ± 5.1	Meyer, 1999; Meyer *et al.,* 2001
	partially rewetted	5.8 ± 7.0	

fertilisation effect can be seen, however (Augustin *et al*., 1998). As in mineral agricultural soils, also in peat soils a remarkable part of the annual flux can occur outside of the vegetation period when there is no plant uptake of nitrogen. The weather conditions, e.g. the frequency of freeze-thaw cycles, thickness of snow cover and possible ice layers, partly determine the magnitude of the winter fluxes (Maljanen *et al*., 2003b, Regina *et al*., 2004).

3.6. The importance of cultivated peat soils in regional budgets of greenhouse gas emissions

Cultivated peat soils play important roles in the greenhouse gas budgets of the many countries with large proportions of organic soils. Several estimates of greenhouses gas emissions have been made for European peatlands. The net annual CO_2 emission rate from non-irrigated peatlands in Poland has been estimated at 14.5 million tonnes, which is around 4% of the country's total annual CO_2 emissions (Czaplak & Dembek, 2000). In Sweden, the contribution of CO_2 emissions from drained peat soils to the national budget of this gas is around 10% (Eriksson, 1991), whilst for the Netherlands the emission of CO_2 from drained peat soils is estimated at 1.3% of the national total (Langeveld *et al*., 1997). In the national inventory of Finland under the United Nations Framework Convention on Climate Change (UNFCCC), peat-based emissions comprise one-third of the total reported net greenhouse gas emissions when sinks and emissions from the land use, land-use change and forestry sectors are included

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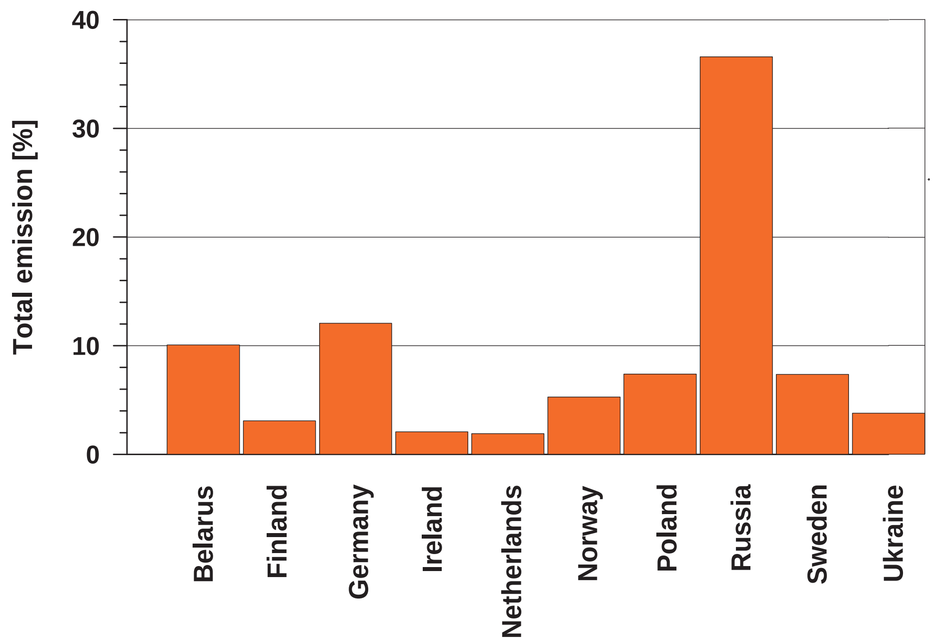

Figure 3.4. *The contribution of 10 selected countries to the total greenhouse gas emissions from peatlands in Europe (Byrne et al., 2004, modified).*

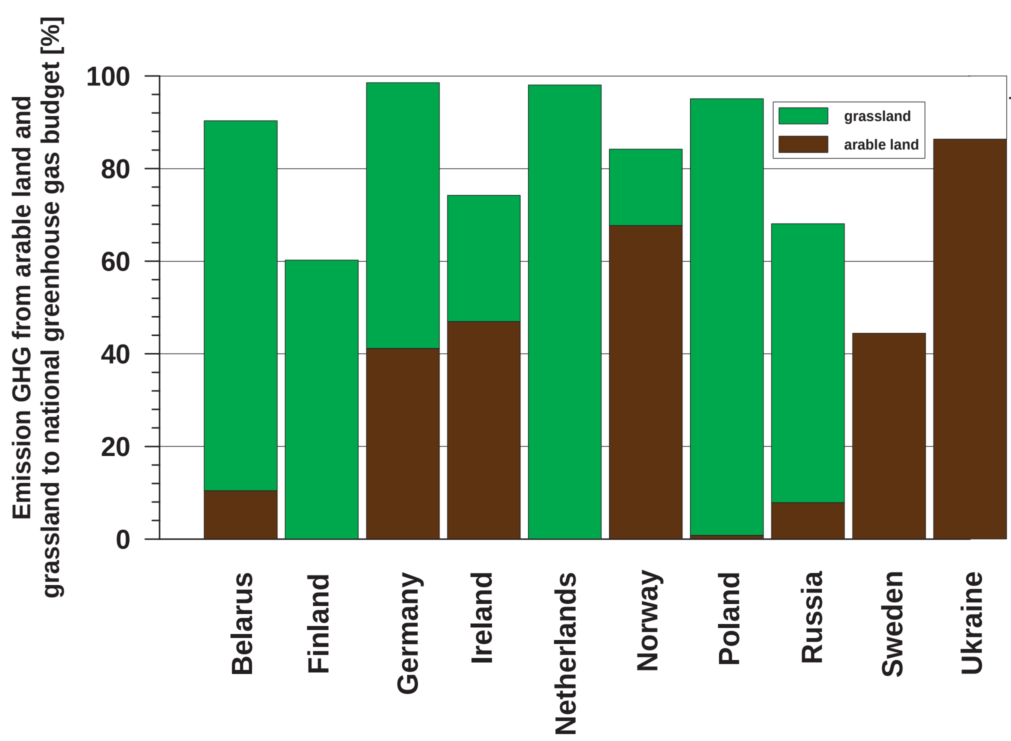

Figure 3.5. *The contribution of emission of greenhouse gases from arable land and grassland to the national peatland greenhouse gas budget (includes emissions from agriculture, natural mires, forestry and peat extraction) from peat soils of selected countries (Byrne et al., 2004, modified).*

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(Lapveteläinen *et al*., 2007). Most of the emissions come from the combustion of peat for energy, but emissions from the agricultural use of peatlands amount to about 25% of the peat-based emissions.

The greenhouse gas budget for bogs and fens under different types of use has been investigated for the European countries and a default estimate of the European peatland greenhouse budget assuming a 100-year time horizon has been published in CarboEurope GHG (Byrne *et al*., 2004). This budget includes emissions from undisturbed mires, and from peatlands used for peat extraction, forestry, grassland and crops. Most European countries are sources of CO_2, but Finland, Sweden, United Kingdom are sinks. However, all of the countries studied are sources of CH_4 and N_2O. The contributions of ten selected countries to the European peatland GHG budget are presented in Figure 3.4.

The results indicate that Russia is responsible for 37% of Europe's greenhouse gas emissions from peatlands. Germany emerges as the second largest emitter (12%), whilst Belarus and Poland respectively produce 10% and 7.4% of the total. The authors of the report express the opinion that the main reason for the high contributions of some countries to the total budget is their intensive utilisation of peatlands as cropland and grassland, the largest CO_2 emitters being countries with large agricultural areas (Russia, Germany, Belarus, Poland). Russia and Germany are also classified as the largest N_2O emitters (Byrne *et al*., 2004). The authors' conclusions are confirmed by the very high contributions of GHG emissions from arable and grassland areas to the national greenhouse gas budgets for peat soils in the countries mentioned (Figure 3.5).

Vomperskij (1994) concluded that the agricultural use of drained peatlands in Russia was responsible for the emission of 4.5 millions tons of carbon per year. Long-term monitoring of Estonian peatlands that are used for agriculture indicates that peat losses caused by mineralization may be up to 10 000-15 000 kg of dry organic matter per hectare per year (Tomberg, 1992). This is at least 405 times the peat accumulation rate in intact mire. CO_2 emissions to the atmosphere are probably in the region of 5 million tons annually. Consequently ameliorated peatlands may be among the most important sources of greenhouse gases (especially CO_2) in Estonia (Paal *et al*., 1998).

Long-term drainage and conversion to farmland of peatlands has repeatedly caused major subsidence and carbon loss, for example in the Sacramento-San Joaquin Delta (California, USA), the Everglades (Florida, USA) and the East Anglian Fens (UK) (Hutchinson, 1980). Oxidation of the Everglades peat has released 13 500 kg CO_2 annually from each hectare of drained soil and the total carbon outflow is on the order 10 million tonnes. Carbon losses from the entire Florida Everglades may be much heavier and this may be one of the world's largest organic carbon sources (Smil, 1985).

3.7. Possibilities for reduction of GHG emissions from agricultural peat soils through management

Lowering the water table and moisture content in the upper layers of peat soils required by agricultural use of peat soils induces biological, chemical and physical changes (decomposition, mineralization, subsidence, greenhouse gas emissions). Future changes in climate may alter the conditions in many peat soils and thus affect the fluxes of greenhouse gases. Adaptation to the future climate may require strategies for managing e.g. the water status, of these soils. According to several climate models, the summertime precipitation is predicted to decrease, at

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least in northern Europe. This combined with higher temperatures would increase mineralization of the organic matter in peat soils. Furthermore, more freeze-thaw cycles during the wintertime in areas where the winters used to be characterized by permanent snow cover will increase the annual N_2O and CO_2 emissions from agricultural peat soils. Proper management must be performed in order to minimise the negative consequences of peatland utilization for soil conservation and greenhouse gas loading of the atmosphere.

Abandoned agricultural peatlands can be rewetted if they are no longer needed for agricultural production. The average emission factors from bogs and fens under grassland, arable and restoration conditions are presented in Table 3.8. The highest emission factors are generally found for drained fens under grassland and arable utilisation, but CO_2 emission is also relatively high for arable bogs. The methane flux values for drained bogs and fens, as well as N_2O fluxes of cultivated bogs, are relatively low; whereas the N_2O flux from drained arable fens is very high. Joosten & Clarke (2002) published similar ranges of greenhouse gas emissions from cultivated bogs and fens taking into account the type of use. Rewetting and restoration of drained and cultivated peat soils reduce emissions of CO_2 and N_2O but increase emissions of methane to the atmosphere.

Joosten & Augustin (2006) reviewed current literature concerning the rewetting of peat soils, and concluded that information on the consequences of this activity as a measure for long-term reduction of greenhouse gas emissions, especially in Central Europe, remains incomplete. On the basis of available data and their own research, these authors distinguished three phases after reflooding of drained peat soils in terms of greenhouse gas (CH_4 and CO_2) dynamics. The first phase, beginning at the time of rewetting, has an extremely negative effect on climate because - according to their simulation - extremely high CH_4 emission occurs in conjunction with low net CO_2 uptake. The second phase has a slightly positive climate effect because methane emission is strongly reduced, whilst carbon dioxide uptake reaches its maximum. During the third and final phase after rewetting, both

Table 3.8. *Average Emission Factors Based on Measured Fluxes from Bogs and Fens under Different Management Types from European Peatland (Byrne et al., 2004, modified)*

Type of management	Emission factor		
	CO_2 (kg C ha^{-1} yr^{-1})	CH_4 (kg C ha^{-1} yr^{-1})	N_2O (kg N ha^{-1} yr^{-1})
Bog (ombrotrophic)			
Grassland	2 350	2.00	0.01
Arable	4 400	0.00	0.00
Restoration	620	15.00	0.02
Fen (minerotrophic)			
Grassland	4 120	0.40	5.05
Arable	4 090	-0.20	11.61
Restoration	-	12.40	0.64

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CH_4 release and net CO_2 uptake are slow, as in pristine mires, so that this phase has a neutral climate effect. The authors analysed the example of the restoration of 17 drained fens and raised bogs covering 42 000 hectares in Belarus. Using published data for net greenhouse gas fluxes from these sites, four hypothetical "emission classes" were identified. The first class estimated the global warming potential for drained conditions and the next three classes described the changes in the global warming potential values during the three phases of rewetting. As the duration of each rewetting phase is unknown, calculations were performed for three scenarios over 100 years of restoration. Even in the most pessimistic scenario (phase 1 - 50 years, phase 2 - 1 year and phase 3 - 49 years) the rewetting process was predicted to reduce the emissions of peat soils in Belarus by more than 30 million tonnes of CO_2 equivalents. So long time of reduction of GHG emissions from abandoned organic soils is confirmed by the research performed for Finnish peatlands. CO_2 and N_2O emissions from abandoned organic cropland soils do not generally decrease with time after agricultural use. For example N_2O emission from croplands on organic soils can still be high even after 20-30 years of abandonmend (Maljanen *et al.*, 2007).

Where peatlands cannot be rewetted because they are still needed for agricultural production, a reduction in emissions of CO_2 and N_2O could be achieved by changing from arable to grassland. In such situation CO_2 emissions from ploughed organic soils can be as much as 20% higher than from grassland and pastures (Augustin, 2001, Maljanen *et al*., 2007). On the other hand, if the grassland is used for grazing this would introduce additional costs: the emission of CO_2 from cows and their excrements. Langeveld *et al.* (1997) estimated that the daily emission of CO_2 from one cow using peatlands as a pasture is around 21.3 kg.

Szymanowski (1999) and Czaplak & Dembek (2000) compared the amount of CO_2 emitted from peat soils at different stages of moorshing with and without irrigation. Irrigation reduced the amount of CO_2 emitted by about 20%. From a literature review of this subject, Ilnicki & Iwaszyniec (2002) concluded that a greater reduction of CO_2 emissions can be achieved by flooding drained German fen soils than by irrigating them. As rewetting of a drained organic area can inhibit the nitrification process, only permanent flooding without a change in moisture saturation, which will prevent the emission of N_2O, was recommended by Meyer *et al.* (2001). This type of management requires a large amount of water, and the vegetation can die off immediately after reflooding. Therefore precise regulation of the groundwater level seems to be crucial (Zeitz & Velty, 2002).

The position of the water table is one of the most important factors influencing peat conditions and soil processes. Renger *et al.* (2002) identified the water level in drained fen grassland soil as the main factor influencing crop production, the rate of peat mineralization and gas emission (CO_2, CH_4, N_2O). They concluded that the optimum water level for these three attributes was 30 cm below the soil surface. In their opinion, about 90% of the best possible plant crop can be attained at this water level, the mineralization rate of peat can be reduced by as much as 30-40% of the maximum value, and greenhouse gas emissions will reach only 50–60% of their maxima.

It has been suggested that afforestation may reduce the emissions of greenhouse gases from cultivated peat soils. Kasimir-Klemedtson *et al.* (1997) reviewed greenhouse gas emissions from farmed organic soils in Sweden, Finland and the Netherlands, concluding for Sweden and Finland that it might be possible to reduce emissions from peatlands by afforesting them. Peatlands drained for forestry emit

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N_2O at a lower rate than peatlands used in agriculture. Forested drained peatlands still lose soil carbon but the growth of the tree stand tends to compensate for this loss. In Finland more than half of the original peatlands were drained for forestry and agricultural use. Since late 1960s the afforestation of agricultural organic soils has been performed in Finland in order to reduce the cultivation area in the country (Mäkiranta *et al.*, 2004). Afforestation of former cultivated soils is considered one of the means of reducing CO_2 emissions from these areas. After changing the type of use from cultivation to forestry the decomposition rate of organic material becomes slower and the carbon emission in the area is lower after afforestation. The time since afforestation, tree species used, as well as peat depth and its quality are the main factors influencing the magnitude of the changes after the new use of drained peat soils. The afforestation of drained peat soils leads to a decrease in CO_2 flux but a serious drawback is that some afforested organic soil emit higher values of N_2O than the cultivated organic soils, even 30 years following afforestation (Maljanen *et al.*, 2004; 2007). The N_2O flux was only slightly lower from abandoned fields than from the peat soils under barley. This issue requires more research to provide data for future land use policies. More details about afforestation and the influence on greenhouse gas balance will be presented in chapter 4.

Reduction in tillage intensity, especially the adoption of no-tillage cropping is recognized as an effective management technique to enhance C storage in soil (Janzen *et al*., 1998). It might be possible to slow down the loss of carbon with reduced tillage also in peat soils. However, with no-till practice there is the risk of higher N_2O emissions which could totally offset the reductions in CO_2 emissions (Li *et al*., 2005). It is well known that tillage practices impact the timing, rates and the distribution of nitrification and denitrification activities in organic and mineral soils (Groffman, 1984). Investigations of the whole soil profile under a wide range of conditions are necessary to completely understand the nature, direction and extent of nitrification and denitrification activity under different tillage systems. It was revealed that no-tillage systems are characterized by higher rates of denitrification and lower rates of nitrification than conventional tillage systems. Lower numbers of nitrifying bacteria in no-tillage soils than in conventional tillage soils from several locations were noted by Doran (1980). Dowdell *et al.*, (1983) suggested lower levels of soil nitrate in direct drilled soils than in ploughed soils and noted that nitrification activity was decreased in direct drilled soils relative to ploughed soils. In contrast, Rice & Smith (1983) noted that rates of nitrification can be higher in no-tillage soils than in conventional tillage soils due to more favorable moisture conditions for nitrification in no-tillage soils. Higher numbers of denitrifying bacteria in no-tillage soils than in conventional tillage soils attributed this effect to the higher water quantity and the less oxidative conditions of no-tillage soils relative to ploughed area. Burford *et al.*, (1981) and Aulakh *et al*., (1982, 1984) as well as Rice & Smith (1982) measured greater amounts of gaseous nitrogen loss from no-tillage soils than from conventional tillage soil. In contrast, Staley (1982) and Vinther *et al*. (1982) did not determine differences in populations of denitrifying bacteria between conventional and no-tillage soils. It is likely that the effect of tillage on gaseous nitrogen losses depends on climate and specific field conditions. There is lack of data on the effects of no-till agriculture on peat soils.

Emissions of greenhouse gases from drained organic soils can be reduced to some extent by practices such as avoiding row crops and tubers, avoiding deep

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ploughing and maintaining a shallower water table (Freibauer *et al.*, 2004). As food production diminishes in many countries, large peat areas are currently taken into bioenergy production. There is also potential for developing the use of crops that can be grown under conditions of high water table (paludiculture), such as reeds for thatching, alder and willow for fuel and biomass, and *Sphagnum* for use in horticulture. Production of such plants is comparable to no-till cultivation and thus possibly can reduce gaseous losses from these fields. However, at present it is lacking research data and it requires field experiment on the greenhouse gas balance of bioenergy production from peatlands.

In many countries agricultural peatlands were needed to ensure food production in the past. Due to current changes in agricultural production it is possible to reduce gaseous emissions or avoid cultivation of peat soils in many areas. The best management and end use depends on local conditions. If peat soils are not needed for production they can be rewetted or burned for fuel. Even if they remain in food production water table management or cultivation technologies can be used to reduce emissions. However, political decisions are often needed to change the management of agricultural soils and there may be socioeconomic consequences that limit these activities in practice.

3.8. Summary

About 14 – 20 % of peatlands in the world are currently used for agriculture and the great majority of these are used as meadows and pastures. For agricultural use, fens and raised bogs have to be drained in order to regulate the air and water conditions in the soil to meet the requirements of cultivated plants. These activities have increased the cultivated areas and the intensity of agricultural production in many countries. Farmed organic soils are productive fields since they have the ability to store a large amount of water and the availability of nitrogen and phosphorus for the plants is high due to peat mineralization in the first years after drainage. The loss of buoyancy of the upper horizons by drainage causes the mechanical compression of the permanently saturated peat layers below the groundwater level. This leads to a lowering of the soil surface level and subsidence. Drainage reduces the emission of methane and increases the emissions of carbon dioxide and nitrous oxide from the peat. The emission rates of the greenhouse gases depend on many factors like soil temperature, groundwater level and soil moisture content. For ploughed fens (arable land), CO_2 emissions in the median of 41.1 Mg CO_2 ha $^{-1}$ year $^{-1}$ can be expected, with a high range of variation. Fens under grassland show carbon dioxide emissions in the median of 15 to 17 Mg CO_2 ha $^{-1}$ year $^{-1}$. Drained peatlands are remarkable sources of nitrous oxide with fluxes varying between 2 and 56 kg N_2O–N ha $^{-1}$ year $^{-1}$. The annual CH_4 fluxes of cultivated peat soils range from a small sink to low emission (from –4.9 to 9.1 kg ha $^{-1}$ yr $^{-1}$).

The position of the water table is one of the most important factors influencing peat conditions and the soil processes, so that precise regulation of the water level seems to be a very important factor. Raising the water level in peat soils reduces emissions of CO_2 and N_2O but increases emissions of methane to the atmosphere. For example, irrigation can reduce the amount of emitted carbon dioxide of about 20% in comparison with the organic soils without irrigation. The rewetting of a drained peatland can inhibit the nitrification process. Sometimes only permanent flooding without a change in moisture saturation, which will prevent the emission of N_2O, is recommended. Abandoned agricultural peatlands can be rewetted if sufficient water is available and they are not needed for agricultural

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production. If they cannot be rewetted, for example because they are still needed for agricultural production, even a change of use from arable to grassland would reduce the emissions. The change from arable land to grassland reduces the emissions of CO_2 and N_2O The comparison of CO_2 emissions from grassland and arable land showed the increase in emission of carbon dioxide of about 20% in case of ploughing organic soils. Taking into account three aims: crop (grass) production, the rate of peat mineralization and gas emission (CO_2, CH_4, N_2O) the groundwater level should be kept at the depth 30 cm below the soil surface under grass utilisation. At this water level about 90% of the optimum plant crop can be reached, the mineralization rate of peat can be reduced even to 30-40% of the maximum values and the greenhouse gas emission values will reach only 50-60% of their maximum amounts.

One of the possible methods to reduce the emissions of greenhouse gases from cultivated peat soils has been thought to be afforestation of these areas especially for Scandinavian countries. Peatlands drained for forestry emit N_2O at a lower rate than the peatlands used in agriculture. Forested drained peatlands still lose soil carbon but the growth of tree stand may compensate for this loss.
The mitigation of the emissions of greenhouse gases from cultivated peat soils can be done by:

- strong limitation on drainage of new natural peatland areas;
- avoidance of arable farming (i.e. favouring grasslands);
- extensive grassland and pasture systems;
- paludiculture
- afforestation;
- rebuilding and renovation of existing drainage systems into sub-irrigation systems;
- rewetting.

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CHAPTER 4:

CLIMATE IMPACTS OF PEATLAND FORESTRY

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4.1. Introduction

Peatlands are an important component of the forested landscape, providing diversity in terms of plant communities, wildlife, hydrologic functions and many environmental services that are valued by society. They also have the capability to be managed sustainably for forest products. Forestry is an important land-use of some peatlands because of the products and productivity potential. The utilization of peatlands for forestry is concentrated in Nordic countries (i.e. Finland, Sweden and Norway) and Russia. In addition peatland forestry has national importance in the United Kingdom, Ireland, Canada and the United States, and the development of tropical peatlands is poised to be an important land use (see Chapter 6).

The prevailing paradigm has been that drainage of organic soils causes carbon (C) loss from the soil to the atmosphere as carbon dioxide (CO_2), accelerating thus the natural greenhouse effect. There are many examples of large C losses from agricultural sites, which are subject to repeated cultivation practices (e.g., tilling, fertilization, re-drainage; see also Chapter 3). It has therefore been reasonable to assume that drainage for forestry would also induce losses of C. This assumption is also used in the Intergovernmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories where default emission factors are provided for peat soils drained for forestry. However, studies from long-term assessments of the C balance on drained peatlands in Finland have shown that soil C storage may in some cases increase as a result of improvements in site productivity. Accordingly, the effects of silviculture, including water management, on the peatland C balance is not unidirectional as in peatland agriculture. Assessing the effects of peatland forestry on global warming potential also requires that the fluxes of the other major greenhouse gases (GHG), methane (CH_4) and nitrous oxide (N_2O), be considered. The climatic impact of peatland forestry is therefore the sum of the changes in the ecosystem C pools and the cumulative changes in GHG fluxes.

Our goal is to review the current knowledge about the climatic impacts of peatland forestry. We review the silvicultural practices used on peatlands in different bioclimatic zones and consider the ecological impacts and the possible changes in C stores and GHG fluxes caused by forestry in peatlands.

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4.2. Forestry in peatlands

4.2.1 Common principles

Forestry in peatlands includes the same silvicultural practices (e.g. felling, site preparation, fertilization) as conducted on upland mineral soils. The fundamental difference is, however, that water management systems (i.e. drainage) are nearly always required when practicing sustainable forestry at these naturally wet sites. Forestry in undrained peatlands is only practiced where there are sufficient periods when the water table is below the surface or frozen allowing operations without degrading the site. Water management is needed to enhance the productivity of the tree stand and to alleviate operational limitations of the peatland. The term "forest drainage" or "forestry drainage" are commonly used to emphasize the idea that the drained sites have been naturally forested, or the reason for drainage is to enable forestry development to take place and that production forestry is practiced at the site.

There are at least three types of drainage systems used in peatland forestry. The most common drainage technique is the pattern ditch system that typically employs a network of open, 80-100 cm deep field ditches that run to a collection ditch. Shallower water furrows are often used in connection with forest regeneration to prevent the water table rising after harvesting. In contrast to these patterned systems, the prescription ditch system employs a single shallow ditch that follows the apparent natural drainage pattern of the land. This system is used mainly in North America. Harvesting, site preparation, fertilization and stand tending practices follow through the rotation.

Peatland types and, consequently, silvicultural practices differ between countries due to the big differences in natural conditions and culture. In the following sections we present the extent and applications of peatland forestry practices in boreal and temperate zones (maritime and continental), with specific examples from Finland, the United Kingdom, Ireland and North America.

4.2.2. Extent of peatland forestry

Drainage is a prerequisite for peatland forestry in most countries. Approximately 15 million ha of peatlands have been drained by ditching for forestry purposes (Paavilainen & Päivänen, 1995), which is ca. 4% of the total area of northern peatlands (ca. 350 million ha, Gorham, 1991). The bulk of the area under peatland forestry is situated in Fennoscandia and Russia, where over 10 million ha of peatlands have been drained for forestry (Table 4.1).

Forestry on undrained peatlands is currently practiced primarily in Canada and the United States. Therefore, in North America, peatland forestry may cover larger areas than indicated by the drainage area in Table 4.1.

The largest area of peatland forestry is found in Finland where ca. 55% of the total peatland area has been drained for forestry. Therefore 25% of Finnish forestry is practiced on peat soils. Drainage of new areas has ceased, but previously drained productive areas are maintained (ditch cleaning and supplementary ditching) annually on an area of ca. 80 000 ha. The tree stand increment in drained peatland forests is 23 million $m^3 yr^{-1}$, being more than a quarter of the total increment of Finnish forests. Standing volume in peatland forests is still increasing, since most of the forest drainage took place between 1960 and 1980, and these forests are beginning to reach maximum growth

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Table 4.1. *Peatland/(wetland[a]) areas drained for forestry in various countries.*

Country	Area drained (10^3 ha)	Reference
Finland	4948	Finnish statistical, 2006
Russia	4000	Minayeva & Sirin, 2005
Sweden	1500	Hånell, 1990
Lithuania	590[a]	Paavilainen & Päivänen, 1995
Latvia	500[a]	Paavilainen & Päivänen, 1995
Estonia	460[a]	Paavilainen & Päivänen, 1995
Norway	420	Paavilainen & Päivänen, 1995
Belarus	280[a]	Paavilainen & Päivänen, 1995
Poland	120	Paavilainen & Päivänen, 1995
Germany	110	Paavilainen & Päivänen, 1995
U.K.	555	Cannell *et al*., 1993; 1996
Ireland	271	Forest Service, 2007
China	70	Paavilainen & Päivänen, 1995
U.S.A.	400	Paavilainen & Päivänen, 1995
Canada	25	Paavilainen & Päivänen, 1995

[a] includes wetlands with no peat or peat forming vegetation

rates. Peatland forests are therefore an increasingly important resource for the forestry sector in Finland.

An area similar to that found in Finland has been drained for forestry in Russia, being concentrated in the northwestern and central-European parts of the country (Minayeva & Sirin, 2005). Drainage has a long history, beginning in the 19th century, and peaking at the same time as in Finland, 1960-1990. The increased forest growth has however, been poorly utilised, and in the absence of ditch maintenance most of the sites have repaludified, often as a result of damming of drains and streams by beavers (Minayeva & Sirin, 2005). Thus, the economic importance of peatland forests in Russia is not as high as it could be.

Peatland forestry has importance in Sweden. There are ca. 5 million hectares of productive forest land on wetlands with mire vegetation and/or peat soil, being ca. 22% of the total forest area of 23 million ha. About 1.5 million hectares of peat-covered soils have been drained for forestry and ca. 1 million hectares of drained soils are classified as productive sites, undergoing normal forestry practices (Hånell, 1990). The sites are generally more nutrient-rich compared to those drained in Finland. In addition to traditional forest drainage, so-called protective ditching is done in wet clear-cut areas to ensure tree stand regeneration.

Peatland forestry is also an important land-use in the baltic countries (Estonia, Latvia, Lithuania), where the proportion of peatlands drained for forestry of the total peatland area is high. However, estimates of the drained area include wetlands with no peat or peat formation, which makes comparisons difficult.

In the Republic of Ireland, peatlands have been disturbed and modified for many purposes, over a long period of time principally for fuel. At present only 18%

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of the original area is in a near-natural condition. The expansion of peatland forestry in Ireland began in the late 1950s. While some of this activity took place on raised bogs in the midlands of the country, most of it was concentrated on the low-level blanket peatlands in the west and the high-level blanket peatlands occurring on mountain ranges throughout the country (Farrell & Boyle, 1990). There are currently ca. 271 000 ha of forestry on peatland (Forest Service, 2007). Of this, 67 000 ha is on raised bogs, 197 000 is on blanket bogs and 7000 ha is on industrial cutaway bogs. The latter is considered to offer potential for the future with 12 000 to 16 000 ha of industrial cutaway peatlands considered to be suitable for afforestation. Drainage of intact peatlands for forestry is no longer practiced and areas of peatland forestry with high conservation value are being restored to functioning wetlands.

The majority of non-permafrost peatlands (13.6 million ha) in North America occur in the boreal zone with permafrost peatlands adding an additional 51 million ha to the inventory. While there is little information differentiating forested versus non-forested peatlands, among all wetland types in the United States, approximately 50% are classified as forested (Trettin *et al.*, 1995). Typically, these peatlands are not managed intensively (e.g. with the use of forest drainage systems). However, in the southeastern US, where there are extensive areas of peatlands with a high productivity potential for *Pinus taeda* (Loblolly pine), forest drainage systems are common. These water management systems are allowed by the law (Sec. 404, Clean Water Act) regulating practices in wetlands, as long as the drainage system does not convert the site to a non-wetland condition. Accordingly, while the water table is altered by the drainage system, the hydrology of the site should still qualify as jurisdictional wetland hydrology. Forestry is also practiced in natural peatlands without artificial drainage systems. The silvicultural system typically consists of clear-cut harvesting followed by natural regeneration.

4.2.3 Silvicultural practices

Boreal zone - Finland

In Fennoscandia drainage by ditching nearly always precedes forestry in peatlands. Forestry in pristine peatlands is rare, but possible, in thin peated spruce swamps, but poor harvesting conditions (soft soil) restrict forestry on these sites. Pattern drainage systems with open ditches (80-120 cm deep, 100-150 cm wide) are used, and the ditch interval is commonly 30-40 metres. In the Baltic countries and Russia much wider intervals (> 100 m) are common.

Peatland forestry is based on the use of natural tree stands (Figure 4.1). Drainage has been concentrated on naturally forested peatlands, and the tree stands are further complemented by natural regeneration after drainage. Afforestation of open, i.e. treeless, peatlands was common practice in connection with drainage in the 1960s, but this practice has ceased because of high planting costs and long rotation times.

The use of natural stands has the benefit of having large initial stand volumes, advancing the possibility for fellings and enabling higher returns. On the other hand, the uneven stand structure of natural peatlands remains throughout the first rotation after drainage, making thinnings less beneficial. The soft soil conditions necessitate harvesting in winter, when the soil is frozen and has greater load bearing capacity. In addition, as the ditches usually have to be cleaned after fellings, thinnings are further postponed. For these reasons only a few thinnings (1-2) usually take place before the final felling, 50-100 years after the first drainage.

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Figure 4.1. *In Finland, forestry in peatlands is based on naturally forested sites. Left, a typical natural pine fen in Central Finland and right, a managed counterpart, drained 50 years ago. (Photos: J. Laine)*

Not all peatland sites are suitable for forestry, because of nutrient limitations. The selection of sites for drainage in Finland has been based on floristic site type classifications that can be successfully used to predict tree stand growth after drainage. Despite this, many peatlands (at least 0.5 million ha) too poor to grow timber were drained in Finland in the 1960s and 1970s. In the future, these sites will most probably be abandoned for production forestry purposes or be actively restored to wetlands. Suitable sites will instead undergo the normal practices of peatland forestry, with fellings and ditch maintenance.

Forest fertilisation was a common practice in Finland until the 1970s when phosphorus and potassium (PK)-fertilisation was usually associated with first-time drainage. Nowadays PK-fertilisation is used when necessary, at oligotrophic sites where deficiencies of these nutrients are common. Water protection measures in connection with drainage systems, such as sedimentation pits and ponds and overland flows, are obligatory. These measures trap solid particles rather well, but leaching of soluble nutrients remains to be a problem.

In Finland, most of the drained peatland forests are still rather young, having been drained between 1960-1980, and having not yet reached regeneration age. Regeneration is based on the use of natural seedlings wherever possible, planting and/or seeding with natural tree species (Scots pine (*Pinus sylvestris*), Norway spruce (*Picea abies*)) is used otherwise. Regeneration normally necessitates site preparation, either mounding or scalping. Ditches are cleaned and/or complementary ditches are dug

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as the last measure after fellings and site preparation to ensure sufficient drainage for the new tree generation.

Temperate, maritime – the United Kingdom and Ireland
In the UK and Ireland, peatlands are naturally treeless and forestry development not only involves drainage but also the use of exotic tree species and fertilisers (Farrell, 1990). The main species used are Sitka spruce *(Picea sitchensis* (Bong.) Carr.) and Lodgepole pine *(Pinus contorta* Dougl.) There are 190 000 ha of forestry on deep peat (>45 cm deep) and 315 000 ha on shallow peat (<45 cm deep) in Great Britain (Cannell *et al.*, 1993) and about 50 000 ha on peats or peaty soils in Northern Ireland (Cannell *et al.*, 1996). In the UK, planting on deep peat ceased in the 1980s, largely to conserve habitats and species.

Site preparation usually involves ploughing although in more recent years this was replaced by mounding using excavators. Mounding involves the use of material from drains to create raised mounds that are used for planting. On blanket peat fertilization with N and P is vital to ensure crop viability. Thinning is not usually carried out on blanket peatland sites due to concerns about windthrow. Rotations are typically less than 50 years. Reforestation is carried out following windrowing of brash and mounding. In recent years there has been a move towards restoration of blanket and raised bogs which is likely to continue in the future.

Temperate, continental - North America
Peatland forestry is practiced with and without water management systems in the temperate-continental zone of North America. Peatlands that support silviculture without the use of water management systems include depressional and riverine wetlands in the southeastern US, and depressional and flat wetlands in the northern or sub-boreal zones of the U.S. and Canada. In the southeastern US, water management systems using pattern drainage systems are common. They often employ control structures that regulate the water level, since artificial drainage may not be needed after the stand is established (e.g. after 4-6 years).

Clear felling is the predominant harvesting system for even-aged forestry for both plantations and naturally regenerated stands. The 'shovel-logging' system is used extensively on un-drained sites and on drained sites where the potential for soil disturbance is a concern. Shovel-logging consists of felling strips and placing the tree boles on the ground perpendicular to the direction of travel, effectively making a trail mat; the feller then clears the trees in-between the strips, with the skidder using the matted trail to remove the trees to a common landing. The boles comprising the trail mat are then removed as the last operation. This system was devised to minimize the amount of soil disturbance and to increase the operability period.

Site preparation methods for regenerating stands on hydric (wetland) soils include bedding, where an elevated planting bed is created by disking soil and debris. This method incorporates organic matter into the planting bed, which has benefits concentrating nutrients. A similar system in concept is mounding. Mounding is a method where an excavator is used to create mounds by excavating a small hole and depositing the soil on the adjacent surface; the planted tree is placed on the top of the mound. In both cases, the elevated beds increase the volume of aerated soil for the planted trees. Other site preparation methods include disk-trenching, which produces a shallow trench and associated berm. It is most commonly used in shallow peatlands (e.g., histic-mineral soils) to facilitate planting the seedling in upper mineral soil. Chemical weed control in advance of planting is common.

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Figure 4.2. *An intensively managed peatland forest plantation in southeastern U.S. The dominant tree species is Loblolly pine (Pinus taeda). Sweet gum (Liquidambar styraciflua) and Wax Myrtle (Myrica cerifera, a woody shrub) form the understorey. (Photo: D. Amatya)*

The fertility of intensively managed peatland forests (e.g. *Pinus taeda* plantations) is commonly managed in the southeastern U.S. with the application of nitrogen (50-150 kg ha^{-1}), and on the coastal plain phosphorus (50-60 kg ha^{-1}) applications are used (Figure 4.2). The combination of improved planting stock, and water and soil fertility management can increase site productivity, reducing rotation length from 40-50 years to 18-24 years. In the northern U.S. and Canada, fertilization is not common because of the muted productivity responses and costs. The commonly managed tree species in the northern regions are *Pinus banksiana* (Jack pine) and *Picea negra* (Black spruce). Both species may be managed in plantations or in naturally regenerated stands. The typical rotation length for northern peatland forest is 60-150 years.

4.3. Ecological impacts of peatland forestry

4.3.1 Peatland carbon cycle

Peatlands represent the long-term and sustained accumulation of carbon, as a result of atmospheric C fixed into organic matter through photosynthesis decaying at a slower rate than it is produced. That concept is conveyed through a complex carbon cycle Figure 4.3. Part of the C photosynthesised by plants is returned to the atmosphere as CO_2 in the maintenance and growth respiration of above- and

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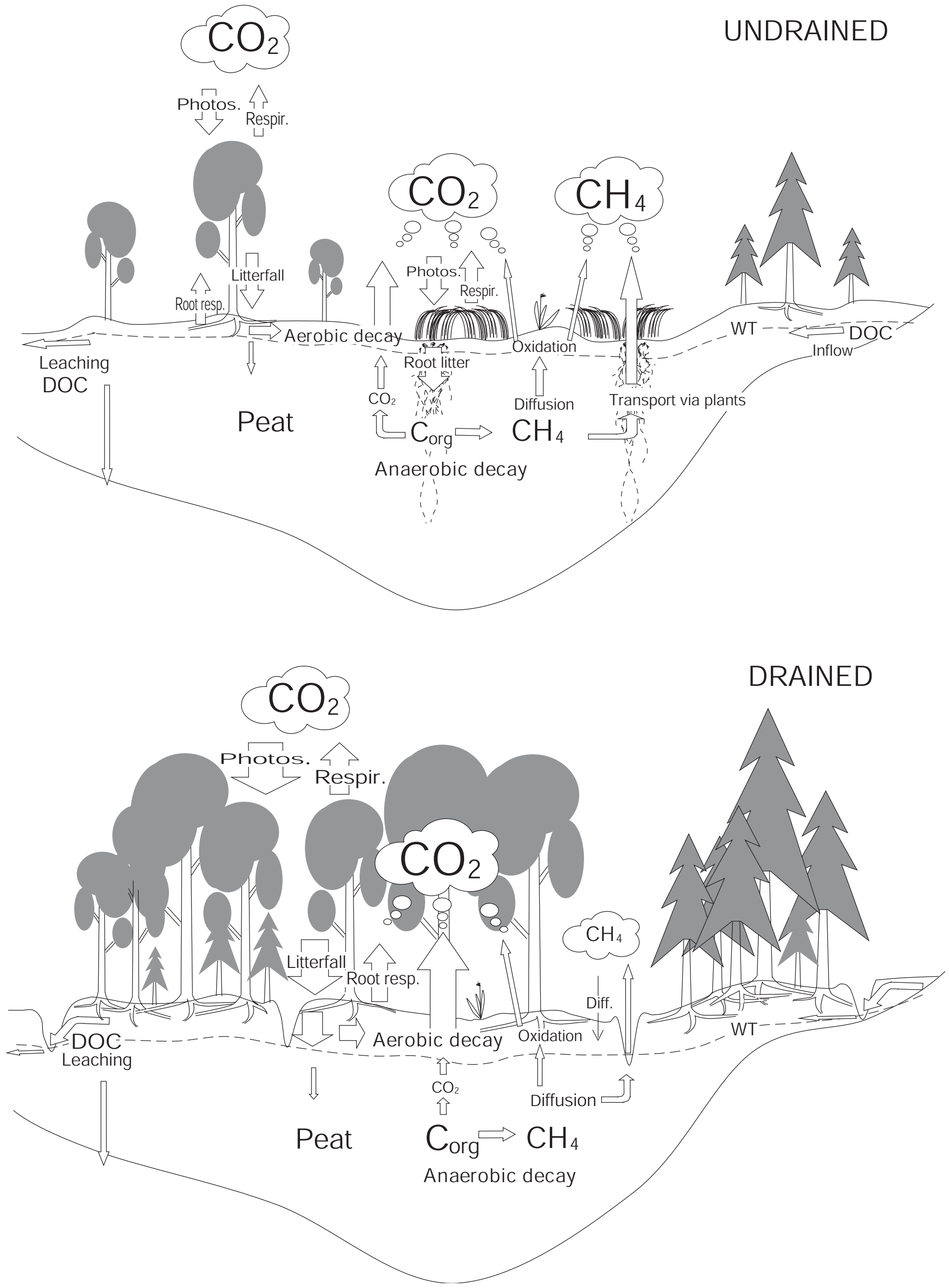

Figure 4.3. *C cycle in undrained mires and peatlands drained for forestry.*

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below-ground biomass. The remaining C is transformed into plant structures and organic compounds, with the rate of sequestration ranging from 300 to 600 g m^{-2} yr^{-1} depending on the bioclimatic zone and peatland type (Trettin & Jurgensen, 2003). A portion of that organic matter is deposited as necromass on (or in) the soil. In the aerobic surface parts of the peat ca. 80-95 % of the litter is decomposed by aerobic bacteria and fungi and released as CO_2, before it is covered by the gradually rising water table. In the water saturated, anaerobic parts of the peat the decomposition processes are very slow, and CH_4 may be produced in this layer , subsequently diffusing to the surface.

In peatlands, the net result of the C sequestration through photosynthesis and the various fluxes is an accumulation of C in the soil. The average present-day accumulation rate of C in northern peatlands is estimated at 21 g m^{-2} yr^{-1} (Clymo *et al.*, 1998). The net sink/source term is the sum of all input and output fluxes and may be quite small compared to any one directional flux component (see also Chapter 2). The balance is quite sensitive to variations in climate (both temperature and precipitation), site properties and land use practices, as discussed below for forestry.

Since the main part of the studies concerning C cycling in drained peatland forests is conducted in Fennoscandia (mostly Finland and Sweden) the discussion in the following sections reflects the ecological impacts of silvicultural methods typical to that area.

4.3.2. Impacts of drainage on the carbon cycle

Physical and chemical changes
Following drainage for forestry and the consequent drawdown of the water level, plant structures collapse and the peat surface subsides rapidly. The surface peat layers are consequently compacted into a smaller volume, and the peat density is increased. At the same time the diffusion of oxygen into peat increases and the aerobic surface peat layer deepens.

There are records of several metres of subsidence in agricultural peatlands where subsidence of the peat surface continues as long as new organic matter becomes available for oxidation through water level drawdown. In peatlands drained for forestry the situation is, however, different. Since plant cover is not removed, deposition of organic matter in the form of litter continues after drainage, and often a secondary ('raw') humus layer is formed on top of the old peat layer. Subsidence does take place, but it is smaller than in agricultural lands and it seems to be mostly caused by physical compaction rather than oxidation of organic matter. Therefore subsidence does not directly indicate C losses from soil. In Finland subsidence in old drainage areas (ca. 60 years) was typically less than 30 cm, being very close to values from the first 10 years after drainage (Minkkinen & Laine, 1998b). This suggests that most of the subsidence takes place soon after ditching due to physical compaction when water support is removed. Later on, the accelerated rate of organic matter decomposition and weight of the growing tree stand may cause further subsidence, but it is counteracted by the growth of the new humus layer.

Subsidence and compaction leads to denser peat. The increase in peat bulk density is highest on the oxic surface peat layers, but it has been observed however, as deep as 60-80 cm (Minkkinen & Laine, 1998a), a layer that is anaerobic most of the time. Although decay processes remain slow in these rather deep, normally anaerobic layers, even a short-term drop in water table (WT) would cause compaction in these deep layers when water buoyancy

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is temporarily lost. As the fluctuation of the water table increases after drainage, the increases in density may also be partly caused by relocation of soluble C from the upper peat layers (Domisch *et al.*, 1998).

The acidity of the peat increases after drainage. Oxidation of organic and inorganic compounds releases protons (H+ ions) into the system and thus increases acidity. In undisturbed minerotrophic peatlands the groundwater flow brings base cations into the mire from surrounding upland mineral soils, neutralizing the organic acidity of the peat. After drainage this influx of water is largely prevented by ditches, and even more cations are taken up by the increasing tree stand, causing thus the peat pH to decrease.

Soil temperature decreases in the long-term after drainage (e.g. Minkkinen *et al.*, 1999), after the initial short-term increase. This is caused by the decrease in thermal conductivity in the drier surface peat and the increasing shading by trees. The decrease in soil temperature may have profound impacts on soil C dynamics, especially on decomposition processes.

Changes in vegetation species composition
Drainage initiates a vegetation succession in which typical mire plant species are gradually replaced by forest vegetation (Laine *et al.*, 1995). The flark (wet location) and lawn level species are the first to disappear, whereas hummock species, being more resistant to the water-level drawdown, persist longer. The rate of change depends mainly on the nutrient level and the quantity of water level drawdown.

On nutrient-poor, deep peat, bog sites, where efficient drainage is difficult to maintain, vegetation succession is slow and often even stops or reverts to original mire vegetation when ditches get choked with mosses and sedges. Trees may die and the site may be restored to a mire ecosystem again if drainage is not maintained by improvement ditching. If the tree stand grows big enough its transpiration will drain the site, even if ditches are dammed. In ombrotrophic sites the tree stand remains sparse and the canopy open, allowing light to penerate to the ground. Ground vegetation remains vigorous, a mosaic of mire and forest species.

On minerotrophic sites with originally high water table (fens), a thin peat layer and high peat nutrient content, the change is much faster. Because of sufficient drainage and nutrients in the peat, tree growth increases rapidly after drainage, and the tree stand soon constitutes the dominant vegetation layer. Later on, shading by the tree stand directs the succession of the ground vegetation towards shade-tolerant flora. Species diversity decreases in the long-term following drainage, along with the disappearance of microtopographical differences.

The changed species composition and stand structure has the utmost importance for the ecosystem C balance through the production of organic matter and the decomposition potential of new litter types. The differences between site conditions (ombrotrophic vs. minerotrophic) must therefore be recognised.

Biomass and primary production
The net primary production and biomass of the vegetation increase after drainage. The greatest increase takes place in the tree stand biomass and production while moss layer biomass may decrease. The lowest levels of biomass, ca. 100 g C m^{-2}, have been reported from undrained treeless fens (Reinikainen *et al.*, 1984). Forestry practices (drainage, thinning) may increase the C stock in tree stands by 6-12 kg C m^{-2} during the first rotation (60-100 years) depending on the site type and climate (Cannell *et al.*, 1993; Minkkinen *et al.*, 2001). This gives an average annual C

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sequestration rate of 45-190 g C m^{-2} yr^{-1} in the tree stand. In addition 300-400 g C m^{-2} litter (average for pine-dominated stands) is produced and deposited in the soil annually, where it partly decomposes producing CO_2. In a closed tree stand 45-65% of the total tree stand C store is located in the stems, and is therefore removed in the cuttings, while the residues are left in the forest, unless they are collected for biofuel (i.e. whole tree harvesting-method).

The C stores in ground vegetation biomass may either increase or decrease depending on site type, but the change is usually insignificant in comparison to that of the tree stand (Minkkinen *et al.*, 1999). As the species composition radically changes with the succession following drainage, the biomass distribution between different plant groups (e.g. mosses, shrubs, sedges, herbs) also changes (Laiho *et al.*, 2003), but as the biomass stays below 500 g C m^{-2}, the C stocks remain small compared to the tree stand. However, the C fixed by the ground vegetation circulates rapidly, and a considerable amount of C flows into the peat through above- and below-ground litter production. Thus the importance of ground vegetation in C dynamics is probably much higher than its biomass would indicate.

The deposition of litter changes strongly as it follows the succession in plant communities (Laiho *et al.*, 2003). The dynamics in litter production rates may be very strong during the first 20 years after drainage, but in the long term it seems to reach a level similar to that in natural mires. A very remarkable change takes place, however, in the quality of the litter, as mire species are replaced by forest species. These changes in the quality of the above- and below-ground litter, which form the organic C flow into the soil, may significantly contribute to the post-drainage C balance of peatlands.

Decomposition

In the changed, more aerobic conditions, peat decay rates increase. In anaerobic conditions the activity of biodegradative enzymes is depressed, partly because of low diffusion of oxygen into peat and partly because of other factors (low pH, low temperature, low organic matter quality) associated with these conditions in peatlands. Increased decomposition rates in association with peatland drainage have been reported, measured as cellulose mass loss (e.g. Lieffers, 1988), or as a change in CO_2 emissions in laboratory conditions (e.g. Moore and Knowles, 1989) and in the field (e.g. Silvola *et al.*, 1996).

Drainage increases the decomposition rates especially in the previously waterlogged surface peat layer, which contains a lot of fresh, poorly decomposed root litter that has been deposited directly into anaerobic conditions. When this layer is exposed to oxygen through drainage, the decomposition rate of that material clearly increases. The situation is, however, different for the above ground litter. In undrained forested peatlands water table varies usually between 10-30 cm, being some 10 to 30 cm deeper in the forestry-drained sites. Litter deposited on the soil surface on a pristine peatland has time to decompose to a relatively high degree before entering anaerobic conditions, making it more recalcitrant for decomposition. The post-drainage change in the aeration of above ground litter, is therefore, not that drastic.

In contrast, some changes, which may retard decomposition in the surface peat take place after drainage. For example, soil temperature, which most strongly regulates decomposition rate in aerobic conditions, decreases in the long-term after drainage (Minkkinen *et al.*, 1999). Increased periods of drought on the drained peat surface may also inhibit decomposition. Laiho *et al.* (2004) found that pine needles and fine roots

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decomposed faster on undisturbed than on drained peat surface, and suggested that drought stress retarded decomposition on the surface peat of the drained site. In addition, increasing acidity retards decomposition through decreases in enzyme activity. Therefore, the effect of increased aeration on decomposition rates is counteracted by changes in other conditions, i.e. lowered peat temperature, increased periods of droughts and decreases in peat pH.

In addition to conditions, the substrate for decomposition changes, as the litter from mire species is replaced by litter from forest vegetation. The decomposition rate of litter is highly dependent on the litter quality with sugars and starches being the easiest organic compound to decompose and lignin being the most difficult. *Sphagnum* mosses are abundant with phenolic compounds, which also are known to be resistant to decay, but there is considerable variation within the genus. On the other hand, overall lignin content of litter increases because of the great increase in woody vegetation, having a retarding impact on decomposition rates.

Following the changes in conditions and produced substrate, the populations of decomposers also change after drainage (Jaatinen *et al.*, 2007). The species-composition and their importance on carbon cycling in the post-drainage ecosystem succession is, however, poorly known.

Leaching
In addition to gaseous compounds, carbon flows in and out of peatlands as dissolved organic carbon (DOC) in the groundwater. As natural mires have very high C densities, the C output is usually higher than with the input, i.e. there is a net loss of C from the mire by the water throughflow. Carbon is also leached downwards in the peat profile (Domisch *et al.*, 1998) and accumulates in the underlying mineral soil (Turunen *et al.*, 1999).

The leaching of organic C increases during and immediately after digging the drainage network, but because the groundwater flow through the peatland is decreased by ditches trapping the inflowing water, the long-term increase in organic C leaching is small (ca. 10%, i.e. 1 g C m^{-2} yr^{-1}; Sallantaus, 1994) or it may even decrease (Lundin & Bergquist, 1990). In addition, the leaching of C downwards in the peat profile may be expected to increase because of the increased fluctuation in the water table after rainfall events. This would form a further outflow of C from the mire as well as more rapid relocation of C downwards in the peat deposit. It is however unlikely that this C flux would have much importance in ecosystem level C balance.

Soil CO_2 efflux
The simultaneous changes in organic matter production and decomposition processes after drainage alter the CO_2 dynamics of peatland soils. The lowered water table increases the volume of aerated soil organic matter. In addition, the increased above- and below-ground litter input from trees and shrubs increases the labile pool of soil organic matter, thereby increasing heterotrophic CO_2 efflux (e.g. Silvola *et al.*, 1996). Autotrophic (root) respiration also increases as a result of increased plant biomass and production, contributing usually 10–50% of total soil respiration in forestry-drained peatlands.

Water table level has often been considered the major control in heterotrophic soil CO_2 efflux from peatlands. In Finnish and Swedish peatlands seasonal CO_2 emissions have been reported to always increase after drainage, and a linear relationship with average water table level and CO_2 efflux has been suggested (Silvola *et al.*, 1996; von Arnold *et al.*, 2005a). That positive relationship of CO_2 efflux with increasing water table was also evident in a recent review across a variety of peatlands, however, there was considerable variation

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in the relationship (Trettin *et al.*, 2006). Contrasting studies do exist, however. For example, Byrne & Farrell (2005) found that on a blanket peatland in Ireland, drainage did not always result in higher CO_2 emissions from peat, despite the positive relationship between water table and CO_2 fluxes within the site. Minkkinen *et al.* (2007a) and Mäkiranta *et al.* (2007) found only a poor correlation with water table and heterotrophic CO_2 efflux within drained peatland forests and afforested fields, where the water table seldom rises above 30 cm. Soil temperature, instead of water table, explained most of the variation in those sites. This same relationship has been shown in undrained peatlands (i.e. bogs) confirming the relevance of the water table affect when it is near the surface, and temperature when the water table is below 30 cm (Lafleur *et al.*, 2005).

It can thus be noted that the impact of water level on CO_2 fluxes greatly differs between studied sites. There are some explanations for this ambiguity. The greatest change in soil respiration occurs when water level varies between 0 and 40 cm, further water table drawdown (within sites) does not result in any further significant increase in heterotrophic respiration. This is most probably caused by a higher degree of humification and lower temperatures in the deeper layers, and also by simultaneous drying of peat surface during the dry periods. It also takes some time for the aerobic decomposers to colonise the previously anaerobic peat, and therefore a short-term drawdown in water level does not necessarily show the same impact as a longer period (which can be seen as a higher impact of water level in between sites comparisons, e.g. Silvola *et al.*, 1996). The inclusion of autotrophic respiration (roots) also affects the relationship. Root respiration increases with tree stand volume, and as stand volume grows, transpiration increases and the water level sinks. In addition, in Finland especially, better site types with bigger tree stands are usually thin-peated with steeper slopes with better drainage than poor types. The sensitivity of CO_2 efflux to short-term changes in water table level within sites is therefore lower and less obvious than can be observed when different sites with varying average water levels are compared.

In drained peatlands the temporal variation in soil CO_2 efflux is fairly well explained by soil temperature alone, if drainage is sufficient (WT lower than 40 cm, Minkkinen *et al.*, 2007a). Spatial variation remains high, even if differences in water table, edaphic factors, and microbial fauna are taken into account. In addition, different measurement methods create more variability, and care must be taken when values from different datasets are compared. It must also be remembered that in forested peatlands chamber CO_2 flux measurements do not represent the full soil C balance, since the organic C input through tree litter production is not accounted for.

Soil C stocks and NEE

The increased CO_2 emissions from peat soil following drainage have sometimes been interpreted to indicate a decrease in the soil C storage. However, as the incoming C fluxes also change after drainage the change in C stocks becomes more complicated to estimate. This can be done by combined gas measurements and modelling, or, for example, by consecutive measurements of soil C pools.

Minkkinen & Laine (1998b) and Minkkinen *et al.* (1999) estimated the changes in peat C pools on the basis of measured and modelled peat C densities and peat subsidence following drainage. In both studies the peat C pools had decreased in the most nutrient-rich sites, especially in the north, but increased in the nutrient-poor sites (Figure 4.4). Annual estimates of soil C balance varied from a loss of 120 to sink

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of 320 g C m^{-2} yr^{-1}. The increase in the peat C pool on the nutrient-poor sites indicates that increased net primary production (NPP) and input of organic matter in the soil as litter on these sites had exceeded the simultaneously increased oxidation of organic matter.

What makes a positive C balance possible at the poor sites? Firstly, root production in the poor sites is higher than on the fertile sites. When nutrient availability is low, trees have to allocate more C to root systems to get the vital amount of nutrients. Secondly, ground layer production, and especially that of mosses, is higher at the poor sites. The tree stand in poor pine sites remains open allowing light to enter to the lower layers of vegetation, which remain vigorous and productive after drainage. The continued high production of lower layers together with increased tree stand growth may have a crucial impact to the positive C balance in these sites. Thirdly, on nutrient-poor sites the decomposition rate is slower than on the more fertile sites, because the decomposition rate depends on the availability of nutrients. Also, drainage on poor sites is usually weaker than on the better sites and the oxidative, aerobic peat layer remains quite shallow even after drainage. The increased production without greatly increased decomposition rates thus enable higher C accumulation rates in the poor sites.

Another factor that may influence the differences in C accumulation between site types is the larger proportion of broadleaved trees (mainly birch, *Betula pubescens* Ehrh.) in the nutrient-rich sites. In nutrient-poor sites a secondary ('raw')

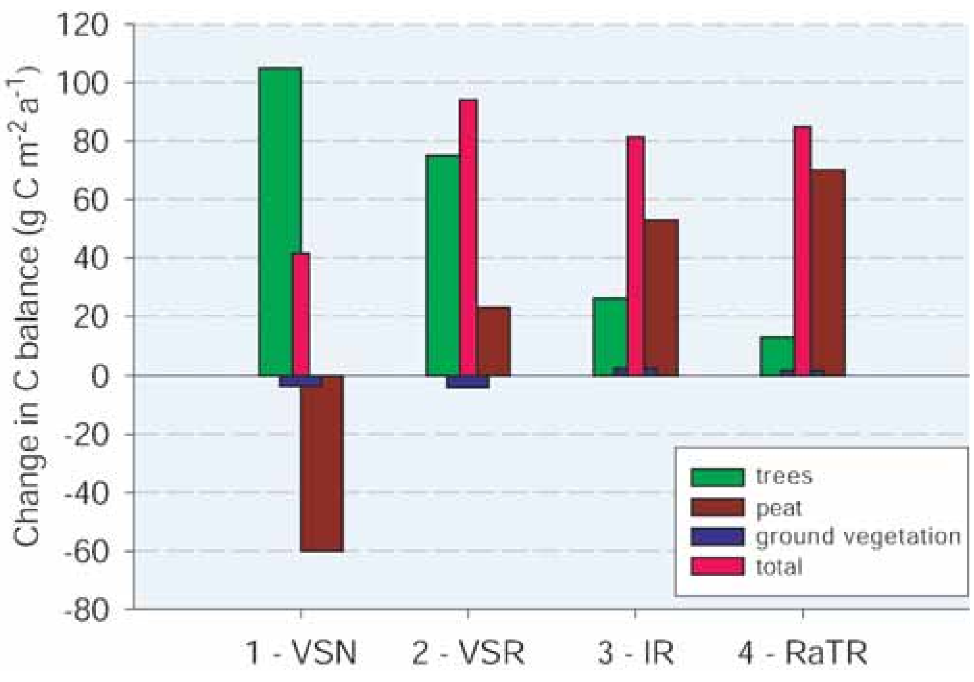

Figure 4.4. *The change in the C balance of the tree stand, ground vegetation and peat soil in four sites on Lakkasuo mire, Central Finland (Minkkinen et al., 1999). C balance of a peatland after drainage for forestry is strongly dependent on the site type and the consequent differences in influx (primary production) and outflux (decomposition) processes. Site 1 - VSN is the most nutrient rich and site 4 - RaTR the most nutrient poor site type.*

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humus layer is often formed on the peat surface when the needle litter from trees is mixed with growing mosses. In the nutrient-rich sites, however, the birch leaf-litter may cover the mosses and decrease their growth, thus slowing down humus formation and consequent C accumulation on the original, pre-drainage peat surface.

In forest drainage areas ditches often get blocked by vegetation, keeping the average drop in the water table rather small but still quite variable between sites. This may partly explain the great variability in peat C balance estimates among separate peatlands. In Lakkasuo mire in Central Finland, peat C stocks had decreased at the site where the average decrease in water table was highest (34 cm), and increased at the sites where the drop in the water table was clearly smaller (13 cm) (Minkkinen *et al.*, 1999). Since the water table level changes relatively little after drainage, the aerobic surface peat layer remains thin, offering still quite hostile conditions for oxidation processes and enabling C accumulation in peat soil after drainage.

Significant losses of peat C following drainage were found only in the northernmost area, Lapland (Minkkinen & Laine, 1998b). In Lapland the impact of drainage on the growth of the tree stand (and litter production) is much smaller than in the south, whereas annual soil CO_2 efflux from decomposing litter and peat is similar to the south (Minkkinen *et al.*, 2007a). This suggests that primary production in drained peatlands would be more climate-dependent than the decomposition of soil organic matter.

Only a few other studies exist that consider the changes in peat C stores in tree-covered peatlands after drainage. Methodological and climatic differences may be behind the variable results in these studies. Sakovets & Germanova (1992) estimated a small loss of old peat C (32 g C m^{-2} yr^{-1}) for a drained herb-rich pine fen in Karelia, but the ecosystem C balance was positive (123 g C m^{-2} yr^{-1}) as the C in the litter layer and trees increased much more than peat decomposition. Vompersky *et al.* (1992) concluded, based on litterfall and litter decomposition studies in Karelian peatlands, that peatlands still accumulated C in soil after drainage despite increased decomposition rates of peat. Byrne & Farrell (2005) found that afforestation of blanket peatland in the west of Ireland increased C efflux from soil in some sites but that on average, the peat appeared to be resistant to decay, despite lowering of the water table. The study suggested that the possible losses of peat C were compensated by C uptake in the biomass.

The only way to directly measure the C balance of a treed ecosystem is the micrometeorological eddy covariance method, but unfortunately such studies in peatlands are rare. Lindroth *et al.* (1998) were the first to report a C loss of 65-220 g CO_2 m^{-2} yr^{-1} from a ditched mineral soil ("sandy till with peaty spots") forest in central Sweden. Silvicultural drainage 20 years previously and the resultant water-level drawdown was hypothesised to be one possible reason for the high C loss from the ecosystem at this site. Later, a similar study in a drained organic forest in southern Sweden (Lindroth et al., 2007) showed an increase in ecosystem C storage with a small loss of soil C (1-47 g C m-2 yr-1). Interestingly, the other two studied forests, growing on mineral soils, were losing soil C at much higher rates than the drained organic soil site! In a micrometeorological study of peatlands in Scotland afforested with Sitka spruce, Hargreaves *et al.* (2003) found that 2-4 years after afforestation (ploughing and planting) peatlands emitted 200–400 g C m^{-2} yr^{-1} but that 4-8 years after afforestation, when ground vegetation colonised the site and planted trees started growing, the site became a net sink for ~300 g C m^{-2} yr^{-1}. After this the peatland

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was a net sink for up to 500 g C m^{-2} yr^{-1}. The authors estimated that the peat soil was a net source of ~100 g C m^{-2} yr^{-1}. Overall they found that afforested peatlands in Scotland accumulate more C in the trees, litter, soil and products than in lost from the peat for 90–190 years. Lohila *et al.* (2007) reported a loss of 50 g C m^{-2} yr^{-1} from a drained peatland forest with an agricultural history in middle Finland. In this fertile (fertilised) pine-dominated site heterotrophic soil respiration released more C than was sequestered to the growing tree stand. Laurila *et al*. (unpublished data) measured NEE in a drained ombrotrophic peatland forest in southern Finland and found that the ecosystem sequestered ca. 270 g C m^{-2} annually. Approximately 60% of that was bound in tree stand biomass, leaving 40% for ground vegetation and soil. As it is improbable that ground vegetation biomass would have increased that much during the measurement period (without any disturbance in light/soil conditions), the most probable sink is the soil through the above and below ground litter input.

Ongoing C flux measurements in Sweden indicate that C dynamics in drained peatlands within the same climatic zone (Finland vs. Sweden) may vary substantially. Studies by von Arnold *et al*. (2005a; b) and some still unpublished studies from Sweden (Leif Klemedtsson, personal communication) suggest much greater losses of C from peat soil than that observed in the Finnish studies. The peatland sites in the middle and southern part of Sweden are typically more nutrient-rich than in Finland. Carbon to nitrogen (C:N) ratios are low (20–26), a great deal of drained forests are closed, spruce-dominated stands with a very sparse (or no) ground vegetation and climate is milder and moister than in Finland, especially during winters. Warmer autumns and winters increase soil C losses since decomposition of organic matter may continue although photosynthesis ceases in the absense of light. These climatic and edaphic factors may partly explain the observed differences between Swedish and Finnish drained peatlands.

Losses of 313 to 602 g C m^{-2} a^{-1} were reported in a three year measurement period from a tropical drained peatland forest in Kalimantan, Indonesia (Hirano *et al.*, 2007), indicating the potentially high C losses after drainage in very warm conditions (see also Chapter 6).

Based on chamber measurements and modelling Alm *et al.* (2007) estimated that drained peatland soils may act as sinks or sources of CO_2, depending on site type. The same methodology was used in the Finnish national greenhouse gas inventory report to the UNFCCC. Overall, the peat soil in Finnish drained peatland forests was estimated to release 6.5 Tg CO_2 (i.e. 1.77 Tg C; where 1 Tg = 10^{12} g) annually, while trees and litter layer accumulated 18.5 Tg CO_2, giving a total sink of 12 Tg yr^{-1}. This sink is of course temporary, since the C stock in the tree stand cannot grow forever. It is also worth noting that the models used show higher soil C source than the default Tier 1 emission factors given by IPCC (for more on greenhouse gas accounting see Chapter 8). Similar conclusion was made by von Arnold *et al*. (2005b) who estimated, based on national measurements and modelling that drained peatland forest soils in Sweden released as much CO_2 (10.8 Tg CO_2 yr^{-1}) as was bound by growing tree stands and litter, thus giving a zero CO_2 balance. The use of IPCC emission factors instead, resulted to a sink of 5.2 Tg CO_2.

Based on available information it appears that although soil CO_2 measurements clearly show increased levels of soil CO_2 efflux after forest drainage, they do not necessarily indicate soil or ecosystem C losses. Differences between studies are, however, huge and the few existing micrometeorological studies have not yet been able to reduce this high variability.

Methane

Methane is formed from organic or gaseous carbon compounds by methanogenic archaea living in the anaerobic, water-saturated peat layers. A major part of the CH_4 formed originates from relatively new carbon compounds brought to the anoxic peat layers by deep-rooted plants, such as sedges. In the upper, more oxic peat layers methanotrophic bacteria oxidize part of the CH_4 diffusing upwards to CO_2. Many wetland plants possess aerenchyma, porous tissue required to provide the roots with oxygen. At sites where such plants dominate (sedge fens especially), most of the CH_4 is transported into the atmosphere via these plants' aerenchyma, thus avoiding the oxidative peat layers.

Because of the anaerobic nature of methanogenesis, CH_4 efflux from the peat surface is sensitive to water table position. Accordingly, drainage invariably decreases CH_4 emissions from peat soils (e.g. von Arnold *et al.*, 2005a) initially as a result of increased oxidation of CH_4 in the enlarged surface oxic layer, and subsequently through the decreased production in the catotelm, when deep rooted mire plants disappear, and input of fresh C to the anoxic peat layers ceases. For example, in a series of Finnish peatland types CH_4 emissions decreased by 30 to over 100% following drainage (Nykänen *et al.,* 1998). The decrease was smallest in ombrotrophic pine bogs where water table lowering remains usually rather small, and greatest in meso-oligotrophic sedge fens, which undergo the greatest change in vegetation and physico-chemical environment and often act as net sinks for atmospheric CH_4 after drainage.

Since CH_4 emissions are closely connected with two interconnected variables, water table level and plant species, mire site types can be used to estimate seasonal CH_4 emissions. This connection is still present at the drained sites, although weaker, since drainage age also affects the fluxes. Minerotrophic sites change more quickly than ombrotrophic, which may remain CH_4 emitting sites long after drainage. As tree growth is closely connected to the post-drainage development of the ecosystem, tree stand volume was found to be a good predictor for CH_4 fluxes in drained peatlands (Minkkinen *et al.,* 2007b). Stand volume is a practical tool in upscaling emissions to national level, since it is generally available from National Forest Inventories.

In drained peatlands drainage ditches form a new kind of wet surface, resembling the conditions of undrained mires. Mire vegetation often colonises ditch bottoms, decreasing water movement, especially in bogs. In such conditions CH_4 emissions may continue and even be enhanced compared to undrained mires. Emissions from ditches are, however, extremely variable. In nutrient poor bogs they are of the same order as under similar vegetation in an undrained mire. Sometimes when a ditch crossing nutrient rich peat soils is dammed, extremely high emissions may develop (Roulet & Moore, 1995) and the impact of ditches may totally counteract the reducing impact of drainage (Minkkinen & Laine, 2006). In these kind of ditches with standing nutrient rich water, ebullition may play an important role, but usually diffusive and plant transported fluxes prevail, comprising the majority of the total CH_4 efflux from ditches.

Ditches also transport CH_4 dissolved in the water away from the drained peatland and it is likely that a major part of this CH_4 will be emitted to the atmosphere outside the peatland. The quantity of this CH_4 source remains unknown.

Nitrous oxide

Nitrous oxide (N_2O) is formed as a by-product of organic N decomposition in nitrification and denitrification processes.

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Nitrate levels are inherently low in natural peatlands since nitrification requires aerobic conditions and the system demand for inorganic N is high. Accordingly, N_2O emissions from natural peatlands are insignificant (e.g. Martikainen *et al.*, 1993). In contrast, drained peatlands have a greater capacity for nitrification and some peatland forests are fertilized with N.

Drainage for forestry has been shown to increase N_2O emissions significantly only at nutrient-rich peatland sites (Martikainen *et al.*, 1993), where the pH is high enough for nitrate formation through nitrification. Tree stand felling and consequent N input from slash may, however, create environmental conditions favorable for N_2O formation and emission even in less nutrient-rich sites, as has been observed in clear-cut experiments in Finland.

Nitrous oxide emissions are more complicated to predict than CH_4 or CO_2 emissions, which follow the temporal dynamics in temperature and water table level. N_2O emissions may change abruptly from one day to another showing peaks that may be tenfold the average levels. These peaks occur often in the cold season being connected to freezing and thawing events.

Most estimates of seasonal N_2O emissions are thus based on integrated (averaged) values from measurements, although process models are being developed. Seasonal emissions from peat soils have been found to be closely connected with the soil C:N ratio (Klemedtsson *et al.*, 2005). The fluxes are close to zero when the C:N ratio exceeds 25-30 and increase exponentially with lower values. Most drained pine bogs have C:N ratios higher than 30 and would therefore emit only small amounts of N_2O. Most drained spruce-deciduous swamps instead have values lower than 25 and thus comprise a potentially high source of N_2O. Attempts have been made to estimate national N_2O budgets based on known soil C:N ratios from national forest inventories. Ernfors *et al.* (2007) estimated average emissions from drained peatlands in Sweden to be 0.31 g N_2O m^{-2} yr^{-1}, which in the area of 1.5 million ha amounted 4700 tonnes, i.e. 4.7 Gg N_2O yr^{-1}. Using a similar approach, national emissions from forest-drained peatlands in Finland were estimated at 11.4 Gg N_2O yr^{-1}, i.e. on average 0.23 g N_2O m^{-2} yr^{-1} (Minkkinen *et al*., unpublished). The lower estimate for Finnish sites is most probably caused by the larger area of drained N poor peatlands in Finland than in Sweden. The estimated emissions in both countries are significant on the national scale and have a bigger impact than, for example, CH_4 flux from drained peatland forests.

4.3.3. Impacts of other silvicultural practices

Fellings

Apart from drainage, there is much less data on the impacts of other silvicultural treatments. During fellings, tree stand biomass is removed from the site and this C stock is, of course, lost from the ecosystem (even though part of it would be stored in products). In thinnings litter production is temporarily reduced but, when the growing space is filled by neighboring trees, pre-harvest production levels are reached again. Final felling creates a much more dramatic change in the ecosystem as all trees are removed and the conditions for light, temperature and moisture become much more extreme.

Even though thinning and clear-felling somewhat increased soil temperatures (about 1°C), no significant increase in heterotrophic soil respiration of the old organic matter (excluding the roots from cut trees), was detected in an experiment in Central Finland (Minkkinen, unpublished data). Instead, the total soil respiration was quickly reduced by the same proportion of root respiration, as was estimated by trenching before the fellings. The small rise of water table (10 cm) slightly reduced CH_4 consumption at the site. Nitrous oxide emissions increased from slash piles, probably because of a liberation of N into the soil from decomposing needles. No impact was seen elsewhere in the clear-felled area.

A big disturbance to the CO_2 balance is derived from the decrease in the primary production. For a few years soil CO_2 efflux is not compensated at all by tree litter, and only a little by the litter from ground vegetation. The ground vegetation greatly suffers from the changed light and moisture conditions and production is close to zero. It takes a few years until plants have adapted to the new conditions and are able to grow and bind C again. Ground vegetation is therefore mainly responsible for C binding at least for the first 10 years after clear-cut, before the regenerated trees may gain dominance again.

The method of harvesting has an impact on stand C stocks. Whole tree harvesting, greatly reduces the amount of C in the ecosystem (Trettin *et al.*, 1992) compared to conventional sawlog or stem-only harvesting, in which residuals are left on (and in) the soil. In mineral forest soils sawlog harvesting on coniferous stands has been reported to increase soil C storage significantly while whole tree harvesting reduces it (Johnson & Curtis, 2001). In peatlands, the C stocks in harvest residues may persist even longer, being rapidly covered by growing mosses and the associated poorly aerated conditions. In particular, stumps and main roots may be preserved in peat for decades after fellings. Removal of slash and stumps for biofuel, a practice which is becoming more common in forestry, will therefore liberate large quantities of C to the atmosphere that otherwise would be stored in the soil. On the other hand, leaching of DOC and nutrients from decomposing residues increases after fellings, only if they are left at the site.

Soil preparation

Soil preparation, by mounding or trenching, for example, is usually necessary to ensure the establishment of a new stand after final felling. The main idea is to reduce the competition of ground vegetation, but at the same time soil preparation increases soil temperature and improves aeration, both of which have the potential to increase the oxidation of organic matter. Mixing of organic layers with underlying mineral soil may further increase decomposition rates because of fertilisation effects. Removal of ground vegetation decreases the overall binding of C, but on the other hand a new tree stand develops faster than on a non-prepared site. However, very little data exists about the impacts of soil preparation on GHG balances or their duration. Trettin *et al.* (1992) reported a rapid decrease in the C store of a histic soil (thin-peated mire) after whole-tree harvesting and site preparation, including trenching and bedding. Five years after treatment, the effect was still evident.

The effects of soil preparation may persist for a long time, if done repeatedly. In Finland, somewhat higher heterotrophic soil respiration has been measured from old afforested agricultural peat soils, which have a long history of soil preparation, compared to mires drained for forestry where ditching has been the only soil disturbance (Minkkinen *et al.*, 2007a; Mäkiranta *et al.*, 2007).

Fertilization

Forests are fertilised in order to increase or ensure tree growth. The impact on soil C balance is many-sided. Fertilisation may decrease peat acidity and increase litter nutrient content, which would increase the decomposition rates. On the other hand, N fertilisation has been suggested to retard decomposition of old organic matter by

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suppression of ligninolytic enzymes of soil microbes and by chemical stabilisation (Jandl *et al.*, 2007). Nitrogen is not usually used as a fertiliser in peatlands, which contain a lot of organic N, but increased pH may increase N mineralisation from peat. Increased availability of nutrients increases tree growth and aboveground litterfall, but decreases biomass and production of tree roots (Helmisaari *et al.*, 2007). In mineral soil forests N fertilisation has usually increased soil C pools, but no experiments are known from peat soils.

When fertilisations increase the amount of mineral N, they also increase the probability for gaseous N losses, including the formation and emissions of N_2O. If, however, fertilisations are restricted to areas where phosphorus and potassium (PK) fertilisers are used to repair nutrient balances, N_2O emissions are not expected to increase.

4.4. Radiative forcing from peatland forestry

Changing concentrations of greenhouse gases (GHG) in the atmosphere cause perturbation in the earth's energy balance. This pertubation is called radiative forcing (RF, unit: W m^{-2}), where positive values indicate a warming effect and negative, a cooling effect.

The climatic impact of peatland forestry can be estimated on the basis of changes in the RF that the ecosystem causes. To do this, the changes in the net fluxes of GHGs are needed. CH_4 and N_2O fluxes can be estimated with direct gas exchange measurements and modelling. The net CO_2 exchange is more difficult to measure in forested ecosystems, which is why measured changes in C pools are often used to estimate the changes in CO_2 balance. As discussed in earlier sections, drainage and other silvicultural practices in peatlands cause multidirectional changes in the components of CO_2 balance, while changes in CH_4 and N_2O exchange is more straightforward, even though variation between estimated net fluxes in different studies is still large.

Global warming potentials (GWP) are a useful and easy tool to estimate climatic impacts, if one wants to compare the integrated RF of pulse emissions over a specified time period, relative to that of CO_2. Thus, for example, CH_4 is 25 times, and N_2O 298 times more effective a greenhouse gas than CO_2, in a 100-year time horizon. This method is however, problematic, if continuous, long-term emissions are studied. While the lifetime of CH_4 is only 12 years, part of the emitted/removed CO_2 remains infinitely in/out the atmosphere and thus their impact depends totally on the time scale used. Therefore in such studies, for example when considering the development of peatlands and their GHG fluxes in the course of time, RF simulation models (e.g. Frolking & Roulet, 2007) are the correct tools for estimating climatic impact.

In the early stages (few hundred to few thousand years) in their development, natural peatlands may have a positive RF (i.e. warming impact on climate), since the RF of emitted CH_4 exceeds that of accumulated CO_2. Later on, the almost infinite cooling impact of CO_2-C sequestration into peat from the atmosphere exceeds the warming impact of CH_4 emissions. Northern peatlands, developed during the Holocene, have therefore a net cooling impact on the climate (Frolking & Roulet, 2007).

Land use changes, such as draining natural peatlands for forestry, may change their RF in either direction. On average, forest drainage decreases CH_4 emissions, increases N_2O emissions and CO_2 emissions from peat, but increases CO_2-C

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sequestration to the ecosystem during the first tree stand rotation. Laine *et al.* (1996) simulated the RF from forest drainage for different site types in southern Finland. The general impact of drainage was cooling, but the magnitude depended on site type. In minerotrophic sedge fens RF decreased greatly after drainage, when high CH_4 emissions virtually ceased, even though the peat layer turned from a C sink to a source. A similar, but smaller, decrease in RF was predicted for the bog (Figure 4.5). On a national scale in Finland, the impact of forest drainage activity was estimated to have a cooling effect on global climate during the first 200 years (Minkkinen *et al.*, 2002). This shift was mainly caused by the large decrease in CH_4 emissions, but also because of increased C sequestration in the ecosystem, and rather small increase in N_2O emissions. The impact of changing albedo was not estimated.

Radiative forcing calculations contain uncertainties, both in the determination of the GHG fluxes and in the modelling of the atmospheric behaviour of the GHGs. If a peatland was permanently changed from a C accumulator to a C source to the atmosphere, the effect of the peatland on RF would inevitably become positive at some stage in the future, despite the possible decreases in CH_4 emissions. Over a potential greenhouse effect mitigation period of 100 years, drainage of nutrient poor peatlands for forestry usually decreases RF, even with small losses of peat C. In nutrient rich sites, where soil C losses and N_2O emissions may be much higher, the situation is likely to be the opposite.

4.5. Conclusions

Peatland forestry, that most often includes drainage by ditching, causes an ecological change in the peatland ecosystem, including changes in physical and chemical conditions and succession of plant and microbial communities. These changes alter the C and GHG fluxes in multiple ways. The rates of CO_2 fluxes in production and decomposition processes increase, while CH_4 emissions decrease and N_2O emissions increase at fertile sites.

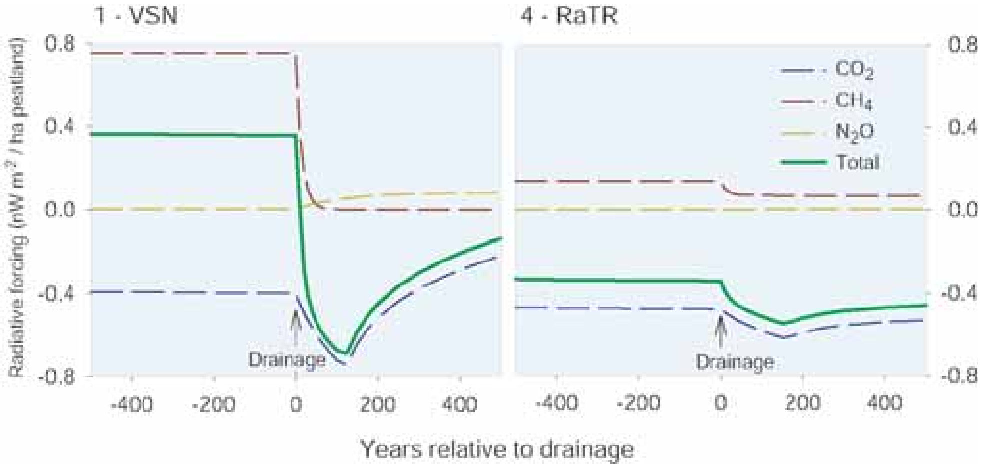

Figure 4.5. *The impact of drainage on the radiative forcing of a mire ecosystem in the minerotrophic (1 - VSN) and ombrotrophic (4 - RaTR) sites in Lakkasuo mire, Central Finland (Laine et al., 1996). The flux estimates behind this figure are slightly different from that in Figure 4.4, but the direction of the change and order of the magnitude is the same.*

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The net impact of forest drainage on the C balance on peatlands depends on site type, climatic conditions and drainage intensity. Soil C (and even ecosystem C) losses may be expected at sites that are rich in nutrients, drainage is intensive and water table level is therefore greatly lowered. Within the boreal zone in Finland, northerly location seems to increase the potential for soil C losses, since the small increase in production (tree stand growth) in the north cannot compensate for the increased decomposition of soil organic matter. The same may be true for the very nutrient-rich sites in southern Sweden. Instead, continued and even increased ecosystem and even soil C sequestration is possible at least in the southern boreal zone in nutrient poor sites, where the intensity of drainage is low and water table drawdown is moderate, but big enough to increase the stand growth and litter production more than the increased soil decomposition. These conclusions are valid for areas where forestry is based on natural tree stands and water and soil management is not very intensive, i.e. the Fennoscandian silviculture. It is, however, difficult to draw conclusions from more southerly and more oceanic areas where forestry is much more intensive and C balance studies are rare. More C exchange measurements on drained peatland forests are needed to verify the findings.

Clear felling disturbs the GHG balance of the site temporally by decreasing primary production and inducing N_2O emissions through liberating nutrients in the soil. Soil preparation further disturbs the soil C dynamics, but the impacts on total C balance in a longer time horizon are unknown. Whole tree harvesting, especially if stumps are removed, greatly reduces the amount of C in the ecosystem compared to conventional harvesting, in which residuals are left on the soil. In general, the potential for soil C losses in peatlands increases with intensity of soil disturbance.

As stated above, the climatic impact of peatland forestry is not unidirectional. It depends on the site type, climatic conditions, and on the intensity of the silvicultural methods used. It is also a function of time. In boreal conditions, in the short run, the C gains of growing tree stands often exceed the possible losses of C from the soil, especially when CH_4 emissions are reduced to zero. This causes a cooling impact on climate. However, if a permanent C loss from soil is created, the climatic impact will eventually be warming.

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CHAPTER 5:

CLIMATE IMPACT OF PEAT FUEL UTILISATION

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5.1. Introduction

Peat has been used for energy purposes for at least 2000 years. It has been an alternative to wood fuel for cooking and heating in temperate and boreal regions of Europe. During the 20th century other fuels replaced the use of peat for domestic heating and cooking. However, the use of peat fuel in industry and in heating and power plants increased. Still today energy peat utilisation is most common in Europe, which accounts for over 95% of both production and consumption world wide. There are a few dominant countries, Finland, Ireland, Russian Federation, Belarus, and Sweden, being responsible for almost 90% of the world's production and consumption of energy peat (WEC, 2001).

Peat cutting and burning cause significant emissions of greenhouse gases, including mainly emissions of carbon dioxide (CO_2) but also, of nitrous oxide (N_2O) and methane (CH_4). Also, sinks of greenhouse gases are impacted. The emissions arise from preparation of the peatland for cutting, production of fuel peat, storage and transportation, combustion and after-use of the cutaway area. There are several alternatives for after use of the cutaway peatland. The carbon balance at the after-used area depends both on soil CO_2 emissions and CO_2 uptake by growing biomass. In addition, emissions of CH_4 and N_2O can be important for the total greenhouse gas balance. Several life-cycle studies on the greenhouse impact of fuel peat utilisation (all for boreal conditions) try to shed light on this complicated matter (e.g. Savolainen *et al*., 1994; Nilsson & Nilsson, 2004; Zetterberg *et al.*, 2004; Kirkinen *et al*., 2007a)

The emissions from peat combustion are reported under the Framework Convention on Climate Change (UNFCCC) by countries and under the European Union Emissions Trading System (EU ETS) by plant operators. The latter considers only the CO_2 emissions from combustion in the energy sector or combustion processes in the included industry if the installation has a rated thermal input of at least 20 MW, and the former requires reporting of all greenhouse gas emissions (CO_2, CH_4 and N_2O) resulting from the different phases of

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peat fuel utilisation. The emissions from the different phases reported under the UNFCCC are entered in different categories so that no picture of the total emissions and sinks are formed within the reporting system (see also Chapter 8).

The objectives of this chapter are to give an overview of fuel peat resources and utilisation as well as fuel peat production techniques in different countries. Furthermore, life-cycle studies of fuel peat utilisation and the climate impact in terms of radiative forcing are presented.

5.2. Energy peat resources and use

The distribution of peat resources and peat production between countries are not directly correlated (Table 5.1). There are nations with vast peat resources that do not use peat for energy at all. In Figure 0.1 (see summary for policymakers) the distribution of peat resources is presented.

Belarus
In Belarus peat has been used as a fuel for many years, with the highest consumption during the 1970's and 1980's. By the mid 1980's the utilisation of fuel peat in power plants ceased and, at present, 85% of the total production of energy peat is made into briquettes used mainly in households. Residential users consume 80% of the briquettes. Only a small fraction of the total peat production (~10%) is delivered to heating plants. The remainder of the peat fuel is either exported or used by other small-scale users (WEC, 2004).

Canada
Canada has a significant part of the world's peat resources. The peat industry in Canada mainly extracts for horticultural use. Historically, fuel peat has been produced in Canada but is not currently produced or utilised. Recently, however, an increasing interest in peat for fuel has evolved as a potential substitute for coal. Initial investigations concerning

Table 5.1. *Countries with Highest Production and Largest Resources of Peat*[a]

Country	Peat area (thousand ha)	Peat area (% of land area)	Production of fuel peat thousand tonnes (2002)	Consumption of fuel peat thousand tonnes (2002)[b]
Belarus	2 900	14.0	2 053	2 000
Canada	111 000	11.1	0	0
Estonia	900	20.0	682	468
Finland	8 900	26.3	6 766	7 838
Indonesia	27 000	14.2	536	520
Ireland	1 180	16.8	2 709	3 605
Russian Federa-tion	154 000	9.0	2 500	2 000
Sweden	6 400	14.2	1 071	1 330
U.S.A.	21 400	2.3	0	0

[a] Source: IPS, International Peat Society (available: www.peatsociety.org); Earth trends, 2003, Environmental Information, World Resources Institute (available: www.wri.org); State report on environment status and conservation in Russian Federation (2005); WEC (2004).
[b] Differences between production and consumption can be due to two factors: (i) import and export of peat and (ii) and the fact that peat often is stored between years since the production can vary significantly between years as a result to differences in the weather conditions during the harvesting season.

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fuel supply, combustion technologies and environmental impact have been performed but production of fuel peat has not started yet (Ontario Ministry of Energy, 2006).

Estonia

A fifth of Estonia's territory is covered by peatlands. Peat is the third most important indigenous fuel in Estonia after oil shale and wood and accounts for about 2% of the primary energy supply. Peat is mainly used for heating; only a small part is used in combined heat and power plants (CHP) where electricity is also produced. Mainly milled peat is produced, which is used either for briquette production or directly as boiler fuel in CHP plants. Briquettes are used domestically but a large amount is exported to other countries; i.e. Sweden, Germany, the Netherlands and Belgium. Sod peat is produced also and exported (WEC, 2004; Soosaar, 2005).

Finland

Finland has the highest proportion of wetlands of any nation in the world. Peat accounts for 6-7% of the total primary energy consumption in Finland. It is an important fuel especially in CHP and district heating plants in which peat accounts for almost 20% of the fuel used. Peat is also used in condensing power generation but, since January 1st 2005, the utilisation is strongly dependent on the price of CO_2 emission allowances of the EU ETS. (For more information about European Union Emissions Trading Scheme, EU ETS, visit http://ec.europa.eu/environment/climat/emission.htm). During recent years large investments have been made in Finnish peat-fired plants in order to enable integrated use of peat and wood (Paappanen & Leinonen, 2005).

Ireland

Domestic consumption of peat for energy purposes in Ireland dates back to prehistoric times, with documentary evidence of its use existing from as early as the 8th century. Centuries of deforestation created the situation where, after the 18th century, peat was the only fuel available to the majority of households.

Still today peat is a very important fuel in Ireland, constituting one of the sparse domestic energy sources. In 2005 peat accounted for 5% of the primary energy consumption in Ireland and almost 8.5% of the electricity was generated by peat firing (McDonnell, 2005).

Mechanical methods of extraction were adopted on a large scale in the decades following World War II, both for the production of milled peat (used as a power-plant fuel and to produce peat briquettes) and to replace manual cutting of sod peat for residential use. Out of current annual consumption of peat for energy purposes, nearly 70% is used in power stations and heating plants, 16% is briquetted and 13% consists of sod peat, used predominantly as a residential fuel. Peat briquettes are almost exclusively used by households.

It is currently estimated that peat extraction for large-scale energy production in Ireland will cease in 15-20 years (Lappi & Byrne, 2003).

Russian Federation

Peat has been harvested in Russia as a source of industrial fuel for centuries, the first factory being built at the end of the 17th century. During the civil war, 1917-1923 peat became a strategic fuel for the country. Peat fuel was a key starting point for the ambitious project to electrify the country and by 1928 over 40% of Soviet electric power was derived from peat. However, the use of peat for power generation has been declining for a long time and since 1980 it has been less than 1% of the total power generation. The total area of cutaway peatlands both for fuel and for other purposes over the whole exploitation period in Russia is estimated to be

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850 000 – 1 500 000 ha. The bulk of current peat production is used for agricultural/horticultural purposes. Approximately 5% of the exploitable peat deposits are used for fuel production, which currently amounts to around 2.5 million tonnes per annum. (WEC, 2004).

Sweden
Extraction of peat for industrial energy use began during the 19th century and, after reaching a peak during World War II, declined to virtually zero by 1970. Use of peat as a fuel for power stations and district heating plants started in the mid-1980's and now constitutes by far the greater part of consumption. According to data reported by Statistics Sweden, fuel peat production in recent years has averaged about 2.3 TWh per annum, varying between approximately 1.5 and 3 TWh per year (where 1 TWh = 3600 TJ). In addition, Sweden has imported on average 1.2 TWh fuel peat annually during the past five years. In 2004, district heating accounted for 73% of total consumption (CHP and heating plants), electricity for 25% (production in CHP plants) and industrial users for the remaining 2% (Parikka, 2005). Peat accounts for approximately 0.7% of total primary energy consumption, but in the CHP-plants and district heating peat has the share of 4% and 6% of primary energy, respectively (Paappanen *et al*., 2006).

5.3. Peat fuel production and utilisation chain

Peat fuel is produced by extraction and drying of peat and the fuel is sometimes refined, for instance, into briquettes before burning. After extraction some type of after-use is applied to the cutaway area. The different stages of the peat production and utilisation chain are described below.

Preparation of production field
The first step in the peat fuel production chain is the preparation of the production field. The peat field has to be drained in order to support the machines used for harvesting. Drainage ditches are dug and the peat field is left to dry. Usually, the moisture content of the peat is lowered from 90-95% to 80-85% during this drainage period (Zetterberg *et al*., 2004), which can vary in length between 0-5 years depending on the initial conditions at the site.

Peat extraction and fuel production
After the area has been drained the vegetation cover is removed and peat extraction can start. There are three different types of commercial peat fuel: milled peat with a moisture content of approximately 40 - 50%, air-dried sod peat with a moisture content of 30 - 40% and artificially dried compressed peat briquettes with a moisture content of 10 - 20%.
The specific extraction and production methodologies of these types are explained below (Box 5.1).

The thickness of peat deposits varies greatly between different sites and countries. In Sweden and Finland peat extraction is not considered economically viable unless the deposit at the site is at least 1.5 - 2 m and at some sites the thickness of the peat layer can be several meters. In Ireland the average thickness in raised bogs is 6 - 7 meters (some having a depth of as much as 14 meters) (Feehan & McIlveen, 1997; Renou *et al*., 2006) whereas the depth of blanket bogs is less. Atlantic (low level) blanket bogs have a depth varying from 1 - 6 m whereas montane (high level) blanket bogs are between 2 - 3 m deep (Hammond, 1981). In Ireland most of the industrial peat harvesting has taken place in raised bogs. In Estonia the average peat depth is 5 - 7 meters. With the commercially used production technologies an extraction area in Finland or Sweden is usually open for 20 - 25 years before the entire peat deposit has been extracted. Naturally, areas with thicker peat deposits will take longer time to cut

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Box 5.1. Peat fuel production methods

Milled peat

0.5 - 2.0 cm of peat is cut from the surface of the peatland and broken up into small particles. The peat is spread uniformly across the peat extraction area and left to dry in the sun. During the drying process, which usually takes 1 - 4 days, the peat is harrowed in order to accelerate the drying process. Harrows generally operate across the full width of the production field and use a mechanism, such as a spoon, to place the wetter peat from the milled layer on top of the drier peat. When the moisture content has decreased to approximately 40%, the peat is ridged and collected to stockpiles. Owing to its low bulk density and relatively high moisture content, the calorific value of milled peat per volume is low. This limits the economic transport distance. Milled peat is mainly constituted of pulverised peat, and it is typically burned in fluidised bed boilers. Milled peat is also used to produce briquettes and pellets.

Sod peat

Sod peat is produced by compressing the extracted peat mechanically (or manually) to sods, which are cylindrical, brick shaped or wave-like. The size and shape of the product is dependent on the production machinery. The peat is cut from a vertical section through the peatland. In Scandinavia the peat is cut from the field with an excavator disc creating a groove approximately 0.5 meters deep and 5 – 10 cm wide. The disc throws the peat directly into a screw press which compacts and shapes it into sods. In Ireland, an excavator operating from a vertical peat face fills tractor-drawn or self-propelled extrusion machines with raw peat that is then mixed, macerated and extruded in sod form. In both cases the sods are left on the extraction area to dry and the sods shrink and harden further during the drying process. Drying sod peat below 40% moisture content requires 10 – 30 days, depending on weather conditions. The sod peat has a higher calorific value per volume than milled peat and can therefore be transported longer distances economically. Sod peat is used in grate or fluidised bed boilers or for domestic burning (Ireland).

Briquettes & pellets

Peat briquettes and pellets are produced by compressing dry pulverous peat. They are uniformly shaped and can therefore be more easily handled than both milled and sod peat. Milled peat is compressed to form the briquettes, which are similar in size to bricks whereas pellets are 3 - 30 mm depending on the machine used. The moisture content of peat briquettes is low, only 10 - 20%, and is achieved by artificial drying. Briquettes are mainly used in stoves and fire-places in private households. The calorific value of both briquettes and pellets is high per volume and this form of peat can therefore be transported economically much longer distances than other types of peat fuel.

Peat production using biomass dryer (new production methodology - under development)

Vapo Ltd and VTT, Finland are developing a new peat production method called "biomass dryer". In this method peat is harvested with an excavator and transported to a separate peat drying field (biomass dryer) by a high power pump. Vegetation cover at the production field can be kept intact until the harvesting starts, and there is no need for effective drainage of the production field. The area of a single production field opened per year is ca. 1 ha. The biomass dryer consists of an asphalted area approximately 2 - 3 ha in size. The peat is spread onto the biomass dryer by a tractor-driven spreader. In optimal weather conditions the drying process is completed within 24 - 36 hours compared to a drying time of 1 - 4 days when using the traditional milling method. Thus, also the weather risks are minimised. The end product of the new method is pieces of sod peat 1 - 4 cm in diameter depending on spreader technology.

(Sources: Mälkki & Frilander, 1997; Leijting, 1999; Silvan, N., personal communication, Preliminary results of the study of new peat production method, 2006; Fitzgerald, P., personal communication, Bord ná Mona, 2006)

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Milled peat *Sod peat* *Peat pellets*

Figure 5.1. *Different types of peat fuel. Source: Reprinted by permission of Vapo Oy.*

completely and in Ireland it usually takes 40 - 50 years to extract the whole economic peat resource in one area.

After drying, the peat is usually collected into stockpiles, which are stored adjacent to the production field. For production of briquettes and pellets the peat is transported to a briquette/pellet factory. Finally the peat is transported to the combustion installation which can be either a CHP- power or heating plant, or individual households.

Combustion
Fuel peat has a carbon content of approximately 50-57% (dry weight) (Nilsson, 2004; Vesterinen, 2003) and hence burning peat results in emissions of CO_2. In Table 5.2 the properties of typical peat fuel are given.

In Finland the most commonly used combustion technology for peat combustion is fluidised bed combustion (CFB) and bubbling fluidised bed boiler (BFB) together with wood in combined heat and power production (CHP). Peat is normally used in middle and large scale installations, which are municipal and industrial heating and power plants (Tsupari *et al*., 2005).
In Ireland the most commonly used combustion technology at commercial power plants is fluidised bed combustion (Lappi & Byrne, 2003). In Sweden there are mainly four types of combustion technology used; fluidised bed combustion (CFB), bubbling fluidised bed boiler (BFB), grate firing, and suspension or pulverized fuel firing. In Sweden the most common technology is fluidised bed combustion (Burvall & Öhman, 2002). In all of the

Table 5.2. *Average Characteristics of Peat Fuel*[a]

Calorific value, dry (MJ kg^{-1})	20–23
Moisture (%)	10 (briquettes)–48 (milled peat)
Ash (%)	2–6
Sulphur, dry (%)	0.05–0.3
Carbon, dry (%)	53-56
H, dry (%)	5–6.5
O, dry (%)	30–40
N, dry (%)	0.6–3

[a] Source: Alagkangas, 2000; also supported by Burvall & Öhman, 2002

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countries mentioned, co-combustion of peat and other solid fuels (biomass or coal) is important.

After-use of extraction area
The currently commercially used peat fuel production methodologies leave a residual layer of peat. In Finland and Sweden this layer is reported to be a few dm thick (depending on the topography of under-laying mineral soil), in Ireland it is reported to be somewhat thicker, 0.5 -1 meter (Fitzgerald, pers. com.). Cutaway peatlands have no viable seed banks and very hostile conditions for plant colonization, (low water levels, large amplitudes of temperatures, danger of peat fires, exposure, etc.) which means that if cutaway peatlands are left without management it will take a long time for plants to re-colonise the site. Different options for after-use are described in section 5.4.3.

5.4. Climate impact of peat fuel utilisation
All activities in the various phases of the peat fuel production and utilisation chain change the rate of emissions and sinks of greenhouse gases in comparison to the non-utilised peat reserve, and all contribute to the climate impact of the peat fuel utilisation. Thus, the climate impact of peat fuel utilisation should be made from a life-cycle perspective, where all the emissions and sinks during the peat fuel life cycle are included, from production of peat to the after-use of the cutaway area. Figure 5.2 shows the different phases of the peat fuel production and utilisation chain.

5.4. Life cycle perspective

Life cycle analysis (LCA) is used in order to consider the environmental impacts associated with the considered activity

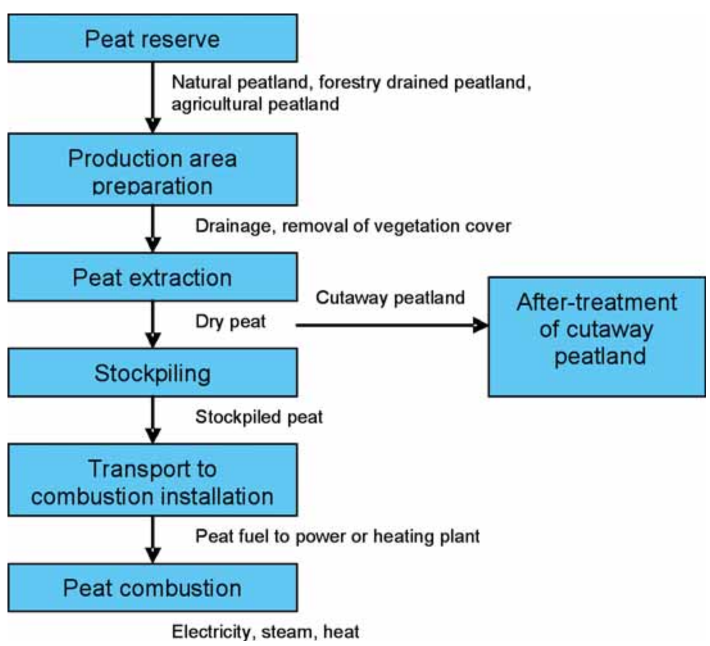

Figure 5.2. *An example of life cycle of peat fuel production and utilisation (edited from Mälkki & Frilander, 1997).*

"from cradle to grave". Life cycle analysis consists of four different phases: goal and scope definition, inventory analysis, impacts assessment, and interpretation. Critical review and reporting are also important parts of LCA. When assessing the greenhouse impact of peat fuel utilisation a life cycle analysis is applied by considering all the phases of the utilisation chain, as described by Figure 5.2. In the life cycle assessment the key factors are the definition of system boundaries e.g. the life cycle time frames, the status of the initial peatland, peat production, and after-use of the cutaway peatland. The impacts caused in the past and impacts that will occur in the future shall be considered if they are linked to the peat fuel life cycle. When making life cycle assessments for other products e.g. food products, the considered time span can be quite short, but in the peat fuel life cycle, the considered time span could be quite long due to the slow renewal rate of peat. However, there are other viewpoints that influence the relevant time frames in the considerations. Firstly, in the reporting of emission under UNFCCC, the time span used in the Global Warming Potentials is 100 years. Secondly, the ultimate objective of the UNFCCC is to stabilise the greenhouse gas concentrations in the atmosphere at a level that prevents dangerous anthropogenic interference with the climate system. If the allowed rise in the global mean temperature is approximately 2 ºC as proposed by the European Union, the atmospheric CO_2 equivalent (CO_2-e) concentration should be of the order of 450 ppm CO_2-e. The allowed concentration level has been estimated based on an assumed level of climate sensitivity, and depending on the actual climate sensitivity the acceptable concentration might be higher or even lower. The 450 ppm_v estimate is based on a best estimate of the climate sensitivity. If a rise of 3 ºC is allowed, the concentration level should be around 550 ppm CO_2-e. Presently the CO_2 concentration is ~380 ppm_v and the additional contribution of other gases is ~50 ppm CO_2-e. The average annual rate of increase of the concentration is ~2 ppm_v. Hence, if the rate of increase is assumed to be constant, it will take about 10 years to reach the 450 ppm CO_2-e level and about 60 years to reach the 550 ppm CO_2-e level. Within these time frames the emissions from the global energy system should be cut strongly in order to stop the concentration increase.

The previous discussion shows that weight should be given to the consideration of a time span of 100 years or less. However, most studies of the climate impact in terms of radiative forcing of fuel peat utilisation have made dynamic calculations and the results for any time from the starting year to 300 years can be read from the result curves.

The greenhouse impact from peat fuel utilisation can simply be expressed by the following equation:

$$I = I_U - I_R$$

where *I* means the total net greenhouse impact during the peat fuel life cycle, I_U means greenhouse impact of the peat utilisation chain (emissions/uptake during drainage, harvesting, combustion and after-use), and I_R means the greenhouse impact of the reference situation (i.e. the greenhouse impact from the peatland if left in its initial state and not used for peat fuel production). This way the avoided/not realized impact due to utilising the peatland is taken into account.

5.4.1. Methods for estimating climate impact from peat fuel utilisation

The potential climate impact due to the increased concentrations of anthropogenic greenhouse gases can be expressed by radiative forcing or by global

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warming potentials. The most important anthropogenic greenhouse gases are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). Peatlands can be sources and/or sinks of these gases and the conditions are impacted by for instance vegetation, level of ground water table, climate and working of the ground (depending on land use), factors of which many are highly impacted by peat fuel production. In section 5.4.3 the greenhouse gas dynamics of different stages of peat fuel production and utilisation are explained in detail.

Radiative forcing describes the disturbance of the Earth's radiative energy balance (expressed in Watts per square metre: W m^{-2}). Increasing greenhouse gas concentrations lead to increased radiative forcing since thermal radiation emitted by the Earth is partly trapped by the greenhouse gases. Less energy is radiated to space, which raises the temperature of the atmosphere-surface system of the Earth (see Figure 5.3).

The different greenhouse gases have different efficiencies of trapping the outgoing infrared radiation. Radiative forcing can be calculated based on the changes in atmospheric concentrations of the greenhouse gases. The concentration changes can be calculated using models that describe emissions, sinks and gas removals. Hence it is possible to calculate the dynamics of the development of radiative forcing due to emissions and sinks of greenhouse gases from the peat fuel life-cycle.

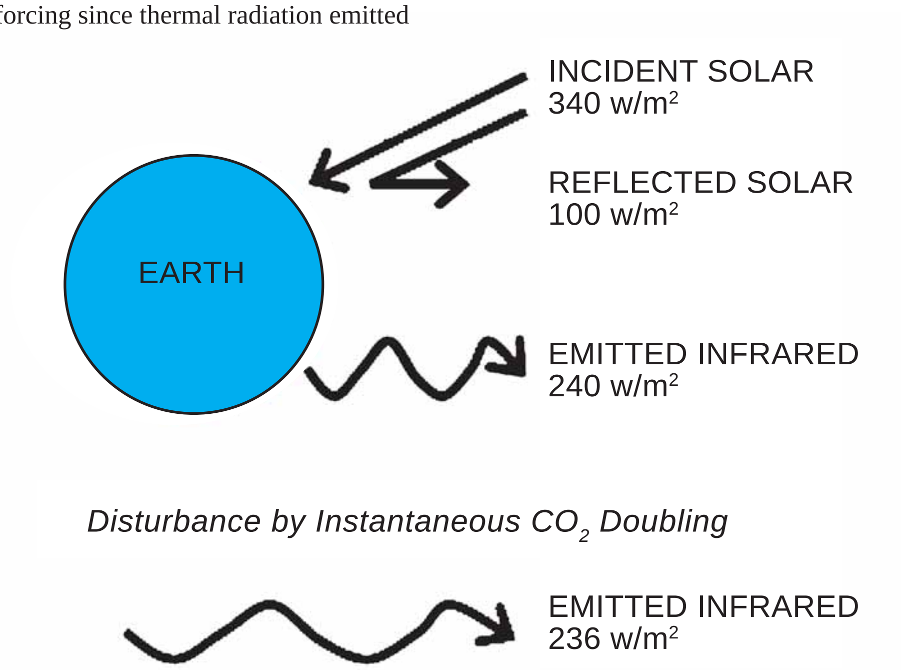

Figure 5.3. *The increased greenhouse gas concentrations by human activities decrease the infrared radiation from Earth to space. The disturbance in the radiation balance leads to an increase of the temperature of the surface/oceans and the lower atmosphere. A doubling of pre-industrial CO_2 concentration would decrease the outgoing radiation by approximately 4 W m^{-2} (IPCC, 2001). At the moment the radiative forcing is +2,30 W m^{-2}, compared to the pre-industrial situation, due to the combined effect of the increase of atmospheric concentrations of the main greenhouse gases (CO_2, CH_4 and N_2O) (IPCC, 2007).*

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Table 5.3. *Direct Global Warming Potentials (GWPs) Relative to CO_2 for Different Time Horizons for Greenhouse Gases Emitted within the Peat Fuel Utilisation Chain*[a]

Gas	Lifetime (years)	Global warming potential (Time Horizon in Years)		
		20 years	*100 years*	*500 years*
Carbon dioxide	100–200[b]	1 (1)	1 (1)	1 (1)
Methane	12	56 (62)	21 (23)	6.5 (7)
Nitrous oxide	114	280 (275)	310 (296)	170 (156)

[a] For UNFCCC accounting GWPs given in the IPCC Second Assessment Report (SAR; IPCC, 1996) are used. Subsequent reports have updated GWPs, for example values from the IPCC Third Assessment Report (TAR; IPCC, 2001) are given in brackets.

[b] The removal of CO_2 from the atmosphere is a complex process with interactions between oceans and the terrestrial biosphere. No single lifetime can be given for CO_2 removal.

Another way of describing the climate impact of the peat fuel life cycle is to use global warming potentials (GWPs). GWPs are indexes describing the relative radiative effect of a given substance compared to CO_2 (for which GWP is 1), integrated over a chosen time horizon. Greenhouse gas emissions of different gases can be transferred into CO_2-equivalents (CO_2-e) by using GWPs (see Table 5.3). Hence, GWPs are used for making simple comparisons and additions of the different greenhouse gas emissions with moderate accuracy. GWPs take into account the lifetime of greenhouse gases in the atmosphere and the absorbed infrared radiation for the chosen time horizon. The GWP values for different greenhouse gases vary with the chosen time horizon. The GWP values used in the reporting to UNFCCC are based on a 100 years time horizon and are given by IPCC (1996) (Table 5.3). The uncertainty of GWPs is relatively high, on the order of +/- 35%.

In the studies of the greenhouse impact of the peat fuel production and utilisation chain presented in this chapter the radiative forcing concept has been used. The main reason for using radiative forcing is that it gives a time dependent and more descriptive and accurate picture of the climate impact than for instance emissions or emission equivalents calculated by using GWPs.

5.4.3. Impacts on greenhouse gas balances during the different stages of peat fuel production

In Finland and Sweden several studies have been performed where the life cycle analysis of peat fuel use and its climate impact in terms of radiative forcing have been determined (Hillebrand, 1993; Savolainen *et al*., 1994; Åstrand *et al*., 1997; Zetterberg *et al*., 2004; Nilsson & Nilsson, 2004; Kirkinen *et al*., 2007a). Several studies have been performed in order to determine the greenhouse gas fluxes from the different stages of the peat production chain in these countries. To date no such studies have been conducted in Ireland although current research is addressing these issues (e.g. Wilson *et al*., 2007). In Canada the first life cycle analysis of greenhouse gas emissions from peat extraction has been performed, however, dealing with extraction of horticultural peat (Cleary *et al*., 2005).

Figure 5.4 is a schematic representation of the different emissions and uptake of greenhouse gases during the various stages of peat production, when peat is

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extracted from a pristine mire and followed by rewetting. Note that the fluxes are not to scale and that N_2O emissions are not included. The size (and sign) of the fluxes will be different depending on type of peat reserve utilised and choice of after-use.

Initial stage

Peatlands can be both sources and sinks of greenhouse gases. Peatlands that are used for peat fuel production are either natural (pristine) peatlands or drained peatlands previously used for forestry, agriculture or some other purpose.

Natural peatlands (i.e. pristine mires) can be divided into two categories, minerotrophic or ombrotrophic, based on the nutrient supply. Fens are minerotrophic peatlands and bogs are ombrotrophic peatlands. Fens and bogs are carbon sinks and CH_4 sources. Under natural conditions peatlands accumulate carbon derived from atmospheric CO_2. Part of this carbon will be converted to CH_4 in the anaerobic layers of the peatland.

Forestry-drained peatlands are common especially in Finland and Sweden, where large peatland areas have been drained in order to improve forest productivity. Forestry-drained peatlands are CO_2-sources due to the lowered water table, which changes the conditions of the upper layers from anaerobic to aerobic. This leads to raised decomposition rates in the aerobic layers, resulting in emissions of CO_2 to the atmosphere (see also Chapter 4).

Agricultural peatlands are previous natural peatlands that have been drained and are used for agricultural purposes and are common in many European countries (see also Chapter 3). The emissions of greenhouse gases from agricultural peatlands differ in magnitude as a result of different cultivation methods and crops. Agricultural peatlands are remarkable sources of CO_2 due to the lowered water table, cultivation and fertilisation. Agricultural peatlands are also sources of N_2O and modest sinks of CH_4.

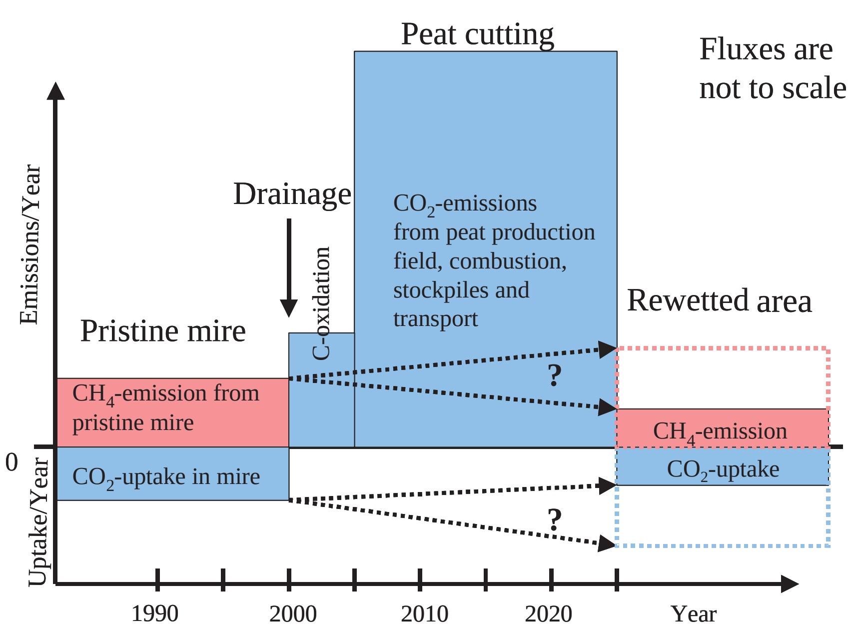

Figure 5.4. *Schematic representation of greenhouse gas emissions during different stages of energy peat production. Also, N_2O emissions can be of importance in peatlands with a low C/N-ratio.*

In Finland the majority of peatlands used for peat production are forestry-drained peatlands (75%). In Sweden peatlands used for peat fuel extraction are pristine mires, abandoned peat production areas or forestry-drained peatland. In Ireland, only pristine peatlands have been used for peat production (Lappi & Byrne, 2003).

The greenhouse gas balances of forestry-drained peatlands have been studied in Finland and in Sweden. The uncertainty or variability range of emissions is large, particularly for CO_2. The influence on the climate impact of peat fuel utilisation of the variability and uncertainty of emissions and sinks of greenhouse gases from forestry

Table 5.4. *Emissions of Greenhouse Gases from Peatlands during Cutting Phase*[a]

	CO_2	CH_4	N_2O
Emissions from peat field	600 (230–1020) gCO_2 m^{-2} yr^{-1} (Sundh *et al.*, 2000)	0.41–4.5 g CH_4 m^{-2} yr^{-1} (Sundh *et al.*, 2000)	Depending on nutrient status of field.
	806 (403–1208) g CO_2 m^{-2} yr^{-1} (Alm *et al.*, 2007)	4.25–5.75 g CH_4 m^{-2} yr^{-1} [b] 8.16–11.04 g CH_4 m^{-2} yr^{-1} [c] (Statistics Finland 2005)	0.08-0.22 g N_2O m^{-2} yr^{-1} (Nykänen *et al.*, 1996[f])
	0.2 tonne C ha^{-1} yr^{-1} = 73.3 g CO_2 m^{-2} yr^{-1} IPCC default factor[d] for nutrient poor industrial peatland[c]	0 g CH_4 m^{-2} yr^{-1} IPCC default factor[d] for drained organic soils	
	1.1 tonne C ha^{-1} yr^{-1} = 403.3 g CO_2 m^{-2} yr^{-1} IPCC default factor[d] for nutrient rich industrial peatlands[e]		
Emissions from stockpiles	1.5 g CO_2 MJ_{peat}^{-1} (Boström *et al.*, 1990)	Up to 0.1–0.2 g CH_4 m^{-2} h^{-1} (during spring melting);	
	1.48 (0.75-2.23)[g] g CO_2 MJ_{peat}^{-1} (Nykänen *et al.*, 1996)	Up to 0.001 g CH_4 m^{-2} h^{-1} (in summer-autumn, wet conditions); (Chistotin *et al.*, 2006)	
Emissions from working machines	1.0 (0.5-1.5) g CO_2 MJ_{peat}^{-1} (Zetterberg *et al.*, 2004)	0.7 mg CH_4 MJ_{peat}^{-1} (Zetterberg *et al.*, 2004)	0.0025 mg N_2O MJ_{peat}^{-1} (Zetterberg *et al.*, 2004)

[a] Note that the emissions do not include the potential emissions from the surrounding area.
[b] Excluding ditches
[c] Including ditches
[d] IPCC GPG LULUCF, 2003.
[e] Emissions at conversion of wetland to peat extraction area in boreal and temperate regions
[f] Drained peatlands previously used for cultivation can have significantly higher N_2O emissions.
[g] Note that this emission estimate is based on measurements during growing season only. The annual emissions are therefore most likely higher.

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drained peatlands and agricultural peatlands have been estimated by Holmgren *et al*. (2006) (see Figure 5.9 and 5.10). The variability in the greenhouse gas balances for these types of peatlands, compared to the variability of emissions and sinks of greenhouse gases during the production phase of peat fuel, is relatively large. For further information on greenhouse gas balances for pristine, forested and agricultural peatlands see chapters 2, 3 and 4, respectively.

Preparation of peatland for extraction and production of fuel peat

Draining a peatland results in a lowered ground water table and an increased oxic zone in the upper part of the peat layer. This results in increased decomposition of peat and emissions of CO_2. The more the ground water table is lowered the more CO_2 will be released. Both the extraction area and some of the surrounding area will be impacted by the drainage. On the one hand, drainage will reduce CH_4 emissions from areas intended for peat extraction, while on the other hand, ditches are constant sources of CH_4 (Christotin *et al*., 2006). Keeping ditches clear of vegetation may limit CH_4 emissions (Sundh *et al*., 2000). During the peat extraction phase there are also emissions (of CO_2 and N_2O) from the machinery used.

Peat cutting

During peat extraction the peat field releases CO_2 since the area is drained and aerobic decay of peat starts. The emissions will be larger when the vegetation cover is removed. The working of the ground by the extraction machinery will further increase the aeration of the upper layers and increase the rate of peat oxidation and CO_2 emissions. Peat decomposition will also continue in the stockpiles and cause additional CO_2 emissions. Finally, there are also CO_2 emissions from the utilisation of fossil fuels by harvesting machines and transport vehicles. The CH_4 emissions from this part of the fuel production chain will be minor and mainly come from drainage ditches and stockpiles.

The emission estimates in Table 5.4 are given per square metre of extraction area. However, the surrounding area will to some extent also be impacted by the drainage. The drainage ditches are effective on both sides and hence also part of the area outside the extraction area will be drained. Currently, there is little or no information on peat layer thickness of the surrounding area and hence it is difficult to estimate the emissions from these areas. Holmgren *et al*. (2006) state that the size of the surrounding area (area outside the extraction area impacted by drainage) is between 0-100% the size of the production area. A larger production area results in a smaller relative size of the surrounding area.

Combustion

Combustion is the greatest source of greenhouse gas emissions in the peat fuel life cycle (up to 90% of climate impact of total peat fuel life cycle). Peat combustion results in CO_2, CH_4 and N_2O emissions. The CO_2 emissions from combustion are the most certainly known in the peat life cycle. The uncertainty is larger for CH_4 and N_2O emissions especially from residential burning.

According to the IPCC (2006), peat is classified into a separate fuel type called peat. In the reporting to the UNFCCC (as further described in Chapter 8) peat is treated just as fossil fuels. Table 5.5 shows the tier 1 combustion CO_2 emission factor given in the IPCC Guidelines (IPCC, 2006) together with the values used for different types of peat in various countries. Also the combustion emission factors for CH_4 and N_2O are given. The tier 1 value of the combustion CO_2 emission factor is set to 106 g CO_2/MJ which means that the specific emissions from energy peat are higher than the specific emissions both

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Table 5.5. *Applied Combustion Emission Factor for Fuel Peat (according to National Inventory Reports and IPCC, 2006)*

Country	Combustion emission factor [g CO_2 MJ^{-1}]	Combustion emission factor [mg CH_4 MJ^{-1}]	Combustion emission factor [mg N_2O MJ^{-1}]	Source
IPCC Tier 1 default	106 (100-108)	1.0 (0.3-3)	1.5 (0.5-5)	IPCC, 2006
Belarus	IPCC default factor	IPCC default factor	IPCC default factor	NIR Belarus, 2005
Estonia	IPCC default factor	30 (energy & manufacturing industry) 300 (other use)	4 (all use)	NIR Estonia, 2005
Finland	105.9 (milled) 102 (sod peat) 97 (pellets & briquettes)	3.0 (average value for fluidised bed combustion. According to NIR, emissions are dependant on size of boiler)	7 (CFB) 3 (BFB + combined technologies) 2 (grate + combined technologies, pulverised comb., gasification)	NIR, Finland 2006
Ireland	140.24 (power plants) 104 (residential peat) 98.86 (residential & commercial briquettes)	0 (power plants) 50 (residential peat, residential & commercial briquettes)	12 (power plants) 5 (residential peat, residential & commercial briquettes)	NIR Ireland 2006
Russian Federation	IPCC default factor	IPCC default factor	IPCC default factor	CENEf, 2004
Sweden	107.3 (power & district heating plants) 97.1 (other use)	20 (power & district heating plants) 30 (other use)	5 (industry, power & district heating plants) 10 (other use)	SEPA, 2006

from coal (anthracite 98.3 g CO_2/MJ), and oil (74.1 g CO_2/MJ).

After-use

The choice of after-use of cutaway peatlands depends on many factors such as thickness of residual peat layer, the properties of the sub-peat mineral soil, water level and drainage conditions, climatic conditions, etc. The most common options of after-use are afforestation and paludification (restoration). Other options include cultivation and lake formation.

In Finland afforestation is the most commonly used after-use. In Ireland the most important after-use alternatives are afforestation, wetland development and dryland recolonisation (Lappi & Byrne, 2003). In Sweden so far not too many areas have been cut completely and hence therefore not been converted to any after-use yet, but afforestation is planned to be the pre-dominant one. There is also interest for restoration in Sweden (Vasander *et al*., 2003). In Estonia there is an increasing awareness that many cutaways are left without treatment and natural recovery is not happening.

According to legislation established already during the Soviet Union, cutaway areas in Russia were to be recultivated for agricultural use, afforestation or used for fish cultivation. Since 1996 rewetting is also an option. However, due to the economic changes in the 1990s and a crisis in the peat industry, large areas of partly cutaway peatlands have been left without recultivation and risks for fires are large at these areas (Minayeva & Sirin, 2005).

The choice of the after-use will also depend on laws and regulations, the interest of the land owner and the local conditions. If the area can not be kept drained without pumping, afforestation might not be a suitable option.

The greenhouse gas fluxes from an afforested area are mainly determined by the following parameters: 1) decomposition of residual peat, which causes CO_2 emissions and 2) uptake of CO_2 in growing biomass. These parameters are impacted by factors such as climate, nutrient supply and hydrology. The peat layer at the extraction area cannot be removed completely; at least 1-3 dm of peat will be left. If afforestation is to be used it is important that the peat layer is sufficiently thick to supply the new tree plants with nutrients (for instance nitrogen) and that it is shallow enough to allow the tree roots to enter the mineral soil. It is also important that the area is kept sufficiently drained in order to not restrain tree growth.

The carbon sequestration into the growing forest is considered differently in different LCA studies of peat fuel production and utilisation. In the comparative study of Finnish and Swedish results (Holmgren *et al*., 2006) this subject was examined and some different ways of how to consider the biomass from the after-used areas were discussed. How system boundaries are set depend on the objectives of the study. One way is to only consider the change in carbon stock. This means that the long term carbon stock changes due to the growth of the forest at the cutaway area could be considered. The average carbon stock over a rotation period could then be considered. Another way is to consider the forest system at the cutaway area as it grows (i.e. the carbon stock builds up but is then released as the forest is cut). Holmgren *et al*. (2006) concluded that the change in carbon stock should be considered but for how long depends on the scope of the study. However, it is very important to describe what system boundaries that have been used in each study.

Also, in the case of restoration there will be a residual layer of peat, however with no viable seed bank. According to Tuittila *et al*. (1999) and Alm *et al*. (2007) rewetting can lead to net uptake of CO_2 whereas CH_4 emissions will increase after rewetting. Canadian studies of restoration of cutaway peatlands, where horticultural peat has been harvested, show that such areas can remain persistent sources of CO_2 emissions (Waddington *et al*., 2002). A significant difference between the Canadian cutaway sites and, for instance, studied areas in Finland and Sweden is the thickness of the residual peat layer. In Canada (where fuel peat is not harvested) the residual layer is significantly thicker (1-2 meters) than the few dm left after harvesting peat fuel in Scandinavia. In Ireland the estimated thickness of the residual peat layer after harvesting fuel peat is 0.5-1 m. Irish studies of rewetted areas also show that they may be net sources of CO_2 (Wilson *et al*., 2007). Restoration of peatlands is considered in more detail in Chapter 7.

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5.5. Results from life cycle studies

A number of studies have been performed in order to describe the potential climate impact of peat fuel utilisation from a lifecycle perspective in terms of radiative forcing (Hillebrand, 1993; Savolainen *et al*., 1994; Åstrand *et al*., 1997; Zetterberg *et al*., 2004; Nilsson & Nilsson, 2004; Kirkinen *et al*., 2007a). The studies show that the climate impact will be different depending on what type of peatland is utilised, what harvesting methods are applied and what after-use of the cutaway peatland is applied.

In Figure 5.5 results from Savolainen *et al*. (1994) are shown. The scenarios show instantaneous radiative forcing of the production of 1 PJ of peat during 300 years. The after-use in the peat scenarios is afforestation. Savolainen *et al*. also made calculations for other choices of after-use such as restoration (paludification). Note that in Savolainen *et al*. (1994) scenario LT 1 (pristine mire – afforestation) it is not considered that the forest could be used in order to produce energy. Instead it is assumed to accumulate carbon until the forest is mature (accumulation during approximately 100 years).

In Zetterberg *et al*. (2004) the scenarios of afforestation, presented in Figure 5.6, include the use of wood for energy production. Nilsson & Nilsson (2004), presented in Figure 5.7, made the same assumption as Savolainen *et al*. (2004) whereas Kirkinen *et al*. (2007a), Figure 5.8,

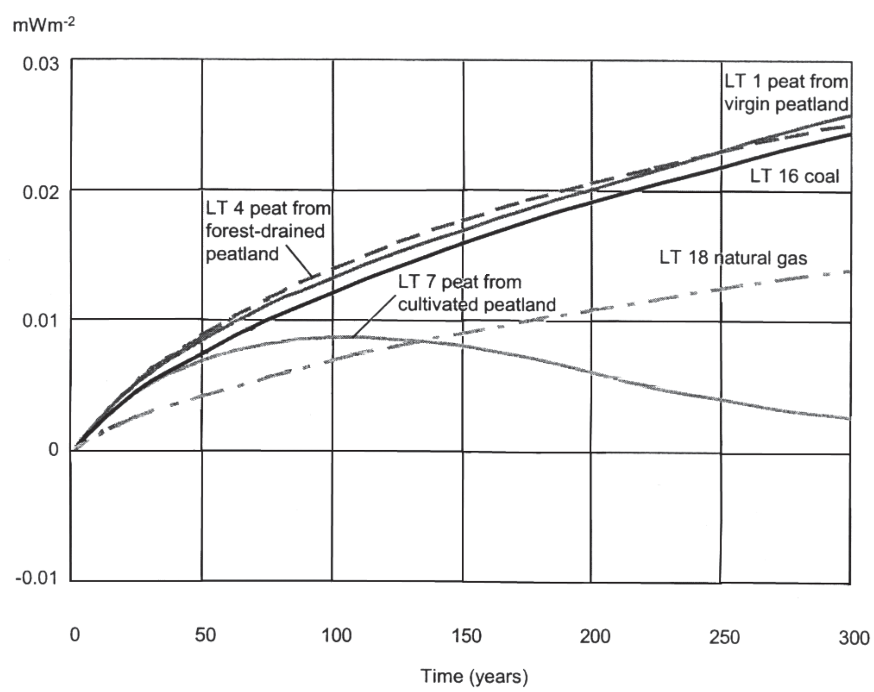

Figure 5.5. *Instantaneous radiative forcing for scenarios calculated by Savolainen et al. (1994). In these scenarios 1 PJ of fuel is produced during each year from year 0-300.*

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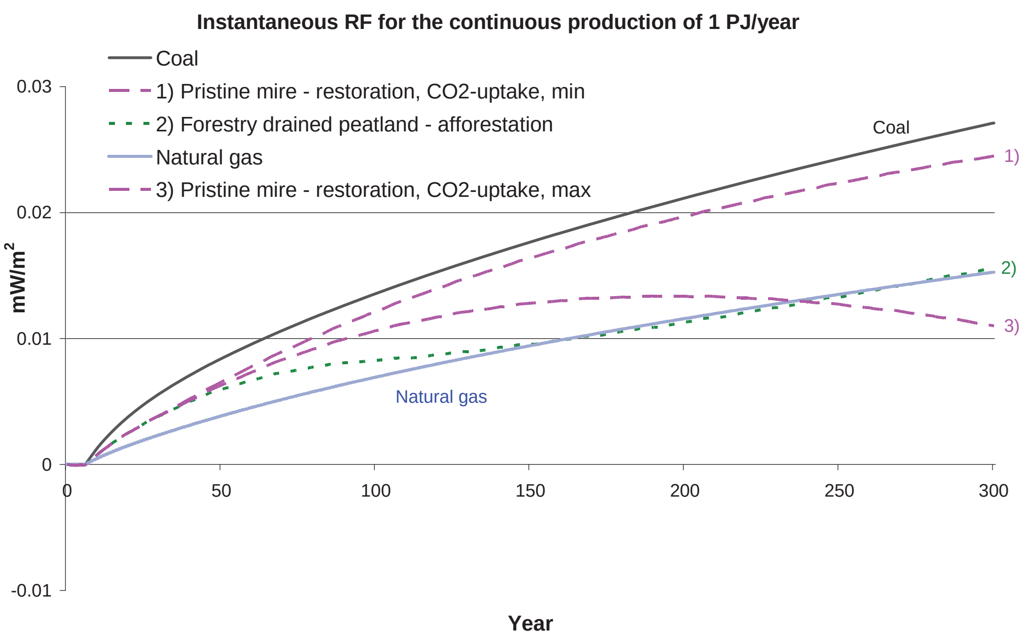

Figure 5.6. *Instantaneous radiative forcing for scenarios calculated by Zetterberg et al. (2004). In these scenarios 1 PJ of fuel is produced during each year from year 6-300. Note that in the afforestation case some of the fuel utilised is wood fuel.*

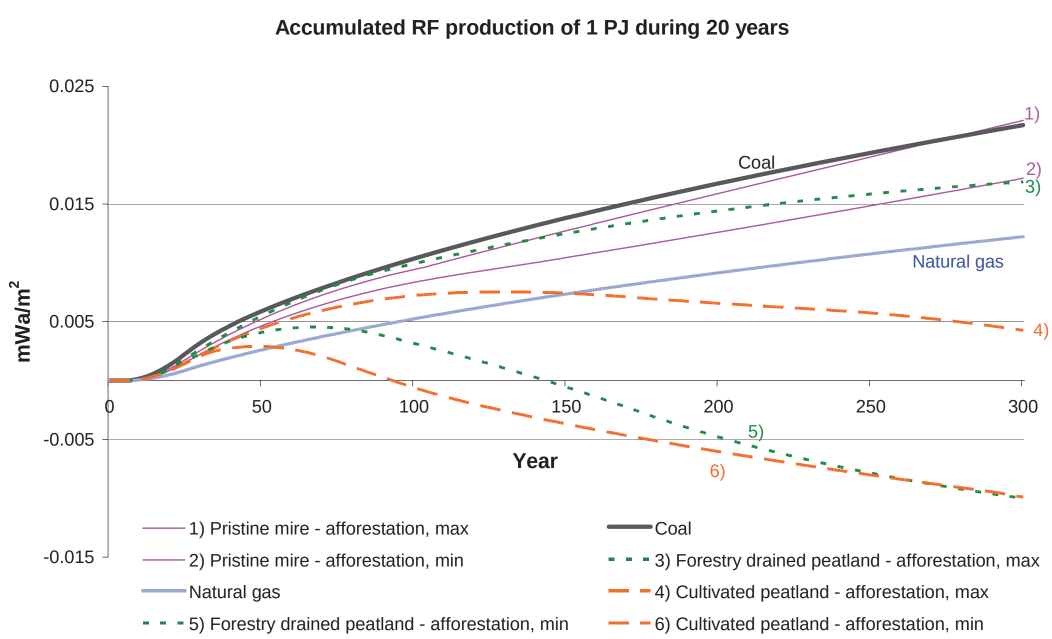

Figure 5.7. *Accumulated radiative forcing (RF) from scenarios where 1 PJ of peat fuel is produced during a period of 20 years. After Nilsson & Nilsson (2004).*

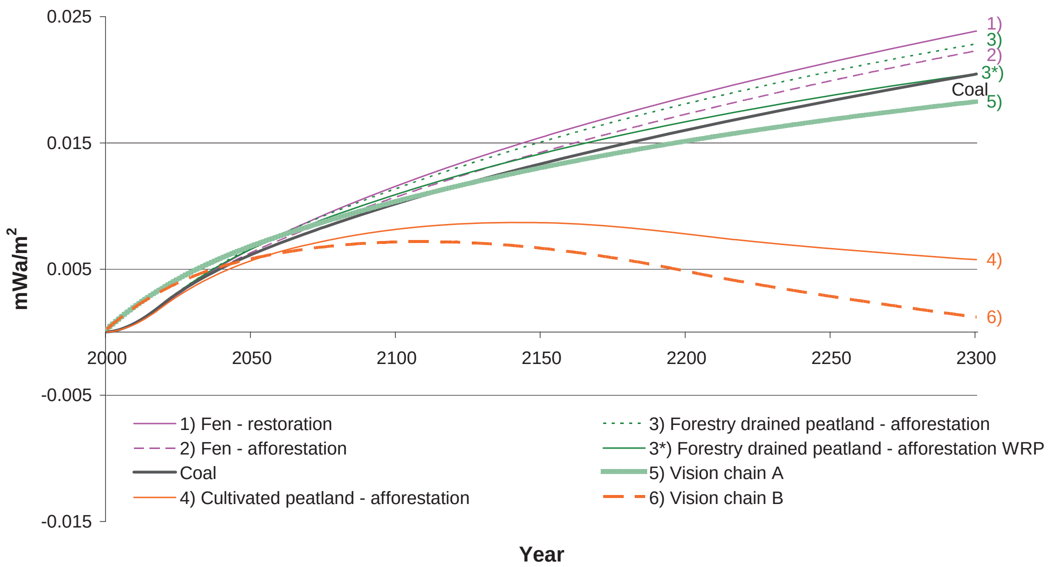

Figure 5.8. *Accumulated radiative forcing (RF) from peat fuel production and utilisation chain where 1 PJ of peat is produced during a time period of 20 years. From Kirkinen et al. (2007a). Reprinted courtesy of Boreal Environment Research.*

have assumed that the forest accumulates carbon until the average of the carbon pool over the forest rotation period is reached. All these different assumptions differ in terms of how the system boundaries of the scenarios have been set. Zetterberg *et al*. (2004) have more of a land-use perspective since they also consider energy produced from wood. Similar assumption has been made by Holmgren (2006) and Kirkinen *et al*. (2007b) where production of reed canary grass at the cutaway area was included in the peat scenarios. In the scenarios where restoration is the chosen after-use the system boundaries are more similar between the different studies.

Just as Savolainen *et al*. (1994), Zetterberg *et al*. (2004) have made calculations for scenarios where 1 PJ of energy is produced during 300 years (note that production begins in year 6 in both Zetterberg et al., 2004 and Nilsson & Nilsson, 2004). Three peat scenarios are shown including two scenarios where pristine mires are used and one scenario where forestry drained peatland is used. For the pristine mires, restoration is the chosen after-use. A maximum and minimum scenario has been calculated. The assumed CO_2 uptake at the restored area differs between scenario 1) and 3) in Figure 5.6, the maximum rate is 590 g CO_2/m^2yr and the minimum rate is 136 g CO_2 m^{-2} yr^{-1}. Savolainen *et al*. (1994) assumed an uptake rate of CO_2 at the restored area of 235 g CO_2 m^{-2} yr^{-1}. Nilsson & Nilsson (2004) assumed a value of 363 g CO_2 m^{-2} yr^{-1} based on Tuittila *et al*. (1999), whereas Kirkinen *et al*. (2007a) have based their estimate of 122 g CO_2 m^{-2} yr^{-1} on new measurements according to Alm *et al*. (2007). The estimated value of the CH_4 emissions differs between 2 - 40 g CH_4 m^{-2} yr^{-1}in the different studies.

The units used in the Figures 5.5 and 5.6 are different from the units in Figures 5.7 and 5.8. However the final numerical values are comparable, since the systems are linear. Only the integration has taken place in different parts of the consideration. In Figures 5.5 and 5.6 the integration is based on production of energy, and in Figures 5.7 and 5.8 it is based on integration of radiative forcing.

In Figure 5.7 results from Nilsson & Nilsson (2004) are shown. Included in the figure are peat fuel production and utilisation scenarios where either pristine mires, forestry drained peatlands or cultivated peatlands are used. Only one type of after-use is represented, i.e. afforestation. Nilsson & Nilsson also made calculations for restoration but recent research has shown that the emission estimates for the restored areas need to be updated and hence the results for restored areas in Nilsson & Nilsson (2004) are underestimates of the climate impact (Holmgren *et al*. 2006). Nilsson & Nilsson have calculated both maximum and minimum values for each of the peatland types. In addition there are also comparative scenarios for coal and natural gas.

In Figure 5.8 the fen chains represent production chains where the peatland was initially a pristine mire. Two different types of after-use have been applied for these chains, either restoration of the area to wetland or afforestation. In Finland the most common after-use is afforestation. Scenario 3* is a scenario where it is assumed that all of the peat can be removed during peat cutting and hence there is no residual peat (WRP = Without Residual Peat).

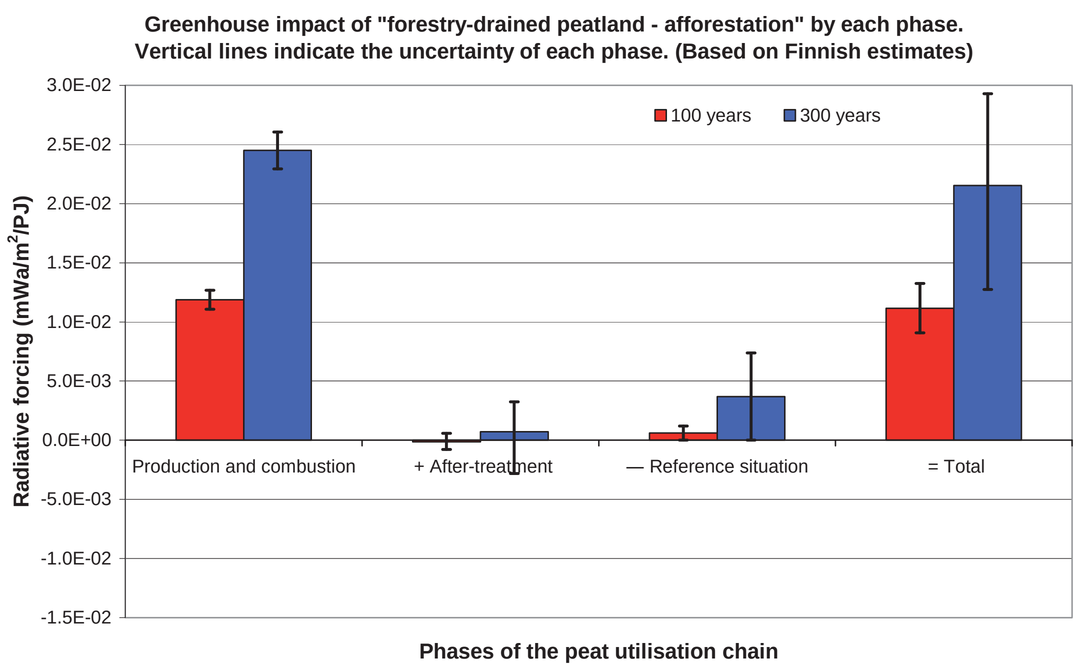

Figure 5.9. *Accumulated radiative forcing of forestry drained peatland – afforestation utilisation chain by each phase. Vertical lines indicate the uncertainty of each phase. The largest contribution to the total uncertainty comes from the reference situation and the after-use. The relative uncertainty increases with time. Source: Holmgren et al. (2006).*

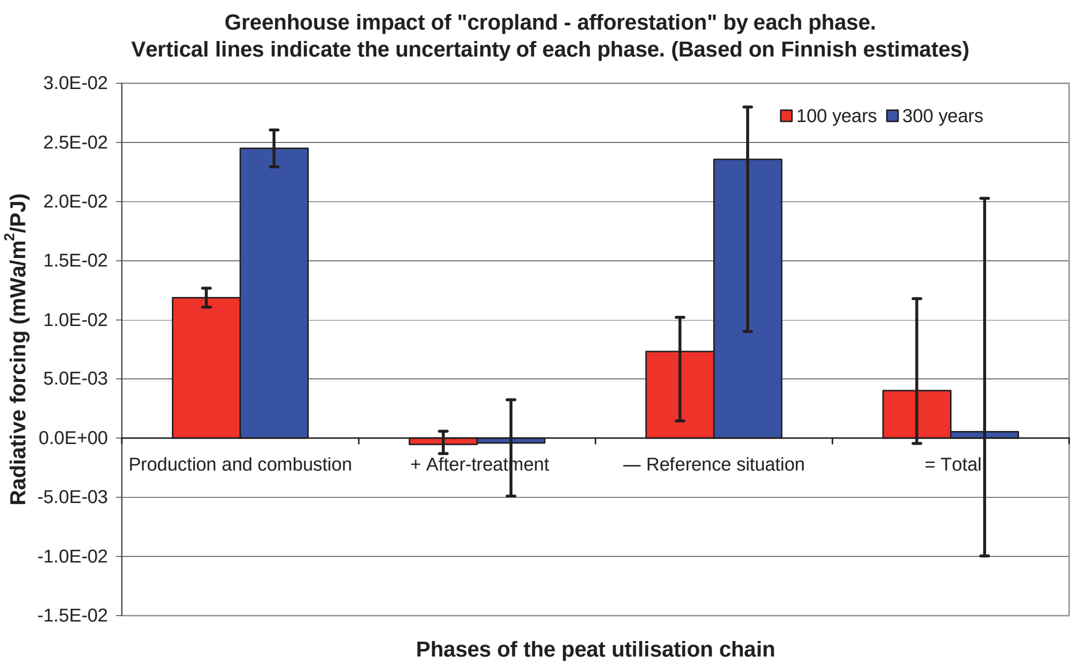

Figure 5.10. *Accumulated radiative forcing of cultivated peatland – afforestation utilisation chain by each phase. Vertical lines indicate the uncertainty of each phase. The largest contribution to the total uncertainty comes from the reference situation and the after-use. The relative uncertainty increases with time. Source: Holmgren et al. (2006).*

The assumption made by Kirkinen *et al*. (2007a) on CO_2 uptake at restored areas is based on new Finnish measurements; 122 g CO_2 m^{-2} yr^{-1} which is significantly lower than the assumptions in Savolainen *et al*. (1994), Zetterberg *et al*. (2004) and Nilsson & Nilsson (2004). For comparison, calculations have been made for the coal utilisation chain, which just like the peat utilisation chains includes emissions from production (coal mining), transportation and combustion. The vision chain B shows the impact of peat cutting at a cultivated peatland with the new production methodology whereas vision chain A shows the impact of peat cutting at forestry drained peatland with the new production methodology. The new production methodology (biomass dryer) is explained in Box 5.1 in this chapter.

Figure 5.9 shows the results from Holmgren *et al*. (2006), a comparative study of Finnish and Swedish results from LCA studies of the climate impact of fuel peat utilisation. Figure 5.10 is from the same study and shows the influence on the climate impact in terms of radiative forcing of the uncertainty in the emissions estimates in the different stages of peat production. The uncertainties of the greenhouse gas emissions from the peat cutting phase are small compared to the uncertainties in emission estimates of the initial (reference) and after-use phases. The after-use phase will last for a long time period (hundreds of years) and there is no data covering such long periods, hence it is difficult to know how the greenhouse gas balance will develop over time and the emission estimates are uncertain.

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5.6. Discussion

5.6.1. Climate impact of peat fuel utilisation

The climate impact in terms of radiative forcing of energy peat utilisation is dependent on the initial greenhouse gas balance (emissions and uptake) of the extracted peatland and the production methodology and after-use. A significant part of the uncertainty of the net climate impact (in terms of radiative forcing) is due to the uncertainty and variability in the estimates of the greenhouse gas balance of the peat reserve in its initial state. Emissions and uptake of greenhouse gases at peatlands varies between sites and years, depending on peatland topography and development history, and on temperatures and levels of water tables. In order to get representative estimates to be used in assessment calculations specific data would be needed including a large number of measurements from different sites, years, etc. To overcome these difficulties several scenario-based and sensitivity studies have been made, however since not all previous results could be included in this chapter we advise interested readers to seek further and more detailed information in the original references

The time spans involved in the development of peatlands are long, of the order of thousands of years. On the other hand, if we consider the mitigation of anthropogenic climate change, the time spans of interest for stabilising greenhouse gas concentrations in the atmosphere are of the order of one hundred years or even shorter. In the studies referred in this chapter the considered time spans extends to 300 years.

Note that the calculations presented in this chapter are made for Finnish and Swedish (boreal) conditions and the conditions in other climatic regions are different. For instance energy peat extraction in tropical peatlands (see Chapter 6), where significantly thicker deposits are removed, the fire risk is enhanced and large scale production methodologies complicate the after-use of the cutaway areas, most likely have significantly higher climate impacts. In addition there are also other environmental impacts (e.g. dust emissions, impacts on terrestrial and aquatic ecosystems), not described here that could be significant and that also should be considered when considering peat fuel extraction. The difference in radiative forcing between different peat fuel production chains is mainly dependent on the estimated greenhouse gas balance at the peat reserve in the reference situation. The climate impact of different peat fuel production and utilisation chains are not directly transferable between different countries, since the natural conditions and the measurement and assessment methodologies of the emissions from the initial situation of peat reserves can differ. Also, the emissions during the other stages of the production and utilisation chain can differ between countries depending on production methodologies, climate, properties of peatlands, etc.

Of the different phases in the peat fuel life-cycle the emissions of greenhouse gases from the combustion and production of peat fuel are known most accurately. However, estimates of the emissions from and the actual size of the surrounding area could be improved. The emissions and sinks of the peat reserves should be studied more. Especially the emissions estimates for already drained areas that are subject to different management could be improved. In addition the after-use alternatives need to be studied further both to develop system boundary definitions and to increase understanding of greenhouse gas balances.

If the cutaway peatland is used for producing renewable fuel, like wood

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biomass, the average greenhouse impact of utilising the peatland for energy production, expressed as climate impact per totally produced fuel energy (fuel peat and wood), will be lower than if only considering the peat fuel utilisation. In most countries using fuel peat it is recognised that there are important advantages in terms of reduced operational disturbances of co-firing peat and biomass fuels. It is therefore claimed that peat can facilitate the utilisation of biofuels in energy production.

5.6.2. Climate impact of horticultural peat

The production and cutting of horticultural peat will also result in greenhouse gas emissions just as for energy peat utilisation, although few studies using life cycle analysis are available (Cleary *et al*., 2005).

In most of the countries cutting and using energy peat, horticultural peat is also cut and used. In addition there are a number of countries where only horticultural peat is cut and used. Just as for energy peat cutting, the peatland has to be drained in order to carry the working machines used for cutting horticultural peat. All drainage of peatlands will result in losses of CO_2 to the atmosphere due to decomposition of aerated peat. However, depending on whether or not both horticultural peat and energy peat is cut at the same area the after use of a horticultural cutaway can be different compared to a traditional energy peat cutting site. In Canada for instance where energy peat is not cut, the remaining peat layer after horticultural cutting is significantly thicker then after energy peat cutting in Europe and Russia. This means that it is easier to re-grow *Sphagnum* moss at these sites.

Even if the horticultural peat is not burnt instantaneously, as is the case for energy peat in power stations, the cut peat will decompose and result in CO_2 emissions to the atmosphere; however this will occur at a slower rate than CO_2 release via combustion (Clearly *et al.*, 2005). Due to these reasons horticultural use of peat will also have an impact on the greenhouse effect. Still, the full extent of this impact remains uncertain due to the limited number of LCA of horticultural peat. Also, the fate of C stored through enhanced productivity resulting from the used of peat as growing media remains unclear, but could act a potential sink for some of the C released through decomposition.

5.7. Conclusions

For boreal conditions it can be concluded that the calculated climate impact in terms of radiative forcing of peat fuel utilisation can be lower when considering the entire production chain of the peat fuel compared to only considering the combustion phase. This is valid at least when extraction is made from areas already drained for other uses and which may be sources of greenhouse gases before peat extraction occurs. On the other hand if the extraction is made from peatlands that initially are sinks of greenhouse gases, the climate impact of the entire production and utilisation chain will be larger compared to only considering the combustion phase. Based on the Finnish and Swedish studies it seems as if peatlands used for agriculture are initially the largest sources of greenhouse gases and therefore when utilised for energy peat production also result in the smallest climate impact in terms of radiative forcing. The uncertainty range of the results is quite large especially due to the variability of emissions from peat reserves in their initial states. The largest contribution to the greenhouse gas emissions from peat fuel is the combustion phase. The combustion phase is in some cases responsible for over 90% of the total climate impact. The production phase has a comparably small impact.

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The climate impacts of different after-use options are less well known, partly due to the possibility of setting different system boundaries and, partly due to the small data availability on emission estimates from after-use areas. The most important after-use choices considered in the Finnish and Swedish studies are restoration and afforestation. Also, cultivation of energy grass (e.g. reed canary grass) has been studied in some recent studies. In Finland, Ireland and Sweden the most important after-use option of cutaway peatlands is afforestation. Afforestation as after-use seems to result in slightly lower climate impact than restoration. In addition to the already mentioned options, lake formation and cultivation at cutaway areas are possible after-use options. Compared to the climate impact of other phases of peat fuel life-cycle, the after-use contributes comparably little to the net climate impact.

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CHAPTER 6:

TROPICAL PEATLANDS: CARBON STORES, CARBON GAS EMISSIONS AND CONTRIBUTION TO CLIMATE CHANGE PROCESSES

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6.1. Introduction

6.1.1 Background

Peatlands worldwide play a vital role in biosphere biogeochemical processes (Immirzi *et al.*, 1992). The majority of the world's peatlands by area occur in boreal and temperate zones where they have formed in low-relief (poorly draining) environments under high precipitation-low temperature climatic regimes. In the humid tropics, however, regional environmental and topographic conditions have enabled peat to form under high precipitation-high temperature conditions (Rieley & Page, 1997). In lowland Southeast Asia, peatlands form part of the mosaic of rain forest types that includes mangrove, lowland dipterocarp, heath, montane and cloud forests (Rieley *et al.*, 1996; Page *et al.*, 1999). Most tropical peatlands are located at low altitudes where peat swamp forest occurs on top of a thick mass of organic matter, to which it has contributed over thousands of years, forming accumulated deposits up to 20 m thick (Anderson, 1983). Other areas of peatland in the tropics occur at high altitude, some of which are forested, while others contain *Sphagnum* mosses (Gore, 1983).

Tropical peatlands can be valued according to their functions, which are either, direct or indirect, products or attributes (Maltby, 1997; Joosten & Clarke, 2002). Direct functions include water flow regulation (water storage, filtration and supply), protection from natural forces (erosion prevention, flood mitigation), macro-climate stabilization, recreation and

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education, and production of food and other resources for local communities. Indirect or ecological functions of peatlands include sediment retention, nutrient detention, carbon (C) balance and storage, and micro-climate stabilization. Peatland products include provision of water to other ecosystems and human communities, forest resources ranging from fuel wood, timber and bark to resins and medicines, wildlife resources, agricultural and horticultural resources, and energy resources (Page & Rieley, 1998). Attributes of tropical peatland are values, other than products, that can be derived directly from the ecosystem or functions that are related closely to the maintenance of environmental quality (Joosten & Clarke, 2002). These include biological diversity since tropical peatlands are important genetic reservoirs of many animals and plants, unique locations for culture and heritage and habitats for the life cycles of flora and fauna. Many of the trees found in tropical peat swamp forest are endemic to the ecosystem, while it is the preferred habitat of some animals, for example, orang utan, often because it is the only location remaining for their survival (Morrough-Bernard *et al.*, 2003).

Tropical peatlands have long provided goods and services for local communities to fulfill their daily, basic requirements, for example, hunting grounds and fishing areas, food and medicines and construction materials. In the marginal areas surrounding tropical peatlands subsistence dry land agriculture has been practised by indigenous people for generations. More recently, timber extraction has been carried out, particularly in the peat swamp forests of Indonesia and Malaysia, providing employment, local income, new industries and business opportunities, and contributing to national exchequers, but at the expense of the ecosystem and the environment (Rieley & Page, 2005).

At the present time, and in the absence of human intervention, many tropical deposits are actively accumulating peat or are in a steady state (Brady, 1997a; b), although evidence suggests that climatic and land use conditions are no longer conducive to continued accumulation at many sites (Sieffermann *et al*., 1988; Page *et al*., 1999; Hirano *et al*., 2007). Land use change has altered some of the deposits and their environments significantly, causing the organic matter that accumulated rapidly in the past to undergo decomposition as a result of lowered water tables. Since the 1970s large areas of lowland tropical peatland in Southeast Asia, especially Indonesia, Malaysia and Thailand, have been converted, usually for single sector purposes, especially agriculture and forestry but to a lesser extent for aquaculture, energy and horticulture following forest clearance and drainage (Notohadiprawiro, 1998). Unfortunately, many projects have been unsuccessful and given rise to major problems as a result of land degradation and fire. In Indonesia, for example, agricultural development on thick peat, beyond tidal influence has failed, largely because planners considered peatlands to be just another type of land and did not take into account the special physical and chemical properties of peat soils. The Mega Rice Project in Central Kalimantan, an attempt to convert about one million hectares of wetland (mostly peatland) to rice fields, failed and similar problems have been experienced on peatland elsewhere in the Southeast Asian region (Muhamad & Rieley, 2002).

Much of the recent increased interest in peatlands globally has resulted from their importance as carbon sinks and stores and their role in carbon cycling between the earth's surface and the atmosphere (Immirzi *et al.*, 1992; Hooijer *et al.*, 2006). There is considerable debate about whether or not peatlands globally are net absorbers or emitters of carbon and under what

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conditions they may sequester or release this environmentally important element. It has been documented that tropical peatland acts as both carbon sequester and producer depending on seasonal changes in precipitation, temperature, the type of vegetation cover and land use. In SE-Asia, in recent years, detailed work has been carried out on carbon gas emissions from tropical peatlands in their natural condition and following degradation and conversion to agriculture and plantations (Hirano *et al*., 2007; Jauhiainen *et al.,* 2004; 2005; Melling *et al.,* 2005 a; b). Tropical forests, especially peat swamp forests, are sensitive, however, to temperature and precipitation changes, and evidence shows that long periods of drought can change peat swamp forests from carbon sinks to carbon sources (Suzuki *et al*., 1999; Hirano *et al.,* 2007).

Tropical peat swamp forest resources and natural functions are being damaged severely as a result of development, illegal logging and fire and may soon be destroyed forever with potentially devastating consequences regionally and globally (Page *et al.,* 2002; Rieley & Page, 2005; Hooijer *et al.,* 2006). Following land clearance and deep drainage the failed Mega Rice Project area became fire prone. Around 56% of this landscape burned in 1997 releasing 150-180 Mt C as gases and particulates to the atmosphere that contributed to climate change processes and also affected human health.

6.1.2. Location and extent of tropical peatlands

Tropical peatlands are found in mainland East Asia, Southeast Asia, the Caribbean and Central America, South America and southern Africa (Table 6.1). The current estimate of the total area of undeveloped tropical peatland is in the range 30 – 45 million hectares, which is approximately 10-12% of the global peatland resource (Immirzi & Maltby, 1992; Rieley *et al.,* 1996). Since most of these peat deposits are situated at low altitude in coastal and sub-coastal locations they are likely to be developed at a faster rate than the peatlands that remain in temperate and boreal zones.

Table 6.1. *Summary Statistics for Tropical Peatlands*[a]

REGION	AREA (Mean) (10^6 ha)	AREA (Range) (10^6 ha)
Central America	2.437	2.276 – 2.599
South America	4.037	4.037
Africa	2.995	2.995
Asia (mainland)	2.100	1.100 – 3.100
Asia (southeast)	26.216	20.205 – 33.211
The Pacific	0.019	0.019
TOTAL	37.80	30.632 - 45.961

[a] Based on Immirzi & Maltby, 1992; Rieley *et al.*, 1996

6.1.3. Location and extent of peatlands in Southeast Asia

Tropical peatlands in Southeast Asia occupy mostly low altitude coastal and sub-coastal environments and extend inland for distances of more than 150 km along river valleys and across watersheds. Most of these peatlands are located at elevations less than 50 m above mean sea level and cover more than 26 million hectares (~70% of all tropical peatlands). Extensive and fully developed tropical peatlands occur along the coasts of East Sumatra, Kalimantan (Central, East, South and West Kalimantan provinces), West Papua, Papua New Guinea, Brunei, Peninsular Malaysia, Sabah, Sarawak, Southeast Thailand and the Philippines (Figure 6.1; Table 6.2).

6.1.4. Age of lowland peat swamps of Southeast Asia

Based on radiocarbon dates, the onset and development of the present day peatlands in Southeast Asia range from the late Pleistocene to the Holocene (Page *et al.,* 2004). Most of the extensive peatlands along the coastlines, however, originated during the middle to late Holocene and are the youngest peatlands in the region. Peat accumulation of these deposits commenced around 3,500-6,000 cal yrs BP (e.g. Anderson & Muller, 1975; Staub & Esterle, 1994), following stabilisation of sea levels of the last transgression (a period of sea-level rise). In comparison, investigations of sub-coastal and inland peatlands, particularly in Borneo, have revealed much earlier dates for peat formation, ranging from Late Pleistocene (~40,000 ^{14}C yrs BP) in the Danau Sentarum basin of West Kalimantan (Anshari *et al*., 2001; 2004) to ~23,000 ^{14}C yrs BP for inland peat in the Sabangau catchment, Central Kalimantan (Page *et al.*, 2004) through to the early Holocene (10,000 – 7,000 ^{14}C yrs BP) for other deposits within Borneo (Neuzil, 1997; Sieffermann *et al*., 1988). Only a few records show, however, extensive and thick peat accumulation since the late Pleistocene and most records contain at least one hiatus as a result of global climatic changes since the last glacial maximum some 18,000 years ago. Clearly, tropical peatlands were involved in the global carbon cycle well before the boreal and temperate peatlands since the latter did not begin to accumulate until around 7,000 to 8,000 ^{14}C yrs BP (Maltby & Proctor, 1996). Several studies of tropical peatlands show, however, that the formation of contemporary deposits has been a dynamic process and that periods of alternating accumulation and degradation have occurred throughout their history (e.g. Anshari *et al*., 2001; Page *et al*., 2004).

Table 6.2. *Summary Statistics for Pre-development Area of Tropical Peatlands of Southeast Asia*[a]

REGION	**AREA** (Mean) (10^6 ha)
Indonesia	20.073
Malaysia	2.730
Papua New Guinea	2.890
Thailand	0.068
Brunei	0.100
Vietnam	0.183
Philippines	0.172
TOTAL	26.216

[a] Based on Rieley *et al*., 1996.

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Figure 6.1. *Distribution of peatlands in SE Asia, where ~70% of global tropical peat deposits occur, and location of selected sites discussed in this chapter.*

6.2. Tropical peatlands and the carbon cycle

In tropical peatlands, both the vegetation and underlying peat constitute a large and highly concentrated carbon pool (Sorensen, 1993). Currently, there is a growing body of information concerning the importance of carbon storage in, and carbon gas emissions from, tropical peatlands and the potential effect on global environmental change processes (Aucour *et al.*, 1999; Page *et al.*, 2004; Kool *et al.*, 2006). The reason for this interest is mainly because land development projects reduce the magnitude of these carbon pools (e.g. Page *et al.*, 2002). Human made ecosystem alterations can cause the natural resource functions of tropical peatlands to fail, converting them from net carbon sinks to net carbon sources.

6.2.1. Peat and carbon accumulation in Central Kalimantan and SE-Asia during the Late Pleistocene and Holocene

Few peatlands in Southeast Asia have been investigated in detail for peat structure, age and development, peat composition, C content, and C accumulation rates (e.g. Neuzil, 1997; Brady, 1997a; Wust, 2001; Page *et al*., 2004) despite the fact that they account for ~70% of the global tropical peat deposits. Information obtained from four sites across SE-Asia (see Figure 6.1 for site locations) illustrates variations in peat age and C accumulation rates and these are compared with high latitude peatlands.

The carbon storage potential of the lowland peatlands of Central Kalimantan was determined from a 9.5 m long core (SA6.5) obtained from the Sungai Sabangau catchment in 1995 (Page *et al.*, 2004; Figure 6.2). The geochronology, established with 27 AMS (radiocarbon) dates, reveals a record of peat and carbon accumulation over a period of 26,000 years. Initially, there was a relatively rapid rate of peat accumulation of 1.0 mm yr^{-1} between 24,000 – 26,000 cal yrs BP (23,000 – 22,000 ^{14}C yrs BP). This is equivalent to a carbon burial rate of about 54 g C m^{-2} yr^{-1} and may have lasted for several thousand years until about 20,000 cal yrs BP, although some peat may have been lost owing to degradation during the somewhat drier last glacial maximum (LGM; e.g. Flenley, 1997). Peat accumulation rates during and after the LGM fell to an average of 0.04 mm yr^{-1} until ~13,000 cal yrs BP (10,830 ^{14}C yrs BP), with an average carbon accumulation rate of 1.3 g C m^{-2} yr^{-1}. These low peat and C accumulation

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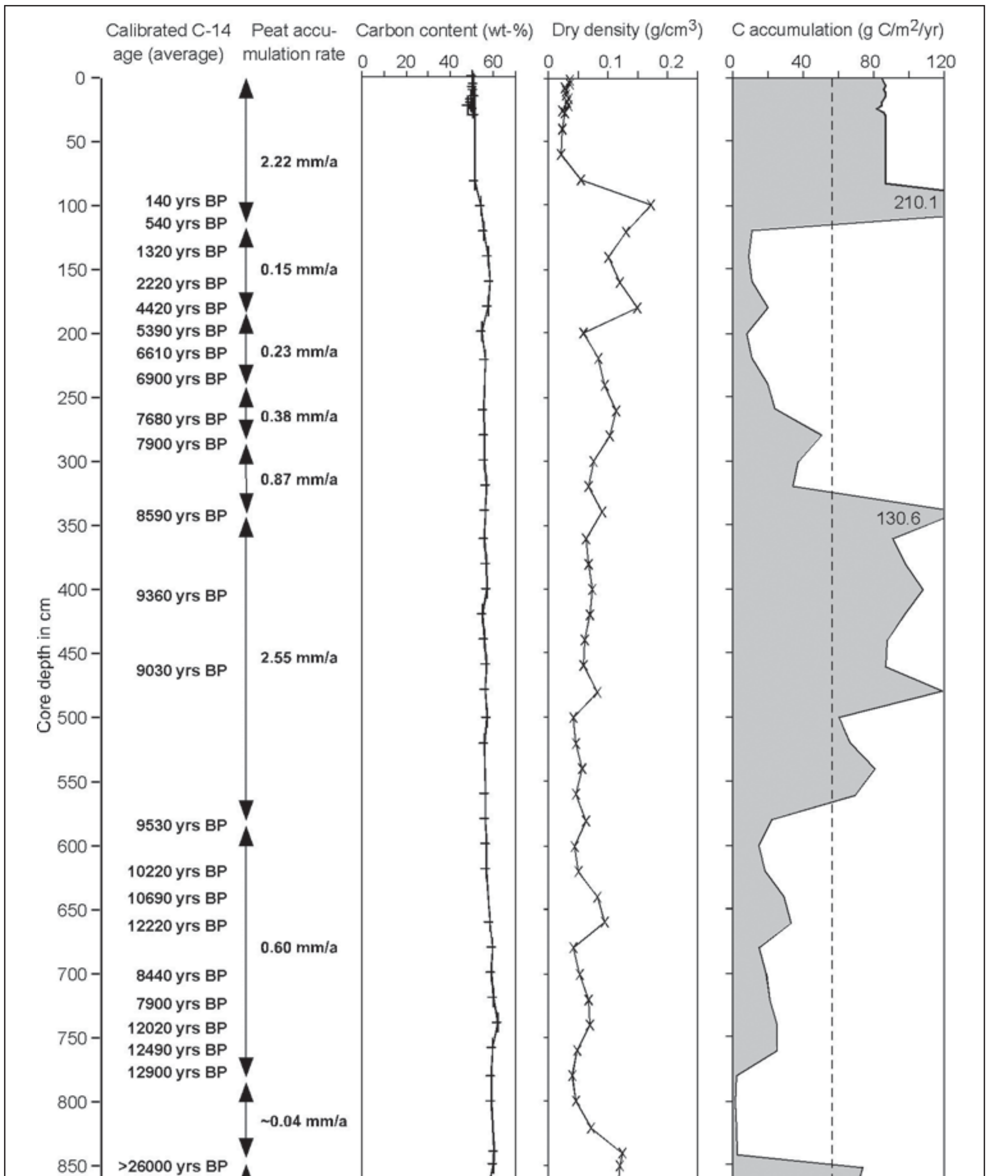

Figure 6.2. *Peat core SA6.5 from Central Kalimantan, Indonesia, showing stratigraphy, stage of peat humification, calibrated 14C ages, peat accumulation rates, carbon contents, dry density and carbon accumulation rates. (Modified after Page et al., 2004).*

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rates need to be treated with caution since it is possible that substantially more peat than currently preserved was deposited prior to the LGM but subsequently oxidised during the following 5,000 to 6,000 years when the climate was unfavourable for peat formation and during which the site likely acted as a C source (Koutavas *et al.*, 2002).

The beginning of the Holocene coincided with increased rates of both peat formation and carbon accumulation at the Sabangau site (Figure 6.2). Between 8,540 and 7,820 cal yrs BP the average peat accumulation rate increased from 0.60 to 2.55 mm yr^{-1} and over a ~2,200 year period, starting from 9,060 cal yrs BP, more than 3.5 m of peat accumulated. These values are similar to the range of 1.4 to 2.4 mm yr^{-1} reported by Sieffermann *et al*. (1988), but are higher than Neuzil's (1997), range of 0.3 to 1.2 mm yr^{-1}. The average rate of carbon accumulation during the early Holocene in the Sabangau peatland was 92 g C m^{-2} yr^{-1}, with a maximum rate of 131 g C m^{-2} yr^{-1}, exceeding the early Holocene rates from Sumatra reported by Neuzil (1997), which are in the range of 47 to 75 g C m^{-2} yr^{-1}, and the mid-Holocene highest accumulation rates (20 to 25 g C m^{-2} yr^{-1}) of temperate and boreal bogs from Canada (Turunen & Turunen, 2003).

Around 7,000 years BP rising sea levels stabilised, resulting in exposure of large, relatively flat areas of new coastal environments throughout the Malesian Region (Tjia *et al*., 1984; Hu *et al.*, 2003). The combination of favourable topographic (i.e. low relief) and wet, humid, climatic conditions (as a result of large shallow seas) led to rapid peat accumulation in coastal and sub-coastal areas throughout the region (Staub & Esterle, 1994; Hesp *et al.*, 1998). In the Rajang Delta of Sarawak, for example, 4.45 m of peat accumulated between 6,400 and 2,060 cal yrs BP (5,610 to 2,070 ^{14}C yr BP), equivalent to an average peat accumulation rate of 1.26 mm yr^{-1} whilst, on the east coast of Sumatra, the peatlands of Riau province also underwent very rapid accumulation with initial rates as high as 6 to 13 mm yr^{-1} between 5,300 – 4,300 cal yrs BP (4,700 and 3,900 ^{14}C yrs BP), reducing subsequently to 0.6 to 2.7 mm yr^{-1} (Neuzil, 1997). These rates are significantly higher than the rates for mid- to late Holocene peat accumulation in the Sabangau peatland at this time, which were only 0.15 – 0.23 mm yr^{-1}.

While the Central Kalimantan area around Sg. Sabangau seems to have received less precipitation after 8-8,500 cal yrs BP, and hence peat accumulation rates were reduced, several other sites in Sumatra and Malaysia commenced accumulation (Staub & Esterle, 1994; Neuzil, 1997) illustrating the significance of local and regional climatic changes, which occurred under the influence of raising sea level, sea surface temperature rise, ocean circulation changes in the Sunda Shelf and the Pacific, and the ice sheet collapses in the Northern Hemisphere. The climate variability during the Holocene led to a complex peat accumulation and degradation pattern across all tropical peatlands, best illustrated in the varying peat accumulation rates discussed in the section below. Peat accumulation rates based on 266 radiocarbon dates from sites across Sumatra, West Java, Kalimantan, Sarawak, Peninsular Malaysia, Thailand, Sulawesi, and New Guinea range mainly between 0 – 3 mm yr^{-1} with a median accumulation rate of ~1.3 mm yr^{-1} (Figure 6.3), which is about 2 – 10 times the rate for boreal and subarctic peatlands (0.2 – 0.8 mm yr^{-1}) (Gorham, 1991) and for temperate peatlands of 0.2 to 1 mm yr^{-1} (Aaby & Tauber, 1975). Very few tropical peatland sites have shown accumulation rates >3 mm yr^{-1}, with the highest, 13 mm yr^{-1}, reported from Bengkalis Island, Sumatra (Neuzil, 1997). Other studies have estimated the average accumulation rate for Indonesian peatlands to be between 1 and 2 mm yr^{-1} (Sorensen, 1993).

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Peat accumulation rates are essential for determining the carbon sequestration potential of tropical peatlands. Accumulation rates and C content allow the calculation of C accumulation rates both in the past and at present. Interestingly, comparison between thin (2 – 3 m) and very thick (>8 m) peat deposits from Malaysia and Kalimantan shows that C accumulation rates are similar despite the former being minerotrophic and the latter ombrotrophic. The mean C accumulation rates range between ~40 – 90 g C m^{-2} yr^{-1} (Figure 6.4).

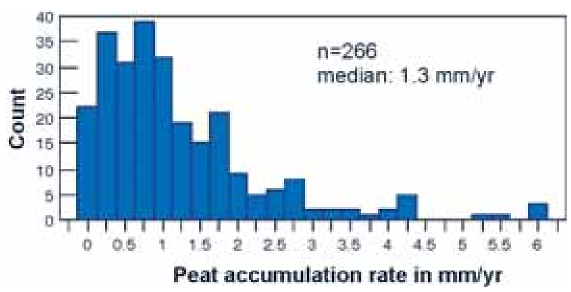

Figure 6.3. *Histogram of peat accumulation rates (showing only the range between 0 – 7 mm yr^{-1}) of 266 samples across sites in Sumatra, West Java, Kalimantan, Sarawak, Peninsular Malaysia, Thailand, Sulawesi and New Guinea. The median accumulation rate is ~1.3 mm yr^{-1} with only a few sites having accumulation rates >3 mm yr^{-1}, while one site on Bengkalis Island, Sumatra has a rate of ~13 mm yr^{-1} (Neuzil, 1997).*

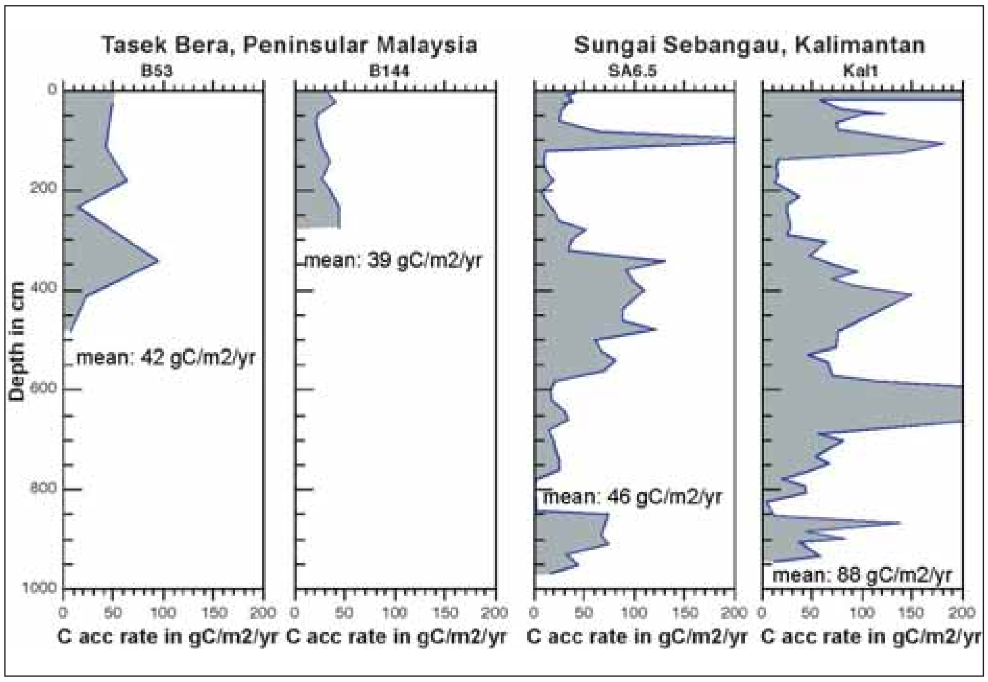

Figure 6.4. *Carbon accumulation rate down cores from two sites in Peninsular Malaysia (Tasek Bera, core B53 and B144) and Kalimantan (Sg. Sabangau; both SA6.5 and Kal1 are located within 1.5 km of each other). Tasek Bera is minerotrophic peat while the Kalimantan site is ombrotrophic.*

Other peat cores from Sumatra, from both the highlands (Tao Sipinggan, Toba Plateau, Figure 6.1) and the lowlands (Siak Kanan, Riau, Figure 6.1) show similar data (Figure 6.5). Peat accumulation rates vary markedly (e.g. between 0.3 – 6.1 mm yr^{-1}) over the history of the peat deposits. The highland peat deposit (Tao Sipinggan) started to accumulate just after Meltwater Pulse 1A (c. 14,500 cal yrs BP) (Liu *et al.*, 2004) while the lowland peat deposits (e.g. SK10) did not initiate until the mid Holocene similar to the peat deposits of Tasek Bera (e.g. B7 and B78). The C accumulation rate ranged from 30 – 50 g C m^{-2} yr^{-1} in the Sumatran highlands, 80 – 270 g C m^{-2} yr^{-1} in the Sumatran lowlands, 30 – 50 g C m^{-2} yr^{-1} in the Tasek Bera area, and 30 – 210 g C m^{-2} yr^{-1} in the Sg. Sabangau region in Central Kalimantan (Figure 6.5).

The peat cores from the various peatlands across SE-Asia and their carbon accumulation rates presented in the above section reveal several important findings. Firstly, they confirm that peat accumulation in both the highlands (Figure 6.5: Toba Plateau; Maloney & McCormac, 1995) and the lowlands (e.g., Kalimantan, Sg. Sabangau peat core SA6.5; Figure 6.4) commenced before the LGM. Since the lowland area of the LGM (when sea level was 125 m below present) is flooded now, the true extent of past lowland peat deposits may never be determined accurately from the limited data and records currently available. In addition, as in the case of the Sg. Sabangau core (Figure 6.2), some of the accumulated organic matter most likely oxidised during the somewhat drier conditions during and shortly after the LGM when the rainforest was markedly reduced in area throughout this region (Flenley, 1998). Other lowland peat deposits, especially coastal peatlands, were inundated during the transgression and this organic matter most likely oxidised except for locations where sediments buried some of the peat. This may be the reason why few submarine thin peat deposits on the Sunda Shelf have been described (Steinke *et al.*, 2003; Liu *et al.*, 2004).

Secondly, the comprehensive radiocarbon record of the Sg. Sabangau core (SA6.5; Figure 6.2), accompanied by a detailed account of peat accumulation, demonstrates that tropical peatlands are complex systems that develop stepwise rather than continuously. Factors influencing the rate of organic matter accumulation include type of vegetation, hydrological setting and climate and environmental changes associated with them. The peat accumulation data implies that, over their lifetime, tropical peat deposits can act naturally as both carbon sinks and carbon sources, although the accumulated peat represents net carbon storage over thousands of years. During the LGM, and towards the end of the Late Pleistocene, for example, the Sabangau deposit probably acted as a carbon source rather than a sink, owing to periods of peat degradation attributable to drier or seasonally variable climatic conditions.

Thirdly, there is evidence of two major periods of peat accumulation in Central Kalimantan during the Late Pleistocene, culminating in a phase of very rapid peat formation and high rates of carbon accumulation during the early Holocene as well as during the last ~600 years (Figure 6.4). Based on this information, the peatlands of Southeast Asia could have played a significant role in the global carbon cycle at those times by acting as a carbon sink from the latter part of the late Pleistocene until about 8,000 – 8,500 yrs BP and then nearer to the present day. This raises the question of how much carbon was stored from the early to mid-Holocene in the peatlands of Kalimantan and in tropical peatlands in general. At the same time, most other lowland and some highland peat deposits commenced peat accumulation only during the mid Holocene (~5,000 cal yrs BP) (Figure 6.5).

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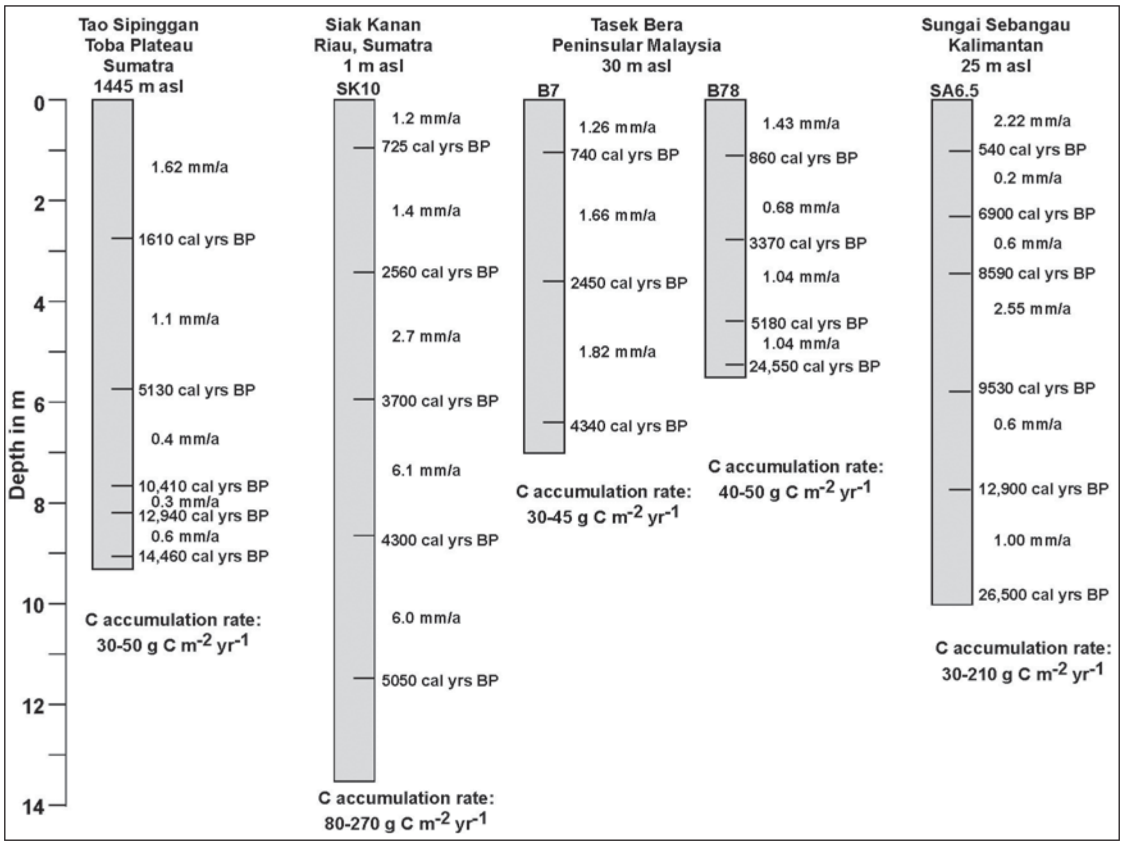

Figure 6.5. *Selected peat sections from various sites in Sumatra, Peninsular Malaysia and Kalimantan showing age of peat accumulation, peat accumulation rates and C accumulation rates that varied between 30 – 270 g C m^{-2} yr^{-1}.*

6.2.2. Contemporary peat and carbon accumulation in the tropics

Peat core-based estimates of present global C deposition within peatlands vary between 0.1 – 0.2 Gt yr^{-1} (Buringh, 1984; Armentano & Verhoeven, 1988), while the share of tropical peat is reported to be about 0.06 – 0.093 Gt yr^{-1} (Immirzi *et al*., 1992; Franzen, 1994).

Estimates of current carbon accumulation rates per unit area in tropical peatlands range from 0.59 – 1.45 t ha^{-1} yr^{-1} (Sorensen, 1993), exceeding the most rapid carbon accumulation rates for boreal and subarctic ombrotrophic bogs of 0.3 and 0.35 t ha^{-1} yr^{-1}, where 1 t ha^{-1} yr^{-1} = 100 g m^{-2} yr^{-1} (Turunen, 2003). Temperate peatlands are likely to be very similar to boreal peatlands (Turunen & Turunen, 2003; Roulet *et al*., 2007). Field research indicates that peat accumulation in the Sabangau peatland of Central Kalimantan, Indonesia is taking place currently at a rate of 2.2 mm yr^{-1} (Figure 6.2) (Page *et al.*, 2004), and thus far exceeds boreal and temperate peat growth rates of 0.2 to 0.8 mm yr^{-1} (Gorham, 1991) and 0.2 to 1 mm yr^{-1} (Aaby & Tauber, 1975), respectively.

The amount of C sequestered by peat swamp forest vegetation and allocated partly to accumulating peat is high in

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climax peat swamp forest although about 80-90% of the annual biomass produced is decomposed quickly and is unavailable for peat accumulation (Brady, 1997a). Suzuki *et al*. (1999), for example, estimated primary tropical peat swamp forest annual net carbon absorption to be 532 g C m^{-2} yr^{-1} in To-Daeng (Thailand), which is similar to net ecosystem carbon uptake of 516 g C m^{-2} yr^{-1} in Surinam (Williams *et al*., 2001). In contrast Whittaker & Likens (1975) provide a much higher value for 'tropical rain forest' of 990 g C m^{-2} yr^{-1}.

By applying the surface peat carbon accumulation rate of 101 g C m^{-2} yr^{-1} derived from the Sabangau peat core (Page *et al*., 2004; Figure 6.2), to the entire peatland area of Kalimantan (6.788 million hectares) (Rieley *et al*., 1996), the potential (pre-disturbance) carbon sink is estimated to be 6.86 Mt yr^{-1}, where 1 Mt = 10^6 t (Table 6.3). Extending this to the area covered by all of Indonesia's peatlands (20.074 Mha) (Rieley *et al*., 1996), gives a potential peatland carbon sink estimate of 20.28 Mt yr^{-1}. If this is extrapolated to the global area of tropical peatland (37.80 Mha) their carbon sink potential is 38.18 Mt yr^{-1}, which is equivalent to about 58% of temperate, boreal and subarctic peatlands (66.2 Mt yr^{-1}) (Turunen *et al.*, 2002), providing a revised estimate of the potential global peatland carbon sink of 104.38 Mt yr^{-1} to which tropical peatlands contribute 37%. In other words, although representing only 10% aerial coverage of total global peatlands, tropical peatlands represent over

Table 6.3. *Estimates of Carbon Pools and Potential Sinks in Indonesian, Southeast Asia and All Tropical Peatlands*

Location	Area (ha x 10^6)	Total carbon pool (Gt)[a,b]	Current carbon sink (Mt C yr^{-1})[b]
Kalimantan	6.788	15.05	6.86
Sumatra	8.253	18.30	8.34
West Papua	4.624	10.25	4.67
Rest of Indonesia	0.409	0.91	0.41
Total for Indonesia	20.074	44.5	20.28
Total for SE Asia	26.216	58.1	26.48
Total Tropical[c] Peatland	37.80	83.8	38.18

[a] 1 Gt = 10^9 t

[b] These estimates are calculated using values of 0.09 g cm^{-3} for bulk density, 56% for carbon content and 101 g C m^{-2} yr^{-1} of tropical peat derived from a 9.5 m metre peat core in Central Kalimantan (Figure 2 and Page *et al.,* 2004); A peat thickness of 4.4 metres was applied in the calculations derived from 126 peat drillings across several peat domes in Central Kalimantan.

Carbon pool = peatland area x peat dry bulk density x peat carbon content x mean peat thickness

[c] These values are provisional because little is known about the thickness, bulk density and carbon content of tropical peat outside of Southeast Asia. They will be updated as new information becomes available.

Potential carbon sink = peatland area x current carbon accumulation rate a third of the potential global peatland C sink. The value for carbon storage in all tropical peatlands is more than the 34 Mt yr^{-1} calculated by Armentano & Verhoeven (1988) but less than the bottom of the range of 41.5 – 93.4 Mt yr^{-1} suggested by Maltby and Immirzi (1993). It should be noted, however, that all of these values refer to potential rather than actual carbon storage whilst the carbon sequestration function of large areas of tropical peatland has been reduced greatly by deforestation, drainage, agricultural conversion and fire, all of which convert peatland ecosystems from carbon sinks to carbon sources.

6.2.3. Role of tropical peatlands as contemporary carbon sinks and stores

There are major difficulties associated with evaluating the role of tropical peatlands as contemporary carbon pools. Firstly, some peatlands, even under natural conditions, are in a steady-state and no longer accumulating peat, whilst others are undergoing degradation (Sieffermann *et al*., 1988; Moore & Shearer, 1997; Hirano *et al*., 2007). Secondly, large areas of tropical peatland have been drained and burned, which has altered the water table as well as water retention properties, shifting the peatland carbon balance from sink to source (Wösten *et al*., 1997; Page *et al*., 2002; Canadell *et al*., 2007). The numbers presented in Table 6.3 are, therefore, a best estimate of pre-disturbance rather than actual contemporary carbon storage values.

Additional problems are encountered when calculating the carbon pool (store) in tropical peatlands, either nationally (e.g. Indonesia), regionally (e.g. Southeast Asia) or globally. National peatland inventories in tropical countries are inaccurate owing to a lack of resources for survey (remote sensing and aerial photographs), difficulty of separating forested peatland from other forest types (e.g. peat swamp forest from other rain forest types, especially heath forest), peat swamp forest from deforested peatland, developed peatland from non-peatland, and inadequate ground truthing. In addition, areas of tropical peatland cleared of their forest cover and then drained and cultivated are usually no longer regarded officially as peatland and are excluded from inventories, although they may continue to have a considerable thickness of peat and a substantial carbon store for a long time after conversion. Information on peat thickness in all tropical countries is very inadequate making it virtually impossible to estimate peat volumes and carbon pools with precision. Despite these limitations, it is possible to provide reasonably accurate values for the total carbon pool contained within the peatlands of Central Kalimantan, from which to derive approximations for the whole of Indonesia and tropical peatlands globally using information derived from remotely sensed and field checked data for the first (Page *et al.*, 2002), official statistics for the second (RePPProT, 1990, cited in Rieley *et al*., 1996) and published information for the last (Immirzi *et al.*, 1992). These areas, when combined with values for peat thickness, bulk density and carbon content, provide estimates of the different carbon pools (Table 6.3). The peatlands of Kalimantan represent a carbon pool of 15 Gt C, those of Indonesia contain 45 Gt C, whilst the global total is in the region of 84 Gt C. The value for the Indonesian peatland carbon pool is much higher than the range of 16 – 19 Gt suggested by Sorensen (1993), and that for all tropical peatlands is greater than the top of the range of 53 – 70 Gt estimated by Immirzi *et al.* (1992).

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6.3. Greenhouse gas (GHG) emissions from natural and degraded tropical peatland

In their natural state, peat swamp forests have the ability to sequester carbon from the atmosphere during photosynthesis, retain this in plant biomass and store part of it in the peat. This process occurs mainly because of the waterlogged condition of the peat, which reduces decomposition significantly and hence the rate of organic matter production exceeds its breakdown. Peatland development, however, requires drainage, brings about changes in the vegetation type, destroys the C sequestration capacity and leads to losses from peat C stores. Carbon is lost especially in the form of carbon dioxide (CO_2) owing to the deeper oxic peat profile caused by water level draw-down. Aerobic conditions and high redox potentials created by drainage are known to favour microbial activity and nitrogen mineralization (Ueda *et al.*, 2000; Jali, 2004), which can enhance C loss by peat oxidation. Peat derived GHG emissions are compared in peat swamp forest and other land use types subject to different cropping vegetation types and peat drainage intensities.

6.3.1. GHG emissions from natural peat swamp forest

Carbon dioxide fluxes

Greenhouse gas release in organic matter decomposition in temperate and boreal peatlands is controlled largely by seasonal changes in temperature, hydrological conditions, and substrate availability and quality (Komulainen *et al.*, 1998; Kettunen, 2002; Vasander & Kettunen, 2006). In the tropics, surface peat temperature is high and promotes rapid organic matter decomposition throughout the year. Although peat temperature can be equable in the tropics, rainfall varies markedly both daily and annually (Takahashi & Yonetani, 1997). The water table in undrained peat swamp forest is above or close to the surface throughout the year and fluctuation is usually small. Draw down of the water table in the dry season is restored quickly at the onset of the subsequent wet season (Takahashi *et al.*, 2000). Periodicity and duration of oxidation-reduction conditions in drying or wetting tropical peat have important roles in the initiation of GHG production (Ueda *et al.*, 2000; Inubushi *et al.*, 2003), but the dynamics of the processes involved are inadequately known.

Few data are available for CO_2 flux rates from the forest floor in peat swamp forests (Chimner, 2004; Jauhiainen *et al*., 2005; Melling *et al*., 2005a). The estimated annual CO_2 flux in undrained selectively logged forest in Central Kalimantan under various hydrological conditions is in the range 953±86 – 1061±83 g C m^{-2} yr^{-1} (Jauhiainen *et al*., 2005), and is comparable with emissions of 1200±430 g C m^{-2} yr^{-1} in a secondary peat swamp forest in South Kalimantan (Inubushi *et al*., 2003). These values differ greatly, however, from the 2130 g C m^{-2} yr^{-1} estimate for drained peat swamp forest in Sarawak (Melling *et al*., 2005a). The large variation in peat swamp forest floor CO_2 emission estimates arises mainly from differences in measurement procedures and methods, variation in environmental conditions (especially peat moisture and water table depth), micro site type selection and vegetation characteristics. Therefore measurement conditions should be specified clearly in the estimates.

The ground (peat surface) in peat swamp forest is a complex continuum of micro sites where differing hydrological conditions and sub-soil structure lead to large differences in CO_2 emissions originating from respiration of heterotrophic decomposers and tree root respiration. Microtopography of the ground surface of peat swamp forest is a mosaic

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of hummocks and hollows. Hummocks consist of tree bases and densely packed small tree roots that accumulate organic debris: their height varies but can be up to 50 cm above the forest floor. The hollows are more open, sparsely vegetated with a variety of pneumatophores and fine breathing roots; their dimension varies in area, shape and depth and they are a prominent feature of the forest floor. The duration of waterlogging differs between hummocks and hollows. The former are water-saturated only after heavy rain while the latter are filled with water frequently, a condition that can last for many months and they can retain moisture throughout the dry season.

According to Jauhiainen *et al*. (2005) CO_2 flux rates in hollows in undrained forest vary from less than 100 mg m^{-2} h^{-1} to almost 900 mg m^{-2} h^{-1} (Figure 6.6). The highest emissions occur in the dry season when the water table falls to 40-50 cm below the surface. Compared to hollows, hummock CO_2 flux rates, under different water table conditions, are relatively uniform (500 – 600 mg m^{-2} h^{-1}) (Jauhiainen *et al*., 2005), (Figure 6.6). Root respiration is an important source of CO_2 especially in hummocks and may form over 50% of the emissions (Hirano, personal communication).

The actual ratio of hummocks to hollows in relation to water table depth over large areas of tropical peatland is complex, variable and difficult to determine. When comparing cumulative forest floor CO_2 fluxes at equal hummock-hollow microtopographical coverage, hummocks are a stronger CO_2 emission source owing to the lower variation in flux rates at different peat water table levels than hollows; high water table conditions clearly lead to reduced cumulative emissions from hollows (Figure 6.7). When the water table is below the peat surface, cumulative flux rates from both hummocks and hollows are similar. Emission rates from both should be included in overall estimates since the ratio of one to the other influences overall CO_2 flux rates. Jauhiainen *et al.* (2005) observed almost 5% change in annual peat swamp forest floor CO_2 emission rate for each 10% change in the proportion of the area of hummocks and hollows.

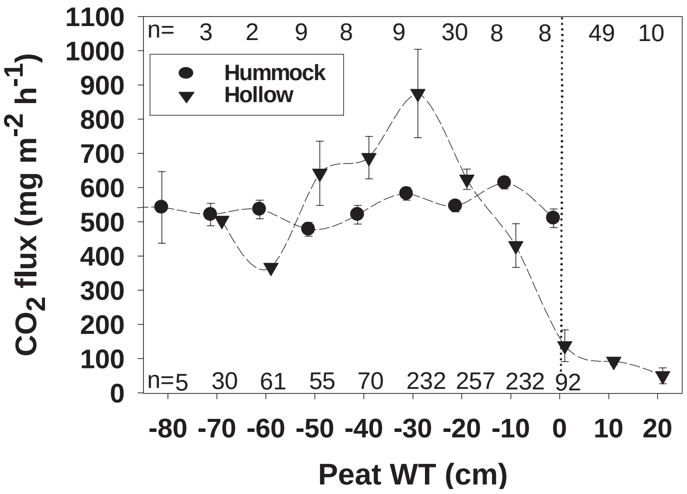

Figure 6.6. *Hummock and hollow mean CO_2 flux rates (mg m^{-2} h^{-1} with standard error) from peat swamp forest in Central Kalimantan, Indonesia at various peat water table (WT) depths. The dotted vertical line shows when the water table is at the peat surface. (Based on Jauhiainen et al., 2005).*

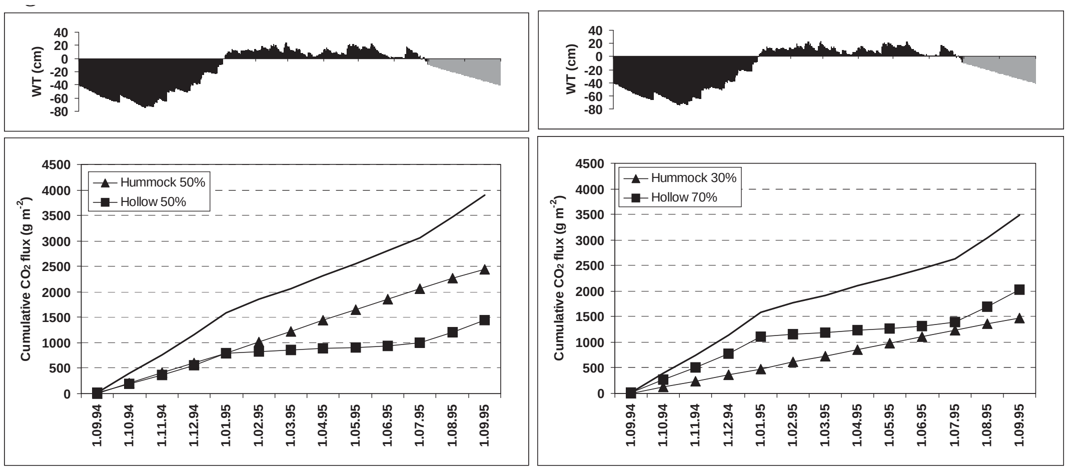

Figure 6.7. *Water table fluctuations (WT) (top) and annual cumulative CO_2 emission (g m^{-2}) (bottom) from mixed peat swamp forest hummocks and hollows (lines with symbols), and combined annual emission (line only) in Sg. Sabangau area, Central Kalimantan. Emissions are presented for 2 hummock-hollow coverage areas; 50:50% on left and 30:70% on right (Based on Jauhiainen et al., 2005).*

Methane fluxes
Methane (CH_4) flux rates from the peat surface in peat swamp forest are between 0.11 and 0.96 mg m^{-2} h^{-1} (Inubushi *et al*., 2003; Jauhiainen *et al*., 2005; Melling *et al*., 2005b). These rates are lower than from boreal *Sphagnum*-dominated bogs, which range from 0.21 to 1.61 mg m^{-2} h^{-1} (Martikainen *et al*., 1995; Nykänen *et al*., 1998). Methane formation in peat requires anoxic conditions (waterlogging). When the depth of the oxic surface peat layer increases, as a result of low rainfall or drainage, methane is oxidized to carbon dioxide and emissions decrease to zero. In dry conditions the direction of CH_4 flux can be from the atmosphere into the peat (Figure 6.8).

Methane fluxes in peat swamp forest hollows can be regarded as approximations or slight over-estimates of overall forest floor fluxes because CH_4 is produced only in waterlogged conditions. The water table in hollows is always nearer to the surface than in hummocks and the peat in hollows therefore affords less space for methane oxidation by methanotrophic bacteria than in hummocks. There are no published data on peat swamp forest hummock CH_4 flux rates. Based on flux measurements in hollows, however, the annual CH_4 emission from peat swamp forest floor is less than 1.06±32 – 1.36±57 g CH_4-C m^{-2} yr^{-1} (Jauhiainen *et al*., 2005), while secondary peat swamp forest in South Kalimantan gave a similar value of 1.2±0.4 g CH_4-C m^{-2} yr^{-1} (Inubushi *et al.*, 2003). The estimate of 0.0183 mg CH_4-C m^{-2} yr^{-1} obtained by Melling *et al*. (2005b) from a peat swamp forest in Sarawak, however, is much less and may reflect its drained and drier condition. Annual CH_4 emissions from tropical peat swamp forest floor are small compared to those from undisturbed boreal *Sphagnum* bogs, which vary between 2 and 15 g CH_4-C m^{-2} yr^{-1} (Martikainen *et al.*, 1995; Nykänen *et al.*, 1998; Alm *et al.*, 1999).

The ratio between CH_4 production in anoxic peat and CH_4 consumption in oxic

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peat determines the gas flux rate at the peat surface (Roulet *et al.*, 1993; Shannon & White, 1994; Komulainen *et al.*, 1998). Maximum CH_4 emission rates from the peat swamp forest floor occur when the water table is near to the peat surface and recently deposited, less decomposed litter (Brady, 1997a) becomes available for anaerobic decomposers, e.g. methanogens (Figure 6.8). In drier conditions oxygen can penetrate into peat pore space and CH_4 consumption by methanotrophic bacteria can balance or slightly exceed CH_4 production, and thus create the potential for CH_4 flux redirection into peat (Figure 6.8). These conditions were increasingly created in peat swamp forest in Central Kalimantan when the water table fell below 20 cm from the surface.

It is possible, however, that some of the CH_4 produced may escape to the atmosphere through vascular plant organs (e.g. leaves and pneumatophores), and thus avoid oxidation in the surface aerobic peat. Remote sensed observations indicate CH_4 is released from tropical forests (Frankenberg *et al.*, 2005; Sinha *et al.*, 2007) but the role of peat swamp forest vegetation in these emissions has still to be clarified. In temperate and boreal wetlands, vascular plants have been noted to have an important role in providing a transport route for CH_4 emissions (Shannon *et al.*, 1996; Frenzel & Rudolph, 1998; Saarnio & Silvola, 1999).

The global warming potential (GWP) of methane emissions from tropical peat is of minor importance compared to that of CO_2. By converting annual CH_4 fluxes into CO_2 equivalents (CO_2-e) by multiplying by 23 (IPCC, 2001; see also Chapter 5), the total CH_4 emissions (31.3 g CO_2-e m^{-2} yr^{-1}) represent only 0.8 – 0.9% of the corresponding CO_2 emissions (3892 – 3493 g CO_2 m^{-2} yr^{-1}) from the ground in undrained forest (Jauhiainen *et al.*, 2005; see also Table 6.4). GWP comparisons using data from Inubushi *et al*. (2003) and Melling *et al*. (2005a; b) provide similar or even smaller contributions by CH_4.

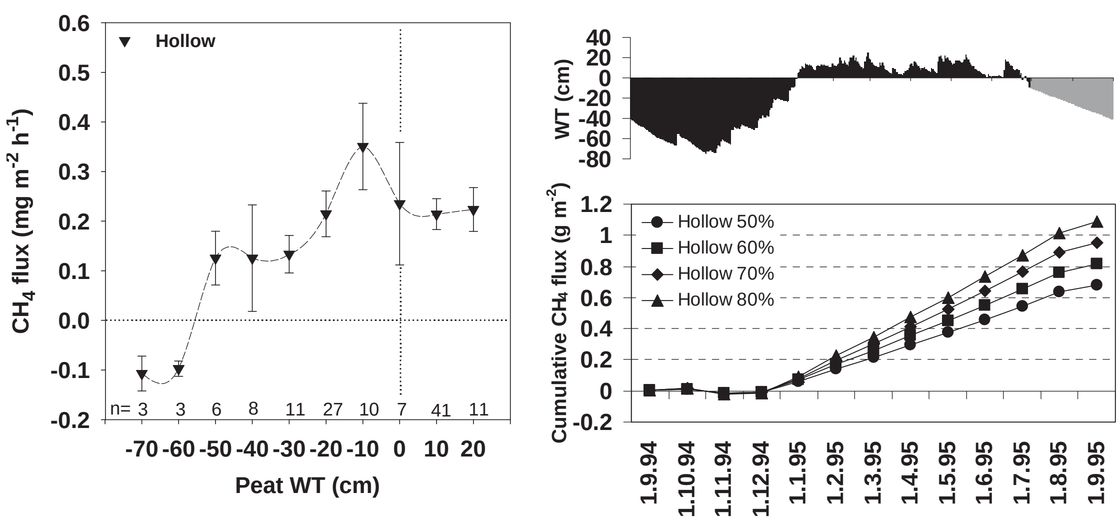

Figure 6.8. *Methane fluxes (mg m^{-2} h^{-1}, with standard error) at various peat water table depths (left) and annual cumulative methane emission (g m^{-2}) (right) from hollows of peat swamp forest in Central Kalimantan, Indonesia. Cumulative emission (1.9.1994 – 31.8.1995) is presented at various hummock-hollow coverage ratios (%). (Based on Jauhiainen et al., 2005).*

Table 6.4. *Carbon Gas Emissions from the Peat Surface in Natural and Degraded Peat Swamp Forest in Central Kalimantan, Indonesia*[a]

	LAND USE	CO_2 EMISSIONS		CH_4 EMISSIONS		TOTAL EMISSIONS
		CO_2 g m^{-2} h^{-1}	CO_2 (CO_2-C) g m^{-2} yr^{-1}	CH_4 mg m^{-2} h^{-1}	CH_4 (CH_4-C) g m^{-2} yr^{-1}	CO_2-e
A	**Undrained peat swamp forest**[b]	43±15 to 689±62	3892±304 (1061±83)	-0.08±0.09 to 0.35±0.01	1.36±0.57 (1.02±0.43)	3892 ± 31.3 = 3923
B	**Drainage affected peat swamp forest**[c]	0±40 to 1404±446	4000 (1091)	-0.08±0.003 to 0.25±0.09	1.3 (0.98)	4000 ± 29.9 = 4030
C	**Clear felled recovering peat swamp forest**	71±18 to 1521±724	3400 (927)	-0.13±0.03 to 0.34±0.06	2 (1.5)	3400 ±46 = 3446
D	**Drained uncultivated agricultural land**	0±44 to 453±19	1928 (526)	-0.04±0.03 to 0.04±0.004	0.12 (0.09)	1928 ± 2.8 = 1931

[a] Temporal and cumulative annual CO_2 and CH_4 flux rates (mean with standard error), and total emissions as CO_2-e global warming potential are given based on data in Jauhiainen *et al*. (2004; 2005).

[b] Numbers for CO_2 are based on 50:50% surface coverage ratio between hummocks and hollows. Numbers for CH_4 are based on hollow emissions assuming 100% surface coverage.

[c] Numbers for CO_2 are based on 30:70% surface coverage ratio between hummocks and hollows.

6.3.2. GHG emissions from degraded tropical peat

Comparison of temporal carbon gas fluxes from three drainage affected tropical peatland site types

Comparison of carbon dioxide and methane fluxes in the upper part of Block C of the former Mega Rice Project in Central Kalimantan, Indonesia shows the effect of different vegetation cover types and land uses (Figure 6.9). This area was the location for intensive land development between 1996 and 1999 and includes drainage affected selectively logged forest, clear felled but recovering (regenerating) forest on drained peat and drained but uncultivated agricultural peatland used previously for growing vegetables. The agricultural area was drained some 20 years prior to the gas flux measurements, and some 15 years earlier than the two other sites. Gas fluxes from these three locations were compared with those from undrained mixed peat swamp forest located in the Sg. Sabangau catchment about 10 km away at a similar distance from the river but on the opposite bank (see Section 6.3.1).

Land use change from peat swamp forest to agriculture affects carbon sequestration markedly because the tree biomass is removed totally and replaced with non peat-forming crop plants. Agriculture requires drainage which creates permanent oxic conditions in peat down to the minimum water table required for optimum crop growth. Variations in precipitation and drainage affect the ability of peat swamp forests to maintain their hydrological

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integrity causing them to emit CO_2 during times of water draw down (Suzuki *et al*., 1999; Hirano *et al*., 2007). Impairing the water holding capacity of peat increases surface peat oxidation and leads to subsidence and loss of carbon (e.g. Wösten *et al*., 1997; 2006; Jauhiainen *et al*., 2004; Furukawa *et al.*, 2005; Melling *et al.*, 2005a). On drained peat, cultivated plants cannot contribute to C sequestration because of their small biomass, compared to natural forest, most of which is removed when cropped or decomposed rapidly as soon as it is deposited on the peat surface.

The highest CO_2 flux rates were obtained in forested and recovering sites, both of which were drained about 5 years before gas measurements were made (Figure 6.9). Gas flux rates at comparable water table depths in the drainage affected forest floor hollows were slightly higher, and fluxes from hummocks were considerably higher, than emissions from undrained forest (Figures 6.6 & 6.9). These differences in gas flux rate response are caused either by markedly higher autotrophic respiration from tree roots and/or enhanced peat oxidation as a result of drainage. CO_2 flux rates in the clear felled recovering area were similar to the forest hollow fluxes at various water table depths. At both sites, CO_2 flux rates increased down to relatively deep peat water table depths, while the maximum rates in undrained forest were obtained at much shallower water tables (Figures 6.6 & 6.9). This may be a consequence of annually repeated deeper drainage during the dry season, which exposes a thicker peat profile to the air and improves the environment for aerobic decomposers in the drained sites. Uncultivated agricultural land with regulated drainage resulted in the lowest CO_2 flux range of the three sites (Figure 6.9).

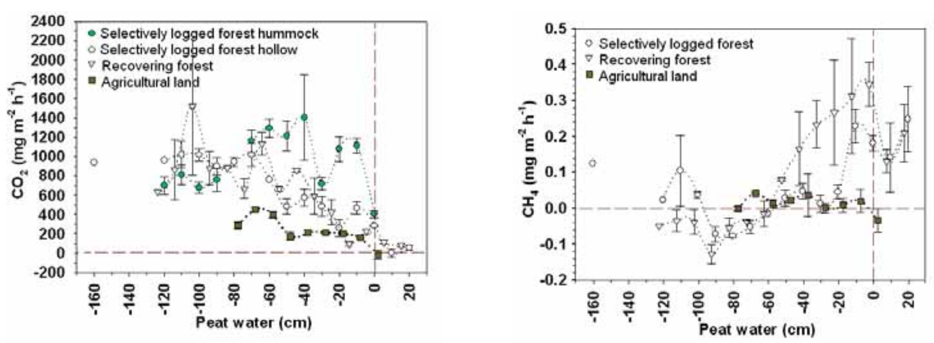

Figure 6.9. *Peat CO_2 flux rates (on left) and CH_4 flux rates (on right) at three tropical peatland sites (mg m^{-2} h^{-1} with standard error) at various peat water table depth classes. The dotted vertical line shows when the water table is at the peat surface and the horizontal line in the CH_4 graph the zero-flux rate. Note the different scales in the graphs. Based on Jauhiainen et al. (2004).*

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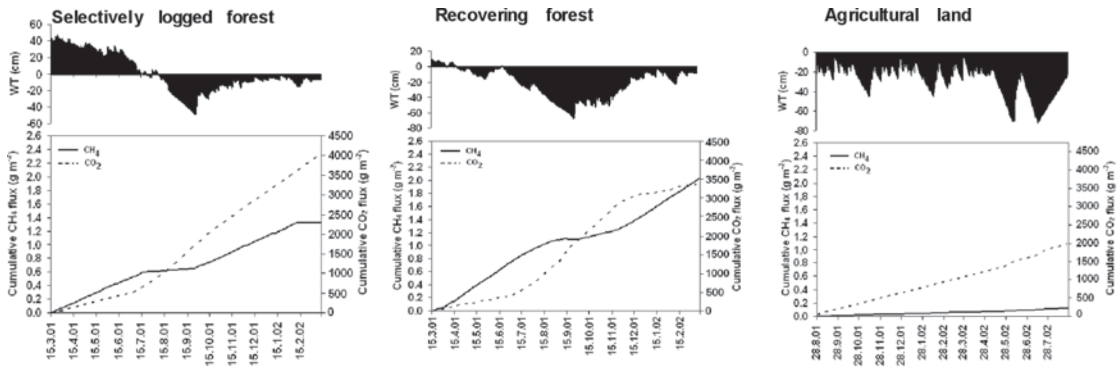

Figure 6.10. *Annual cumulative CH_4 and CO_2 fluxes (g m^{-2}) at the peat surface of three drained/drainage affected sites in Central Kalimantan. The water table (cm) below the surface at the sites is relative to the hollows in peat swamp forest and the almost level surface in the agricultural area. (Based on Jauhiainen et al., 2004).*

Following drainage, the volume of peat experiencing oxic conditions increases considerably and the potential for CH_4 oxidation by methanotrophic bacteria is much greater than in undrained forest peat. Methane flux rate ranges were greatest in the drainage affected forest hollows and in the recovering forest floor (Table 6.4). In these sites, peak CH_4 emissions occurred when the water table was near or above the peat surface (Figure 6.9). On the agricultural land, CH_4 flux rates were almost zero at all peat water table depths (Figure 6.9). The differences in these flux rates can be attributed to a low organic carbon supply and controlled drainage on agricultural land, whereas abundant vegetation of trees and bushes at the two other sites supply litter continuously into the peat (roots) and onto the surface of the peat (leaves and branches) where it can be consumed by methanogenic bacteria in water saturated (anoxic) conditions. CH_4 fluxes at these three site types (see Table 6.4) are similar to those from other land use types on drained peat. For example, CH_4 fluxes from the peat surface in an oil palm plantation are from -0.044 to 0.0056 mg m^{-2} h^{-1} (water table maximum ~-60 cm), whilst in a sago plantation on peat, which requires shallower drainage (water table maximum ~-27 cm) the range is -0.0010 to 0.14 mg m^{-2} h^{-1} (Melling *et al*., 2005b).

Cumulative (annual) gaseous C emissions from three drainage affected site types in 2001/2002

The highest annual CO_2 emission (4000 g m^{-2} yr^{-1}) was recorded in the drained forest (Figure 6.10, Table 6.4) where the CO_2 flux rate from hummocks was very high even in wet conditions (see Figure 6.7, Table 6.4). Annual CO_2 emission in the recovering forest site was slightly lower than in undrained peat swamp forest (Figure 6.10, Table 6.4). There were abundant trees growing at both of these sites, resulting in considerable litter production, so that both root respiration and decomposition would be taking place. Annual fluxes in the drainage affected forest site were similar to the undrained forest site because hydrological conditions on both sites were almost the same. In the drainage affected forest, however, the water table, which was 75 cm below the surface in 2001, fell to much lower depths of -153, -113, -167 and -108 cm in the years 2002 – 2005, respectively, probably as a result of progressive peat

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degradation. The highest CO_2 emission rates in drainage affected sites occurred where drainage channels were deepest.

The annual CO_2 flux at the agricultural site was considerably lower than at the other two sites (Figure 6.10, Table 6.4). The agricultural land emission (526 g CO_2-C m^{-2} yr^{-1}) was about the same as has been estimated to result from annual subsidence of developed tropical peatland in Johor and Sarawak, Malaysia (Wösten *et al*., 1997). According to Melling *et al*. (2005a), oil palm plantation emissions, including peat decomposition and plant root respiration, were 1540 g CO_2-C m^{-2} yr^{-1} and sago plantation soil emissions 1110 g CO_2-C m^{-2} yr^{-1}, which were more than twice as high as those from bare peat at the Central Kalimantan site but similar to undrained forest soil CO_2 losses.

Cumulative annual CH_4 fluxes were highest in the drainage affected forest and recovering forest sites (Figure 6.10, Table 6.4), in which both periodical anoxic wet conditions in peat and easily decomposable litter were available. At these two sites, peat water table was within 20 cm of the peat surface for similar periods of about 230 days, but higher annual CH_4 emission occurred in the recovering forest because of higher fluxes in these water table conditions. Deep water table conditions at these two sites did not cause notable CH_4 influx to the peat (Figure 6.10). At the uncultivated agricultural site the peat was practically CH_4 neutral on an annual basis (Table 6.4) with cumulative emissions similar to CH_4-C fluxes from grassland (0.073 g m^{-2} yr^{-1}) and vegetable fields (0.046 – 0.192 g m^{-2} yr^{-1}) in the vicinity (Hatano *et al*., 2004). According to Melling *et al*. (2005b), annual peat CH_4 emissions from oil palm and sago plantations in Sarawak are -0.015 g CH_4-C m^{-2} yr^{-1} and 0.18 g CH_4-C m^{-2} yr^{-1}, respectively.

Drainage leads to permanently drier conditions in peat and, consequently, CH_4 emissions decrease. Studies carried out so far have concentrated, however, on peat surface methane fluxes and thus do not take into account other potentially important gas sources in drained tropical peat, for example, the large drains and the drainage water they contain can become anoxic and may constitute a CH_4 source because of the solid and soluble organic residues they contain.

Carbon release can also take place via waterways (streams, rivers and drainage channels) in the form of dissolved organic carbon (DOC), particulate organic carbon (POC), dissolved inorganic carbon (DIC) and dissolved CO_2. Studies of these potential carbon release pathways from tropical peatlands are very limited but a recent one by Baum *et al*. (2007) suggests that Indonesian rivers, particularly those draining peatland areas, transfer large amounts of DOC into the oceans, with an estimated total DOC export of 21 Gt yr^{-1}, representing approximately 10% of the global riverine DOC oceanic input. The character and magnitude of fluvial carbon release from tropical peatlands are the subjects of current, detailed investigations. They are likely to be influenced by a range of biotic and abiotic processes, including land use change. Research on temperate peatlands (Worrall & Burt, 2004), for example, indicates that increases in DOC concentration and flux are associated with major droughts and decreases in the peatland water table, which has implications for carbon release from tropical peatlands under different land management and climate change conditions.

Net peatland C flux is determined largely by the net balance between CO_2 uptake in photosynthesis and C release by ecosystem (autotrophic and heterotrophic) respiration (Table 6.4). Peat carbon gas flux measurements are important in order to provide information on peat C dynamics, but cannot be obtained yet for

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all major sinks and sources in a forested peatland ecosystem. For example, the lack of accurate data on the amount of CO_2 sequestered by green plants in photosynthesis is a major problem for which suitable methods have yet to be developed. The complex structure of tropical rain forest canopies adds to the magnitude of this problem. The large amount of CO_2 emitted from peat swamp forest floor is likely to be mostly or completely reabsorbed by the vegetation it supports making it CO_2 neutral whilst, if it is accumulating peat, the ecosystem must be CO_2 negative. On the other hand, in degraded and drained peat swamp forest, although it appears to be releasing similar large amounts of CO_2 as undrained forest, its greatly reduced canopy will not be absorbing as much CO_2 and will therefore be a net emitter of this greenhouse gas. The same applies to the recovering forest except, in this case, an even larger proportion of the CO_2 released will enter the atmosphere because it is virtually devoid of trees and the low growing vegetation of ferns and scrub absorbs relatively little CO_2. Virtually all of the smaller amount of CO_2 emitted from the agricultural land will be transferred to the atmosphere because the biomass is removed, one or more times a year in the case of arable crops and after the life cycle time of 8-25 years in the case of plantation crops (e.g. pulp trees and oil palm), and any CO_2 fixed in crop photosynthesis will also be released eventually as products are consumed or used and eventually decompose.

Surface peat CO_2 emissions contribute considerably to total ecosystem respiration, and are influenced greatly by water table depth. Stability of hydrology, forest floor microtopography and vegetation structure are other factors influencing peat CO_2 dynamics in undrained peatland. In contrast, CO_2 dynamics in drained peatland are determined by time from initial drainage, vegetation type, and drainage depth. The role of CH_4 in the tropical peat carbon balance is relatively small on the basis of currently available results, but potentially important CH_4 sources have been identified both in undrained and drained peatland types (Table 6.4), for example, emissions from vegetation in the former and drainage channels in the latter. Carbon losses from fire and groundwater flow were not addressed, although these may be important especially on drained peatlands. Ecosystem-level measurement of gaseous carbon fluxes are needed in order to determine the true overall C gas balances on undrained, degraded and developed tropical peatland.

6.4. Impact of CO_2 emissions from drained peatlands in Southeast Asia on climate change processes

Scientists have understood the link between peatland development and CO_2 emissions for some time (Figure 6.11), but policy makers and peatland managers are still hardly aware of the global implications of local and national peatland management strategies and actions. As a result, CO_2 emissions from SE Asia's drained and burning peatlands are slow to be recognized in the climate change debate, and the decisive international action required to help these countries to manage their peatlands better has yet to start. According to Page *et al.* (2002) the total carbon store in Indonesian peatland is 26-50 Gt, a conservative estimate that could rise considerably when peat thickness variation is known with greater accuracy (Hooijer *et al.*, 2006). A peat carbon content of 50 kg C m^{-3} is considered to be representative for SE Asian peatlands in general and combining this value with peatland area and thickness (Table 6.3), suggests that carbon storage in SE Asian peatlands is in the order of 58 Gt.

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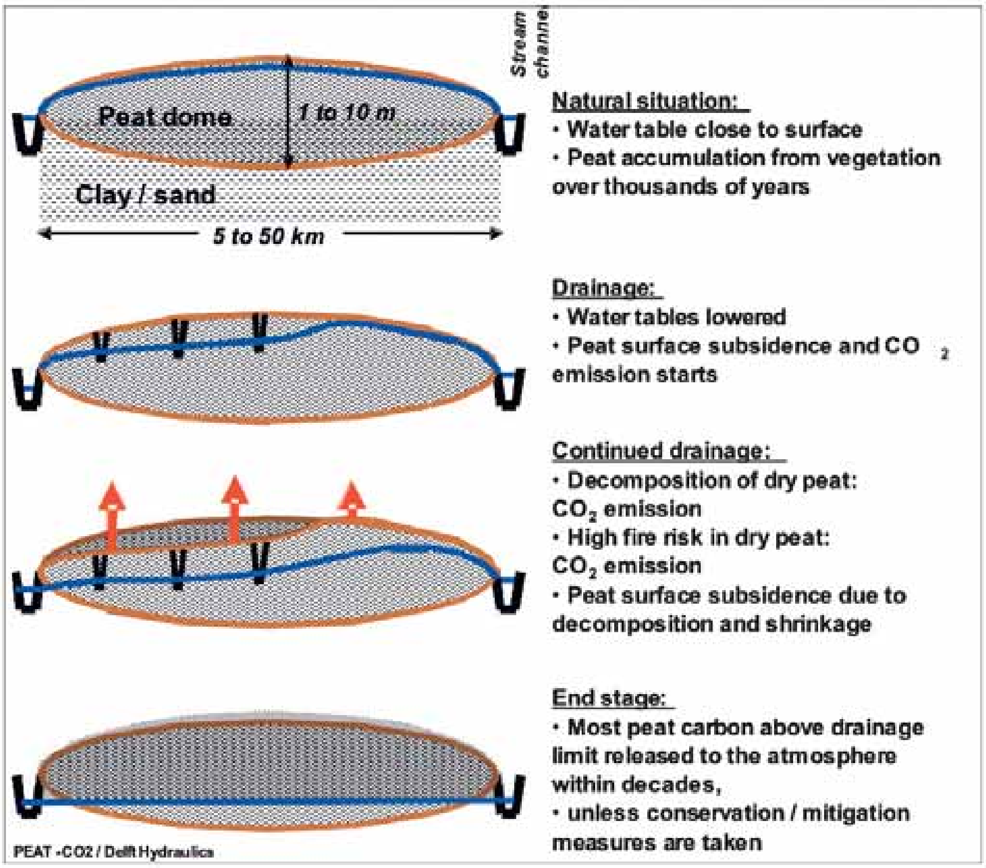

Figure 6.11. *Schematic illustration of CO_2 emission from drained peatland (from PEAT-CO2 Report, Hooijer et al., 2006). The PEAT-CO2 assessment is based on a peatland area of 27.1 Mha, 22.5 Mha of which are in Indonesia, 2 Mha in Malaysia and 2.6 Mha in Papua New Guinea; peat thickness ranged from 0.5 to over 12 metres. In 2000, 61% of these peatlands were covered in forest (JRC, 2003). Only peatlands located less than 300 m above sea level (lowland) were included in the assessment. About 3% of peatland in SE Asia occurs in the highlands of Irian Jaya and PNG but these are shallow and represent less than 1% of the peat carbon store.*

6.4.1. Impact of land use change on tropical peatland sink and store capacities

In their natural condition most tropical peat swamp forests function as C sinks and stores, but forest clearance and drainage can convert them rapidly to C sources. This happens because destruction of the forest cover leads to a decrease in the amount of C allocated into the ecosystem. Agricultural practices require tilling, fertilization, and lowered water tables within the peat, all of which increase surface peat oxidation, leading to subsidence (Figure 6.11).

In the late 1980s, 3.7 million hectares of Indonesian peat swamp forest were developed (Silvius *et al*., 1987), leading to an 18% decrease in peat swamp forest area with a consequent reduction in the C-fixation capacity of 5-9 Mt yr^{-1} (Sorensen, 1993). Deforestation, drainage

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and conversion of peatland in Indonesia and Malaysia continued throughout the 1990s and are still occurring. The Mega Rice Project (MRP) in Central Kalimantan was the largest of these peatland development schemes (Muhamad & Rieley, 2002). It was linked to the Indonesian Government's transmigration programme and led to the clearance of lowland peatland forest in order to contribute to the national food supply and accommodate part of the country's rapidly growing population. The MRP commenced in January 1996 and lasted until 1999 when it was abandoned as a failure. By this time around one million hectares of wetland, mostly peatland, had been cleared, more than 4,500 km of drainage and irrigation canals constructed and 60,000 transmigrants settled in one part of it (Rieley & Page, 2005). Almost half a million hectares of this degraded peatland burned in 1997 with the loss of 150-180 Mt C and, since then it has leaked carbon to the atmosphere constantly through peat subsidence and fire. In addition, the carbon sequestration function of this vast peatland has been destroyed.

The basic relationship between peat subsidence and CO_2 emission is that every centimetre of peat subsidence results in a CO_2 emission of approximately 13 t ha^{-1} yr^{-1} (Wösten *et al*., 1997). This value can be combined with information on long-term average relationships between peat subsidence and water table depths for different regions of the world in order to obtain estimates of CO_2 emissions under different environmental conditions (Figure 6.12). An increase in soil temperature and the absence of winter-summer periodicity explain the increase in subsidence rates at all groundwater levels from temperate through to tropical areas in the world. At the same time, lowering of the water table causes a dramatic increase in the release of CO_2 for all peatland locations. In SE Asian peatlands, average water table depth requirements in drained areas are typically between 95 cm in large intensively drained croplands (including oil palm and pulp wood plantations) and 33 cm in smallholder farms (Hooijer *et al.* 2006).

Role of fire in losses of carbon from tropical peatland

Owing to poor management, fires ignited on peatland used for agriculture can spread to adjacent logged forests, destroying the vegetation (Uhl & Kauffman, 1990; Verissimo *et al*., 1995), igniting the peat and increasing the vulnerability of the landscape to subsequent fires (Nepstad *et al*., 1995; Page *et al*., 2002). Emissions from peat fires can release 5,000 – 10,000 g C m^{-2} (Page *et al*., 2000; Siegert *et al*., 2001) but may be as much as 30,000 g C m^{-2} (Page *et al*., 2002). Based on estimates of the total area of fire-affected peatlands in Indonesia in 1997/1998, between 0.8 and 2.6 Gt C were released to the atmosphere as a result of burning peat and vegetation in the 1997 El Niño Southern Oscillation (ENSO) (Page *et al*., 2002; Langenfelds *et al*., 2002). The 1994/1995 ENSO-related fires produced 0.6 – 3.5 Gt C (Langenfelds *et al*., 2002) while a further 0.25 – 0.5 Gt C and 0.66 – 0.77 Gt were released during fires in 2002 (Bechteler & Siegert, 2004) and 2006 (Langner & Siegert, 2007), respectively (Figure 6.13). If CO_2 emissions from drained peatland are included this makes this region, especially Indonesia which is responsible for more than 90% of these emissions, one of the largest contributors to global CO_2 emissions (Hooijer *et al.*, 2006).

Severe and long-lasting regional smoke-haze and fire episodes during extreme droughts associated with ENSO events have occurred repeatedly in Southeast Asia over the last two decades (Maltby, 1986; Page *et al*., 2000). The danger of an increased frequency of major fires in years with prolonged dry seasons, and future climate scenarios, suggest there will be an increase in the number of days with high

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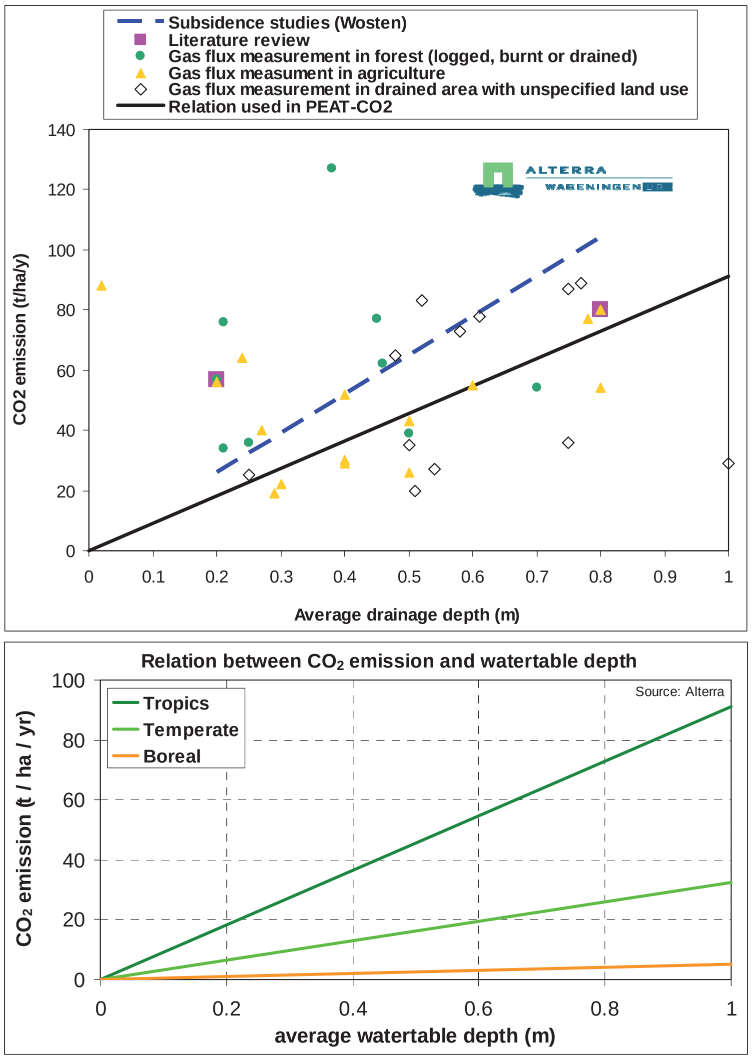

Figure 6.12. *Relationships between drainage depth and CO_2 emission from decomposition (fires excluded) in tropical peatlands (Hooijer et al., 2006). Note that the average water table depth in an undrained peatland is near the peat surface (by definition, as vegetative matter only accumulates to form peat under waterlogged conditions). Top: The relationship for tropical areas, including SE Asia, is based both on long-term subsidence studies and shorter-term gas flux emission studies applying the 'closed chamber method' (Jauhiainen et al., 2005). Results of different methods were combined to derive a linear relationship. This needs to be investigated further since there is evidence to suggest it is probably curved. In reality CO_2 emissions are known to be suppressed at drainage depths up to 0.2 m-0.3 m. Also, CO_2 emissions for a given drainage depth change over time. However, use of a constant and linear relation is deemed acceptable for long-term assessments and for drainage depths between 0.25 m and 1.1 m as applied in this study. Bottom: Tropical drained peatlands have far higher CO_2 emissions than temperate and boreal drained peatlands at the same drainage depth, because of higher decomposition rates in permanently hot and humid climates. Moreover, peatlands in SE Asia are generally drained to much greater depths than is common in temperate and boreal peatlands.*

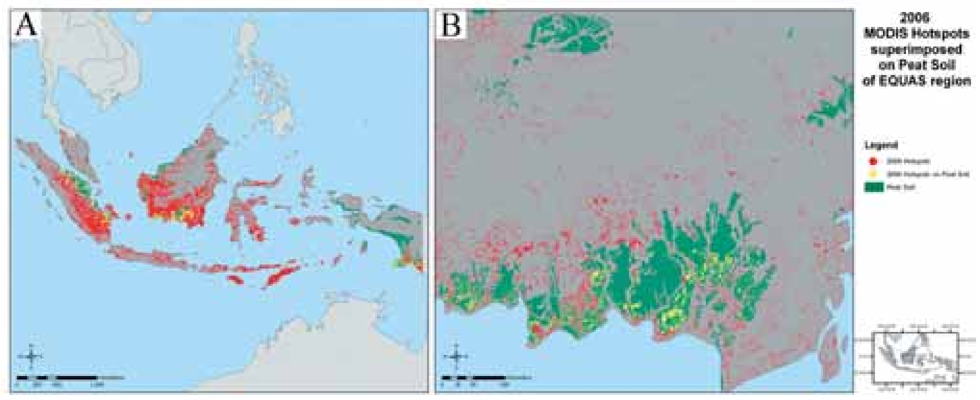

Figure 6.13. *(A) Fires recorded in 2006 superimposed on EQUAS region (Indonesia, Malaysia, Brunei). Peat soils are displayed in green color. Fires on peat soil are shown in yellow color while fires on non peat soil are displayed in red. (B) Subset area of extensive peat areas in Central Kalimantan (Borneo).*

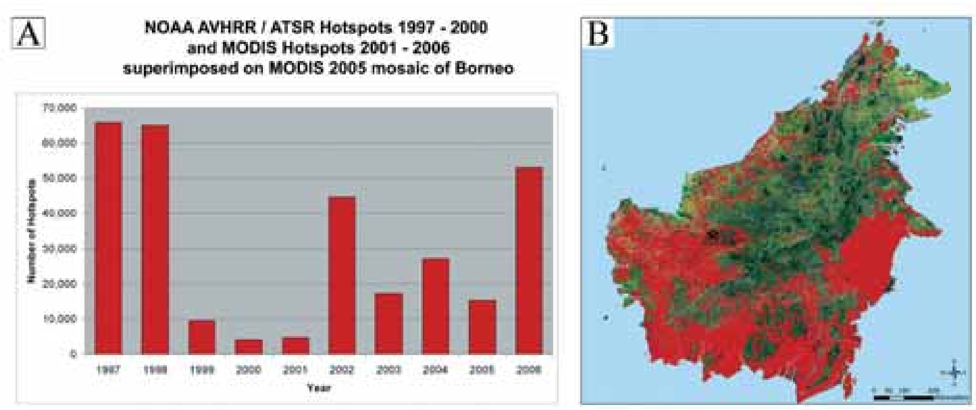

Figure 6.14. *(A) Total number of NOAA AVHRR / ATSR hotspot data from 1997 – 2000 per year and MODIS hotspot data from 2001 – 2006 per year. (B) Total number of hotspots recorded between 1997 and 2006 superimposed on multitemporal composite of Borneo 60 MODIS Surface Reflectance images recorded in 2005.*

risk of fire in tropical areas (Goldammer & Price, 1998; Stocks *et al*., 1998). Some studies suggest an increase in ENSO related tropical storm intensities with CO_2-induced warming in the Asia Pacific region in the future (Royer *et al*., 1998). This is likely to increase the abundance and size of gaps in fragmented forest canopies and areas with long, convoluted boundaries or degraded forest structure resulting from previous fires and logging activities. Storm damage and logging both reduce leaf canopy coverage, thus allowing sunlight to dry out the organic debris on the ground, leading to increased amounts of flammable undergrowth and frequency and intensity of fires with consequent release of large amounts of stored carbon from biomass and peat (Verissimo *et al*., 1995; Holdsworth & Uhl, 1997; Siegert *et al*., 2001). Compared to the severe El Niño of 1997/1998, the 2002 and 2006 ENSOs

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were weaker but they still led to severe fire events on the island of Borneo, especially in Central Kalimantan (Figures 6.13 & 6.14). Seventy three per cent of the forest area affected by fire in 2002 occurred in peat swamp forests although their total area is just 23% of the total forest cover. During 2006, fires destroyed 0.76 Mha of peat swamp forests corresponding to 75% of all forest burnt in that year. By focusing on the actual number of fires, the situation becomes even more evident. In 2002, more than ten thousand hotspots occurred in peat swamp forest, corresponding to 81% of all forest fires; in 2006 the number increased to nearly fifteen thousand, which equates to 82% of all forest fires (Langner & Siegert, personal communications). These results show that peat swamp forests are much more prone to fire than any other forest type, probably owing to land clearing activities for the establishment of oil palm and pulp wood plantations. Fires on peatland release much larger amounts of CO_2 into the atmosphere than fires in forests on mineral soils because both the surface vegetation and the underlying peat layer can burn. This is of global importance because the extensive undisturbed peat swamp forests of Southeast Asia are a major C store and sequester of CO_2.

6.4.2. Current and projected CO_2 emissions from drained peat in Indonesia

Present and future emissions from drained peatlands in Indonesia have been quantified recently using the latest data on peat extent

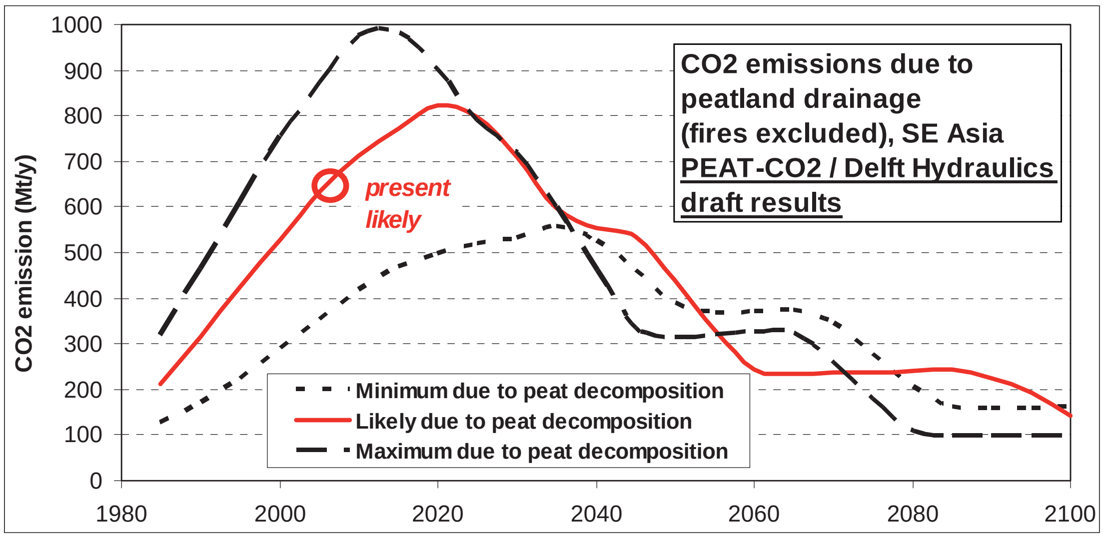

Figure 6.15. *Historical, current and projected CO_2 emissions from peatlands, as a result of drainage (fires excluded). The increase in emissions is caused by progressive deforestation and drainage of peatlands. The decrease after 2020 ('likely' scenario) is caused by shallow peat deposits being depleted, which represent the largest peat extent. The stepwise pattern of this decrease is explained by the discrete peat depth data available (0.25m, 0.75m, 1.5m, 3m, 6m, and 10m). Peat depth data are only available for Indonesia; for other areas conservative estimates were made: 3m for Malaysia and Brunei (similar to Kalimantan) and 1.5m for Papua New Guinea (similar to Irian Jaya). Note that peat extent and thickness data for 1990 (Sumatra) and 2000 (Kalimantan) have been assumed at the starting year of the analysis, in 1985. Considering the uncertainty margin around these data, and the likely systematic underestimation of peat depths, this does not introduce a large additional error in the analysis.*

and depth, present and projected land uses and water management practices, decomposition rates and fire emissions (PEAT-CO_2 Project: Hooijer *et al.*, 2006). Current CO_2 emissions (2005) caused by decomposition of drained peatlands are estimated to be 632 Mt yr^{-1} (range 355 – 874 Mt yr^{-1}), which will increase in coming decades, and will continue well beyond the 21st century, unless land management practices and peatland development plans are changed. In addition, between 1997 and 2006 an estimated average of 1400 Mt yr^{-1} of CO_2 emission was caused by fires associated with peatland drainage and degradation. The total current CO_2 emissions from tropical peatland of approximately 2000 Mt yr^{-1} equals almost 8% of global emissions from fossil fuel burning. Emissions are likely to increase every year for the first decades after 2000 (Figures 6.15 & 6.16). As shallow peat deposits become depleted, however, and the drained peatland area therefore diminishes, peat subsidence emissions are predicted to peak sometime between 2015 and 2035 at between 557 and 981 Mt yr^{-1} (likely value 823 Mt yr^{-1}), and will decline steadily thereafter. As the deeper peat deposits will take much longer to be depleted, significant CO_2 emission will continue beyond 2100.

6.5. Conclusions and recommendations

The total amount of C in peatlands in Southeast Asia is at least 58 Gt (depending on assumptions of peat thickness and C content), equaling at least 212 Gt of potential CO_2 emissions. Current CO_2 emission rates (fires excluded) from drained tropical peatlands are estimated at between 355 and 874 Mt yr^{-1}, with a most likely value of 632 Mt yr^{-1}. If current rates of tropical peatland conversion and practices of peatland development and degradation continue, this will increase to 823 Mt yr^{-1} in 10 to 30 years time, followed by a steady decline over ensuing centuries as the thicker peat deposits become depleted.

Current CO_2 emissions from Indonesia alone are 516 Mt yr^{-1}, which is equivalent to: 82% of peatland emissions in SE Asia (fires excluded); 58% of estimated global

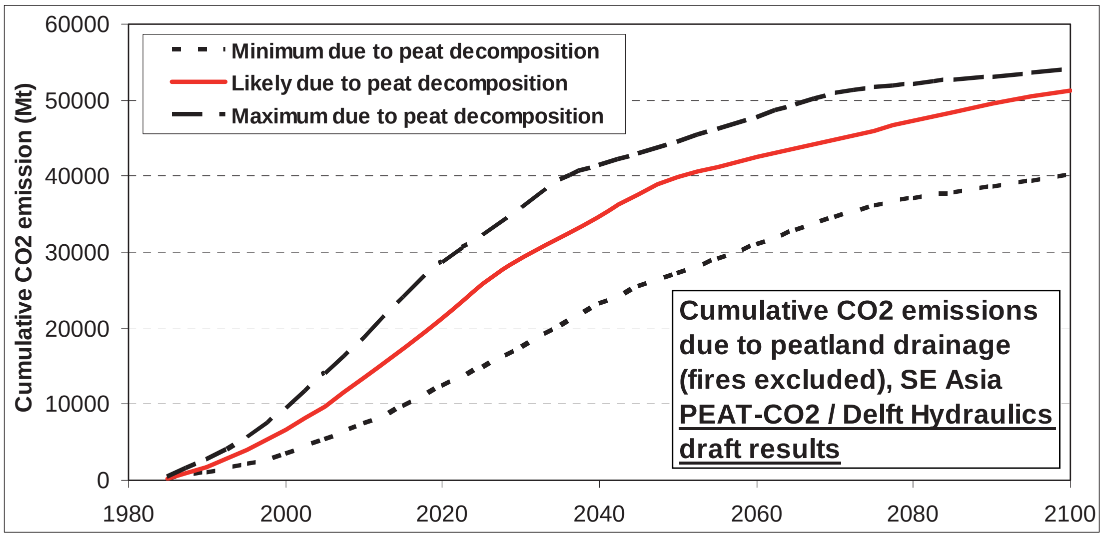

Figure 6.16. *Cumulative CO_2 emissions from SE Asia over 100 years. Note that total storage is at least 206 Gt CO_2 (56 Gt C).*

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peatland emissions, fires excluded; and almost two times the emissions from fossil fuel burning in Indonesia.

Recent developments give little cause for optimism because, while deforestation rates on non-peatlands in SE Asia have decreased somewhat (at least in part owing to depletion of forest resources), those on peatlands have been stable (on average) for up to 20 years. The current (2000-2005) average deforestation rate is 1.5% yr^{-1}. In 2005, 25% of all deforestation in SE Asia was on peatlands. Apart from logging for wood production, an important driver behind peatland deforestation is development of palm oil and timber plantations, which require intensive drainage and cause the highest CO_2 emissions of all land uses. A large proportion (27%) of palm oil concessions (i.e. existing and planned plantations) in Indonesia is on peatlands; a similar percentage is expected to apply in Malaysia. These plantations are expanding at a rapid rate, driven in part by the increasing demand in developed countries for palm oil as a biofuel. Production of 1 tonne of palm oil causes a CO_2 emission of between 10 and 30 tonnes through peat oxidation (assuming production of 3 to 6 tonnes of palm oil per hectare, under fully drained conditions, and excluding fire emissions). The demand for biofuel, aiming to reduce global CO_2 emissions, may actually be causing an increase.

It is essential that future land use of tropical peatland incorporates the principles and practices of sustainable 'wise use', especially with respect to hydrology, water and carbon management. Unfortunately, governments of developing countries in the tropics have higher priorities than maintenance of the natural resource functions of peatlands (Rieley & Page, 2005), despite their proven important role for the global C cycle. This is particularly pertinent given the latest climate change predictions for the Southeast Asian region. The IPCC Fourth Assessment Report predicts that the region will experience a median warming of 2.5°C by the end of the 21st century, accompanied by a predicted mean precipitation increase of about 7% (IPCC, 2007). The strongest and most consistent precipitation increases are forecast to occur over northern Indonesia and Indochina in June, July and August, and over southern Indonesia and Papua New Guinea in December, January and February. The pattern is essentially one of wet season rainfall increase and dry season decrease, with areas experiencing a mean rainfall decrease also likely to experience increases in drought and fire risk. Models indicate that these changes will be strongest and most consistent across south Sumatra and southern Borneo, where there are extensive peatlands. The effect of higher temperatures, decreased dry season rainfall and longer dry periods will be to lower peat water tables more frequently and for longer periods of time, thus exposing remaining peat C stocks to enhanced aerobic decomposition and increased risk of major fires (Hooijer *et al*., 2006).

Inevitably, every type of human intervention on peatland leads to impairment or even loss of natural resource functions (ecology, hydrology, biodiversity, carbon storage). The challenge facing those involved in the management of tropical peatlands in the 21st century is to develop integrated planning and management mechanisms that can balance the conflicting demands on the tropical peatland heritage and its environmental feedback mechanisms to ensure its continued survival to meet the future needs of humankind. This strategy of wise use of tropical peatlands involves evaluation of their functions and uses, impacts caused by and constraints to development so that, by assessment, reasoning and consensus, it should be possible to highlight priorities for their management and use, including

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mitigation of past and future damage. There are, however, still some critical gaps in our understanding of the carbon dynamics of tropical peatlands. For example, ecosystem-level C budgets under differing hydrology, land management and disturbance regimes require further investigation and elaboration; likewise, we have limited knowledge of how climate change will impact upon the C dynamics of tropical peatlands. Further scientific enquiry will add to our understanding of the role that tropical peatlands have played and continue to play in global environmental change processes, but effective peatland management will also require improved engagement between scientists, policy makers and stakeholder groups, whether they be land development companies or smallholder farmers. New opportunities for protection of the tropical peat carbon store may arise from current negotiations on financial payments for reduced emissions from avoided deforestation and forest degradation (REDD). This could put an economic value on the remaining tropical peat swamp forests and their globally important C stores, and provide an incentive for their protection. It is to be hoped that these initiatives will herald a new era for the fragile tropical peatland ecosystems of Southeast Asia.

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CHAPTER 7:

RESTORATION OF PEATLANDS AND GREENHOUSE GAS BALANCES

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7.1. Introduction

As recently as the 1970s and 1980s in Europe and North America the use and drainage of peatlands was the primary objective of peatland management. However, over the last two decades the restoration of peatlands has gained importance. The role of peatlands in global biodiversity and their importance for endangered species of fauna and flora has been more and more realized. Wetlands, including peatlands, have been rewetted as habitats, especially for migratory and breeding birds following the European Natura 2000 convention.

An accurate estimate of the area of restored peatlands in Europe and Northern America is not available. Nevertheless, some facts may underline the growing importance of peatland restoration in the upcoming decennia. In Lower Saxony, one of the Northern states of Germany where two thirds of German bogs are situated, a bog protection programme was started in 1985. To date, about 20 % of the 2500 km² initial bog area has been included into nature protection areas. Special emphasis was placed on the after-use of peat excavated areas. Currently, 42 % of former cutaway areas are nature conservation areas. With the expiration of extraction rights, a further 23 500 ha (corresponding to 82 % of the current peat extraction area) will be allowed to regenerate (NLWKN, 2006). In Sweden, about 500 to 1000 ha, i.e. 5 % of the cutover area, are presently restored and a prognosis of 5000 ha or 30 % of the peat cut area is given for 2010 (Vasander *et al.*, 2003). For Ireland it is estimated that around 30 000 ha could be available

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for rewetting or restoration over the next 20-30 years. These will also come out of industrial peat production. In Canada 1 800 ha of excavated peatland are currently in process of restoration or reclamation and 3 100 ha are projected for restoration or reclamation until 2011 (Canadian Sphagnum Peat Moss Association, CSPMA, unpublished). The restoration of fens is still less widespread, as fens are frequently very fertile agricultural land. Nevertheless, about 10 000 ha of grassland fens have been rewetted in Northeast Germany (LUNG, 2006) and 2 500 ha in North West Germany (NLWKN, 2007). Similarly, in the Netherlands and Colorado efforts have been made to restore fens (Cooper & MacDonald 1998; 2000).

Restoration of a disturbed peatland is the attempt to establish, as far as possible, the conditions for peat growth. However, disturbance may often lead to irreversible changes in peat structure, peat composition, peat (land) hydrology and peatland position in the landscape (Schouwenaars, 1993). This makes it difficult to re-establish the conditions that are essential for the formation and growth of the peatland. Additionally, factors such as climate change and eutrophication could impede or reduce peat formation at the local or global level. Thus, in contrast to past conditions, it is not always certain that the conditions favourable for peat growth will lead to sustainable peat formation in the forthcoming centuries or even millennia. Moreover for the assessment of the net climate effect of peatland restoration, peat growth alone is not sufficient as an indicator, as the exchange of climatically relevant trace gases rules the climate effect.

A systematic overview on the exchange of greenhouse gases in restored peatlands is not available as direct measurements of gas emissions are scarce. Additionally, the starting conditions are very heterogeneous from one site to the other, as has been shown for former peat cutting areas (Blankenburg & Tonnis, 2004). Thus, in this chapter, several examples will be given, covering the most important regions (temperate and boreal) and peat types (fens and bogs) of non-tropical peatlands.

In this chapter the impact of peatland restoration on greenhouse gas fluxes will be discussed based on a literature review. As the starting conditions for restoration experiments and restoration techniques are very diverse, not all combinations of the involved factors can be taken into account. Moreover, direct measurements of greenhouse gas emissions are very laborious and expensive, and, therefore, only few examples on greenhouse gas emission measurements from restored peatlands exist. In the following, case-studies will be presented covering different peatland types, different regions and different starting conditions.

7.2. Restoration techniques and factors involved in greenhouse gas emissions

The success of peatland restoration and its impact on greenhouse gases fluxes is strongly dependent on the time elapsed since the end of peatland use or excavation, the starting conditions, and on the restoration techniques. Gorham & Rochefort (2003) suggest that peatland restoration should occur immediately after extraction to reduce degradation at the peat surface, preventing irreversible hydrological changes and potentially returning the carbon sink functions of the ecosystems. In regard to the starting conditions, two general types are distinguished here (1) the restoration of peat cutting areas (section 7.3) and (2) the restoration of peatlands formerly used for agriculture or forestry (section 7.4).

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Concerning restoration following peat extraction, Blankenburg & Tonnis (2004) defined a number of starting conditions based on residual peat depth and peat type, landscape situation and cutting method. Concerning the cutting method, a different restoration success was observed for block-cut and milled peat production or vacuum extracted peatlands: Block-cut cutover peatlands can regenerate to an extent due to *Sphagnum* diaspores that remain combined with residual micro topography of the baulks and trenches which aid in sustaining adequate soil moisture and soil-water tension for *Sphagnum* reestablishment. However, only 17.5% of abandoned blocked-cut bog trenches in Quebec had *Sphagnum* covers greater than 50% (Poulin *et al*., 2005). The potential for natural regeneration of vacuum extracted cutover peatlands is much lower because of greater degradation of hydrological conditions from mechanized extraction and complete removal of plant material. Ferland & Rochefort (1997) observed that the vacuumed peat surfaces dry out quickly even if the water table is close to the surface hindering the ability of *Sphagnum* to re-establish on the peat surface.

Self-regeneration of peatlands can occur but often leads to limited recovery of pre-extraction functions. Recent North American studies even suggest that non-restored cutover peatlands represent a persistent source of atmospheric CO_2 (e.g. Waddington & Price, 2000). Therefore, human activities are frequently involved in order to accelerate the regeneration process. The most commonly used restoration techniques include reduction of land-use intensity or ending of land-use, artificial topography, rewetting by blocking drainage ditches or by flooding from passive or pumped seepage reservoirs, introduction of peat forming plants (e.g. *Sphagnum* species) or companion species, and straw mulch application. Concerning water supply, Schouwenaars (1988) suggested that effective *Sphagnum* re-establishment in cutover peatlands should occur where the water table does not drop more than 40 cm below surface. *Sphagnum* growth in a cutover peatland is limited not only by water availability but also by drying and wetting cycles (McNeil & Waddington, 2003). These moisture cycles can suppress photosynthesis for a prolonged period of time and enhance respiration losses. The evaporative water loss over summer can especially impact carbon dynamics within a peatland (Waddington & Price, 2000). Moreover, the degree of degradation of the peat hydraulic properties (i.e. hydraulic conductivity and soil-water tension relationships) under past drainage and land-use measures determines the swelling and the porosity of the peat after rewetting, and, thus, the moisture conditions for *Sphagnum* colonization. Companion species might be important as it has been observed that the growth of *Sphagnum* is strongly dependent on the presence of vascular plants (i.e. ericaceous shrubs; McNeil & Waddington, 2003), or with *Polytrichum strictum* at the early stages of restoration (Groeneveld *et al*., 2007).

Of course, the starting conditions (i.e. degree of peat degradation, water supply, evaporative water loss, presence of companion species) and the above mentioned restoration techniques have an impact on the emission of greenhouse gases. Firstly, they affect the productivity of the peat forming plants and, thus, the process of peat accumulation which represents a continuing sink for carbon dioxide (CO_2). Secondly, rewetting inhibits methane (CH_4) oxidation at the peat surface and CH_4, which is formed in the anoxic zones, is emitted directly to the atmosphere. This depends strongly on the water table, as a mean water table below 10 cm seems to be sufficient in preventing accelerated CH_4 emissions. Nevertheless, CH_4 emissions can remain high from former drainage ditches (Waddington & Price 2000). As bogs are

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generally nutrient poor and receive no fertilizer additions, nitrous oxide (N_2O) emissions are expected to be negligible. Nevertheless, nitrogen input from deposition has increased in the last decades in regions where peatlands are distributed in Europe (e.g. Northwest Germany, The Netherlands), which can lead to higher N_2O emissions, nowadays.

Different starting conditions have also to be considered regarding the restoration of peatlands formerly used for agriculture or forestry, e.g. type of land-use, degree and duration of drainage, peat type, depth and peat degradation, fertilization, liming, and forest productivity (here especially drainage by tree transpiration). For restoration of peatlands used for forestry two general pre-rewetting situations can be distinguished in Northern Europe. In the first, forest is only slightly drained with small hydrological effects. The land often forms imperfectly drained soils with great similarities to natural peatlands. Here restoration, e.g. rewetting, will cause only small changes. The nutrient cycling processes, including greenhouse gas fluxes, will fairly soon be similar to a natural peatland, low sedge fen or bog type (Kasimir-Klemedtsson *et al*., 1997). In the other case, forest drainage is more efficient and medium to high productive forests have developed. The decomposition of the peat after drainage is accelerated but, at the same time, a new humus layer of ca. 10 cm develops at the soil surface. Restoration, i.e. forest harvesting and rewetting, will raise the water table close to the surface. The peat decomposition will be reduced but at the same time the fairly easily decomposable humus layer and fine roots in the peat will be oxidized. Thus, an extra release of carbon will be observed during an initial restoration phase of 2-10 years before new wetland plants colonize. Finally, a medium to fairly species rich wetland will develop, quite different from the original peatland eco-system (Vasander *et al*., 2003).

Frequently, the structural, hydrological and chemical properties of peat layers remaining after agricultural and forestry use have been changed much more than those of peat layers remaining after peat extraction (Andersen *et al*., 2006; see also Chapter 3).

7.3. Greenhouse gas fluxes from cutover peatlands under restoration

7.3.1. Boreal peatlands in North America

Peat extraction in North America

Within North America approximately two-thirds of the peat extraction for horticultural purposes occurs within Canada and the demand for horticultural peat in Canada over the past century has led to the drainage and extraction of over 12 000 ha of peatlands (Cleary *et al*., 2005) of Canada's estimated 171 million ha of peatlands (Gorham, 1991). Drainage and extraction of peatlands create conditions that disturb the natural hydrological and carbon cycling regimes of the ecosystem. Block-cutting and vacuum extraction techniques have been used primarily for the peat horticulture industry in Canada. Early peat horticulture was typically performed by block-cutting where drainage of the peatland occurred with a series of ditches and subsequent extraction trenches. The acrotelm was removed and discarded over the shoulder exposing the catotelmic peat used for horticulture. The extraction of this deeper peat was accomplished by hand-cutting into blocks on average to 60 cm depth (Girard *et al*., 2002) consequently leaving the landscape in an arrangement of alternating baulks (raised mounds) and trenches. The remaining acrotelm material discarded in the centre of the extracted trench was a form of unconscious plant reintroduction. Rarely used commercially

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today, remnants of these systems still remain in eastern North America. By the mid-1970s mechanized cutting became the dominant peat extraction practice for commercial use. Occurring at a larger areal scale than block cutting, deeper and more frequent drainage ditches are used in order to facilitate adequate drainage to support heavy extracting machinery. Similar to block-cutting, the acrotelm is removed; however, rather than discarded adjacent to the extraction site, the stripping spoil or skag (i.e. vegetation layer) is discarded completely. The peat surface is then milled to facilitate drying and peat fragments are typically vacuumed from the surface (i.e. vacuum extraction) using heavy tractors to depth of ~75 to 100 cm. Consequently, the peat extraction process creates unfavourable conditions at the peat surface especially for species such as *Sphagnum* moss, the main peat forming vegetation, to re-establish.

Greenhouse gas emissions from restored cutover peatlands

While studies have shown that decreased CO_2 efflux can occur due to rewetting of the surface, respiration can remain quite high post-restoration (Waddington & Warner, 2001). Waddington *et al*. (2003) also observed that the addition of mulch to the surface can represent a short-term source of atmospheric CO_2 due to its decomposition over time. New mulch decomposition accounted for between 17 and 30 % of total respiration. Similarly, Petrone & Waddington (2001) determined that a restored peatland was a larger source of CO_2 than an adjacent cutover site due to mulch decomposition exceeding the production of the newly emergent mosses and vascular vegetation. Mulch addition, blocking of ditches with old vegetation, and new emergent vegetation with plant reintroduction will also likely contribute to an increase of DOC leaching. In a restored peatland, DOC concentrations increased in the outflow, which is likely attributed to the wetter conditions post restoration (Waddington *et al*., 2007). However, while CO_2 fluxes may have increased post restoration (1753 g CO_2 m^{-2} yr^{-1}), DOC export represented only a small portion (0.7%) of the total CO_2 flux (Waddington *et al*., 2007) from the ecosystem. Additionally, restoration of peatlands leads to an increase in CH_4 production where fluxes can be significantly larger than cutover sites. It is suggested that a rise in water table and establishment of vascular vegetation post-restoration increases CH_4 flux due to increased labile carbon sources and enhanced CH_4 transport through the vegetation. An increase in CH_4 was observed from 0.02 g CH_4 m^{-2} yr^{-1} to 1.3 g CH_4 m^{-2} yr^{-1} three years post-restoration (Waddington & Day, 2007) representing over 70 times increase in CH_4 emissions. These findings are consistent with observations of natural peatlands being sources of atmospheric CH_4.

7.3.2. Boreal peatlands in Northern Europe

Peat accumulation at a self-regenerating cutover pit in Sweden

Study site and methods

Three boreal cutover peatlands where peat cutting by hand (peat pits, 20 x 100 m) had ended in the period 1950 - 1975 underwent a self-regeneration process (Box 7.1). The layer of the new formed peat was investigated 25-50 years after the end of peat cutting by sampling of undisturbed peat cores.

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Box 7.1. Regenerating cutover peatland, Sweden

Site:
- Country: SW Sweden
- Location: Björnmossen bog
- Co-ordinates: N 59°05’, E 14°39’

Climate:
- Precipitation: 800 mm yr^{-1}
- Evapotranspiration: 500 mm yr^{-1}
- Runoff: 300 mm yr^{-1}
- Annual mean temperature,: + 6 °C
- Vegetation period (T > +5°C) 205 days

Peat properties:
- Peat type: *Sphagnum* peat
- Climatic region nemo-boreal bog

Former land use and restoration:
- Former land-use: hand-cut peat pits, 20 x 100 m
- Abandoned since: 1950 / 1975
- Management type: self-regeneration, poorly drained
- Investigation: 2000

Study methods:
- Peat accumulation determined using undisturbed peat cores

Reference: Lode, 2001

<u>Results</u>
The rate of peat accumulation varied between the sites with a low rate where 10 cm accumulated during 50 years giving 2 mm yr^{-1}, and a high rate at pits being under short regenerating period with 25 cm in 25 years resulting in a rate of 10 mm yr^{-1}. Using an average bulk density of 0.05 g cm^{-3} the organic matter accumulation could be estimated to 100-500 g m^{-2} y^{-1}. On the assumption of 20-50% of the biomass ending in peat (Ilomets, M., personal communication) the low rate would result in a peat accumulation of 20-50 g m^{-2} yr^{-1} while the high rate would result in 100-250 g m^{-2} yr^{-1}. With a carbon content roughly being 50% this would correspond to sinks in the range of -37 to -460 g CO_2 m^{-2} yr^{-1} (Lode, 2001). Methane and nitrous oxide emissions were not measured at this site.

Spontaneous regeneration of a cutover peatland in Finland
<u>Study site and methods</u>
Both Yli-Petäys *et al*. (2007) and Kivimäki *et al*. (in press) carried out studies in central Finland as part of the European RECIPE project (Boxes 7.2 & 7.3). Peat harvesting in the area started in 1942. The harvesting method used first was block-cutting, which created several 3 to 4 m deep trenches, surrounded by ca. 5 m wide dry baulks of peat. The harvesting was ceased in 1948 at the study site. The drainage system was fairly inefficient and the trenches started to regenerate spontaneously soon after the harvesting had ceased.

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Box 7.2. Old self-regenerated cutover peatland, Finland

Site:	
– Country:	Central Finland
– Location:	Aitoneva, Kihniö
– Co-ordinates:	N 62°12`, E 23°18`
Climate:	
– Precipitation:	700 mm y^{-1}
– Annual mean temperature,:	+ 3.5 °C
– Vegetation period (T > +5°C)	160 days
– Temperature sum (+5°C)	1100 d.d.
Peat properties:	
– Peat type:	*Sphagnum* peat
– Climatic region	southern boreal to middle boreal
Vegetation:	
– *Eriophorum vaginatum,. Carex lasiocarpa, C. rostrata, Sphagnum riparium, S. papillosum, S. pulchrum*	
Former land use and restoration:	
– Former land-use:	peat block cutting since 1942
– Abandoned since:	1948
– Management type:	self-regeneration, poorly drained
– Investigation:	2000-2001
Methods:	
– Gas exchange measured with closed chambers	
– Modelling of net ecosystem CO_2 exchange	
Reference: Yli-Petäys *et al*., 2007	

In the study of Yli-Petäys *et al*. (2007) sites of four regenerating plant communities were chosen for C flux measurements. The water table at the sites differed slightly. Instantaneous CO_2 exchange rates in the plots were determined using the closed chamber method described by Alm *et al*. (1997). Kivimäki *et al*. (in press) compared CO_2 dynamics in different vegetation types over 2 years (Box 7.3). They established altogether 19 sample plots in four types of patches along a moisture gradient at the site. Additionally, they laid out three bare control plots where they removed all vegetation. The seasonal CO_2 exchange was determined with the closed chamber method from June to September.

Results and discussion

In the study of Yli-Petäys *et al*. (2007) four plant communities revegetated the sites (average water table during growing season in cm below ground in parentheses): *Eriophorum vaginatum* and *Sphagnum riparium* ("Ripa": 12), *S. papillosum* ("Papi", 8), *Carex lasiocarpa, S. papillosum* and *S. pulcrum (*"PaPu": 4) or *Sphagnum pulcrum* ("Pulc": 2). All acted as sinks for CO_2 during the two growing seasons. Nevertheless, the CO_2 emission

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Box 7.3. Actively restored cutover peatland, Finland

Site, climate and peat properties: (see Box 7.2)

Former land use and restoration:

– Former land-use:	peat harvesting (Finnish HAKU harvesting method) until 1975 (Frilander et al. 1996)
– Restoration since:	1994
– Management type:	blocking the ditches, additional water supply
– Investigation:	2003-2004

Vegetation:
- monostands *Eriophorum vaginatum*
- monostands *Carex rostrata*
- mixed stands *Eriophorum vaginatum* and *Sphagnum*
- mixed stands *Carex rostrata* and *Sphagnum*
- bare plots

Methods:
- Gas exchange measured with closed chambers
- Modelling of net ecosystem CO_2 exchange

Reference: Kivimäki *et al*., in press

during winter was in most cases as high as the CO_2 uptake during the growing season. Thus, the estimated one-year net CO_2-C balance was low or even clearly negative and amounted to mean values of 35, 31, 53 kg CO_2 m^{-2} yr^{-1} in the "Ripa", "Papi" and "PaPu" communities for the two years of the study. Only in the "Pulc" community, which had the highest water table, a net mean carbon dioxide uptake of -143 kg CO_2 m^{-2} yr^{-1} was observed. Thus, the low seasonal production of most plant communities in this study was not sufficient to exceed losses of winter time respiration, which may partly result from suboptimal weather conditions during the study period.

Likewise, pristine mires may also undergo large interannual variations in CO_2 balance (e.g. Alm *et al*. 1999a,). Unlike in the studies of Bortoluzzi *et al*. (2006) from Central Europe, the advanced regeneration stage in Aitoneva did not represent a strong sink of CO_2. Besides weather conditions it is also possible that the residual peat decomposition in general exceeds the production rate of new biomass of the trenches. The results of Yli-Petäys *et al.* (2007) may suggest decreased carbon sink of restored site after the observed *Eriophorum* peak in productivity (Tuittila *et al*., 1999). Methane emissions were high in treatments with high water table and amounted to 19, 29, 36 and 45 kg CH_4 m^{-2} yr^{-1} in the "Ripa", "Papi", "PaPu" and "Pulc" plots.

Kivimäki *et al*. (in press) studied the carbon dioxide exchange of four vegetations types (mean water table in cm in parentheses, negative values mean water table above ground): monostands of *Eriophorum vaginatum* ("EV": 0) or *Carex rostrata* ("CR": -3) and mixed stands of *Eriophorum* or *Carex* with *Sphagnum* ("EV+S" and "CR+S": -10). A bare control had a mean

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water table of 7 cm below ground. The seasonal net CO_2 exchange resulted in a carbon uptake of the peatland both years in all the vegetated plots. Assuming a CO_2 emission of 161 g CO_2 m^{-2} outside the growing season (Yli-Petäys *et al*., 2007), the annual net ecosystem exchange (NEE) was calculated. Mean values amount to -97 and -63 g CO_2 m^{-2} y^{-1} in the monostands "EV" and "CR", respectively, and to -347 and -293 g CO_2 m^{-2} y^{-1} in the mixed stands "EV+S" and "CR+S", respectively. The interannual variation of the mean water table and the slight moisture gradient had no significant effect on the CO_2 exchange. The plots with *Sphagnum* had higher seasonal NEE than the pure sedge plots resulting from the higher seasonal gross productivity (GEP). It thus seems that increased number of species leads to increased productivity and either vascular plants become more efficient when growing with *Sphagnum* or *Sphagnum* becomes more efficient when growing with vascular plants.

The bare peat plots were seasonal sources of 123 g CO_2 m^{-2} into the atmosphere. Alm *et al*. (1999b) estimated that CO_2 release from peat in winter would represent 21 to 23 % of the annual total CO_2 release from peat. Thus, the bare plots emitted 150 g CO_2 m^{-2} yr^{-1}. These values were much smaller than those measured by Waddington *et al*. (2002) from non-restored cut-away peatland. They had values of 1331 g CO_2 m^{-2} y^{-1} in dry year and 411 g CO_2 m^{-2} yr^{-1} in a rainy year. It seems that the restoration has also decreased the respiration of the non-vegetated areas at Aitoneva. By colonization by *Sphagnum* and vascular plants, the respiration of the bare peat surface decreases and with the growing functional diversity the ecosystem becomes more efficient in carbon accumulation. Because different *Sphagnum* species are favoured by different water levels, their species richness enhances the forming of *Sphagnum* coverage to the whole area. Spreading *Sphagnum* diaspores by the North American method (Quinty & Rochefort, 2003) may further speed up this process.

7.3.3. Temperate peatlands in Central and Western Europe

Rewetting of hand-cut bog peat pits in the Netherlands

The Fochtelooërveen area is a disturbed raised bog in the north of the Netherlands (Nieveen *et al*., 1998; Box 7.4). The vegetation is natural tussock grassland, with an averaged height of approximately 40 cm. A layer of 10 cm of dead organic material from the previous growing seasons covered the tussocks and the hollows in between. The dominating plant species is *Molina caerulea* (>75%) but also species like *Eriophorum vaginatum, Calluna vulgaris* and *Erica tetralix* could be found. Throughout the seasons, the water table varied, depending on the weather, from 0 to 20 cm below the tussock soil interface but the soil remained saturated. CO_2 flux measurements were conducted, using the eddy correlation technique, during June 1994 and October 1995. For this period the site acted as a source of CO_2. Total NEE was estimated at 97 g CO_2 m^{-2} yr^{-1}.

Restoration of a rewetted industrial cutaway peatland in Ireland

Study site

The study was carried out in an Irish rewetted industrial cutaway peatland (Wilson *et al*., 2007; Box 7.5). Turraun was one of the first bogs to undergo industrial peat extraction in Ireland. Prior to harvesting, the average depth of peat at Turraun was 6.2 m (Rowlands, 2001). When milled peat harvesting ceased in the 1970s, the residual peat depth ranged from 0 - 1.8 m. Over the following decades, the cutaway area was allowed to revegetate naturally. In 1991, a 60 ha lake was constructed, the drainage ditches

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Box 7.4. Rewetting of a hand-cut bog, the Netherlands

Site:	
– Country:	NE Netherlands
– Location:	Fochtelooër bog
– Co-ordinates:	N 53°00', W 6°23'
Climate:	
– Precipitation:	853 mm y^{-1}
– Evapotranspiration:	531 mm y^{-1}
– Runoff:	n.a.
– Annual mean temperature,:	+ 9.0 °C
– Vegetation period (T > +5°C)	264 days
Peat properties	
– Peat type:	*Sphagnum* peat
– Climatic region	mid-latitude marine
Former land use and restoration:	
– Former land-use:	hand-cut peat pits
– Restoration since:	1985
– Management type:	damming
– Investigation:	1994/1995
Methods:	
– Eddy correlation technique	

Reference: Nieveen *et al*., 1998

were blocked, a mineral soil/peat bund was formed and the cutaway area was reflooded. Since that time, a wide range of vegetation communities have become established representing both dryland and wetland ecosystems. The dryland communities are dominated by *Betula* and *Salix* spp., *Calluna vulgaris*, *Molinea caerulea*, and *Juncus effusus*. Within the wetlands a hydroseral gradient, i.e. the sequence of vegetation communities which occur during the transition from shallow open water at the edge of the lake to drier terrestrial ecosystems, has developed. These include extensive stands of *Phragmites australis*, *Typha latifolia*, *Phalaris arundinacea*, *Eriophorum angustifolium* and *Carex rostrata* communities.

The residual peat deposit is mainly *Phragmites australis* or fen type peat overlying undulating calcareous marl or clay sub-peat mineral soils or limestone bedrock. There is a range in pH values from 4.5 to 7.9 at Turraun closely related to the depth of the underlying calcareous sub-stratum (Rowlands, 2001). Bulk density values range from 180 ± 48 kg m^{-3} (0 – 15 cm peat depth), 120 ± 7 kg m^{-3} (15 – 30 cm) and 120 ± 11 kg m^{-3} (30 – 45 cm).

From the different communities that revegetated the site, four wetland communities were selected for C flux measurements called after the dominating plant species: *Typha (T. latifolia); Phalaris (P. arundinacea), Eriophorum/Carex (E.*

Box 7.5. Rewetted industrial cutaway peatland, Ireland

Site:
- Country: Irish Midlands
- Location: Turraun, Co. Offaly
- Co-ordinates: N 53º14' - 53º19', W 7º42' - 7º48'

Climate:
- Precipitation: 804 mm yr^{-1}
- Annual mean temperature: + 7.5 °C

Peat properties:
- Peat type: fen (*Phragmites australis*) over calcare–ous fen-peat
- Climatic region temperate, marine

Former land use and restoration:
- Former land-use: milled peat harvesting
- Restoration since: 1970
- Restoration type: re-vegetating naturally; 1991: blocking of ditches
- Investigation: Jan. 2002- Dec. 2003

Methods:
- Gas exchange measured with closed chambers
- Modelling of net ecosystem CO_2 exchange

Reference: Wilson *et al*., 2007

angustifolium and *C. rostrata)* and *Juncus/ Holcus (J. effusus and H. lanatus).*

Gas exchange measurements and balances
CO_2 fluxes were measured between April 2002 and December 2003 using the closed chamber method with light and dark chambers. CH_4 measurements took place between July 2002 and December 2003. The hourly time series of environmental variables recorded by a weather station and data loggers located within the communities were used to reconstruct the annual gas balances for each sample plot separately (Figure 7.1).

Results
All communities were sources of CO_2 and CH_4 in 2002 and 2003 (Figure 7.1). Large losses of CO_2 occurred in all communities driven partly by considerable losses during the wintertime periods (Wilson *et al*., 2007) and also by deeper water tables throughout the peatland in 2003. Emissions of CH_4 were highest in the *Typha* communities in both years. No CH_4 fluxes were detected in either the *Juncus*/*Holcus* communities or in the bare peat plots as a consequence of deep water tables (*Juncus*/*Holcus*) or absence of vegetation (bare peat plots).

Conclusions
Restoration of the C sink function at Turraun was not observed in the two years of the study. Interannual variation in climatic inputs had a significant impact on

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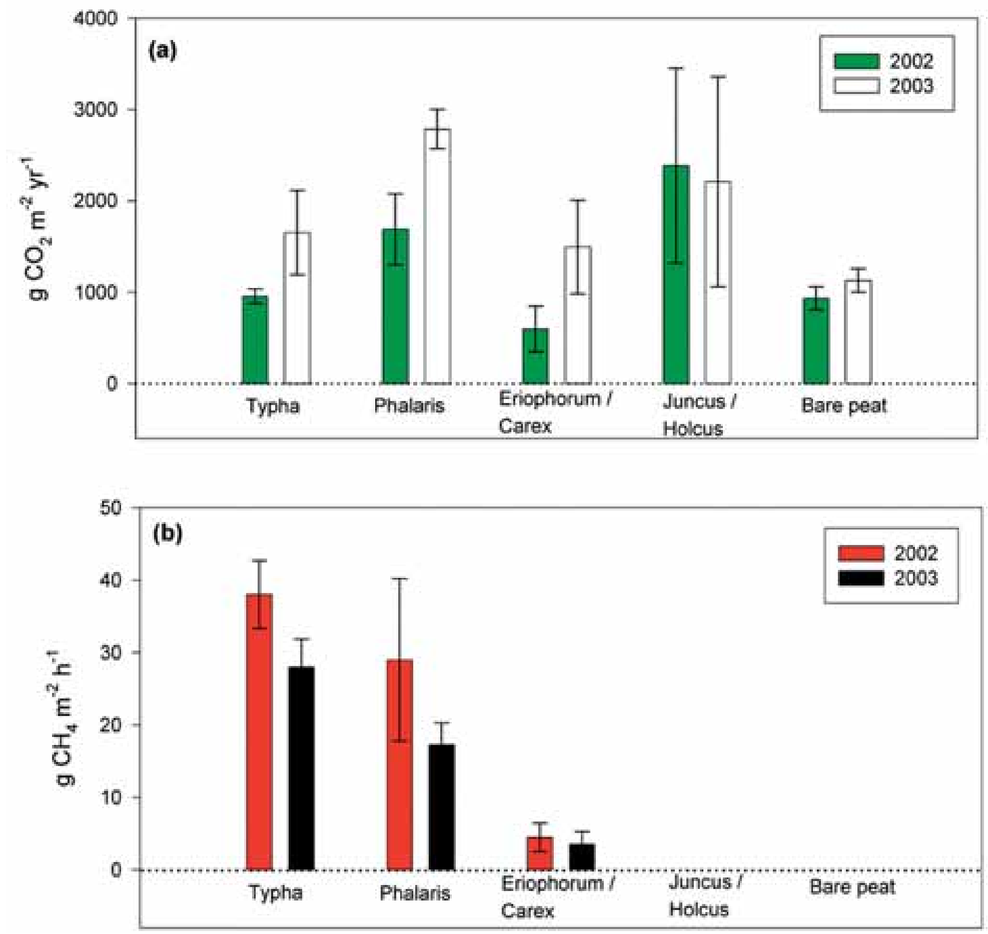

Figure 7.1. *Annual (a) Net ecosystem exchange of CO_2 (g CO_2 m^{-2} yr^{-1}) and (b) CH_4 fluxes (g CH_4 m^{-2} yr^{-1}) for 2002 and 2003 at Turraun, Co. Offaly. For more details see Box 7.5.*

water tables within the peatland resulting in large losses of CO_2 particularly in late summer / early autumn of 2003. Without a functioning acrotelm layer, it is difficulty for a cutaway peatland to maintain a high water table and large losses of CO_2 may be inevitable. Furthermore, the mild, oceanic climatic conditions experienced in Ireland permit the degradation of organic matter even throughout the winter period.

7.4. Greenhouse gas fluxes from restored peatlands formerly under forest, agricultural use or drained fallow land

7.4.1. Boreal bogs and fens in Northern Europe

Estimation of greenhouse gas emissions from Scandinavian bogs and fens
Information on gas fluxes from Scandinavian restored sites where no peat excavation had taken place and which have been under forest or agricultural use before

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is limited. If no peat has been excavated and if the sites have only weakly been drained or fertilized for the past land-use, gas exchange from the restored sites should soon reach the level of natural mires after restoration. Therefore, here some results of Scandinavian studies on greenhouse gas exchange from pristine peatlands are shown.

For poor sites, such as bogs, values up to -367 g CO_2 m^{-2} yr^{-1} have been reported, while fens are in the lower range, i.e. ca. -55 g CO_2-C m^{-2} yr^{-1} (Tolonen & Turunen, 1996). For CH_4 emissions from natural mires a most comprehensive investigation was carried out on a large number of peatlands in Sweden, however mainly during one year (Nilsson *et al*., 2001). The CH_4 emission values were stratified on three trophy (nutrient) levels from poor to rich and also for geographical location. The values are presented here for the poor and rich sites together with the range of the emission values. The poor sites mainly bogs show a range of 1.3 to 10.7 g CH_4 m^{-2} yr^{-1}. The richer sites being sedge fens are in the range of 4 to 22.7 g CH_4 m^{-2} yr^{-1}, tall sedge fens having the highest values.

The N_2O emissions from pristine peatlands are very low. Values do not differ between poor and rich sites. At investigations on two sites, a range of 0.02 to 0.03 g N_2O m^{-2} yr^{-1} was found (von Arnold, 2004). With high nitrate availability, e.g. on formerly fertilized peatlands, the values could reach higher levels.

The global warming potential calculated from CO_2 accumulation, CH_4 and N_2O emissions result in a sink of -354 to -283 g CO_2 equivalents (CO_2-e) m^{-2} yr^{-1} for poor sites, while richer sites tend to result in a source of -20 to 122 g CO_2-e m^{-2} yr^{-1} (global warming potential on a 500-year-basis; see Chapter 5 for more details).

Calculations and compilation of data from Swedish rewetted forested peatland show both C sequestration and CO_2 emission after rewetting (Nilsson & Nilsson, 2004). On poor sites, C sequestration is observed (-294 g CO_2-e m^{-2} yr^{-1}). This is due to a relatively high accumulation rate and a low decomposition rate. But, with time the accumulation rate might decrease. Methane emissions are low (3 g CH_4 m^{-2} yr^{-1}). On rich sites, on one hand, the sequestration rate is lower than on poor sites amounting to -110 g CO_2 m^{-2} yr^{-1}. On the other hand, high CH_4 emissions occur. The latter are estimated as 27 g CH_4 m^{-2} yr^{-1}. N_2O emissions from rewetted sites are negligible. The loss of C stored in tree biomass is not included in this calculation. In summary, rich peatlands are about neutral with respect to greenhouse gas emissions while poor sites are generally sinks.

7.4.2. Boreal and temperate peatlands in Eastern Europe

Rewetting experiments in Eastern Germany

Study site

The experimental site is a part of the Peene river valley mire in Northeast Germany (Box 7.6). It is situated within a large-scale nature conservation project called “Peene-Haffmoor / Peenetal” (20 000 ha). It is a fen mire, partially of the percolation mire type and partially of the spring mire type. Total peat depth is up to 10 m. The carbon: nitrogen (C/N) ratio and peat degradation according to von Post are 12.2 and 10 in 0-20 cm, 18.5 and 6 in 20-50 cm, and 21.4 and 3 in 50-110 cm, respectively.

There has been low-intensity use for pasture and local peat cutting since the middle of 18th century. After 1960 deep drainage and periodic ploughing with renewal of the grass sward was carried out to ensure an intensive use as grassland. In 1990 the intensive land-use was abandoned.

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Box 7.6. Restoration of grassland fens, NE Germany

Site:

– Country:	NE Germany
– Location:	Zarnekow Polder
– Co-ordinates:	N 53º52´ E 12º53´

Climate:

– Precipitation:	544 mm y^{-1}
– Annual mean temperature:	+ 8.1 °C
– Vegetation period (T > +5ºC):	222 days

Peat properties:

– Peat type:	fen (percolation or spring fen)
– Climatic region	cool, temperate

Former land use and restoration:

– Former land-use:	intensive grassland since 1960
– Restoration since:	2005
– Restoration type:	a.) low intensity pasture b.) flooding (0.2-1 m) in 2005
– Investigation:	April 2004 until 2007

Methods:
- Gas exchange measured with closed chambers
- Partially automated chambers

References: Augustin & Joosten, 2007; Augustin, unpublished

One part of the polder area was used for low-intensity pasture. The other part was flooded again in the course of the restoration project in the beginning of 2005.

The water level varies between 20 and 100 cm above surface on the flooded area. At the non-flooded low intensity grassland the groundwater level is at soil surface during winter time and up to 60 cm below surface during summer. The hydrologic regime is influenced in a complex way both by the river Peene which has a very small slope before flowing into the Baltic Sea and by the groundwater of the adjacent valley edges.

The restoration was started by opening the dykes which limit the river Peene. Since the water level of the river is higher than a part of the fen area today, as a result of the peat decline, it was flooded after dyke opening. Two treatments were established: recently rewetted low-intensity grassland (control) and recently flooded grassland (Joosten & Augustin, 2006; Augustin & Joosten, 2007; Augustin, unpublished).

Results

Opening of the dykes resulted in changes in the gas exchange on both the flooded area and the rewetted grassland because of the risen groundwater level. Contrary to the originally drained sites the rewetted grassland (control) is a weak source of N_2O (0.22 g N_2O m^{-2} yr^{-1}) and CH_4 (0.001 g CH_4 m^{-2} yr^{-1}). Completely surprising is that the rewetted grassland already functions as a strong CO_2 sink, too (-917 g CO_2 m^{-2} yr^{-1}). On the flooded site the sink function for CO_2 was also established very rapidly and in a very strong manner (-2383 g CO_2 m^{-2} yr^{-1}). Simultaneously, however, extremely high methane emissions of, up to 267 g CH_4 m^{-2} y^{-1} occurred. The N_2O release was lower than in the rewetted grassland.

In the first year after flooding the treatments behaved contrary to the expectations. In a 500 year perspective the control showed a strongly positive climate effect or GWP (-883 g CO_2-e m^{-2} yr^{-1}). Despite the high CO_2 assimilation the flooded treatment had a smaller GWP than the non-flooded rewetted grassland due to the high CH_4 emission (-352 g CO_2-e m^{-2} y^{-1}). The high CH_4 emission in the flooded treatment is due to the fact that fresh and easily degradable plant material is submerged and a fermentation process is initiated by flooding.

Conclusions

Information from literature and observations indicate that after flooding of degraded fen mires three phases with very different characteristics may occur. In the *first phase* extremely high CH_4 emissions will be observed in connection with a low net CO_2 uptake (accumulation). This initial phase has an extremely negative climate effect. The experiments discussed above are presently in this phase. In the *second phase*, CH_4 emissions are strongly reduced, whereas CO_2 uptake shows its maximum. This phase has a slightly positive climate effect. For the final *third phase* both low CH_4 releases and low net CO_2 uptakes are expected, similar to the situation in pristine mires. In this phase the climate effect of a rewetted peatland is close to that of a natural one.

Regretfully no information exists on the duration of the individual phases and how emissions develop within and between the phases. Moreover, there is only little known about the effect of different water levels after flooding on the gas exchange. Therefore comprehensive long-term field studies on gas fluxes are urgently needed for designing optimally effective methods and to evaluate the effects of flooding.

7.4.3. Temperate bogs in Central and Western Europe

Restoration of drained South German bogs under fallow

Study site

The Kendlmühlfilze is a representative bog area for this extensive mire belt (Box 7.7). Total peat depth is up to 10 m, with fen peat in the lower 3 m and bog peat in the top 7 m. The C/N ratio of the upper peat layer was between 26.5 (restored former drained sites) and 40.5 (natural *Sphagnum* hollow).

Former land-use at the studied sites was fallow land under drainage as preparation for peat cutting (but without extraction of the peat) and small-scale domestic peat cutting. Both activities ceased around 1950. Restoration works were undertaken in 1990 and 1999. Therefore, two time steps could be included in the assessment of the restoration effect on greenhouse gas exchange. The gradient from natural to restored to degraded sites is reflected in the mean water table. Natural sites show water tables between 0 and 10 cm, degraded sites between 12 and 29 cm and restored sites between 5 and 12 cm below surface. Maximum oscillation could by found at the degraded sites (54 cm) and minimum at the natural sites (17 cm).

Restoration included blocking the ditches (former drained areas) and damming (former peat cut areas) to reduce the discharge from the sites. Drained sites where small-scale peat cutting had taken place were (partly) flooded whereas drained sites without peat cutting were just rewetted. No active introduction of

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Box 7.7. Rewetting of bogs drained for domestic peat-cutting, South Germany

Site:
- Country: S Germany
- Location: Kendlmühlfilze
- Co-ordinates: N 47°20´ E 12°25´

Climate:
- Precipitation: 1483 mm y^{-1}
- Annual mean temperature: + 8.3 °C

Peat properties:
- Peat type: bog
- Climatic region: cool, temperate

Former land use and restoration:
- Former land-use: fallow land, drained for peat cutting , no or domestic peat cutting until 1950
- Restoration since: a) 1990, b) 1999
- Management type: blocking of ditches and damming
- Investigation: 1999 – 2000

Methods:
- Gas exchange measured with closed chambers
- Modelling of net ecosystem CO_2 exchange

Reference: Drösler, 2005

vegetation was undertaken. Natural versus restored versus degraded (drained only and peat cut) sites were compared.

Results

Degraded former peat cut sites had CO_2 emission of 1472 g CO_2 m^{-2} yr^{-1}, degraded drained sites without peat cutting showed lower losses of 864 g CO_2 m^{-2} yr^{-1}. Restored sites had emissions of 466 g CO_2 m^{-2} yr^{-1}. Only the natural sites had a significant uptake with a mean CO_2 sink rate of -260 g CO_2 m^{-2} yr^{-1}. Methane emissions at the former peat cut sites were insignificant (0.07 g CH_4 m^{-2} yr^{-1}), whereas at the drained but not peat cut sites, emissions were moderate at 1.9 g CH_4 m^{-2} yr^{-1}. The restored sites showed higher emissions of 4.8 g CH_4 m^{-2} yr^{-1}. As expected, the highest emissions were found at the natural sites with a mean of 25.9 g CH_4 m^{-2} yr^{-1}. Nitrous oxide emissions were only significant at the former peat cut sites with a mean of 0.17 mg N_2O m^{-2} yr^{-1}. Looking at the C balance, calculated as the difference between NEE, CH_4-C losses and estimated DOC losses, the mean at the natural sites was -45.6 g C m^{-2} yr^{-1}, which is around the double of the long-term rate of carbon accumulation for northern bogs.

As shown, natural bogs in this study are sequestering C. However, for the assessment of the climatic relevance, the global warming potential (GWP) is the key indicator and is presented here on the 500 year perspective. Degraded former peat cut sites had a GWP of 1499 g CO_2-e m^{-2} yr^{-1}, respectively. Drained but not peat cut sites showed 878 g CO_2-e m^{-2} yr^{-1}. At the restored

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sites, the GWP was estimated to be 502 g CO_2-e m^{-2} yr^{-1}. Only natural sites act as a sink for the greenhouse gases at the long term (500 years) perspective with GWP of -67 g CO_2-e m^{-2} yr^{-1}.

Conclusions

Rewetting of temperate peat bogs is shown to reduce C losses in comparison to drained bog sites. However, restoration does not immediately lead to a C sink within the studied first ten years. But, it helps to reduce C losses significantly. Natural sites were the only ones showing C uptake in this study. Restoration of peat bogs for climate mitigation should avoid flooding the sites but instead should establish a water table slightly below peat surface to reduce the dominating effect of CH_4 emissions. As the peat bogs along the Alps are no longer under land use pressure, restoration can be seen from a functional aspect, not provoking too many conflicts with the land-owners. Conflicts between the objectives of climate mitigation and species protection are not prominent. The typical peat bog species have evolved under natural conditions which are favourable in terms of greenhouse gas exchange.

7.4.4. Temperate fens in Central and Western Europe

Restoration of a previously drained fen area in the Netherlands

Study site and methods

The village Zegveld is located in the centre of the peat area in the Western part of the Netherlands (Box 7.8) (Langeveld *et al*., 1997). The cultivation of the area around the Zegveld started in the 11th century. Drainage by digging ditches was the main measure to enable the mining of peat. This resulted in a typical landscape with long stretches of land and numerous ditches.

Box 7.8. Relict of a previously drained partly restored fen area, the Netherlands

Site:	
– Country:	SW Netherlands
– Location:	Zegveld
– Co-ordinates:	N 52°07’, E 4°51’
Climate:	
– Precipitation:	790 mm y^{-1}
– Evapotranspiration:	543 mm y^{-1}
– Annual mean temperature,:	+ 9.8 °C
– Vegetation period (T > +5°C)	280 days
Peat properties:	
– Peat type:	fen (woody sedge)
– Climatic region	temperate, marine
Former land use and restoration:	
– Former land-use:	grassland
– Restoration since:	1970
– Management type:	raising of water table
– Investigation:	2001/2002

Reference: Langeveld *et al*., (1997)

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After the peat mining stopped, a 5-6 m thick peat layer remained. The main land use type became production grassland with interchanging periods of grazing by cows or sheep and mowing. The water content of the soil is regulated by maintaining the water levels in the ditches at a fixed height. The traditional depth of the water table in the ditches is 70 to 90 cm below the surface. The consequence is a subsidence rate of approximately 1.1 cm yr^{-1} of the land surface (Beuving & van den Akker, 1996). In the early 1970s an experiment was started to investigate possibilities to reduce the rate of subsidence by manipulating the water table depth. At a research farm near Zegveld, two water table regimes were maintained for a number of fields. The first water table regime was identical to the traditional regime, i.e. 70 cm below the surface (hereafter called "dry" treatment), and the second regime was 20 cm below the surface (hereafter called "wet" treatment). This resulted in water table depths in the centre of the fields varying between -15 cm in winter and -60 cm in summer for the dry, and for the wet treatment between 0 and -35 cm.

To estimate the effect on the greenhouse gas emissions of this difference in water table depth, measurements of CO_2 and N_2O exchange were performed simultaneously on both fields. Based on earlier work by van den Pol-van Dasselaar *et al*. (1999) it was assumed that for these water table depths the emission of CH_4 is negligible. For the measurement of CO_2 exchange the eddy correlation technique was applied. N_2O emissions were measured at irregular time intervals (varying from 1 week during the growing season to 1 month in winter).

Results and conclusions

In both treatments there is a net emission of CO_2, where the wet plot showed a 25 % lower emission than the dry plot (Fig; Jacobs *et al*., 2003). The emission of N_2O was approximately 34 % lower for the wet relative to the dry treatment.

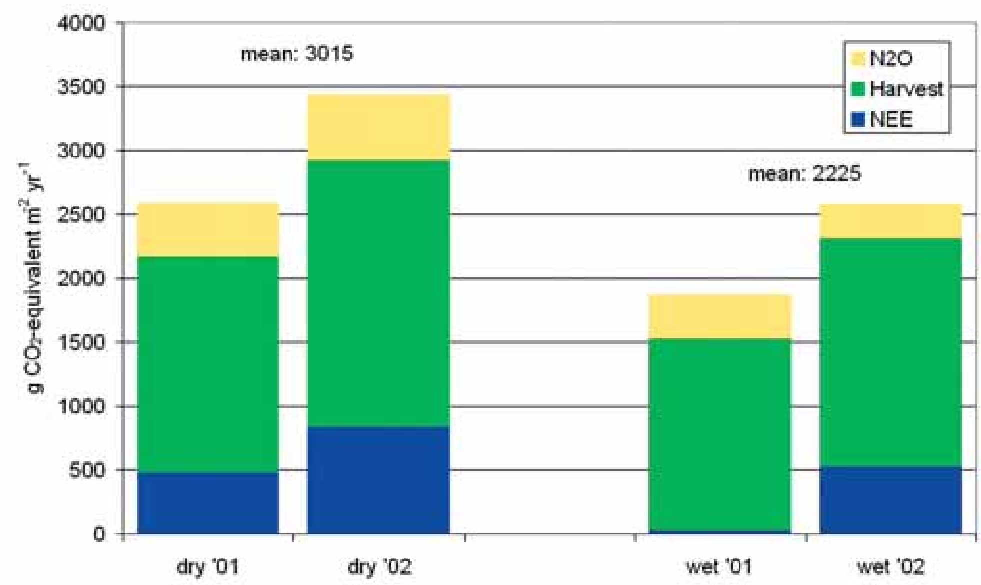

Figure 7.2. *The total emission of CO_2 and N_2O for the dry and the wet fields of partially restored peatland in the Netherlands following use as grassland. For more details about the study site see Box 7.8.*

Emission of N_2O is very much governed by management, i.e. the timing and amount of fertilizer used and the grazing density. However, this did not explain the differences found. The total amount of fertilizer and manure applied was 15 % higher at the wet field. A more likely explanation is the higher water table of the wet field limiting the aeration of the top soil and thus stimulating the production of nitrogen gas (N_2) at the expense of N_2O (see also Chapter 3). Data were lacking in this experiment to verify this hypothesis. In addition to NEE measured with the eddy correlation system, losses of C in biomass removed by mowing or grazing were taken into account. In Figure 7.2 the greenhouse gas balance of the fields is depicted as the sum of NEE, the biomass removed and the emission of N_2O. This shows that the total emission is 26 % less for the wet treatment than for the dry treatment, i.e. 2225 versus 3015 g CO_2-e m^{-2} yr^{-1}.

To evaluate the net effect of land use of an area on the total emission of greenhouse gases other sources of emission should be taken into account as well. For the experiment discussed above, other possible contributions exist which were not quantified, e.g. the outflow of carbon by hydrological pathways and, at the farm level, the emission from the stables and the fuel used to manage the land. More information of emissions at the farm level for this site may be found in Langeveld *et al*. (1997).

Most of the peat areas in the Netherlands are traditionally in agricultural use, mainly as grassland. There is still a lot of debate about the future land-use for this area. If land subsidence and CO_2 emissions are to be stopped, the water table should be even higher than as described above for the wet treatment. This would make the traditional concept of agriculture impossible in this area. At the moment the most likely solution is a limited raising of the water table for a large area and a water table close to the surface only for small areas. At present, possible benefits of a higher groundwater table in summer by applying subsurface-infiltration using drains are being investigated. Also, research into possible negative side effects on CH_4 emissions in the case of a groundwater table at or close to the surface is being performed.

Results of a rewetting experiment on a shallow grassland fen in northwest Germany

Study site

A rewetting experiment on a fen was carried out in northwest Germany in the Dümmer region (Meyer *et al*., 2001; Box 7.9). The experiment was part of a nation-wide project on rewetting of fens. The fen is of lacustrine origin (in the vicinity of the Dümmer Lake). A *Phragmites australis*, small sedge or brook forest peat layer of 30 to 60 cm overlies 30 cm of calcareous, clay or organic mud. The pH value (measured in $CaCl_2$) in the peat layer ranges from 4.5 to 5.3. The bulk density is about 480 kg m^{-3} (0 – 30 cm peat depth) and 200 kg m^{-3} (30 – 55 cm). The ratio of organic carbon to total nitrogen (C/N ratio) varies between 14 and 18.

Experimental design

The site had been used as intensively fertilized grassland until 1992. In 1993 fertilization was stopped, but harvesting was continued once to three times a year, if possible.

The following treatments were established:
1. *Non-rewetted (dry):* In this treatment no change in water regime was done. The mean water table was 50 and 30 cm below the surface in the winter months (November to April) and 80 and 70 cm below the surface in the summer months (May to October), in the years 1996 and 1997, respectively, where the gas measurements were carried out. The grassland was not

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Box 7.9. Rewetting of a shallow grassland fen, NW Germany

Site:
- Country: NW Germany
- Location: Dümmer
- Co-ordinates: N 52º47', E 8º30'

Climate:
- Precipitation: 698 mm yr^{-1}
- Annual mean temperature: + 8.7 °C

Peat properties:
- Peat type: fen (*Phragmites australis*, sedge, wood), lacustrine origin
- Climatic region temperate

Former land use and restoration:
- Former land-use: intensive grassland
- Restoration since: 1993
- Management type: no fertilization, a) rewetting with ditches, b) flooding
- Investigation: 1996 and 1997

Reference: Meyer *et al*., 2001

fertilised and harvested twice a year. The grassland vegetation is dominated by *Phalaris arundinaceae, Poa pratensis* and *Alopecurus pratensis*.

2. *Rewetted by ditches (moist):* In 1993, a 2.5 ha fen area was rewetted by ditches which had been dug at a distance of 40 m. The water level in the ditches was maintained constant at 30 cm below the surface through the year by supplying river water from the nearby Hunte River. The mean water table was observed at 35 and 15 cm below the surface in the winter months and at 50 and 50 cm below the surface in the summer months, in the years 1996 and 1997, respectively. 347 mm and 357 mm of river water had been pumped into the rewetted area in the years 1996 and 1997, respectively.

3. *Rewetted by flooding (flooded):* In 1995, another area of 2.5 ha was rewetted by flooding with water from the Hunte River at 10 cm above ground level during the whole year. Vegetation changed from the formerly dominating species *Phalaris arundinaceae* and *Deschampsia cespitosa* to increasing spread *of Glyceria fluitans* and *Typha latifolia.* Harvesting was not possible anymore.

Twelve plots were established on each treatment, six with and six without vegetation. Plots with vegetation were used to measure the gas exchange of N_2O and CH_4. Plots without vegetation, where the vegetation had been eliminated in the beginning by taking off the upper 2 cm layer and was eliminated during the experiment by burning upcoming plants, were designated to determine CO_2 emission

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from the peat. Gas measurements were done weekly between March 1996 and March 1998 using the closed chamber technique.

Results
All plots were sources of CO_2 independently of the rewetting measure (Figure 7.3a). Under flooding the highest emissions were observed in the first two years after the beginning of the experiment. The dry and moist treatment behaved almost neutral with respect to CH_4 emissions. On the contrary, flooding lead to high CH_4 emissions of 61 to 131 g CH_4 m^{-2} yr^{-1} (Figure 7.3b). The N_2O emissions were slightly higher in the moist plot compared to the dry treatment (Figure 7.3c). In the flooded treatment, N_2O emissions were reduced to 0 and even a small sink function for N_2O was observed.

Conclusions
Raising the water table by ditches which are filled 30 cm below surface did not reduce the CO_2 emissions of the fen and even slightly increased the N_2O emissions by favouring nitrogen mineralisation and denitrification due to higher moisture contents. Thus, this measure, designed to rewet under continuing land-use as grassland, is not suited to reduce greenhouse gas emissions and peat mineralisation. The underlying mud impedes water rise from the sandy subsoil and, thus, in summer the water table falls in the plots between the ditches due to high water consumption by transpiration.

Also, flooding did not restore the function of the fen as a C sink in the first two years. Even though not the entire NEE was assessed, it can be concluded from the CO_2 emissions from the bare plots, that flooding lead to high C emissions from aerobic and anaerobic processes. Only the N_2O emission was strongly reduced and the site converted into a net N_2O sink. Nevertheless, the flooding seems to be the best method, to keep water on the plots during the whole year, and vegetation changes indicate a shift towards peat forming plants (*Typha latifolia*, *Phragmites australis*). New field measurements planned for the coming years are designed to show a decrease in CO_2 emissions from the flooded site. It is yet unclear, whether in this highly degraded peat a C sink function will be re-established one day.

7.5. Conclusions on the most efficient restoration techniques with respect to greenhouse gas emissions and/or peat growth

The results presented in this chapter are summarized in Table 7.1. There is a large variation in greenhouse gas emissions between the different restored peatland sites. The emissions of CO_2, CH_4 and N_2O and the C accumulation rate depend on the geographical situation (i.e. temperate vs. boreal peatlands) and peat type (i.e. nutrient poor bogs vs. nutrient rich fens).

Most investigations of gas emissions are from North European bogs. The global warming potential greenhouse gas emissions from Finnish or Swedish spontaneously regenerated cutover bogs or bogs used for agriculture or forestry is slightly negative to neutral, covering a range between -354 g CO_2-e m^{-2} yr^{-1} (sink of greenhouse gases) and 253 g CO_2-e m^{-2} yr^{-1} (source of greenhouse gases) on a 500 year basis. Nevertheless, information on CH_4 emissions is sometimes missing and could worsen the GWP by 10 to 81 g CO_2-e m^{-2} y^{-1} (Nilsson *et al*., 2001).

North European fens tend to show gas emissions with a positive global warming potential due to lower C accumulation and higher CH_4 emissions than the bogs. The GWP is in the range of -20 to 122 g CO_2-e m^{-2} yr^{-1} (500 years).

Restored Canadian cutover bogs show exceptionally high gas emissions, especially

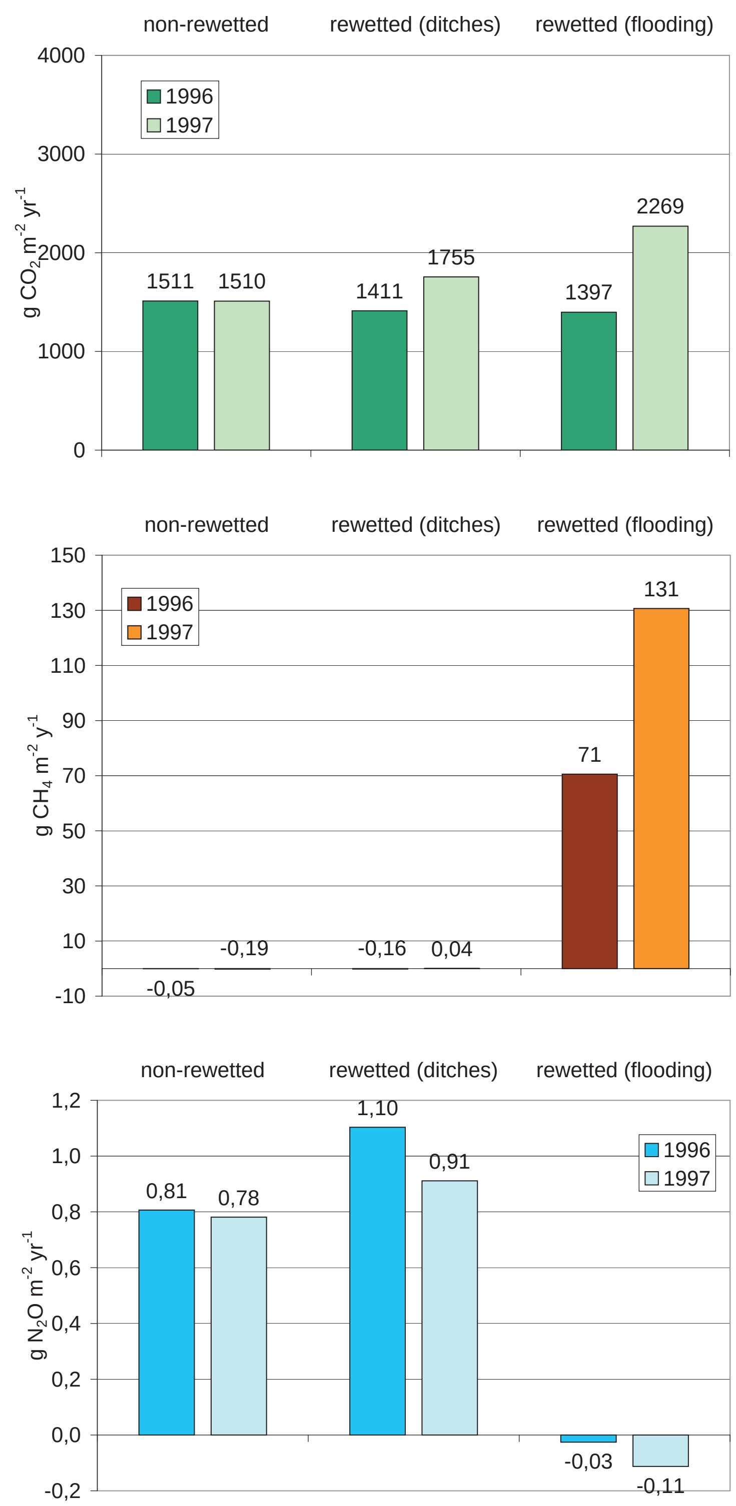

Figure 7.3. *Annual (a) CO_2 fluxes from bare plots; (b) CH_4 fluxes and (c) N_2O fluxes from vegetated plots for 1996 and 1997 from a shallow fen at Schäferhof, Hunte, Germany.*

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Table 7.1. *Greenhouse Gas Emissions from Restored and Non-restored (control) Sites of Different Peat Types, Regions and Restoration Types*

Country	precip/ temp (mm yr^{-1}/ ºC)	Restoration type, vegetation	Time after restoration (yr)	NEE (CO_2) (gCO_2 m^{-2} yr^{-1})	CH_4 (gCH_4 m^{-2} yr^{-1})	N_2O (gN_2O m^{-2} yr^{-1})	GWP[a] 100 yr (gCO_2-e m^{-2} yr^{-1})	GWP[a] 500 yr (gCO_2-e m^{-2} yr^{-1})	References
Cutover bogs (see section 7.2)									
Boreal									
Canada		Blocking of ditches, mulching with straw	3	1753[b]	1,3	0	1785	1763	Waddington *et al*., 2002
		Peat cutting area, non-restored		871	0,02	0	871	871	
Sweden	800 / 6	Self regeneration, poorly drained, mean estimate	50	-460 to -37[c]			-460 to -37	-460 to -37	Lode, 2001
Finland	700 / 3.5	All: blocking ditches, additional water supply	10						Kivimäki *et al*., in press
		Pure stands of *E. vaginatum*, or *C. lasiocarpa*		-80[d]			-80	-80	
		Mixed stands of *E. vaginatum* and *C. lasiocarpa*		-320[d]			-320	-320	
		Bare plots		150[d,e]			150	150	
Finland	700 / 3.5	All: self regeneration, poorly drained	52						Yli-Petäys *et al*., 2007
		Wet: *S. pulcrum*		-143[d]	45		982	199	
		Dry: *S. papillosum, E. vaginatum, C. lasiocarpa*		40[d]	28		740	253	
Temperate									
the Netherlands	853 / 9	Damming of area	10	97	0	0	97	97	Nieveen *et al*., 1998
Ireland	804 / 9.3	All: blocking of ditches, flooding	10						Wilson *et al*., 2007
		Juncus, Holcus (drier places)		2281	0	0	2281	2281	
		Phalaris, Typha (wetter places)		1755	27.9	0	2453	1967	
		Eriophorum, Carex (wetter places)		1039	4.0	0	1140	1070	
		Bare soil		1019[e]	0	0	1019	1019	

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Peatlands used for agriculture or forestry (see section 7.3)									
Boreal bogs									
Finland		Restored		-367[f]					Tolonen & Turunen, 1996
Sweden		natural		f	1,3 to 10,7	0,02	-329 to -94	-354 to -283	Nilsson et al., 2001; von Arnold et al., 2004
Sweden		Rewetted forest sites		-294	3	0,02	-213	-268	Nilsson & Nilsson, 2004
Boreal fens									
Finland		Restored		-55[f]					Tolonen & Turunen, 1996
Sweden		natural		f	4 to 22,7	0,03	54 to 521	-20 to 122	Nilsson *et al.*, 2001; von Arnold et al., 2004
Sweden		Rewetted forest sites		-110	27	0,03	574	100	Nilsson & Nilsson, 2004
Temperate bogs									
S Germany	1483 / 8.3	Drained and peat cutting (control 1)		1472	0,07	0,17	1525	1499	Drösler, 2005
		Drained (control 2)		864	1,9	0	911	878	
		Blocking ditches, damming	1-10	466	4,8	0	586	502	
		Natural		-264	25,9	0	383	-67	
Temperate fens									
the Netherlands	790 / 9.8	Non rewetted control		2545[g]	0	1.47	2983	2770	Jacobs *et al.*, 2003
		Raising water table	31	1915[g]	0	0.97	2204	2063	
NE Germany	544/8.1	Flooded (0.2 to 1 m)	1	-917	0	0.22	-851	-883	Augustin, unpublished
		Low intensity pasture, raising water table	16	-2383	267	0	4300	-352	
NW Germany	697/8.7	Non rewetted control, unfertilized		1511[e]	-0.1	0.79	1744	1631	Meyer *et al.*, 2001
		Rewetting by ditches	4-5	1584[e]	-0.1	1.01	1882	1737	
		Flooding (0.1 m)	1-2	1833[e]	101	-0.07	4329	2588	

[a] GWP after Forster *et al.*, 2007. For 100 year time horizon GWP is 25 and 298 g CO_2-e for CH_4 and N_2O, respectively. For 500 year time horizon GWP is 7.6 and 153 g CO_2-e for CH_4 and N_2O, respectively.
[b] CO_2 emission from straw included
[c] net long-term peat accumulation
[d] values measured for growing season (June-Sept) were corrected by respiration in winter 70 g CO_2 m^{-2} yr^{-1} (Alm *et al.*, 1999b).
[e] bare soil
[f] values for all three gases were combined from Finnish and Swedish studies to calculate GWP
[g] carbon exportation by harvesting was included

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of CO_2, if compared to North European restored cutover bogs. One reason is the decomposition of mulch straw, used in restoration to favour the establishing of *Sphagnum* mosses. As this straw would also be decomposed following other uses (e. g. ploughing under or organic fertilization) and as it is a renewable C source at a short term, it needs not be considered in the GWP of the restored cutover bog. Furthermore, this case study was done three years after starting restoration, and it is known that successful restoration needs more time. There is a need for more studies on greenhouse gas emissions on restored North American peatlands.

Greenhouse gas emissions and GWP from restored temperate cutover peatlands seem to be much higher. Nevertheless, contradicting results are reported. Whereas Dutch cutover bogs show rather low GWP (97 g CO_2-e m^{-2} yr^{-1}) very high values are reported for Irish bogs, lying one order of magnitude higher than the Dutch or North European results. In temperate peatlands, peat mineralisation is favoured by mild winters, possibly enhanced by slightly increasing temperatures in the last decade due to climate change. Therefore the need for optimal conditions for the growth of peat forming plants in the summer months is much higher under temperate than under boreal conditions because higher winter C losses have to be compensated for. For restored temperate bogs in South Germany, under high precipitation rates, a GWP of 502 g CO_2-e m^{-2} yr^{-1} was determined on a 500 year basis. Thus, these restored bogs emit greenhouse gases into the atmosphere in contrast to natural bogs. After 1 to 10 years of restoration the optimal conditions for low greenhouse gas emissions had not been achieved at the restored site. Nevertheless the greenhouse gas emissions were already much lower than from non-restored peatlands (Drösler, 2005).

From restored temperate fens contradicting results are reported. In Northeast Germany flooding or raising the water table lead to CO_2 uptake of the peatland and the GWP, at least on a 500 year basis, was positive. In contrast, in Northwest Germany high CO_2 emissions were observed after rewetting by ditches or flooding. Eventhough this study was performed on bare soil and photosynthesis as a C sink process was excluded, it cannot be assumed that this process together with the root respiration will lead to a large accumulation of carbon in soil. Also, Wilson *et al*. (2007) (Figure 7.1) detected lower NEE for CO_2 from bare than from vegetated plots. Both German studies confirm that flooding leads to very high CH_4 emissions up to 267 g CH_4 m^{-2} $y4^{-1}$. Methane emission can be avoided, if a small oxic zone (10 cm) is maintained at peat surface where CH_4 oxidation will take place.

Nitrous oxide emissions from restored boreal bogs and fens were generally very low. The GWP of the determined emissions is about 0 to 5 g CO_2-e m^{-2} yr^{-1}. For restored temperate fens N_2O-based GWP up to 155 g CO_2-e m^{-2} yr^{-1} on a 500 years basis was measured.

Several factors conditioning or limiting the success of restoration measures and their impact on greenhouse gas emissions were not assessed in direct measurements. First of all there are technical limitations for optimal rewetting conditions. For example, due to loss in buoyancy, shrinkage and peat mineralization the peatland surface is not flat anymore and it becomes difficult to establish the optimal flooding level for peat growth together with low CH_4 emissions. The spatial distribution of greenhouse gas exchange between the peatland and the atmosphere needs further examination. Also, the availability of water for rewetting may be limited. In summer, the water loss by evaporation has to be compensated for, which is difficult if the hydraulic

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conductivity of the peat is reduced by degradation and especially if the peatland is grown on low permeable mud, impeding water supply from groundwater.

Secondly, political, social and global factors play an important role in restoration success and climate change mitigation by peatland restoration. For example, rewetting may need landscape planning involving different land owners. Compromises have to be made, permitting agricultural land-use in summer, e.g. at least temporary draw down of the water table. This may strongly limit the success of peatland restoration for climate change mitigation. In the summer months peat mineralisation is most intensive and water most limited. Moreove, the predicted temperature rise and mild winters will favour peat mineralisation and increase the emissions of greenhouse gases of peatlands on a long term basis.

Peatland restoration might be a very cost-efficient solution for greenhouse gas mitigation compared to technical solutions, e.g. insulation of buildings, renewable energy sources or wind energy (Joosten & Augustin, 2006). Nevertheless, a cost-efficiency analysis has to be done for each case individually. Costs should include one-off costs, e.g. restoration measures and land acquisition, and on the other hand running expenses, e.g. maintenance and annual interests or capitalization of investments. One-off costs of 400 € per ha in Ireland (restoration measures, Box 7.5) and 1 000 to 2 000 € per ha in Sweden (Lundin, personal communication) are reported. In Canada the costs of restoration including *Sphagnum* transfer, mulching and blocking the ditches also varies from 1 000 to 2 000 € per ha (Rochefort, personal communication). For the restoration of 10 000 ha of fens in Northeast Germany about 30 million € is planned to be spent between 2000 and 2008, corresponding to 3 000 € per ha, for land acquisition, water management, planting of trees, opening of dams and infrastructural measures, e.g. construction of bridges and lanes (LUNG, 2006). Assuming a long-term interest rate of 3 %, the annual costs due to capitalization interests amount to 12 - 90 € per ha. If a reduction in greenhouse gas emissions of about 1 000 g CO_2-e m^{-2} yr^{-1} is assumed (e.g. Box 7.5 or Box 7.7) the cost per tonne of CO_2-e mitigation is between 1.2 and 9 € per year. To date (appointed date: 05.03.2008) the EU Emission Allowances are listed at 0.03 € per tonne on the stock market CARBIX (Available http://www.eex.com) and are too low to cover the restoration costs. Nevertheless, the Second Period European Carbon Futures are listed at 21.15 € per t (2008) to 24.17 € per tonne (2012) at the derivatives market, indicating increasing prices in the forthcoming years. This will greatly improve the cost efficiency of peatland restoration projects if the greenhouse gas sequestration potential is included.

Of course, additional effects of peatland restoration should be considered as well. Peatland restoration is, in general, not necessarily designed for mitigation of greenhouse gas emissions. There are different objectives for peatland restoration, e.g. protection of rare species or biodiversity (birds, plants and animals), ecosystem restoration, or tourism. The different objectives necessitate different measures and different conditions. For example, it might be difficult to establish a water table equally favourable for waterfowl, breeding birds and *Sphagnum* growth. Nevertheless, it has to be kept in mind that peatland ecosystems are unique in their function as a sink for atmospheric CO_2. If rewetting is planned the re-establishing of this function should be of pre-eminent importance.

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CHAPTER 8:

PEAT IN INTERNATIONAL CLIMATE CHANGE CONVENTIONS

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8.1. United Nations Framework Convention on Climate Change

Two international treaties, the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol (KP) supplementing it, aim at mitigation of human induced climate change. The aim of the UNFCCC is to "stabilize greenhouse gas emissions at a level that would prevent dangerous anthropogenic (human induced) interference with the climate system." The UNFCCC does not include mandatory emission reduction or limitation commitments, but industrialised countries (listed in Annex I of the Convention and referred later as Annex I Parties) committed themselves to the aim of returning individually or jointly their emissions of anthropogenic greenhouse gases (GHG) to the 1990 level. Developing countries are not listed in the Annex I of the UNFCCC and are referred to as Non-Annex I Parties. At the same time Annex I parties were committed to develop, periodically update, publish and make available national inventories of anthropogenic emissions by sources and removals by sinks of all GHGs not controlled by the Montreal Protocol. Thus, the UNFCCC obligates Annex I Parties to report annually their GHG emissions from the year 1990 to the current inventory year. Currently 192 countries have ratified the UNFCCC, which was adopted in Rio de Janeiro in 1992.

Kyoto Protocol was adopted in 1997. It sets quantitative and legally binding GHG emission limitation or reduction targets for industrialised (Annex I) countries. Only Parties to the UNFCCC that have also become Parties to the Protocol are bound by the Protocol's commitments. The ultimate objective of the Kyoto Protocol is to curb GHG emissions of industrialised countries at least 5% from the 1990 level during its first commitment period in 2008-2012. So far, 37 Annex I countries and the European Community have ratified the Protocol and committed themselves to the specified quantified emission limitation or reduction targets listed in the Annex B of the Kyoto Protocol. The maximum amount of emissions (measured as equivalent carbon dioxide equivalents) that a Party may emit over the commitment period in order to comply with its emissions target is calculated as the Party's assigned amount. Parties, which have so far ratified the Protocol make up a 64% share of the total annual emissions of Annex I Parties in 1990. The USA, which produces a substantial share of global GHG emissions, has not ratified the Protocol. The Kyoto Protocol does not contain quantified emission limitation targets for non-Annex I parties (developing countries).

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If a Party fails to meet its emission reduction or limitation commitment under the Protocol, it must make up the difference in the second commitment period, and in addition a penalty of 30%. Its eligibility to participate in the emissions trading will also be suspended. The Protocol includes provisions for the review of its commitments.

Flexibility mechanisms
Cost of reducing emissions varies much from region to region. The Kyoto Protocol defines three so called "flexibility mechanisms" which aim at lowering the overall costs of emission abatement by Annex I countries. These mechanisms are Emissions Trading, Joint Implementation and Clean Development Mechanism. They enable Annex I Parties to implement activities and projects to reduce emissions or to remove carbon from the atmosphere either in another Annex I country (Joint Implementation) or in a non-Annex I country (Clean Development Mechanism).

Emissions Trading (see Box 8.1) allows Annex I Parties to acquire carbon units from other Annex I Parties and use them towards meeting their emission targets under the Kyoto Protocol. Only Annex I Parties with quantified emission limitation and reduction commitments may participate in the emissions trading under the Protocol.

Box 8.1. European Union greenhouse gas Emission Trading Scheme (EU ETS)

EU member states started their emission trading system (EU ETS) at the beginning of 2005, before the commencement of the global emission trading scheme under the Kyoto Protocol. The scope of EU ETS is more limited than the emission trading (ET) under the Kyoto Protocol. During the first trading period from 2005 to 2007, the ETS covered only CO_2 emissions from large emitters in the power and heat generation industry and in selected energy-intensive industrial sectors: combustion plants, oil refineries, coke ovens, iron and steel plants and factories making cement, glass, lime, bricks, ceramics, pulp and paper. The EU ETS covers more than 11 000 installations in EU member states.

EU ETS includes combustion installations that use peat as a fuel with rated thermal input of more than 20 MW and smaller combustion installations connected to the same district heating network. IPCC default emission factor for peat is 106 g CO_2 MJ^{-1} and peat is treated in the same way as fossil fuels in the trading system.

The EU ETS considers biomass as CO_2 neutral and an emission factor of 0 is applied to biomass fuels. Biomass is defined as non-fossilised and biodegradable organic material originating from plants, animals and microorganisms. For fuels or materials containing both fossil and biomass carbon, a weighted emission factor shall be applied, based on the proportion of the fossil carbon in the fuel's overall carbon content. Biomass fraction refers to the percentage of mass combustible biomass carbon according to the biomass definition. The specific procedure to determine the biomass fraction of a specific fuel type including the sampling procedure shall be agreed with the competent authority before the start of the reporting period in which it will be applied. If the determination of the biomass fraction in a mixed fuel is technically not feasible or would lead to unreasonably high costs, the operator shall either assume a 0% biomass share or propose an estimation method for approval by the competent authority (Commission Decision of 29/01/2004 establishing guidelines for the monitoring and reporting of greenhouse gas emissions pursuant to Directive 2003/87/EC of the European Parliament and of the Council, Brussels, 29/01/2004).

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8.1.1. National greenhouse gas inventories

The UNFCCC and Kyoto Protocol cover six greenhouse gases (CO_2, N_2O, CH_4, HFC, PFC and SF_6), the anthropogenic emissions and removals of which are to be estimated and reported annually to the UNFCCC secretariat. Reliable greenhouse gas inventories create the basis for assessing whether Parties will meet their emission limitation and reduction commitments under the Kyoto Protocol. The national GHG inventories are prepared according to guidelines accepted by the Intergovernmental Panel of Climate Change (IPCC) and agreed upon by the Conference of the Parties to the UNFCCC and Kyoto Protocol. Inventories should be transparent, consistent, comparable, complete and accurate. They are subject to annual review conducted by international expert teams. Good quality inventories, which meet the standards set by the UNFCCC and the Kyoto Protocol, are a requirement for Parties to be eligible to participate in Emissions Trading and the other Kyoto mechanisms.

The general framework, coverage and format of reporting are specified in the UNFCCC reporting guidelines. Actual methodologies for calculation of the emission estimates are provided in IPCC guidelines. All Parties to the UNFCCC and Kyoto Protocol have to prepare their national GHG inventories in accordance with the IPCC guidelines.

Emissions and sinks are to be reported in six different sectors; 1) Energy, 2) Industrial processes, 3) Solvents and other product use, 4) Agriculture, 5) Land Use, Land Use Change and Forestry (LULUCF) and 6) Waste. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 1997) constitute the basic guidance for the inventory calculations. Due to the development of methodologies and advances in research over time these guidelines have been elaborated by the IPCC Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories in 2000 (IPCC, 2000) and the Good Practice Guidance for Land Use, Land Use Change and Forestry (GPG-LULUCF) in 2003 (IPCC, 2003). The good practice guidance reports aim at producing inventories that are neither over nor underestimates as far as can be judged, and in which uncertainties are reduced as far as practicable. The good practice guidance reports use the same methodological approaches as the 1996 IPCC Guidelines, but they give a more systematic approach for the choice of "tier" level for the methodology to be used in estimating the emissions and removals. Three hierarchical tiers of accounting methods were introduced by the good practice guidance reports that range from default data and simple equations to the use of country specific data and models to accommodate national circumstances. They also include additional guidance on quality assurance and control in preparing the inventory, quantification of uncertainties, development of consistent time series and reporting documentation. The GPG-LULUCF also introduced a new, more systematic reporting category structure with the aim of more consistent representation of land areas in the inventory.

In 2006, the IPCC adopted the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). These new guidelines constitute a revision of previous guidelines and include new sources and gases, revised methodologies taking into account new scientific and technical information, and an updated sector structure for reporting: the Industrial Processes and Solvent and Product Use sector have been combined to a new sector called Industrial Processes and Product Use, and the Agriculture and Land Use, Land-Use Change and Forestry sectors have been

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combined to a sector called Agriculture, Forestry and Other Land Use. During the first commitment period of the Kyoto Protocol the 1996 IPCC Guidelines and Good Practice Guidance reports (IPCC, 2000; IPCC, 2003) will set the standards for the reporting requirements under the Kyoto protocol. According to the Kyoto Protocol, any revised methodologies can be adopted only for future commitment periods.

8.2. Peat in international climate change conventions

8.2.1 UNFCCC reporting

Peat-based emissions to be reported under the UNFCCC are divided in the 1996 IPCC guidelines and the GPG-LULUCF under several sectors and categories. Reporting of all known anthropogenic emissions and removals is encouraged, but only those emissions sources for which the IPCC has been able to provide generic and scientifically sound methodologies are under mandatory reporting. Only human-induced emissions are included in the reporting, thus emissions and removals from virgin peatlands are not part of the reporting. For practical reasons, the IPCC GPG-LULUCF uses “managed land” as a proxy for identifying anthropogenic emissions by sources and removals by sinks. All emissions and removals occurring on managed land are included in the reporting.

1996 IPCC guidelines and GPG-LULUCF give methodological guidance for estimating peat-based emissions from three different reporting sectors: 1) Energy, 2) Agriculture, and 3) Land Use, Land Use Change and Forestry (LULUCF). Details for each sector are described below and given in Table 8.1.

Methane (CH_4) emissions and removals associated with drainage, and rewetting of organic soils are discussed in 1996 IPCC Guidelines and the GPG-LULUCF, but no methodological guidance is given.

Energy

Emissions (CO_2, N_2O, CH_4, NMVOC, NO_x, and CO) from peat combustion for energy and heat are to be reported in the Energy sector (Table 8.1). Carbon dioxide emissions from fuel combustion for electricity and heat are calculated on the basis of annual fuel consumption, fuel specific emission factors and fuel specific oxidation factors. The IPCC default emission factor for peat is 106 g CO_2 MJ^{-1}.

In 1996 IPCC Guidelines peat is classified as a fossil fuel, while in the new 2006 IPCC guidelines peat fuel has its own category between fossil fuels and biomass fuels. The 2006 IPCC Guidelines further specify that although peat is not strictly speaking a fossil fuel, the CO_2 emissions from combustion of peat are included in the national emissions as for fossil fuels.

Meanwhile CO_2 emissions from combustion of biofuels are reported under the Energy sector as a memo item, which means that they are not calculated into the total emissions of the Energy sector or included in national total emissions. This is justified by avoiding double accounting, since these emissions are already reported as emissions from stock changes due to human activities in the Land Use, Land Use, Change and Forestry sector (e.g harvesting plantations, commercial fellings, fuel wood gathering and other management practices). In addition, carbon stock changes due to natural losses on managed lands like fires and storms are reported in the LULUCF sector.

Methane and N_2O emissions from fuel combustion are reported in the Energy sector and included in the sector and national totals. This applies also to emissions from biofuels, because the effect

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Table 8.1. *Emitted Greenhouse Gases and Sources for Peat Related Reporting Sectors*

Reporting sector	Emission source	**Gas(es)**
Energy - Fuel combustion	Peat combustion	CO_2, N_2O, CH_4, NMVOC[a], NO_x[b] and CO[c]
Agriculture - Cultivation of organic soils	Organic agricultural soils	N_2O
LULUCF - Forest land	Drained, organic forest soils	CO_2
LULUCF -Cropland (inc. liming) -Grassland	Organic agricultural soils	CO_2
LULUCF - Wetlands	Peat extraction areas	CO_2, N_2O
LULUCF - Biomass burning	Biomass on peat soils (Forest land, Cropland, Grassland)	CO_2, N_2O, CH_4, NO_x and CO
LULUCF - Direct N_2O emissions from fertilisation	N fertilization in peaty forest soils	N_2O
LULUCF - N_2O emissions from disturbance associated with land use conversion to cropland	Organic agricultural soils	N_2O
LULUCF - N_2O emissions from drainage of soils	Drained, organic forest soils	N_2O (optional, in appendix in GPG-LULUCF)
Virgin peatlands		Not included

[a] NMVOC = non-methane volatile organic carbon
[b] NO_x = nitrogen oxides
[c] CO = carbon monoxide

of these emissions is an addition to the stock changes reported in the LULUCF sector.

Agriculture
The only source of peat-based emissions to be reported in Agriculture sector according to 1996 IPCC guidelines and good practice guidance is N_2O emissions from organic agricultural soils. These emissions are calculated in general by multiplying an emission factor with area data. All the other land use related, peat-based emissions are reported in the Land Use, Land-Use Change and Forestry sector.

Land Use, Land-Use Change and Forestry (LULUCF)
Current reporting requirements in the LULUCF sector follow GPG-LULUCF and aim at covering emissions and removals of GHGs from all managed land and land-use changes. Emissions and removals of CO_2 should be estimated from all the relevant

C pools (that is biomass, dead organic matter and soil) divided to six different land-use categories; *Forest land, Cropland, Grassland, Wetlands, Settlements* and *Other land*. Particularly, changes from one land use to another and the following changes in C stocks are to be reported. Emissions and removals from mineral and peaty (organic) soils should be reported separately. In addition to reporting of human-induced changes in C stocks, also emissions from non-CO_2 greenhouse gases are reported. Nitrous oxide and CH_4 are relevant greenhouse gases especially on peaty soils and wetlands. IPCC guidelines provide guidance for estimation of non-CO_2 emissions from biomass burning, N_2O emissions from disturbance associated with land-use conversion to cropland and N_2O emissions from forest fertilization and drainage (Table 8.1).

Peat-based emissions from land use have to be reported under the land use category where they occur. For example, emissions from organic agricultural lands should be reported under Cropland and emissions/removals from forests growing on peaty soils should be reported under Forest land. For organic forest soils, default methodology is provided only for drained organic soils, not for managed undrained soils.

The land-use category Wetlands is defined in the GPG-LULUCF as “land that is covered or saturated by water for all or part of the year and does not fall into the forest land, cropland, grassland or settlement categories”. Actual guidance for estimation of emissions and removals from Wetland category is provided only for industrial peat extraction areas and reservoirs. The 2006 IPCC Guidelines recognize also emissions from peat extracted for horticultural and other purposes, while current guidance takes into account emissions only from peat extracted for energy use.

IPCC default methodology for reporting of CO_2 emissions from peat soils in different land use categories is area data multiplied with a land-use-specific emission factor. More sophisticated methods require derivation of emission factors from country-specific data and more detailed classification of land-use categories to different land-use types (e.g. forest type) and/or different management and climate regions. This requires that region/country specific experimental data and research results are available. Also dynamic, process-based models can be developed for estimation purposes.

In the reporting under the UNFCCC, land-use categories should be subdivided into managed and unmanaged according to national definitions. The national definitions need to be used consistently over time. Different countries with similar conditions may report the emissions differently depending on their interpretation of managed land and national land-use definitions. However, artificial manipulation of soil water level is considered in the guidelines as human-induced management and emissions/removals from all drained areas are thus part of the reporting requirements.

8.2.2. Kyoto reporting and peat

The Kyoto Protocol contains quantified emission limitation targets for Annex I countries. Reporting and accounting of emissions/removals under the Kyoto Protocol differ somewhat from the reporting under the UNFCCC. Total national emissions are reported to the UNFCCC both including and excluding the Land Use, Land Use Change and Forestry sector from the sum. The total emissions excluding the LULUCF sector are the basis for the emission reduction and limitation commitments under the Kyoto

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Protocol. Emissions and removals from the LULUCF sector are considered only partially when assessing the fulfillment of the commitment. Emissions and removals during the commitment period caused by afforestation, reforestation and deforestation since 1990 are added to or subtracted from the Party's assigned amount according to the rules and modalities under Article 3, paragraph 3 of Kyoto Protocol, and part of the mandatory reporting and accounting under the Protocol. The Party's assigned amount is the maximum amount of emissions (measured as CO_2 equivalent; CO_2-e) that the country may emit over the commitment period in order to comply with its emission target. The emissions or removals from forest management, cropland management, grazing land management and revegetation will also impact the accounting, if elected by the Party (Article 3, paragraph 4 of the Protocol). If these LULUCF activities are net sinks, they will generate so called removal units (RMU) that can be used to meet the Kyoto commitments. One RMU is equal to one metric ton of CO_2-e. If they constitute a net source, the corresponding amount will be subtracted from the Party's assigned amount under the Kyoto Protocol (emission limitation burden of the country increases accordingly).

Reporting under the Kyoto Protocol also includes some additional requirements for reporting of area data (spatial data); the units of land subject to the activities under Article 3, paragraphs 3 and 4 need to be identifiable and geographical boundaries of land subject to these activities have to be reported.

Peat-based emissions in the Agriculture sector, that is N_2O emission from agricultural soils, and emission from peat combustion in the Energy sector are reported under the Kyoto Protocol as under the UNFCCC. In the accounting, the emissions during the commitment period are compared to the base year emissions (net-net accounting).

Reporting under Article 3, paragraph 3 relates to emissions and sinks from changes in forest cover which result in a change in the land-use category, e.g. conversion of forest land into settlements. For area considered, "Kyoto Forest" is a minimum area of land of 0.05-1.0 ha with tree crown cover (or equivalent stocking level) or more than 10-30% with trees with the potential to reach a minimum height of 2-5 m at maturity in situ. Parties estimate and report under the Article 3.3 of the Kyoto Protocol emissions and removals during the first commitment period in 2008-2012 from afforestation, reforestation and deforestation (ARD) activities since 1990. When estimating these emissions so called gross-net accounting method is used. This means that only the net change in C stocks over the commitment period (gross emissions less removals) is taken into account.

Peat-based emissions under Article 3.3 relate to situations where forests grow on the top of the layer of peat or organic soil and tree cover is removed permanently, e.g. to use land for agricultural cultivation. This would be accounted as deforestation under Article 3.3. Also situations where, for example, drained organic forest land would be restored back to its natural state by blocking the ditches and removing the existing tree cover would be accounted as a deforestation activity (providing that the tree cover before the conversion meets the definition of Kyoto Forest).

Afforestation of, for example, abandoned peat extraction areas or organic arable land would also be included (depending on time since the last forest cover) in the reporting and accounting under Article 3.3. Normal forest regeneration after harvesting is not included. Afforestation and reforestation sites may also be net

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sources of GHG emissions due to high emissions from organic soils (all C pools are taken into account in the reporting and accounting). For example, afforestation of a former organic agricultural land can constitute a net GHG source if emissions from soil C oxidation exceed the removals created by biomass growth. Afforestation and reforestation activities were intended in the Kyoto Protocol as means to create sinks and thus ease the compliance with the emission reduction commitment. Despite this, possible net emissions due to afforestation and reforestation also have to be reported under the Article 3.3.

Article 3, paragraph 4 of the Kyoto Protocol provides countries an opportunity to elect into the accounting certain activities in LULUCF sector, which can produce GHG removals and help countries to meet their commitments. These activities are Forest management (FM), Cropland management (CM), Grazing land management (GM) and Revegetation (RV). Possibilities to use removals from these activities to meet the commitments under the Kyoto Protocol are limited with different rules (Marrakesh Accords, 2002; a set of agreements reached at the 7th Conference of the Parties in 2001 about the rules for meeting the targets set in the Kyoto Protocol). Sinks from Forest management are calculated using the gross-net accounting method when sinks from agricultural activities (CM, GM and RV) are calculated with the net-net accounting method. As in the case of article 3.3 also article 3.4 activities have to be reported (once elected), even if they constitute a net source of emissions.

Forest management under article 3.4 is defined very broadly as "a system of practices for stewardship and use of forest land aimed at fulfilling relevant ecological (including biological diversity), economic and social functions of the forest in a sustainable manner". The total impact of forest management to C stock changes depends on all the C pools including biomass, dead organic matter and soil. Organic soils can be either net sinks or net sources of GHGs depending on site fertility, climatic conditions and intensity of management (e.g. drainage), among others. Drainage of organic soils normally accelerates decomposition of peat and increases CO_2 emissions. Also, N_2O emission may increase somewhat, but CH_4 emissions decrease or stop depending on the level of drainage. A Party can issue credits from forest management activity, even if organic forest soil pools constitute a source, if sinks from the other pools (biomass and dead organic matter) exceed the soil emissions.

Organic agricultural soils are usually sources of greenhouse gas emissions. Cropland management is the system of practices on land on which agricultural crops are grown and on land that is set aside or temporarily not being used for crop production. Under the Kyoto Protocol, net-net accounting is used for emissions from Cropland management, i.e. emissions in inventory year are compared to emissions in base year. This enables Parties to issue RMUs from cropland management, even if the activity constitutes a source in the commitment period providing that emissions in the commitment period are smaller than corresponding emissions were in the base year.

Grazing land management is defined as the system of practices on land used for livestock production aimed at manipulating the amount and type of vegetation and livestock produced. For grazing land management net-net accounting is also applied.

Revegetation under Article 3.4 is defined as "direct human-induced activity to increase carbon stocks on sites through the establishment of vegetation that covers a minimum areas of 0.05 hectares and does

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not meet the definitions of afforestation and reforestation". GPG-LULUCF gives activities such as restoring/reclaiming herbaceous ecosystems on C depleted soils, environmental plantings, and planting of trees, shrubs, grass or other non-woody vegetation on various types of land as examples of revegetation. Revegetation does not necessarily imply a change in land use.

Planting of new vegetation (e.g. energy crops) on previous peat extraction areas could be interpreted as revegetation under Article 3.4. The net emissions/removals of the area would depend on the decay of the residual peat and C sink in the vegetation. Also, restoration of wetlands that have previously been drained and taken into, for example, agricultural use, or restoration of wetlands unsuccessfully drained for forestry (existing forest cover does not meet the Kyoto Forest definition), could also be accounted as revegetation activities. According to the definition, revegetation should "increase the C stocks on site". If restoration of a previously drained site includes rising of the water table, e.g. as a result of blocking the ditches, the CO_2 emissions tend to decrease, whereas CH_4 emissions increase. At present, the research results on this subject are scarce and inadequate for generalization of what would be the net impact of restoration on the C balance of different kinds of drained peatlands (see also Chapter 7). As well, the time scale to be considered when assessing whether the activity would result in an increase or decrease in the C stock on site is not defined in the Protocol.

Devegetation/degradation of Wetlands is not considered under the Protocol. Thus, for example, C loss due to degradation of Wetlands (e.g through intensive drainage and/or peat extraction) on sites where original vegetation cover did not meet the Kyoto Forest definition would not affect the accounting under the Protocol.

Parties will include different activities in the LULUCF sector under the Kyoto Protocol depending on their choices under Article 3.4. Also, country-specific definitions of these activities may vary between the Parties. However, once a certain activity has been selected into the Kyoto accounting, the land area subject to that activity needs to be included in the reporting continuously and consistently, even if a sink becomes a source of emissions.

8.3. Non-Annex I Parties (developing countries)

Non-Annex I countries do not have emission limitation targets under the Kyoto Protocol. The group of non-Annex I Parties is heterogeneous, including least developing African countries to partly industrialised countries like China, India, South-Africa and certain Latin American countries. Also, countries with high GDP per capita like the United Arab Emirates, Kuwait and Israel are included in the group. Under the UNFCCC, non-Annex I Parties are also obligated to provide information on their emissions periodically in so-called national communications. National communications shall include national GHG inventory figures and a general description of steps taken by the Party to implement the Convention. Most of the non-Annex I countries have reported inventory figures for the year 1990 and/or 1994. Annex I Parties also submit national communications, in addition to the annual inventory submissions, but the requirements about the contents and the timetable are different from those of the non-Annex I Parties. Non-Annex I countries cannot participate in emissions trading, since they do not submit annual inventories which are subject to reviews.

Peatland fires and degradation (drainage) of natural wetlands for agricultural (crop

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production and grazing), forestry, and peat extraction purposes is a problem in many non-Annex I Parties with large peatlands (e.g. South-East Asia, China; see also Chapter 6) (Figure 8.1). It has been estimated that annual CO_2 emissions from peatland fires and drainage of tropical wetlands alone in Indonesia can account 2000 million t CO_2 (Hooijer *et al*., 2006).

The Clean Development Mechanism (CDM) introduced in the Kyoto Protocol allows industrialised Annex-I Parties to fulfill part of their emission commitments by taking actions that reduce GHG emissions in developing countries. Annex I Parties may use certified emission reduction units (CERs) created from implementing projects in non-Annex I Parties to meet their commitments. CERs are generated by climate-friendly, sustainable development projects in developing countries. They can be used by developed country governments and companies to meet their reduction commitments under the Kyoto Protocol. Under the emissions trading scheme set up be the 1997 landmark agreement, CERs can be traded and thus help to combat climate change in the most cost-effective way. A CER amounts to one ton of CO_2-e. The CDM should assist non-Annex I Parties to achieve sustainable development. According to CDM rules, the project activity should result in a reduction in anthropogenic emissions by sources of GHGs that are additional to any that would occur in the absence of the proposed project activity. However, it has been decided in Marrakesh Accords, that only afforestation and reforestation activities in the Land Use, Land-Use Change and Forestry sector are applicable to the CDM in the first commitment period of the Kyoto Protocol. This limits the type of projects which could be implemented in non-Annex I countries to reduce GHG emissions from tropical peatland degradation.

8.4. Post 2012 and peat

For the first commitment period of the Kyoto Protocol (2008-2012) procedures, modalities and rules have been agreed. Reporting follows the 1996 IPCC Guidelines and Good Practice Guidance

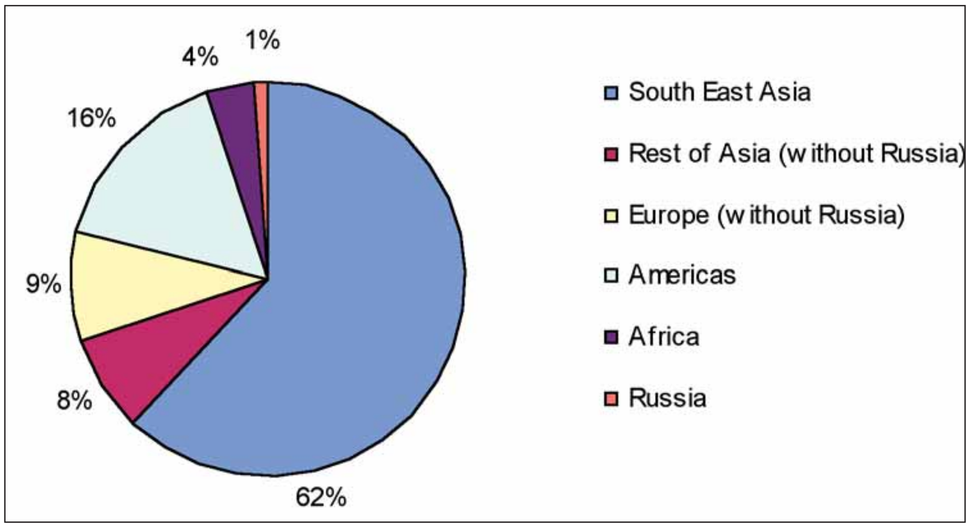

Figure 8.1. *Global CO_2 emissions from peatland drainage (estimated total of 800 Mt CO_2/year). (Source: Silvius et al., 2006)*

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reports (IPCC, 2000; 2003). Negotiations on future commitments beyond the 2012 period have started. The Post 2012 framework may not be identical to current framework under the Kyoto Protocol. New types of commitments, new countries, methodologies and approaches, as well as a different set of emission sources and sinks may be included.

The curbing of global peat-derived emissions would benefit from inclusion of the mitigation of emissions of wetland degradation and fires to the future climate framework. These emissions are significant globally, especially emissions from some non-Annex I countries. Depending on the future framework, this could be done either via enlargement of the scope of LULUCF activities under the CDM or through voluntary or other commitments by non-Annex I countries to limit their emissions at some level.

Utilisation of the latest research results and lessons learnt from the current methodologies should be carefully reviewed to assess whether current methodologies, approaches and rules are applicable and adequate to assess human-induced sinks and sources related to peat and peatlands. The review should give input to how these emissions and sinks could be best included in the emission limitation targets of a future climate framework.

Research results from life cycle analyses (see Chapter 5) and site measurements can guide Parties to promote/choose the most climate friendly ways for peatland land use and management. For example, life-cycle analyses of industrial peat production would take into account the overall climatic impact of the product (peat fuel or horticultural peat or products derived from peat) over time taking into account all the emissions and removals from the product during its life-cycle (e.g the whole production chain starting from the preparation of the peat extraction site to combustion, and the after-treatment of the extraction site). Life cycle analyses (e.g. Savolainen *et al*., 1994; Mälkki & Frilander, 1997; Uppenberg *et al*., 2001; Nilsson & Nilsson, 2004; Holmgren, 2006; Kirkinen *et al*., 2007) comparing the GHG impact of peat and fossil fuel combustion have shown that the GHG impact of peat is comparable to fossil fuels. Emissions from peat extraction can vary much depending on site characteristics and extraction technology. Some new technologies have been estimated to have the potential to reduce the emissions from the extraction site considerably. The overall greenhouse impact also depends on the emissions/ removals from the site in question before the extraction is initiated, as well as the type of after-use (Kirkinen *et al*., 2007).

There is little guidance on how rewetted/ restored peatlands should be treated in GHG inventories in the long term. Some researchers have proposed that restored peatlands in which the water table has reached the original level could be considered as unmanaged lands and be excluded from GHG inventory reporting, including cases where net emissions after restoration would be higher than before restoration (see e.g. Ministry of Agriculture and Forestry, 2007). The actual restoration activity would be included in the inventory until the original water table and conditions are reached.

On the global scale knowledge of peat-derived emissions is still very limited. The data availability and quality of data varies for different climatic regions and countries. While the location of peatlands is, on a large scale and general level, known rather well data on e.g. spatial distribution of soil carbon densities within different peatland types and regions, are inadequate.

The methodologies and guidance for estimating peat-based emissions in the

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GPG-LULUCF and 2006 IPCC Guidelines are rather scarce. The guidance is not very detailed and, due to limitations in data and knowledge, only in a limited way considers country/region- and site-specific conditions. The IPCC has not been able to provide comprehensive default methodologies, which would include all anthropogenic activities likely to alter the hydrological regime, surface temperature and vegetation composition. Major disturbances like fires would also need a more detailed methodology to comprise the specifics of peatland fires. Fires can combust both aboveground biomass and soil surface peat. It is estimated that on boreal organic soils with large combustible surfaces, fires usually consume peat soils to depths less than 10 cm, whereas some deep-burning fires can consume peat much deeper, even down to 1 m (Zoltai *et al*., 1998). The IPCC does not provide methodology for the estimation emissions from peaty soils due to fires.

According to the IPCC guidelines and good practice reports, new inventory methods and data should be used when they become available, if it improves the reliability and accuracy of the inventories. The development of scientifically sound emission factors for land use activities on peaty and wet soils is complicated and resource demanding due to the variation in site types and climatic conditions, different land uses and different intensity of the management (e.g. drainage depth), all of which have impact on soil water level and GHG fluxes. More research and long-term measurements on sites representing different conditions are needed for the development of reliable emission factors applicable for inventory purposes, and for understanding the GHG impacts of the different activities in different circumstances.

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