11-2004

EU Peatlands: Current Carbon Stocks and Trace Gas Fluxes

Kenneth A. Byrne
University College Cork, Ireland

Bogdan Chojnicki
Department of agrometeorology, Poznan, Poland

Torben R. Christensen
Lund University

Matthias Drosler
Department of Plant Ecology, Bayreuth, Germany

Steve Frolking
University of New Hampshire - Main Campus, steve.frolking@unh.edu

See next page for additional authors

Recommended Citation

Christensen TR, Friborg T (lead authors) with Byrne KA, Chojnicki B, Drösler M, Freibauer A, Frolking S, Lindroth A, Mailhammer J, Malmer N, Selin S, Turunen J, Valentini R, Zetterberg L, Vandewalle M. 2004. EU peatlands: Current carbon stocks and trace gas fluxes, Report 4/2004 to 'Concerted action: Synthesis of the European Greenhouse Gas Budget', Geosphere-Biosphere Centre, Univ. of Lund, Sweden.

Authors

Kenneth A. Byrne, Bogdan Chojnicki, Torben R. Christensen, Matthias Drosler, Steve Frolking, Anders Lindroth, Julia Mailhammer, Nils Malmer, Pirkko Selin, Jukka Turunen, Riccardo Valentini, and Lars Zetterberg

EU Peatlands: Current Carbon Stocks and Trace Gas Fluxes

Authors:
Kenneth A. Byrne
Bogdan Chojnicki
Torben R. Christensen
Matthias Drösler
Annette Freibauer
Thomas Friborg
Steve Frolking
Anders Lindroth
Julia Mailhammer
Nils Malmer
Pirkko Selin
Jukka Turunen
Riccardo Valentini
Lars Zetterberg

Edited by:
Torben R. Christensen
Thomas Friborg

Editorial assistant:
Marie Vandewalle

This discussion paper originated from a workshop in Lund, Sweden, October 2003, a contribution to the project *Concerted Action CarboEurope-GHG* which is part of the CarboEurope Cluster

Published November 2004

• Contents

• WORKSHOP PARTICIPANTS		**4**
• ABSTRACT		**5**
1. INTRODUCTION		**6**
2. CURRENT AREAL DISTRIBUTION AND C STORAGE OF EUROPEAN PEATLANDS		**7**
2.1	Reliability of the estimates in C storage studies	7
2.1.1	Dry bulk density and C concentration	7
2.1.2	Depth and areal distribution of mires	7
2.2	Carbon storage	9
3. MAIN DRIVERS CONTROLLING KEY ASPECTS OF THE GHG BALANCE OF PEATLANDS		**11**
3.1	The system	11
3.2	Rates of primary production, litter decay and carbon accumulation	14
4. DIFFERENCES IN METHODOLOGIES FOR ASSESSMENT OF STOCKS AND FLUXES		**17**
5. CURRENT ANNUAL RATES OF GHG EXCHANGES IN UNDISTURBED PEATLANDS		**19**
6. PRESENT EXTENT OF PEATLAND MANAGEMENT IN THE EU		**23**
6.1	The general trend	23
6.2	Management and sustainable use of peatlands in EU as influenced by environmentaland economic policy	25
7. MANAGED PEATLANDS		**27**
7.1	Influence of management practices on Peatland functioning	27
7.2	Consequences for the GHG exchange	27
7.2.1	Effects of drainage	27
7.2.2	Effects of changing plant composition	28
7.2.3	Nutrient effects	29
7.2.4	Emission factors and Global Warming Potential	29
8. EUROPEAN PEATLAND GHG BUDGET		**30**
8.1	Area estimates	30
8.2	Emission factors	31
8.3	European peatland budget	33
8.4	Sensitivity tests	35

9. GAPS AND FUTURE RESEARCH NEEDS **38**

• APPENDICES **40**

Appendix I: Potential in peatland management for reducing GHG in EU and its accountability in relation to the Kyoto protocol **40**

A.1. Background and purpose 40

A.2. Scenario: Natural mire - > Drained mire -> Peat Energy production/Cut- off -> Re-wetting 41

A.2.1. Schematic description of changes in GHG-fluxes and potential climate impact 41

A.2.1.1. Schematic description of how GHG-fluxes change over time due to this scenario 41

A.2.1.2. Total climate impact 42

A.2.2. Accountability in relation to the Kyoto protocol 42

A.3. Drained peatland that is forested -> Deforested area -> Peat Energy production/cut-off -> Afforestation 43

A.3.1. Schematic description of changes in GHG-fluxes and potential climate impact 43

A.3.1.1. Schematic description of how GHG-fluxes change over time due to this scenario 43

A.3.1.2. Total climate impact 43

A.3.2 Accountability in relation to the Kyoto protocol 44

Appendix II: Tables 45

• GLOSSARY **49**

• DETAILS ON PARTICIPANTS **50**

• REFERENCES **51**

• Workshop participants

The workshop was organised by Torben R. Christensen & Thomas Friborg

The participants were:

(from left)

Back: Steve Frolking, Erminia Sezzi, Riccardo Valentini, Bogdan Chojnicki, Torben R. Christensen, Matthias Drösler
Front: Thomas Friborg, Pirkko Selin, Julia Mailhammer, Lars Zetterberg, Anders Lindroth, Jukka Turunen, Annette Freibauer, Kenneth A. Byrne, Nils Malmer

University of Lund
Sweden

GeoBiosphere Science Centre
Lund, Sweden

• Abstract

Peatlands in Europe has formed a significant sink for atmospheric CO_2 since the last glacial maximum. Currently they are estimated to hold ca. 42 Gt carbon in the form of peat and are therefore a considerable component in the European carbon budget. Due to the generally wet soil conditions in peatlands they are also significant emitters of the strong greenhouse gas (GHG) methane (CH_4) and in some cases also of nitrous oxide (N_2O). The EU funded CarboEurope-GHG Concerted Action attempts to develop a reliable and complete greenhouse gas budget for Europe and this report aims to provide a review and synthesis of the available information about GHG exchanges in European peatlands and their underlying processes. A best estimate for all the European countries shows that some are currently sinks for atmospheric CO_2 while others are sources. In contrast, for CH_4 and N_2O, only the sources are relevant. Whilst some countries are CO_2 sinks, all countries are net GHG emitters from peatlands. The results presented, however, carry large uncertainties, which cannot be adequately quantified yet. One outstanding uncertainty is the distribution of land use types, particular in Russia, the largest European peat nation. The synthesis of GHG exchange, nevertheless, indicates some interesting features. Russia hosts an estimated 41% of European peatlands and contributes most to all GHG exchanges (CO_2: 25%, CH_4: 52%, N_2O: 26%, Total: 37%). Germany is the second-largest emitter (12% of European total) although it contains only 3.2% of European peatlands. The reason is the use of most of the peatland area for intensive cropland and grassland. The largest CO_2 emitters are countries with large agricultural peatland areas (Russia, Germany, Belarus, Poland), the largest N_2O emitters are those with large agricultural fen areas (Russia, Germany, Finland). In contrast, the largest CH_4 emitters are concentrated in regions with large areas of intact mires, namely Russia and Scandinavia. High average emission densities above 3.5 t C-equiv. ha^{-1} are found in the Southeast Mediterranean, Germany and the Netherlands where agricultural use of peatlands is intense. Low average emission densities below 0.3 t C-equiv. ha^{-1} occur where mires and peatland forests dominate, e.g. Finland and the UK. This report concludes by pointing at key gaps in our knowledge about peatland carbon stocks and GHG exchanges which include insufficient basic information on areal distribution of peatlands, measurements of peat depth and also a lack of flux datasets providing full annual budgets of GHG exchanges.

• 1. Introduction

A peatland is a type of ecosystem where carbon along with nitrogen and several other elements has been accumulated as peat originating from the plant litter deposited on the site.

A logical consequence of the above definition of peatlands is that they are ecosystems, which by way of nature are a sink for atmospheric carbon dioxide (CO_2). This is the case because more carbon is accumulated through photosynthesis than is released through respiration. As a consequence of this organic matter accumulates as peat. The carbon accumulated in peatlands is equivalent to almost half the total atmospheric content and a hypothetical sudden release would result in an instantaneous 50% increase in atmospheric CO_2. While this is unrealistic it nevertheless highlights the central role of peatlands in the global climate system in that these huge amounts of CO_2 have almost entirely been "consumed" since the last glacial maximum. Peatlands have over the past 10000 years helped to remove significant amounts of CO_2 from the atmosphere.

A complicating factor in this respect is that in terms of the major greenhouse gases (GHGs) peatlands are not only acting as a sink for CO_2. The wet conditions that lead to the slow decomposition, and enable peat accumulation to occur, also cause significant amounts of the powerful greenhouse gas methane (CH_4) to be formed. Indeed global wetlands (predominantly peatlands) are considered to be the largest single source of atmospheric CH_4 even when considering all anthropogenic emissions. Peatlands are therefore also a key player in the atmospheric CH_4 budget and as a result also influence the global climate.

Peatlands therefore play a vital and integral role in regulating and influencing atmospheric concentrations of CO_2 and CH_4. It is essential that attention is given to these important peatland exchange functions for CO_2 and CH_4 when considering a synthesis of greenhouse gas fluxes at the large scale. The EU funded CARBOEUROPE-GHG project has set out to produce a full synthesis of greenhouse gas fluxes in Europe and this report is forming the input regarding European peatlands. The report will start by reviewing the current areal distribution of peatlands in Europe and their associated carbon (C) stocks. We will then provide some background information on the processes determining the GHG balances in peatlands and the methodologies used to study the exchanges. Then the issue of peatland management with special attention to the EU will be reviewed. The report will conclude with an attempt at providing the best estimate of a full greenhouse gas budget for European peatlands. In appendices we provide further information on peatland GHG exchanges in a post-Kyoto context and we also provide a series of tables on statistics relating to peatlands in the individual European countries.

Although the natural focus is the main peat countries within EU15 we will adopt a flexible definition of Europe as many important peatland areas now are added as we have moved to EU25. Where appropriate we will draw on information and data beyond geographical Europe (only where relevant from Canada, US and Siberia).

• 2. Current areal distribution and C storage of European peatlands

Organic peat deposits are characterized by C contents of approximately 50% of dry matter (Table 1). Therefore, the abundance of peat signals a significant net transfer of C from the atmosphere to soils. Gorham (1991) has estimated the global area of northern peatlands to be 346 million ha out of which European peatlands cover about 51 million ha (Joosten & Clark, 2002). Northern mires have accumulated about 200 to 455 Pg C as a peat during the Holocene (Gorham, 1991) an estimate that was narrowed to 270 to 370 Pg C by Turunen *et al.* (2002). The large range in the global C storage estimate mainly reflects uncertainty in the depth of peat deposits. The same uncertainty arises from the total C stock estimates for European peatlands because detailed results of depths are available only for a few European countries.

Despite the uncertainties in the storage estimates, peatlands are a substantial global reservoir of C, constituting at least one-fifth of the total soil C pool in the world (Post *et al.*, 1982), which is equivalent to approximately half the amount of CO_2-C in the atmosphere (Houghton *et al.*, 1990). This section will primarily focus on the present knowledge of C storage in European peatlands. Factors causing variation and uncertainties in the C storage estimates will be discussed.

2.1. Reliability of the estimates in C storage studies

2.1.1 *Dry bulk density and C concentration*

A significant source of overestimation in C accumulation and the total C storage calculations of northern peatlands/mires may have been the estimates used for dry peat bulk density. The estimate most often referred to of 455 Pg stored as peat (Gorham, 1991) is based on the value of 112 g dm^{-3}. This estimate is a significantly higher estimate for dry bulk density when compared to most published studies (Table 1). According to Mäkilä (1994, based on 49 953 samples), the mean dry bulk density of Finnish geological mires is 91 g dm^{-3}, similar to those obtained for other peat deposits in the boreal region (Table 1). C concentrations of different peat types are relatively well studied and range from 40% to 58% (Table 1).

2.1.2. *Depth and areal distribution of mires*

The datasets on peat depth collected in most countries are small and may not represent the mean depths of peatland/mires while they at the same time do not take into account areal information of different mire types. The bias in mean depths is also evident through the mire classification in northern latitudes based on the artificial minimum depth of peat deposits. The minimum depth for geological mires in Europe, Canada and Russia is 30-70 cm (Zoltai *et al.*, 1975; Botch & Masing, 1979; Lappalainen, 1996). This results in serious under-representativeness of shallower mires, and is reflected in the total C storage estimates. Also, peat depth data of permafrost areas is scarce because of the difficulties of coring frozen peat. In Russia, the average peat depth is reported to be 2.2-2.6 m (Botch *et al.*, 1995). Gorham (1991) used a mean of 2.3 m for boreal and subarctic mires. However, uncertainty arises from the representativeness of these depth estimates for all northern mires. The mean depth measured at 900 000 sites spread over 5.1 Mha of Finnish mires is 1.52 m (Lappalainen & Hänninen, 1993). Also, the mean depths in Germany, Norway, Poland, Russian Karelia, Sweden and Ukraine have been reported as 1.5-2.0 m (Lappalainen, 1996). However, the question of mean/median depths remains open until more systematically surveyed data becomes available for comparison.

Detailed results of different mire types (bogs/fens) are available only for a few countries, such as Finland, based on National Forest Inventories (Minkkinen *et al.*, 2002; Turunen *et al.*, 2002). Also, a problem in global mire mapping may be the uncertain identification of forested mires from upland forest sites in vast remote areas of Canada and Russia. In West Siberian mire mapping, this uncertainty has been recognized (Kremenetski *et al.*, 2003). However, in Europe this source of error may be marginal. Overall, more accurate investigations of total areal distribution and the mean depths of different mire types (including forested/unforested mires) are required.

Table 1: *Mean of dry bulk density, carbon concentration and the long-term apparent rate of C accumulation (LORCA) of mires in North America, Asia and Europe. * = C concentration estimate used for LORCA calculations.*

Region	Mire type	Dry bulk density $g\ dm^{-3}$	Number of sites n	Carbon concentration %	LORCA $g\ C\ m^{-2}\ yr^{-1}$	Reference
European/Asian sites						
Russia	bogs	80	?	56.0	31.4	Botch *et al.* (1995)
Russia	bogs	93	11	52.7	17.2	Turunen *et al.* (2001)
Russia	fens/bogs	–	2	51.6	13.0	Oksanen *et al.* (2001)
Sweden	bogs	81	5	40.0-51.0	–	Malmer & Holm (1984)
Sweden	fens	66	11	52.0	16.0	Klarqvist (2001)
Finland	bogs	89	3670	50.3	–	Mäkilä (1994)
	fens	93			–	Virtanen *et al.* (2003)***
Finland	bogs	74	548	50.0*	24.0	Tolonen & Turunen (1996)
	fens	81	373		15.1	
Finland	bogs	72	6	44-58	16.7-22.3	Mäkilä (1997)
Finland	fens	82	180	53.9	–	Minkkinen & Laine (1998)
Finland	bogs	80	20	50*	10.2	Pitkänen *et al.* (1999)
	fens	78	7		5.6	
Finland	bogs	74	927	50.0*	20.8	Turunen *et al.* (2002)
	fens	81	375		16.9	
Ireland	bogs	69	11	51.1	–	Tomlinson & Davidson (2000)
North American sites						
USA	bogs	71	4	50*	22.9	Tolonen *et al.* (1988)
Canada	bogs and fens	112	?	51.7	29.0	Gorham (1991)
Canada	bogs	94	40	45.7	17.6	Robinson & Moore (1999)
	fens	88	60	46.2	17.0	
Canada	bogs, open fens	94	475	51.8	19.4	Vitt *et al.* (2000)
	wooded fens	105				
Canada	bogs	64	13	50*	20.9	Turunen *et al.* (unpublished)
Canada	bogs	98	12	49.3	6.3	Turunen &Turunen (2003)

2.2. Carbon storage

The estimated C storage for European peatlands is presented in Table 2 and 3. The present areal data is based on Joosten & Clark (2002). The peat resources (Gm^3 or Mt) reported in Lappalainen (1996) are converted into C storage estimates based on the average dry bulk density of 91 g dm^{-3} (Mäkilä, 1994) and C concentration of 51.7% (Gorham, 1991). The depth estimates used in the calculations are based on Lappalainen (1996) when reported. However, for most countries the average peat depths were not available and a conservative estimate of 1.75 m was used as the base of volume calculations. It is obvious that there is a lot of uncertainty in the total C storage estimates, for example, in United Kingdom the peatland C store is estimated as 4523 Mt, and the standard error of C storages about 2200 Mt (Milne & Brown, 1997). In C store calculations, the conservative minimum value of 2323 Mt is used for the C store of the United Kingdom. Also, the intensive use of mires for forestry and peat extraction (horticulture, agriculture, domestic heating and energy generation) has changed the original C storages. Countries like Denmark, Netherlands and Germany have lost a significant part of their original mire area/C storage (see section 3).

The total peat C storage of Europe was estimated as 42 Pg (Pg = 10^{15} g = Gt). This is about 10-15% of the total northern peatland C storage. However, it is notable that the total mire area in Europe has decreased considerably during the past decades (Table 3), and therefore the area of undisturbed C storage is also considerably lower compared to the total peatland area.

In mire ecosystems, additional C storage is also found in the live vegetation and in the mineral subsoil beneath peat. Gorham (1991) estimated that about 1.5% (2 kg m^{-2}) of the total C is found as biomass in the live vegetation, while 98.5% occurs in the form of peat. Using the same estimate, European peatlands would have about 515 Tg C in live vegetation. The mineral subsoil under peatlands is an additional C sink that has been overlooked and could account for some 5% of the unaccounted C in the global C budget (Turunen *et al.*, 1999; Turunen & Moore, 2003). The global minimum estimated C storage for mineral subsoil beneath peat is about 10 Pg. For European peatlands the corresponding value is about 1.6 Pg. The estimated C storage of northern mineral subsoil would be 2-5% of the corresponding C reservoir in peat.

Table 2: *Estimates of European C storages. Rough C storage estimate for the entire Russian and Canadian peatlands included for comparison. Up until 2004 EU consisted of 15 countries: Belgium, Germany, France, Italy, Luxembourg, Netherlands, Denmark, Ireland, United Kingdom, Greece, Spain, Portugal, Austria, Finland and Sweden. In 2004 the enlargement took place with 10 new countries joining - Cyprus, the Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, the Slovak Republic and Slovenia.*

Region	Area (Mha)	Peat depth (cm)	Dry bulk density (g dm^{-3})	C concentration (%)	C store (Pg = 10^{15} g = Gt)
Russia	140	2.3	91	51.7	152
Canada	110	2.3	91	51.7	119
European peatlands	51.5	1.5-2.0	91	51.7	41.8
EU15-countries	19.9	1.5-2.0	91	51.7	14.9
EU25-countries	23.2	1.5-2.0	91	51.7	17.2

Table 3: *Peatland/mire area and total C store estimates in Europe. Peatland is an area with or without vegetation with a naturally accumulated peat layer at the surface, including mires drained for forestry, agriculture, horticulture and energy production. A mire is a peatland where peat is currently being formed (Joosten & Clark, 2002). a) Joosten & Clarke (2002), b) Lappalainen (1996), c) Milne & Brown (1997), d) Minkkinen et al. (2002), e) Freibauer et al. (in prep), f) estimated.*

Country	Peatland area 2002 (km^2)	Mire area 2002 (km^2)	Peatland C store $Tg = 10^{12}$ g = Mt
Albania	179	4	15[f]
Andorra	5	2	0.4[f]
Austria	200	100	16[f]
Azores	1	1	0.08[f]
Belarus	23 500	11 412	879[b]
Belgium	160	3	13[f]
Bosnia and Herzegovina	150	10	12[f]
Bulgaria	25	5	2.1[f]
Croatia	1	1	0.08[f]
Czech Republic	200	50	19[b]
Denmark	1 400	50	89[b]
Estonia	10 000	3 000	693[b]
Faroe Islands	30	25	2.5[f]
Finland	85 000	42 500	5500[d]
France	1 500	100	91[b]
FYRO Macedonia	30	5	2.5[f]
Germany	13 000	100	422[e]
Greece	71	13	196[b]
Hungary	330	30	27[f]
Iceland	8 000	3 500	1000[b]
Ireland	11 500	2 100	947[f]
Italy	300	30	25[f]
Latvia	6 600	4 663	490[b]
Liechtenstein	1	1	0.08[f]
Lithuania	3 520	750	239[b]
Luxembourg	3	1	0.2[f]
Moldova	10	1	0.8[f]
Netherlands	2 350	150	193[f]
Norway	28 000	22 000	2239[f]
Poland	12 500	2 000	794[b]
Portugal	20	2	1.6[f]
Romania	1 000	500	82[f]
Russia European part	213 000	150 000	20 042[f]
Slovakia	26	13	2.1[b]
Slovenia	100	10	8[f]
Spain	60	10	5[f]
Svalbard/Spitsbergen	10	10	0.8[f]
Sweden	66 000	55 000	4550[b]
Switzerland	300	200	25[f]
Ukraine	8 000	5 800	554[b]
United Kingdom	17 500	1 000	2323 [c]
Yugoslavia	300	50	25[f]
Total	**514 882**	**305 202**	**41 835**

• 3. Main drivers controlling key aspects of the GHG balance of mires

3.1 The system

In any undisturbed pristine mire the upper, aerated and more or less oxic layer, the acrotelm (usually <0.4 m in depth), contrasts with the underlying, much deeper (usually >1 m) anoxic layer, the catotelm (Fig. 1). In the acrotelm, the organic material is a kind of decaying litter, mixed with living mosses and vascular plant roots, and not peat in a strict sense. Most of the microbial activity and the flow of water in mires take place there (Ivanov, 1981). The permanently water saturated catotelm (in arctic and sub-arctic areas often with permafrost) has a low microbial activity (Clymo, 1984), contains only a few living roots and the water movement is negligible. Therefore, it is only the organic matter in the catotelm that can be looked upon as peat. Consequently, the C accumulation rate in a mire refers to the rate at which the C becomes part of the catotelm.

The first step in the C accumulation in a mire is the sequestering of CO_2 from the air by the vegetation through photosynthesis (gross production, Fig. 1). This process is light dependent and takes place at the very surface of the mire. The active plants in this process are in most mires *Sphagnum* mosses together with dwarf shrubs and sedges, but less of trees and herbs. The *Sphagnum* mosses can be looked upon as ecological engineers (Jones *et al.*, 1994) since they usually occur with high abundance and much of their specific qualities will characterize the whole ecosystem. One part of the sequestered C appears as an increased plant biomass (primary production), another part is given off from vascular plant roots as non-structural, easily decomposed organic matter (Wallén, 1983, Joabsson *et al.,* 1999; Ström et a., 2003), while the respiration of the plants releases a lot of CO_2 back to the atmosphere (Christensen et al., 2003; Olsrud et al., 2004).

Most of the dead plant material forming the litter originates from the mosses and the aboveground parts of the vascular plants and accumulates near the surface. Root litter is deposited further down in the acrotelm but only small amounts are deposited into the catotelm. The surface litter from mires dominated by *Sphagnum* mosses is loose in structure (dry bulk density ~ 0.02 g cm^{-3}) but as the decay process proceeds (decay rates between 5–15 $*10^{-3}$ a^{-1} or as half-lives 50-140 years (cf. Clymo, 1984; Johnson & Damman, 1993; Malmer & Wallén, 1993; 2004) it is covered by new litter and becomes increasingly compacted (dry bulk density at the bottom of the acrotelm is typically >0.05 g cm^{-3}). The loss of mass continues until the litter becomes part of the anoxic catotelm as a result of the slow rise of the ground water (Ingram, 1982; cf. Fig. 1). Excluding the peat layers with permafrost, the inclusion might best be characterized as semi-permanent (Clymo 1984) since anaerobic decomposition is still going on in the catotelm albeit at a slow rate (~$0.02*10^{-3}$ a^{-1} or as half-life ~35000 years, cf. Clymo, 1984; Clymo & Pearce, 1995).

In the acrotelm (Fig. 1), the C losses are mainly as CO_2 and much less as CH_4 (Joabsson & Christensen, 2001) while the C concentration in the litter increases (Malmer & Wallén, 1993). In the catotelm, the C is released as equivalent amounts of CO_2 and CH_4. In addition to the C losses to the atmosphere a varying but usually a rather small proportion is lost in the run-off as dissolved organic carbon (DOC, Olsrud and Christensen, 2004) and through erosion by wind and water from exposed patches without plant cover, e.g., due to disturbances (Proctor, 1997).

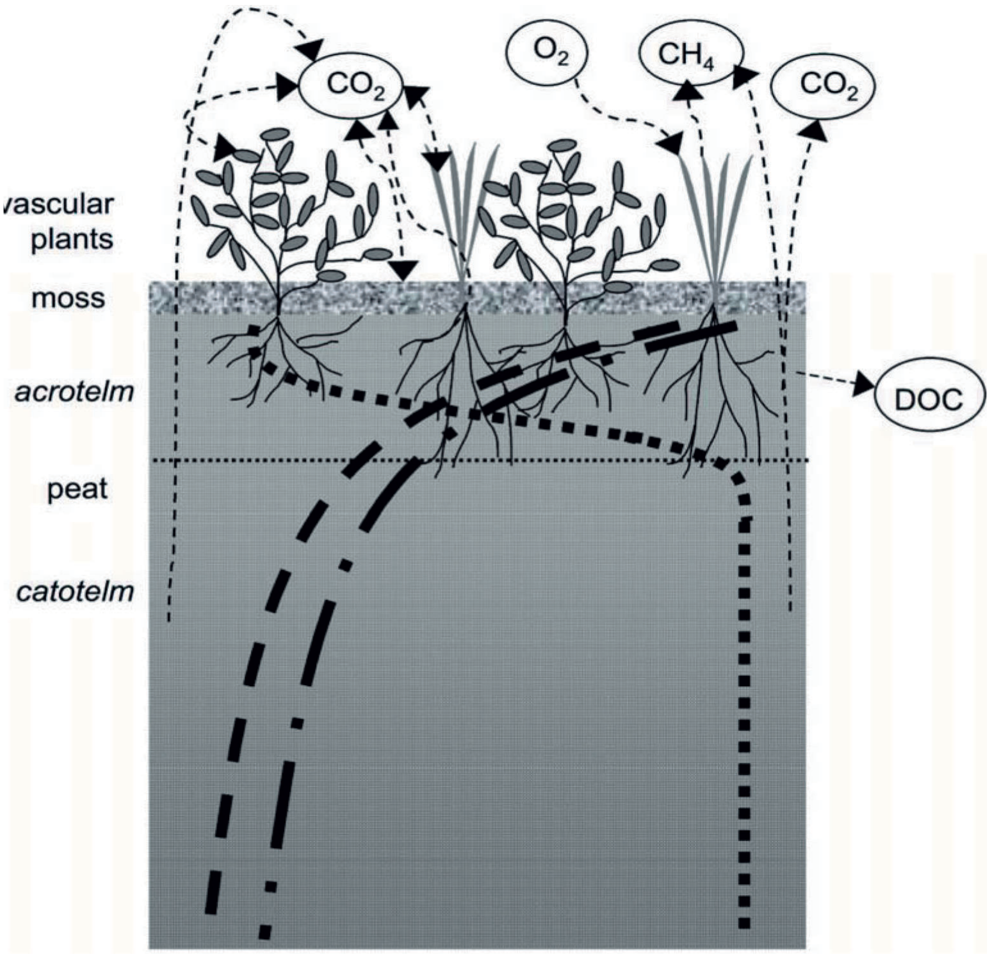


Figure 1: *Controls on CO_2 and CH_4 fluxes from mires. The carbon balance represents the net difference between photosynthetic C uptake by vascular plants (e.g., shrubs and sedges) and non-vascular plants (e.g., mosses); respiration by plant foliage, stems, and roots; and decomposition of plant litter, plant root exudates, and peat. The horizontal dotted line represents the approximate boundary between the acrotelm and the catotelm. Controls on plant photosynthesis and respiration include temperature, water status, incoming solar radiation, and seasonal phenology. Litter and peat decomposition is controlled by interacting gradients down the peat profile of temperature (heavy dashed line), tissue quality (heavy dot-dash line), and anoxia (heavy dotted line). Note that these lines in the figure represent the general character of these gradients, which are temporally and spatially variable. C losses from decomposition will be primarily as gas fluxes of CO_2 and CH_4, and as drainage losses of dissolved organic carbon. Methane fluxes are controlled by factors influencing decomposition, and by additional factors controlling methane transport to the atmosphere and the degree of oxidation during this transport. Transport is via diffusion, bubbling, and through gas-conducting plants. A key control for transport and oxidation is the density, root distribution and phenology of gas-conducting vegetation (e.g., sedges) that can transport both oxygen from the atmosphere to the rhizosphere (enhancing CH_4 oxidation) and CH_4 from the rhizosphere to the atmosphere (enhancing CH_4 flux).*

The two determinants of the total litter decay losses in the acrotelm are the residence time and the decay rate (Malmer & Wallén, 1993; Belyea, 1996). In general, the mass loss (including C) in the acrotelm during the decomposition process can be described by the exponential equation:

$$\ln M_t = \ln M_0 - kt$$

where M_t and M_0 designate mass at time t and at the formation of the litter, respectively, and k the decay rate. The limit between the acrotelm and catotelm appears as a distinct decrease in decay rate (Fig. 2). The residence time depends on the position of the water level, which is determined by the equilibrium between the supply of water and the resistance to the flow of water in the acrotelm (Belyea & Malmer, 2004). Lowering the water level will increase the residence time and the decay losses while a rise would shorten it and decrease the decay losses (Fig. 2). The decay rate is directly proportional to the temperature but also strongly dependent on the litter quality (Coulson & Butterfield, 1978; Johnson & Damman, 1993). *Sphagnum* litter, woody material and the basal parts of *Carex* and *Eriophorum* species are much more decay resistant than the ordinary plant litter. Decay resistant litter also contributes a much greater share to the peat than would be expected based on the primary production values. The litter will also become more decay resistant as time goes on and the decay rate will decrease towards the bottom of the acrotelm (Clymo, 1984).

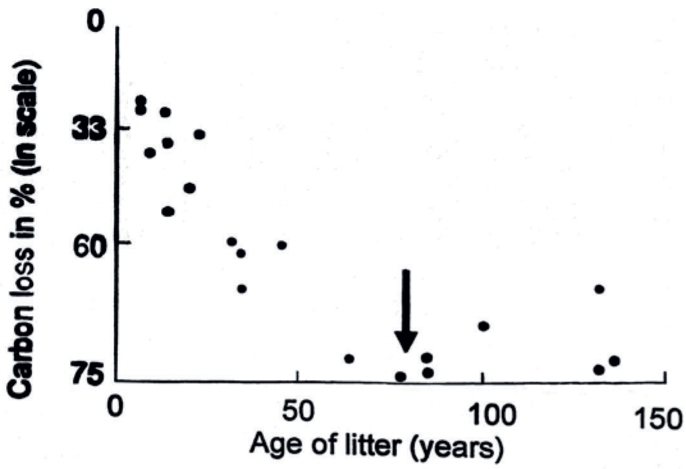


Figure 2: *An example of the percentage cumulative decay loss (ln-scale) with age (years) in the litter through the acrotelm and the uppermost part of the catotelm on an ombrotrophic site with a Sphagnum-dominated plant community. Samples (thickness 2.5 cm) combined from four cores (depth c. 0.3 m, moss layer excluded). The arrow indicates the approximate position of the limit between acrotelm and catotelm, although it is rather to be regarded as a transitional zone than a distinct level (Clymo, 1992). – From Malmer & Wallén, 2004, revised.*

The key processes that determine the rate of C input into the semi-permanent pool of the catotelm are thus the rate of litter formation by the vegetation and the decay losses in the acrotelm. The decay loss usually has a greater effect on the input rate than the rate of litter formation (Clymo, 1984; Belyea, 1996). Since both the litter formation and the decay losses are highly dependent on the plant species and environmental factors triggered by the climatic conditions, the C accumulation rates in a mire vary over time (Belyea & Malmer, 2004). In addition, because of the small but continuous C losses in the catotelm, the apparent C accumulation rate observed in a given peat layer is slightly less than the original input rate, and this difference increases with increasing peat age. An overall net C accumulation in a mire requires that the C input into the catotelm exceed the integrated C losses from the whole catotelm. Since the total release of C from the catotelm is proportional to the accumulated amount of peat, the C losses from the catotelm sooner or later will equal the annual input from the acrotelm and no further net carbon accumulation take place in the mire although the surface looks quite healthy (Clymo, 1984).

3.2 Rates of primary production, litter decay and carbon accumulation

In mires, primary production is traditionally measured by harvest methods. However, root production, although considerable, is normally neglected. Productivity estimates from such an incomplete method usually range from 50 to 700 g m^{-2} a^{-1} of dry matter (Moore, 2002; Malmer *et al.*, in press). For non-forested *Sphagnum* mires the productivity estimates range from 100 to 300 g m^{-2} a^{-1}. Annual estimates of the whole carbon sequestering using atmospheric flux measurements have been increasingly used in recent years and will be discussed in the next section. Since the productivity varies a lot between the small-scale microtopographical patterns found in most mires, extrapolations of the traditional small-scale estimates from m^2 to hectares and km^2 are often uncertain. However, for the C balance of a mire the absolute value of the plant productivity is usually of less interest than the type of litter produced since the proportion of decay resistant litter formed, e.g., by *Sphagnum* mosses has a greater effect on the C balance than the productivity of the site (Belyea 1996).

Experimental elevations of atmospheric CO_2 have usually resulted in enhanced productivity that subsequently has declined (Oechel *et al.,* 1994), probably because of a nutrient limitation in most mire ecosystems. *Sphagnum* mosses and most other plants in mires are well adapted to low nutrient conditions, e.g., nitrogen and phosphorus. However, high temperatures might increase the vascular plant productivity since the release of the plant mineral nutrients will increase and summer droughts might not have serious impacts on them (Malmer *et al.*, in press). On the other hand, summer droughts might hamper the growth of the *Sphagnum* mosses unless a longer vegetation period (temperature >0°C) can compensate for it. The recent increased anthropogenic nitrogen deposition also hampers the growth of the peat-forming *Sphagnum* mosses in heavily polluted regions and might facilitate an increased growth of vascular plants (Lütke Twenhöven, 1992; Gunnarsson & Rydin, 2000; Gunnarsson *et al.*, 2002; Malmer *et al.*, 2003). Therefore, a shift towards a more rapidly decaying litter may happen in the near future over large areas of Europe.

Peat formation at northern latitudes began after the end of the last glaciation about 10 000 years ago. The estimates of the global average apparent C accumulation rate in peat since then vary from 13 to 21 g m^2 a^{-1} (Clymo *et al.,* 1998; Robinson & Moore, 1999; 2000; Vitt *et al.,* 2000; Turunen *et al.,* 2001; 2002). This means an average increase of 0.07 Gt a^{-1} in the C storage in all northern mires which is equivalent to ~2% of the present global anthropogenic emissions. Studies on the apparent C accumulation rate in Finnish mires show that the average rates are decreasing with increasing latitude from 26 to 17 g m-^{2} a^{-1}, indicating a close relationship to the climate (Clymo et al. 1998, Turunen *et al.,* 2002). Ombrotrophic mires, which generally have more decay resistant litter**,** show greater C accumulation rates than minerotrophic mires, e.g., in southern Finland 29 g m^2 a^{-1} and 19 g m^2 a^{-1}, respectively, while the general differences seem to be less between continental aapa mires and more sub-oceanic mire types (Clymo *et al.*, 1998, Turunen *et al.* 2002).

The apparent C accumulation rate has also varied considerably over the time as a result of either changes in the peat forming vegetation, climate changes or autogenic processes associated with the rise of the peat surface (Barber, 1981; Malmer & Wallén 2004). In an ombrotrophic mires in southern Sweden, the apparent C accumulation has varied from 14 to 72 g m^{-2} a^{-1} (Malmer *et al.*, 1997). The increases in C accumulation rates were always rapid and contemporary with vegetation changes and lake water level rises of the region while the decreases were much slower and probably associated with the rise of the peat surface inducing an increasing runoff resulting in a deeper acrotelm and a longer residence time for the litter (Belyea & Malmer, 2004). At the end of the 19th century the apparent C accumulation rate was about 27 g m^{-2} a^{-1} but the recent changes in the peat forming vegetation have reduced its annual litter production by about 60%, partly because of nitrogen deposition (Malmer & Wallén, 1999; 2004). As a result, the future influx of C into the catotelm might be about 8 g m^{-2} a^{-1}. Since the release of C from the catotelm (depth ~5 m) may be about the same, this mire cannot be expected to act as a C sink any longer. Similar results have been obtained from a variety of other mires (Oechel *et al.,* 1993; 1995; Malmer & Wallén, 1996; Mäkila *et al.,* 2001; Klarqvist *et al.,* 2001). Both the present and future C balance of mires is therefore hardly possible to estimate from the historical peat deposits. Moreover, many European have already or may soon switch from sinks to sources of atmospheric C.

At present the residence time for the litter, at least on ombrotrophic sites, is between 60 and 120 years resulting in a total decay loss of 60–75% of the original organic matter before entering the catotelm (Malmer & Wallén, 1993). In a general perspective (Fig. 3), both the rate of litter formation and the decay loss in the acrotelm will decrease with increasing latitude since the decreasing temperature will reduce both the plant growth and the decay rate and result in low C accumulation rates at the high latitudes (Clymo *et al.,* 1998). At ombrotrophic sites, the rate of litter formation is significantly higher at oceanic sites compared to more continental ones, because of a higher productivity. However, the difference in C accumulation rate is much less since the peat rarely becomes frozen in the oceanic areas and the percentage of decay losses is greater than in more continental regions (Malmer & Wallén, 1993).

The observed changes in the mire vegetation alter the litter composition, higher temperatures increase the decay rates, and shorter periods of frozen peat may increase the release of C from the acrotelm and thus reduce the future C accumulation rate in mires (Moore, 2002; Malmer *et al.,* in press). Higher temperatures may also increase the slow release of the already accumulated C in the catotelm**,** particularly in mires with melting permafrost. These changes will tend to reduce the C accumulation rate and increase the emissions of CO_2 and CH_4 compared to earlier periods (Christensen *et al.*, 2004). There is, however, a possibility that a higher precipitation and an increased climatic humidity would increase *Sphagnum* productivity and shorten the residence time for the litter in the acrotelm, both changes that could be expected to reduce the total decay losses. That would to some extent counteract the effects of the vegetation changes and the higher temperatures.

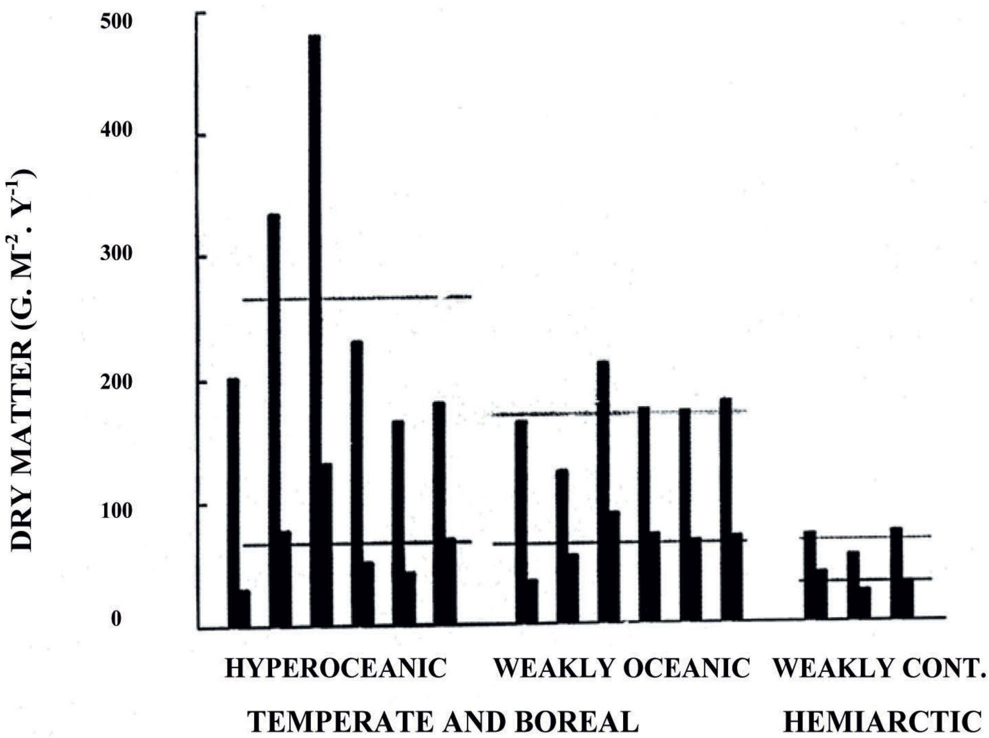


Figure 3: *The mass balance in hummocks at ombrotrophic sites with different climatic conditions. The difference between litter input to the acrotelm (left column in each pair) and the peat accumulation in the catotelm (right column in each pair) gives the decay losses in the acrotelm. The horizontal lines indicate the mean values for litter production and peat accumulation under hyperoceanic (left group), weakly oceanic (middle group) and weakly continental conditions (right group), respectively. From Malmer & Wallén, 1993, and unpublished.*

• 4. Differences in methodologies for assessment of fluxes and stocks

Two different methods are used to measure the gaseous fluxes of carbon between the surface and the atmosphere. The most commonly used is the chamber method, which has been applied for measurements of CO_2 and CH_4 for decades, and has been developed gradually over time. This type of measurement is performed by placing a chamber (steel/aluminium or transparent enclosure) on the soil surface and monitoring the exchange between the soil/vegetation and the atmosphere for a limited length of time (minutes). The change in concentration of the gas inside the chamber is hereby directly corresponding to the gas exchange from the surface on which the chamber is placed. Advances in this technique have made the sampling from chambers automatic, thereby allowing continuous measurement series of fluxes from a number of different gases (CO_2, CH_4, N_2O).
Overall the chamber method is excellent for providing instant flux measurements from smaller well defined surface areas, and can be scaled up to landscape coverage by operating replicated chambers for each of the ecosystem types present in the area of interest (e.g. Christensen et al., 2000). Chamber measurements are often the best choice where fluxes are small (dry environments or winter conditions) and when direct environmental response to the fluxes are of interest (e.g. treatment studies).

The other commonly used method for measurements of gas end energy exchange between surface and atmosphere is the eddy correlation (= eddy covariance) method. The method provides a non-intrusive and continuous measure of the flux at landscape scale level (hectares). Flux calculations are based on the principle of turbulent exchange in the lower atmospheric boundary layer, where fast response instruments are applied to measure fluctuations in vertical wind exchange and gas or energy concentrations (Moncrieff *et al.*, 1997). The method is widely used for CO_2 flux measurements but requires fast responding (1 Hz) instrumentation of high precision, which makes it relatively costly, especially for measurements of gases with low concentration in the atmosphere or where fluxes are small (e.g. CH_4 or N_2O). Overall the eddy correlation method is ideal when gas exchange budgets covering longer time periods (months or years) for fairly homogeneous surfaces are of interest. By not affecting the environment in which measurements are carried out, the fluxes obtained in this way are truly comparable and suitable for comparisons of e.g. carbon budgets or GHG effects (Friborg *et al.*, 2003).

Long-term apparent rate of C accumulation (LORCA) is calculated for easier comparison with measured flux rates. This value is derived from the total mass of stored C in the peatland of interest divided by the age (^{14}C dating) of the basal peat layer (Turunen *et al.*, 2002). A comparison between LORCA and present day flux data for a specific peatland often show that LORCA results in much smaller C uptake rates than what could be expected from the flux measurements. Several reasons can be found for this discrepancy: Most flux measurements only represent summer time where the C uptake rates are much higher that during off- season which is often characterized by emission. As only few wintertime measurements have been carried out over peatland, there is a risk that the importance of this part of the year is underestimated or ignored. As the LORCA data represent full year accumulation rates for longer time periods (100-1000 years) it is likely that the estimates do not show a good match. Further, CH_4 measurements are often absent in flux measurements which will also lead to an over-estimation of the accumulated C. Further, flux measurements only accounts for the exchange between surface and atmosphere and not the C export as DOC (dissolved organic carbon) to ground water, rivers and lakes, which also will lead to lower C accumulation rates (LORCA) than the actual gaseous C uptake as measured through the flux measurements.

• 5. Current annual rates of GHG exchanges in undisturbed mires

As mentioned several times, by definition peatlands are associated with an uptake of CO_2 from the atmosphere in the long-term time perspective. The formation of peat is a result of an accumulation rate of organic material (mainly vegetation), which is higher than the decomposition rate, mainly due to a high water table resulting in anaerobic conditions in the soil column. However, peatlands are formed over very long time periods (>1000 years), and may during especially drier and warmer climatic periods lose carbon as CO_2. Despite the fact that the overall pattern is a CO_2 uptake in peatlands it can be problematic to evaluate the actual present day exchange rate of these gases. A distinct feature of peatlands is an uptake of CO_2 and a corresponding emission of CH_4, especially during periods of peat accumulation. As both species are GHGs, the actual effect in terms of radiative properties on the atmosphere can only be determined by measuring the exchange of both gases simultaneously.

In this section we summarize the current knowledge on GHG exchange from natural peatlands in Europe. We adopt the atmospheric perspective on the exchange of these gases by evaluating the fluxes of CO_2 and CH_4 individually and also take into account the radiative properties of these gases. This is done by using the most commonly used global warming potential for CH_4, namely as being 21 times more powerful than CO_2 in the 100 year time horizon (IPCC, 2001).

The vast majority of undisturbed peatland in Europe are found in European Russia and in the Nordic countries (Sweden and Finland). Minor areas are found in Norway, Ireland, Scotland, England, Germany and Poland (Table 3 and Figure 4).
Year-round measurements of CO_2 exchange are available from Sweden and Finland and only for a few consecutive years. Methane exchange from peatlands have been measured in a limited number of studies and mainly during summer season which is problematic since the location of peatlands at high latitudes makes winter the longest season of the year, adding to the uncertainty of the annual estimate of the CH_4 emission.

In Table 4 available data on annual flux estimates are given for unmanaged wetlands grouped by country and functional type (positive numbers indicate emission). Although most of the available data originates from the countries holding the largest area of peatlands in Europe, a high degree of uncertainty exists both with respect to ecosystem functional types within these countries and with respect to European countries not covered by annual measurements. Even for Finland and Sweden, which are fairly well documented with respect to gas flux measurements, there is a high degree of uncertainty especially on the annual estimates of CH_4 fluxes and for a number of ecosystem functional types. In general, flux data for natural ecosystems (not least wetlands) are less robust than for managed ecosystems, e.g., forests. For peatlands only very few year-round data on CO_2 exchange exist globally (less than 10 sites) and none for CH_4 exchange. The data in Table 4 should be interpreted with caution, as most data originates from studies carried out over a minor part of the year, typically during summer, and the yearly fluxes are estimated or modelled for the remaining part of the year. Further, for those studies where annual fluxes originates from year-round measurements (marked with * in Table 4) measurements have typically been undertaken for less that five years and so the inter-annual variation may therefore not be revealed to the full extent.

Table 4: *The range and mean of annual and seasonal fluxes of CO_2, CH_4 and N_2O, summary column with conversion to CO_2 equivalents in (kg C ha^{-2} y^{-1}) using GWP100 (N_2O-N to CO_2-C equ: 133, CH_4-C to CO_2-C equ: 7.64). * Year-round measurements*

Country	Wetland type	Annual CO_2 (t C ha^{-1} y^{-1})		Annual CH_4 (kg C ha^{-2} y^{-1})		Annual N_2O (kg N ha^{-2} y^{-1})	GHG balance in CO_2 eqv. (kg C ha^{-2} y^{-1})	References
Sweden		Range	Mean	Range	Mean			
Arctic or sub-Arctic	Ombrotrophic (Nutrient poor, bogs)							
	Mixed (Aapa mire)	-0.14 – -0.27	-0.21*	23 – 38	30		-36 - 20	Friborg, unpublished
	Minerotrophic (Nutrient rich, fens)							
Boreal	Ombrotrophic (Nutrient poor, bogs)			20 – 73	37 (whole country)			Nilsson *et al.*, 2001
	Mixed (Aapa mire)			11 – 22	19 (whole country)			Nilsson *et al.*, 2001
	Minerotrophic (Nutrient rich, fens)	-0.49 – -0.53	-0.51*	30 – 304	90 (whole country)		-260 - 1793	Nilsson *et al.*, 2001
Finland								
Arctic or Sub-Arctic	Ombrotrophic (Nutrient poor, bogs)							
	Mixed (Aapa mire)	–	-0.19*	–	42		131	Aurela, 2002; Harding *et al.*, 2001
	Minerotrophic (Nutrient rich, fens)							
Boreal	Ombrotrophic (Nutrient poor, bogs)	-0.17 – -0.26	-0.20	23 – 225; 20 – 220	120; 100	0.000	-16 - 1459	Minkinnen, 2002; Joosten & Clark, 2002

Country	Wetland type	Annual CO_2		Annual CH_4		Annual N_2O	GHG balance in CO_2 equ.	References
Finland		**Range**	**Mean**	**Range**	**Mean**			
	Mixed (Aapa mire)							
	Minerotrophic (Nutrient rich, fens)	-0.17 – -0.25	-0.21	1 – 225; 135 – 480	75; 308	0.112	-147 - 3432	Minkinnen, 2002; Joosten & Clark, 2002
Eur. Russia								
Arctic or Sub-Arctic	Ombrotrophic (Nutrient poor, bogs)		0.03		12		62	Heikkinen *et al.*, 2002
	Mixed (Aapa mire)							
	Minerotrophic (Nutrient rich, fens)	-0.17 – -0.43	-0.32	12 – 120	60		-78 - 487	Heikkinen *et al.* 2002
Boreal	Ombrotrophic (Nutrient poor, bogs)							
	Mixed (Aapa mire)							
	Minerotrophic (Nutrient rich, fens)							
Germany	Ombrotrophic (Nutrient poor, bogs)		-0.71		174	-0.0112	618	Drösler *et al.* 200X
	Minerotrophic (Nutrient rich, fens)		-0.4	142 - 293	218		1266	Glatzel S. and Stahr, K, 2002; Joosten & Clark, 2002
UK	(Nutrient poor, bogs)			10 – 40				Joosten & Clark, 2002
	Minerotrophic (Nutrient rich, fens)							

The overall impression from Table 4 is that most wetlands in their natural state are sinks of CO_2 and carbon and sources of CH_4, whereas the N_2O exchange is limited regardless of functional type despite with large variations. With respect to the GHG effect on the atmosphere most functional types vary between a small sink and a moderate source of GHG, which can be related to a substantial CH_4 emission. None of the functional types listed in Table 4 show unambiguous uptake of GHG but the interpretation of the data depends on the time horizon adopted for the calculation of the GHG budget. The short lifetime of CH_4 in the atmosphere results in very different Global Warming Potentials for this gas depending on the time horizon considered. Here we have adopted the most commonly used 100-year time horizon from IPCC 2001.

• 6. Present extent of peatland management in the EU

6.1 The general trend

Peatlands are being managed for a variety of different reasons, and in most cases this causes a dramatic change in the ecosystem. Over the last century management has caused the loss of peatland area in many European countries, mainly to agriculture and forestry. The high organic content of peatlands in general makes these areas fertile and therefore well suited for food or fibre production, which is more profitable to society. Out of a total mire and peatland area of 617.000 km^2 in Europe, 52 % has been converted over the last century. Main areas of conversion include agriculture (50%), forestry (30%), peat extraction (10%) with the remains lost to urbanisation, inundation and erosion (Joosten & Clarke, 2002) Compared with other continents the relative proportion of peatland area converted in Europe is by far the largest. In some countries such as the Netherlands and Germany more than 80% of the peatlands have been converted to agriculture. In Finland more than half the natural peatlands have been converted to forestry. Peat production is important in certain countries but rarely exceeds a use of more than 1% of the total peatland area (only Ireland 8% and Germany 2% in EU-15). In absolute terms, Russia and Finland represent the countries where the largest area of original peatlands have been converted into other use over the last century, but these are also the two countries holding the largest total area.

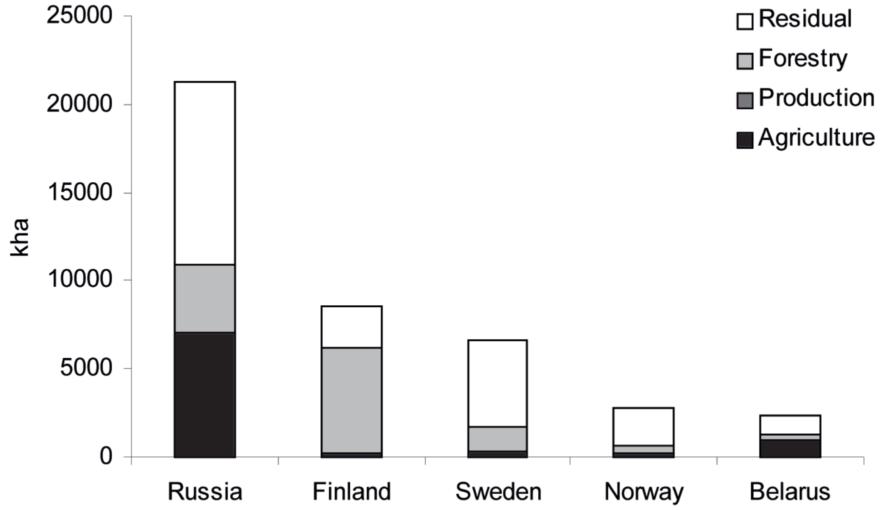
Figure 4 summarises data on the use of peatlands in Europe (including European Russia). The total area is from the 2002 estimates also quoted in Table 3 and the use terms are divided into the major three categories, i.e. agriculture, forestry and peat production. The residual include virgin peatlands.

Common utilisation and products of managed peatlands:

Agriculture: European peatlands converted into agriculture are most commonly utilized as meadows and pasture for grazing of cattle and sheep. In general a lowering of the ground water table to 0.4-0.8 meters below surface is required for peatlands to be used as meadows and pasture and 1.0 to 1.2 meters for arable land (Joosten *et al.* 2002).

Forestry: The largest areas of European forest production on peatlands are found in Finland, Russia, Sweden, Norway and the Baltic countries. Most commonly forest products from peatlands or drained peatlands are used in paper production with only a smaller proportion used for furniture and construction materials. The economically profitable forestry production often requires drainage.

Peat extraction: In Europe the most important peat resources are found in Russia, Finland, Sweden, Norway, Iceland, Ireland, Estonia and Latvia (please refer to table 3). Since the middle of last century peat production for energy use has generally decreased but is still an important energy source mainly in rural districts of Finland, Ireland, Sweden, The Baltic states and Russia. Peat extraction used for energy is still the main use in Finland (6.8 million tonnes), Russia (2.8 million tonnes) (www.worldenergy.org) and Ireland (4.7 million tonnes) (Joosten *et al.* 2002). In most other European countries peat extraction is associated with a variety of other products of which organic fertilizer in agriculture and substrate in horticulture (sphagnum) are the most important. Peat products are also used in chemical and medical/cosmetic industry and as insulation material in housing.



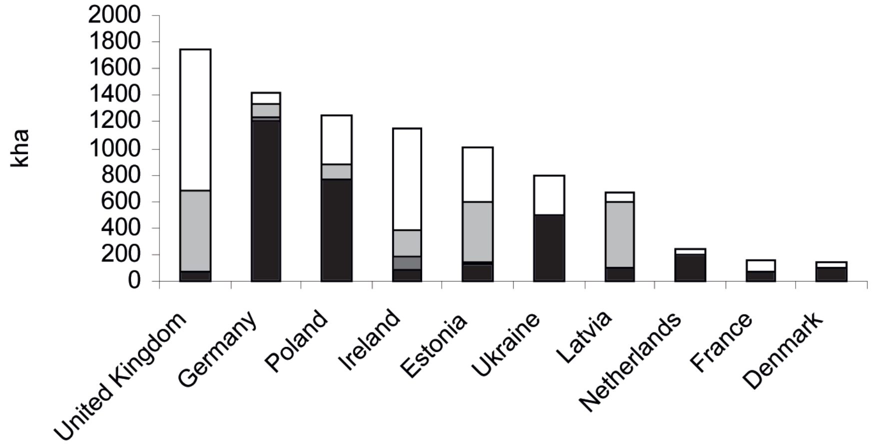


Figure 4: *Reported peatland area as in Table 3 broken into published numbers of areal use of peatlands and the residual for individual European countries with >100 kha of peatlands (Joosten and Clarke, 2002; The International Peat Society). The residual may either be virgin peatlands or under other types of use than the three listed (which are the major ones). Certain countries are excluded from this figure due to inconsistency in data sources (e.g. reported used land exceeding total peatland area).*

Natural conservation areas:

Most management practises including agriculture, intensive forestry, and peat extraction require trenching and lowering of the peatland water table, which causes dramatic changes of the peatland ecosystem (floristic composition, micro-climate, hydrology, etc.). Over the last decades there has been a rising concern over the changes in environment that drainage and other land-use changes cause in peatland ecosystems. Wetlands are in general important habitats for wide range of wildlife not least birds, and in order to preserve the unique biotopes of the peatlands a range of natural conservation and restoration actions have been initiated over the last years.

As a result of these concerns a number of international conventions and regulations for wetland use and conservation have been adopted by most European countries. An overview of the international conventions, which are most important for peatland management is given at the end of this section. One of the most prominent such conventions is the RAMSAR convention which was formed in 1971, in order to protect wetland areas worldwide. In Europe there are at present(2004) 780 protected sites comprising approximately 20% of Western European and approximately 6% of Eastern European wetland areas, which are protected under the RAMSAR convention (Finlayson *et al.*, 1999. The RAMSAR convention is mostly focused on protection of present day wetlands as habitats for waterfowl, but a number of national and international conservation programmes exists for a wider range of peatlands. However, no overview of the areal extent of such programmes in Europe exists at present.

The consequences of management schemes in relation to GHG budgets where all relevant gases are evaluated is likewise highly uncertain, but here in section 7 and 8 typical changes in land-use practices are related to GHG emission factors.

6.2 Management and sustainable use of peatlands in EU as influenced by environmental and economic policy

During the last few centuries many peatlands in Europe have been converted to uses from which economic benefit can be derived such as agriculture, forestry and peat production for energy and horticulture. However the conservation value of peatlands is increasingly recognised and many European countries have conservation measures in place. There are good reasons to promote the conservation of peatlands, these include:

1. Peatlands are important for landscape level processes such as catchment level water balance.
2. Peatlands are important habitats and are home to organisms, which can be specifically adapted, and confined, to them.
3. Peatlands are archives of climate and vegetation change, which scientists can use to reconstruct past environmental and climatic changes.

There are a number of international conventions and agreements concerning nature conservation and environmental protection, many of which have relevance to peatland conservation.

The RAMSAR Convention on Wetlands (1971) is an intergovernmental treaty, which provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources. There are presently 138 Contracting Parties to the Convention, with 1317 wetland sites, totalling 111 million hectares, designated for inclusion in the Ramsar List of Wetlands of International Importance.

The United Nations Framework Convention on Climate Change (UNFCCC) was agreed in 1992 and signatories are encouraged to conserve and enhance sinks of greenhouse gases. The Kyoto Protocol to the UNFCCC allows for carbon sequestration in vegetation and soils in certain land use categories to be used to offset greenhouse gas emissions.

Agenda 21 (1992) provides a blueprint for sustainable development and encourages the careful use of non-renewable resources. Peatland may be classified as such. The concept of sustainable development was incorporated into European law through the Maastricht Treaty (1992) and later made one of the overriding objectives of the European Union through the Amsterdam Treaty (1997).

At EU level there are a number of schemes, which influence peatland management and conservation. The Birds Directive (1979) is concerned with the long-term protection and management of bird species. The Habitats Directive (1992) is the main EU instrument for safeguarding biodiversity. Member states are responsible for identifying and designating as Special Areas of Conservation, sites which are important for the protection of species and habitats covered by the directive. The Directive specifically mentions peatland habitats. All areas under these directives make up the Natura 2000 European network of sites. The EU is also developing a thematic strategy for soil protection. The long-term aim is to develop a legislative base for soil monitoring in order to provide the information necessary to underpin soil protection. The EIA Directive (1995) requires that the environmental effects of projects be assessed. This covers development on peatland such as afforestation, peat extraction and agricultural development.

In 1995, the European Commission adopted a Communication COM (95) 189 on the 'wise use and conservation of wetlands'. This was also included in the Fifth Environmental Action Plan. Conservation projects in peatlands have received financial support through the LIFE programme. Peatland restoration projects have also received support; for example, in Ireland a number of peatlands drained for forestry are being restored to functioning peatland ecosystems. The cessation of peat harvesting is also likely to provide areas for peatland restoration.

Within the EU 15 there is almost no drainage of pristine peatlands for conversion to other uses. Peat mining is unlikely to expand onto pristine areas. For instance, in Finland new peat mining areas need to be previously drained for forestry or agriculture. In Ireland, no new peat mining areas will be developed. Changes to the Common Agricultural Policy have stimulated a move towards more environmentally friendly agricultural practices as well as efforts to reduce agricultural production. This has reduced the pressure to drain peatland for agriculture. This is supported by national schemes in many member states.

Peatland restoration is likely to increase in the future as peat extraction ceases in many areas and as unproductive forest drainage areas are restored.

Future changes in energy policy are likely to impact on peat use for energy. This particularly applies to carbon taxes. If peat is subject to higher taxes than other energy sources this may make peat less competitive. Peat is an important strategic energy reserve in countries, which are largely dependent on imported fossil fuels for energy production. These countries, such as Ireland and Finland, may continue to support the peat industry in order to maintain this secure energy supply.

• 7. Managed peatlands

Peatlands offer a wide set of ecological functions to the society (Joosten & Clark, 2002), which include e.g. the regulation function for the local or regional water balance or the production function for human welfare. Huge areas of European peatlands are managed for optimization of the production function, namely for agriculture, forestry or the peat industry (section 6). This section focuses on the effect of these management practices on the processes of carbon- and GHG-exchange in peatlands, and the consequences for the climatic relevance of managed peatlands.

7.1 Influence of management practices on peatland functioning

Peatlands are semi-terrestrial systems the natural functioning of which depends on the interaction between water level, nutrient status and plant composition at the soil surface. The balance between the rate of C–fixation and decomposition determines the C-accumulation rate (section 2). This rate together with time (hundreds to thousands of years) determines the amount of peat deposited and hence the actual carbon store in the peatlands. Management practices are distinctively influencing these interacting compounds (water level, nutrient status and plant composition) with consequences on the GHG-exchange and subsequent on the carbon store (see qualitative characterisation of these impacts in Table 5).

7.2 Consequences for the GHG exchange

7.2.1 *Effects of drainage*

The aeration of the upper peat layers enhances aerobic decomposition processes, which provoke higher CO_2 respiration fluxes (mainly microbial respiration). The CO_2 fixation rate of the active plant layer may gradually reduce, because of water limitation. Depending on the balance between higher respiration and lowered fixation, accumulation rates gradually decrease and finally the ecosystem switches from a carbon sink to a carbon source.

The main methane production zone is the anaerobic peat layer 20-50 cm below the water table. If the water table drops, this zone follows downwards. A gradually decreasing CH_4 production rate may be the consequence, as old organic matter is more resistant to decomposition processes. But the notably reduced emission of methane from the peatland after drainage is mainly a product of accelerated oxidation of produced CH_4 during transport through the thicker aerobic peat layer.

The rate of N_2O production is driven by nitrification and denitrification processes. This complex balance and interaction is not easy to determine on site. However, drainage generally leads to higher N_2O emissions, if nutrients do not limit N_2O production. For details on processes please refer to section 3.

Table 5: *Qualitative characterisation of the influence of different management practices on the principal compounds of peatland functioning.*

	Water level	Plant composition on the surface	Nutrients	Peat deposit – C-store
Preparation for forest introduction and peat-cutting	moderate to severe drainage	normally still untouched but changing due to changed water level	mineralization - but no input of fertilizers	moderate loss
Forest	moderate to severe drainage	introduction of tree species, partly removal of original vegetation	mineralization - only limited input of fertilizer (Liming?)	Moderate to severe change – C store shifts from Peat (loss) to Biomass (gain)
Grassland	severe drainage	original vegetation removed, introduction of productive grass-species	nutrient input as fertilizer at moderate to high levels	fast loss
Arable	severe drainage	original vegetation removed, introduction of productive field species	nutrient input as fertilizer at high levels	Very fast loss, because of enhanced aeration of the peat via ploughing. Fertilisation may also accelerate decomposition
Peat cut	severe drainage	original vegetation totally removed	mineralization - but no input of fertilizer	peat removed -immediate loss
Abandonment after peat cut	drainage remains active over long periods	succession of mostly woody species, which are tolerant to water stress	mineralization - but no input of fertilizer	constant loss at moderate to fast rates
Restoration	rewetting or even flooding	succession or introduction of matrix species, establishment of original vegetation needs time	aerobic mineralization limited or stopped depending on water level	loss reduced to stopped – new built-up depends on species, water level and nutrients – and on time

7.2.2 *Effects of changing plant composition*

During vegetation succession following a lowered water table, species disappear which normally comprise the active CO_2-fixation layer for peat formation (mosses and sedges). The composition shifts to species, which prefer drier stand conditions (grasses, woody species). But generally in the beginning, net CO_2-fixation at community level may decrease, as decomposition processes are dominating the fluxes. Of course, total removal of the vegetation layer (as in peat harvesting), takes off the growing layer for the peatland, stopping any CO_2-fixation process. But agricultural and even grassland species, introduced to the drained peat soil, do not contribute as well to C fixation: they are grown normally with low water table and the production of roots and litter (as remaining C pool after harvesting or grazing) is not able to compensate for soil respiration CO_2-emissions. CO_2-exchange of forests established on peatlands is more complex: CO_2-fixation rates are significantly higher than those of undisturbed peatland vegetation. If this fixation rate is higher than the rising C-loss via decomposition processes on drained forest soils, the newly established forests may even lead to slight net sinks for CO_2. But here, the timescale is very important, with forests in temperate and boreal climate reaching normally C-equilibrium in around 100 years. Peatlands on the other hand can accumulate carbon over several thousands of years (see section 3).

As for CH_4 emissions, the major impact of changing plant composition is the percentage of species, which have aerenchyma and act therefore as bypass for oxidation processes. These species also have a whole range of other impacts on potential CH_4 emissions. Sedges (like *Eriophorum vaginatum*) normally disappear during drainage, but can rapidly re-appear, when restoration is done via rewetting (not flooding) of bare peat soils.

Nitrous oxide emissions are not known to react directly to a specific plant composition, though vegetation cover can be used as indicator for sites with potential N_2O-emissions.

7.2.3 *Nutrient effects*

Higher nutrient availability together with drained soil is expected to accelerate decomposition processes, leading to higher CO_2-respiration fluxes. On the other hand, higher nutrient availability may lead to a higher productivity of the (changed) plant cover (especially in forests). So for CO_2 exchange, negative effects are foreseen for all but forested sites, which may react differently. Effects on CH_4 exchange are not expected. N_2O may be the gas which reacts more closely to nutrient levels. In nutrient poor bogs, the potential N_2O-production is limited, even under drained conditions. In nutrient rich fens, the potential for N_2O-emissions is much higher, and is additionally forced by fertilizer applications to these productive systems.

7.2.4 *Emission factors and Global Warming Potential*

The impacts on the processes of GHG production and exchange described above should lead to differentiated emission factors (fluxes) from bogs (nutrient poor) and fens (nutrient rich) underlying different management practices. A review of published flux measurements of CO_2, CH_4 and N_2O on European peatlands gives a detailed picture of best estimates of expected emission factors for the three gases (Table 6 in Section 8). Summing up and multiplying the emission factors individually with their GWP values (100-yr timescale) gives an estimate for the overall climate warming effect of the management types. As shown above, the balance for forests should be interpreted with caution, as timescales may change accumulation rates.

It is clear that data are unevenly distributed, both in geographical representation as in covered types. We are still lacking CO_2-exchange data for restored fens, while we have a notable amount of studies on N_2O-exchange on fen grasslands.

• 8. European peatland GHG budget

Peatland GHG fluxes are governed by a much more complex interplay of external and autochthonous drivers than GHG fluxes in mineral soils. This leads to high site-to-site and within-site variability and renders the generalisation of measurements difficult. Peat type (nutrient status), disturbance (drainage, peat cut) and climate are key parameters for estimation and understanding of peatland GHG fluxes. For the purpose of a peatland GHG budget, we distinguish therefore between:

Climate regions: sub-arctic – boreal – temperate

Peatland type:
- "bog": ombrotrophic and oligotrophic peatlands, including all subarctic peatlands such as aapa mires or flarks. This differs from the usual "bog" definition, but turned out as most appropriate for the characterization of emission factors;
- "fen": minerotrophic, mesotrophic and eutrophic peatlands in boreal and temperate climate.

Land use:
- mire (undisturbed)
- forestry (drained)
- newly drained for forestry or peat cut
- grassland (drained)
- cropland (drained)
- peat cut (drained)
- abandoned peat cut
- restoration.

8.1 Area estimates

Literature provides three different estimates of total peatland areas at the national level (Joosten and Clarke (2002), www.worldenergy.org, Appendix II). As the reason for the differences is unclear, we used the areas of Joosten and Clarke (2002), which turned out most consistent with regard to used areas, mires and totals. Information about the extent of fens versus bogs is often available (Joosten and Clarke, 2002), but not attributed to climate zones nor to land use types. Therefore, assumptions based on indications in Lappalainen (1996) and other literature and expert judgement were necessary. The classification of peat type/use areas is further complicated by ambiguities in the literature. In many cases it is unclear whether "agricultural use" means cropland or cropland plus drained grassland. In the category "forestry", we assume that all forest is drained. This may not be always true. However, the emission factors for forestry assume a very moderate drainage only. On the other hand, drainage is repeatedly done, and the extent of recently drained as opposed to older drained areas in unknown. "Peat cut" areas vary considerably in literature. There is some ambiguity what types of areas are included in the estimates because some areas are drained and prepared for cut, but not in present use, other abandoned lands after peat cut still may show the effects of drainage. Joosten and Clarke (2002) and Worldenergy seem to include to some extent areas drained for cutting and after cutting, while Selin (1999; http://www.peatsociety.fi/) seems to report the active cutting areas only. The status of the land not in active cutting is unclear. There is too little area data to quantify the extent of the most recent land use "restoration". Normally, the allocation of land uses to the total peatland area left a residual area (within an error range of <9%), which could not be allocated, to any specific use. This residual could be used, as undrained land, e.g. for rough sheep grazing, or abandoned cut-over land.

The major uncertainty in the estimates originate from the attribution of land uses to peat types, and the distribution of agricultural use between croplands and drained grasslands. Our detailed but rough estimates are given in Appendix II. Without geo-referenced information about peat types and areas it is impossible to incorporate climate information in the GHG budget calculation.

8.2 Emission factors

Summarizing the data shown in Table 4, we can derive emission factors for European undisturbed mires. We use a conservative estimate by taking the median of the measured data. The probability density function of the data is often positively skewed. The mean would therefore overestimate emissions due to few extremely high data points. The median is also more robust than the mean when new data are included (Table 6). Despite the small number of studies, which often cover only some of the GHGs, the uncertainties and the large variation in flux data, the emission factors appear to be relatively robust and consistent with expected patterns in dependence of drainage status and CH_4 emissions. Summing up the emission factors as C-equivalents per hectare, peatlands generally turn out as source of GHGs. Emission densities increase in the order:

- bog: forestry < mire < restoration < new drainage for forest/peat cut < peat cut < abandoned after harvest = grass < crop
- fen: (restoration <) forestry <= mire < new drainage for forest < grass < crop

The fact that forestry peatlands emit less GHGs than undisturbed mires has to be viewed with caution. The underlying studies suggest a mild drainage only, at which CH_4 emissions are reduced but peat formation still goes on. This is an optimal scenario, which disregards the fact that drainage is repeated periodically. Furthermore emissions vary with stand age as transpiration by trees affects the water table of the peat in a way that can range from high CH_4 emissions under young stands to significant drainage and CO_2 emissions under older stands. The studies therefore show only snapshots rather than equilibrium emissions.

There are no data on CO_2 fluxes in restored fens, so it is unclear whether and when these areas turn into peat-forming lands again.

The most robust emission factors based on the largest data sets are as follows:

- bog: new drainage for forest/peat cut, forest (snapshots in some age classes), CH_4/CO_2 mire, CO_2 peat cut.
- fen: grass, forest, new drainage for forest, N_2O crop.

There is particular need for more data about

- bog: grassland, cropland, land abandoned after peat cut, restoration, forest chronosequences, N_2O fluxes in general.
- fen: abandoned after harvest, restoration, CO_2 fluxes in general.

Table 6: *Emission factors based on measured fluxes from different bog and fen management types from European peatlands (median. Range and number of data sets n are given in brackets.).*

Managed - types of peatlands	**Emission factor**			**GWP 100 (CO_2-C Equivalents kg ha^{-1} yr^{-1})**
	CO_2 (t C ha^{-1} yr^{-1})	**CH_4 (kg C ha^{-1} yr^{-1})**	**N_2O (kg N ha^{-1} yr^{-1})**	
BOG (ombrotrophic)				
Forest (drained)	-0.19 [1,17,18,26] (-1.85 – 2.26, n=8)	11.15 [1,18,25,26,29] (-0.10 – 26.9, n=14)	0.04 [1,18,22,26] (0 – 1.10, n=7)	-105*
Drainage (for forest, peat cut)	1.10 [7,18,26,30] (-0.81 – 2.46, n=6)	20 [6,18,23,25,26,29] (0.02 – 78.9, n=9)	0.04 [6,18,23,26] (-0.01 – 0.60, n=16)	1253
Grassland	2.35 [10,11,21,31] (1.50 – 3.50, n=5)	2 [11,13] (2, n=2)	0.01 [11,13] (0 – 0.02, n=2)	2367
Arable	4.4 [11] (n=1)	0 [11] (n=1)	0 [11] (n=1)	4400
Peat cut**	1.75 [9,11,28,32] (0.63 – 13.3, n=11)	17.25 [28,32] (2.25 – 33.7, n=9)	0.4 [28] (n=1)	1930
Abandoned after harvested	2.35 [7,33] (0.41 – 4.43, n=4)	1.3 [6,33] (0.30 – 3.98, n=4)	1.09 [6,33] (0.5 – 1.69, n=2)	2501
Restoration	0.62 [7,33] (-0.65 – 1.92, n=5)	15 [6,33] (2.25 – 71, n=5)	0.02 [6,16] (-0.01 – 4.45, n=4)	736
FEN (minerotrophic)				
Forest (drained)	-0.2 [26,39] (-0.21 – 0.48, n=4)	-0.05 [1,2,3,14,15,20,26] (-1.8 – 44.5, n=13)	1.83 [1,3,14,15,20,21,22,26,30] (0.09 – 26.9, n=20)	42*
Drainage (for forest)	0.4 [18,23,26] (0.14 – 3.6, n=4)	1.0 [18,23,25,26,29] (0 – 11.3, n=7)	1.05 [18,23,26,30] (0.08 – 2.23, n=10)	547
Grassland	4.12 [19,24,27] (0.82 – 6.58, n=5)	0.4 [3,8,19,24,28,34] (-1.04 – 105, n=13)	5.05 [2,3,4,5,12,14,21,22,24,28,36,37] (0.30 – 38.8, n=60)	4794
Arable	4.09 [13,21,28] (1.09 –10.6, n=3)	-0.2 [8] (-0.20, n=2)	11.61 [5,12,14,21,30,38] (4.0 – 56.4, n=15)	5634
Restoration	-	12.4 [3] (6.5 – 18.3, n=2)	0.64 [16] (n=1)	179 (without CO_2-C)

Table 6: *continuation*

Natural - types of peatlands	Emission factor CO_2 (t C ha^{-1} yr^{-1})	CH_4 (kg C ha^{-1} yr^{-1})	N_2O (kg N ha^{-1} yr^{-1})	GWP 100 (CO_2-C Equivalents kg ha^{-1} yr^{-1})
BOG (ombrotrophic)				
All climate zones	-0.20 (-1.57 - 0.14, n=12)	57 (12 – 382, n=16)	-0.02 (-0.02 – 0, n=3)	192
Boreal/sub-arctic	-0.20 (-0.32 – 0.03, n=6)	37.5 (12 – 217.5, n=10)	0.00	88
Temperate	-0.71 (-1.57 – 0.14, n=6)	174 (54 – 382, n=6)	-0.0112 (-0.02 – 0, n=2)	615
FEN (minerotrophic)				
All climate zones	-0.44 (-0.53 – -0.21, n=4)	131 (60 – 150, n=4)	0.05 (-0.01 – 0.112, n=2)	559
Boreal/sub-arctic	-0.49 (-0.53 - -0.21, n=3)	120 (60 – 150, n=3)	0.112	427
Temperate	-0.40 (n=1)	142 (n=1)	ND	691

* Should be interpreted with caution due to much earlier equilibrium of C exchange in forests (~100a) than in peatlands (thousands of years). ** Harvested peat is not taken into account.

[1]Alm *et al.* (1999); [2] Augustin *et al.* (1996); [3] Augustin *et al.* (1998 a); [4] Augustin *et al.* (1998 b); [5] Dörsch *et al.* (in prep); [6] Drösler *et al.* (in prep.); [7] Drösler *et al.* (submitted); [8] Flessa *et al.* (1998); [9] Glatzel *et al.* (2003); [10] Hillebrand (1993); [11] Höper & Blankenburg (2000); [12] Jaakkola (1985); [13] Joosten & Clark (2002); [14] Kasimir Klemedtsson (1997); [15] Klemedtsson (1997); [16] Komulainen *et al.* (1998); [17] Laine & Minkkinen (1996); [18] Laine *et al.* (1996); [19] Langenveld *et al.* (1997); [20] Lustra (2002); [21] Maljanen *et al.* (2001); [22] Martikainen *et al.* (1993); [23] Martikainen (1995); [24] Meyer (1999); [25] Minkkinen *et al.* (2002); [26] Mundel (1976); [28] Nykänen *et al.* (1995); [29] Nykänen *et al.* (1998); [30] Regina *et al.* (1996); [31] Silvola (1986); [32] Sundh *et al.* (2000); [33] Tuittila (2000); [34] Van den Pol-van Dasselaar (1999); [35] Velthof & Oenema (1993); [36] Velthof & Oenema (1995); [37] Velthof *et al.* (1996); [38] Weslien *et al.* (in prep.); [39] (Augustin 2001).

8.3 European peatland GHG budget

Table 7 shows the conservative default estimate of the European peatland GHG budget. The calculation does not distinguish between climate zones and uses uniform emission factors per land use type as follows: Mire, residual according to “Natural” in Table 6, and forestry as “forest, drained”, grassland, cropland, peat cut found under “Managed in Table 6. Due to lack of adequate area statistics, recent drainage for forestry or peat cut, abandoned peat cut, and restoration could not be calculated.

Table 7: *Conservative default estimate of the European peatland GHG budget in Gg CO_2-C equivalents (C-equiv.) assuming a 100-year time horizon.*

	Total Gg CO2-C	Total Gg CH4-C	Total Gg N2O-N	Total Gg C-equ.	Mire Gg C-equ.	Forestry Gg C-equ.	Crop Gg C-equ.	Grass Gg C-equ.	Peat cut Gg C-equ.	Residual Gg C-equ.
Albania	71	0.05	0.20	99	0.23	0.00	99	0.00	0.00	0.00
Andorra	1.1	0.03	0.00	1.8	0.11	0.00	1.7	0.00	0.00	0.00
Austria	-6.8	1.9	0.00	7.6	3.8	0.00	0.00	0.00	0.00	3.8
Azores	-0.04	0.01	0.00	0.06	0.06	0.00	0.00	0.00	0.00	0.00
Belarus	3494	118	6.0	5198	484	12	542	4154	6.6	0.00
Belgium	41	0.05	0.00	41	0.06	0.00	7.0	34	0.00	0.00
Bosnia and Herzegovina	57	0.13	0.16	79	0.56	0.00	79	0.00	0.00	0.00
Bulgaria	-0.60	0.14	0.00	0.49	0.10	0.00	0.00	0.00	0.00	0.39
Croatia	-0.04	0.01	0.00	0.06	0.06	0.00	0.00	0.00	0.00	0.00
Czech Republic	28	0.63	0.04	38	1.4	0.00	17	17	1.9	0.83
Denmark	404	2.5	0.50	490	1.9	0.00	0.00	479	1.9	6.6
Estonia	325	40	1.2	782	128	-7.6	0.00	623	17	21
Faroe Islands	-0.72	0.17	0.00	0.58	0.49	0.00	0.00	0.00	0.00	0.10
Finland	-971	209	7.3	1592	621	-94	0.00	959	106	0.00
France	250	5.4	0.77	393	1.9	0.00	372	0.00	1.9	18
FYRO Macedonia	10	0.07	0.03	14	0.28	0.00	14	0.00	0.00	0.00
Germany	5083	8.6	8.1	6230	28	2.5	2564	3577	58	1.1
Greece	23	0.17	0.07	33	0.73	0.00	33	0.00	0.00	0.00
Hungary	19	3.4	0.06	52	1.7	0.00	14	24	0.00	13
Iceland	238	88	0.69	1001	197	0.00	0.00	623	0.00	180
Ireland	622	41	1.1	1075	42	-22	505	293	174	83
Italy	109	0.39	0.31	154	1.7	0.00	152	0.00	0.00	0.00
Latvia	236	46	1.2	741	175	-6.3	563	0.00	3.9	5.3
Liechtenstein	-0.02	0.01	0.00	0.02	0.02	0.00	0.00	0.00	0.00	0.00
Lithuania	431	19	1.3	740	36	0.68	563	94	3.9	42
Luxembourg	-0.07	0.02	0.00	0.06	0.02	0.00	0.00	0.00	0.00	0.04
Moldova	-0.44	0.13	0.00	0.56	0.06	0.00	0.00	0.00	0.00	0.51
Netherlands	808	4.7	1.0	978	8.5	0.00	0.00	959	0.00	11
Norway	1773	126	-0.04	2729	427	0.00	1848	450	3.9	0.00
Poland	2943	44	4.0	3812	107	0.18	31	3595	1.9	77
Portugal	7.3	0.03	0.02	10	0.11	0.00	10	0.00	0.00	0.00
Romania	-2.9	0.83	0.00	3.5	1.7	0.00	0.00	0.00	0.00	1.7
Russia European part	4930	1564	15	18904	5584	-149	1487	11388	20	574
Slovakia	3.4	0.17	0.01	5.6	0.62	0.00	1.7	3.2	0.00	0.10
Slovenia	-4.4	1.3	0.00	5.6	0.56	0.00	0.00	0.00	0.00	5.1
Spain	7.8	0.49	0.03	15	0.56	0.00	13	0.00	0.00	1.5
Sweden	-859	525	4.9	3804	2137	-44	1690	0.00	21	0.00
Switzerland	33	2.5	0.06	60	11	0.00	11	38	0.00	0.00
Ukraine	1016	62	3.5	1957	265	0.00	1690	0.00	1.6	0.00
United Kingdom	-135	73	0.34	468	44	-55	206	73	7.7	193
Yugoslavia	100	0.65	0.29	144	2.8	0.00	141	0.00	0.00	0.00
Total	**21081**	**2988**	**58**	**51660**	**10316**	**-362**	**12653**	**27384**	**431**	**1239**
EU-15	**5383**	**872**	**24**	**15291**	**2891**	**-212**	**5550**	**6375**	**371**	**317**
EU-25	**9363**	**1025**	**32**	**21468**	**3342**	**-225**	**6740**	**10730**	**400**	**482**

The CO_2 fluxes in Table 7 result from sinks in some peatlands and sources in others. In contrast, for CH_4 and N_2O, only the sources are relevant. Whilst some countries turn out as CO_2 sinks, all countries are net GHG emitters from peatlands. The results in Table 7 are associated with large uncertainties, which cannot be adequately quantified yet. One outstanding uncertainty, however, is the distribution of land use types, particularly in Russia, the largest European peat nation. The results indicate some interesting features. Russia hosts 41% of European peatlands and contributes most to all GHGs (CO_2: 25%, CH_4: 52%, N_2O: 26%, Total: 37%). Germany turns out as the second-largest emitter (12% of European total) although it contains only 3.2% of the European peatlands. The reason is the intensive cropland and grassland use of most of the peatland area. The largest CO_2 emitters are countries with large agricultural peatland areas (Russia, Germany, Belarus, Poland); the largest N_2O emitters are those with large agricultural fen areas (Russia, Germany, Finland). In contrast, the largest CH_4 emitters are concentrated in regions with intact mires of Russia and Scandinavia. High average emission densities above 3.5 t C-equiv. ha^{-1} are found in the Southeast Mediterranean, Germany and the Netherlands where agricultural use is intense. Low average emission densities below 0.3 t C-equiv. ha^{-1} occur where mires and bog forests dominate, e.g. Finland and the UK.

8.4 Sensitivity tests

We performed sensitivity tests to quantify the effects of the most important uncertainties in the estimates. The default assumptions were varied as follows:

- Forestry: the default emission factors indicate that the peat forests have been drained only slightly, if at all. However, the area data indicate the areas of drained forest where drainage is likely to be repeated periodically. Therefore, we compare the default result with the emission factor for "peat forestry" with the emission factor "peat drained for forestry" (Table 6). The reality will lie in between these extremes.
- "Residual": The actual status of residual peatland is unclear. We assumed conservatively in the default calculation that the residual is undrained and behaves like a "mire". Alternatively, the "residual" could indicate abandoned land, so we use the emission factors for "peat abandoned after harvest" (bog) and "peat drained for forestry" (fen).
- Cropland and grassland: The default represents the best possible estimate of area distribution between cropland and grassland. In two extreme scenarios, we assume 1) 100% cropland, and 2) 100% grassland.

Table 8 displays the results of the sensitivity analyses for the major emitters and three countries with high sensitivity to the variation in assumptions. Obviously, CO_2 is most sensitive to changes in assumptions, followed by N_2O in countries with large agricultural fen areas. Changes in the assumptions for forestry and residual peatland can in some cases convert the national GHG budget from a sink to a source. Clearly, effects in the sensitivity tests exaggerate the uncertainty in the estimates because for many countries considered the data base of areas and land use is better than assumed here. Countries will be somewhere in-between the default and the variation in the sensitivity analyses, depending on drainage status and whether fens or bogs are dominant. The total uncertainty in the GHG budget of Russia and UK is exaggerated because of the large residual, which is likely to be undrained as in the default estimate. Similarly, forests in Finland are likely to be sinks of C as in the default estimate. Russia is the largest GHG emitter and contributes most to overall uncertainty of the European GHG budget.

Table 8: *Sensitivity analyses: variation in % of original value*

	Maximum variation in CO2 (%)	Maximum variation in CH4 (%)	Maximum variation in N2O (%)	Maximum variation in C-equ. (%)	Forestry C-equ. (%)	Agriculture C-equ. (%)	Residual C-equ. (%)
Russia European part	139	-9	118	43	1303	-2 - 15	735
Germany	12	-4	50	18	5966	-8 – 16	486
Belarus	4	0	91	17	1200	-2 – 15	NA
Sweden	155	1	-11	35	2996	-15 – 0	NA
Poland	10	-41	124	22	53014	0 – 18	101
Finland	514	11	-18	314	5141	0 – 18	NA
Netherlands	1	-58	132	17	NA	0 – 18	-3
United Kingdom	2450	-69	310	654	1398	-21 – 23	1154
Total Europe	**100**	**-9**	**62**	**46**	**2769**	**-7 – 15**	**650**
EU-15	**226**	**-6**	**24**	**82**	**3459**	**-8 - 15**	**1087**
EU-25	**144**	**-9**	**36**	**67**	**3529**	**-7 – 15**	**795**

Peatlands in a Kyoto perspective

In contrast to the emission factors in Table 6, which are used in this report, emissions factors according to the Kyoto protocol are standard values for different management practises. It is evident that the use of the emission factors in table 9 give a very different result from what is found in chapter 8, mainly because CH4 is not counted in the Kyoto emission factors. Drainage and use of peatlands is reportable under the UNFCCC. IPCC GPG (2004) provides new, and updated, emission factors for net anthropogenic emissions (drainage fluxes minus original mire fluxes) of CO2 and N2O. Drainage reduces natural CH4 emissions, which are not reported under the UNFCCC, so the net anthropogenic effect is not reported either (see table below). Comparing these emission factors with the net effect of land use in our calculation reveals interesting differences. The effects of forestry in our calculations are smaller than in IPCC GPG (2004), except for N2O from fens. Grassland and cropland on bogs lead to much higher CO2, but much lower N2O emissions than in IPCC GPG (2004), but are based on a single study only. Our emission factors for grassland, cropland and new drainage on fens are comparable with those in IPCC GPG (2004).

Table 9: *Net anthropogenic emissions factors for, wetlands remaining wetlands and land converted to wetlands in boreal and temperate climate (IPCC GPG 2004, IPCC GPG 2000)*

Management types of peatlands	Emission factors		GWP 100 (CO2-C Equivalents kg ha –1 yr-1)
	CO2 (t C ha-1 yr-1)	N2O (kg N ha-1 yr-1)	
BOG (ombrotrophic)			
Forest (drained)	0.16 (0.08-1.09) a,e	0.1 (0.02-0.3) e	0.17 (0.08-1.13)
Grassland	0.25 ± 0.225 c,e	8 (1-80) f	1.31 (1.09-1.54)
Arable	1 ± 0.9 c,e	8 (1-80) f	2.06 (1.16-2.96)
Drainage for peat cut	0.2 (0-0.63) e	0.1 (0-0.3) e	0.21 (0-0.67)
FEN (minerotrophic)			
Forest (drained)	0.68 (0.41-1.91) b,e	0.6 (0.16-2.4) e	0.76 (0.43-2.23)
Grassland	2.5 ± 2.25 d,e	8 (1-80) f	3.56 (1.31-5.81)
	2.7 ± 1.4 g	7 (6-9) g	3.63 (2.10-5.30)
Arable	10 ± 9 d,e	8 (1-80) f	11.1 (2.1-20.1)
	4.1 ± 1.4 g	7 (6-9) g cereals	5.03 (3.50-6.70)
		20 (10-30) g vegetables, roots and tubers	6.76 (4.03-9.49)
Drainage for peat cut**	1.1 (0.03-2.9) e	1.8 (0.2-2.5) e	1.34 (0.06-3.23)

a boreal climate
b temperate climate
c cold temperate climate (mean annual temperature 0-10°C)
d warm temperate cropland (mean annual temperature 10-20°C)
e IPCC GPG 2004
f IPCC GPG 2000
g Freibauer, 2003

• 9. Gaps and future research needs

Presentations and discussions at the workshop identified several key gaps in data and understanding and several key research needs, which can be grouped into broad categories: physical characteristics, greenhouse gas fluxes, management, and future trends. Almost all data gaps become more problematic as the spatial domain increases from EU15 to EU25 to geographical Europe.

Physical characteristics of peatlands:

- Peat depths are not well-known in much of Europe, and are a key item in calculating current carbon stocks and long term accumulation rates.
- Peat dry bulk density profiles are not well-characterized across Europe

Peatland greenhouse gas fluxes:

- There is particular need for more data about GHG budgets of peatlands under particular land uses: a) bog: grassland, cropland, land abandoned after peat cut, restoration, forest chronosequences, N_2O fluxes in general. b) fen: abandoned after harvest, restoration, CO_2 fluxes in general.
- There are very few year-round data sets of greenhouse gas fluxes. Non-growing season fluxes are inadequately quantified, yet may be important to annual budgets.
- There are very few sites with comprehensive, multi-year greenhouse gas flux measurements (CO_2, CH_4, and N_2O).
- There are insufficient data to characterize greenhouse gas flux sensitivity to the full potential of weather variability (e.g., droughts or exceedingly wet years, very warm or very cold winters).
- The scale of interannual variability in peatland greenhouse gas budgets is not well quantified, but has the potential to be large due to significant and highly variable methane fluxes.

Peatland management:

- Peatland management can significantly affect vegetation cover (e.g., afforestation, peat harvest for fuel or horticulture), and may have impacts of the surface energy budget, through changes in albedo, surface roughness, and energy partitioning between sensible and latent heat losses; these changes in the surface energy budget may exceed the radiative forcing impacts of changes in greenhouse gas emissions, particularly in the short term. These non-greenhouse gas climate impacts have not been adequately quantified.
- There is a need for a full life-cycle assessment of the climate impacts of peat use (direct emissions from use, changes in site greenhouse gas emissions over the timescale of recovery, embedded energy costs, avoided emissions, etc…).
- Is it possible to manage intact peatlands for low greenhouse gas impacts? At what cost to other environmental and/or economic services a peatland provides? If this can be done, should policies be developed for peatland restoration, which encourage low greenhouse gas impacts? Would it be sustainable under climate change scenarios?

Future Trends:

- While some studies have investigated the short-term impact of a changing environment on greenhouse gas fluxes in selected peatlands, little is known about how peatland vegetation distributions will respond to climate change and other global change forcings (increasing atmospheric CO_2, increasing N-deposition, increasing tropospheric ozone, permafrost degradation)?

- Dynamic vegetation models that otherwise have been used to make projections for the potential distribution in vegetation types into the future based on climate change scenarios are unfortunately this far not applicable to peatlands as they have not been identified as specific plant functional types (PFTs) in these models. However, some recent climate predictions have been run in combination with improved wetland/hydrology schemes suggesting significant changes in potential wetland distribution over the next 100 years.

• Appendices

APPENDIX I: Potential in peatland management for reducing GHG in EU and its accountability in relation to the Kyoto protocol

A.1. Background and purpose

Peatlands currently exist as natural mires or as anthropogenically affected peatlands:

1. Virgin mires
2. Drained peatland to be cut-off for energy production
3. Drained peatland that is forested
4. Drained peatland used for agriculture
5. Afforested cut-offs
6. Rewetted cut-offs
7. Abandoned cut-offs
8. Restored peatlands from forest
9. Restored peatlands from agriculture

Both natural mires and anthropogenically affected peatlands emit GHGs. For instance, virgin peatlands emit methane and serve as a sink for atmospheric CO_2. Using peat for energy leads to combustion and emissions of carbon to the atmosphere. Drained peatland used for cut-off, forestry or agriculture emit CO_2 and N_2O due to the oxidation of peat while CH_4 emissions cease. Different options of after-treatment of cut-off or restoration will create specific GHG- fluxes (both emissions and sinks) of CH_4, CO_2 and N_2O.

When addressing the issue of management of peatlands one important consideration should be how GHG fluxes are affected. These changes will also have importance in relation to the national reporting to the UNFCCC. However, according to current reporting standards not all of these flux changes can be accounted for. Moreover, it should be noted that in the context of peatland management there are a number of other issues besides GHG fluxes that are important to consider such as the preservation of habitats, the sequestration of water runoff and safeguarding of regional water supply.

Peat harvesting for energy purposes is an important use of peatlands today. This use of peatlands will lead to emissions of CO_2 from the combustion of peat. But the total climate impact from using peat for energy is more complex. Harvesting peat means that the GHG fluxes from the harvested area are changed. For instance, the previous emissions of methane from the virgin mire will cease as the mire is drained. Depending on after-treatment, new fluxes will be created that under favourable conditions can have a lower climate effect than before drainage. Studies have shown that the total climate effect of using peatlands for energy, including both the combustion stage and the land use changes, can be lower than the figure associated with the combustion, depending on selection of mire types and after-treatment, if methane emissions from the virgin mire are set as reference.

In the context of UNFCCC reporting peat is associated with an emission factor from combustion only, 106 g CO_2/MJ which is higher than the emission factor for coal. In contrast, “fugitive” emissions from production and processing are so far only considered for coal and natural gas. The potential changes of GHG emissions from land use changes are normally not considered. The Good Practice Guidelines for LULUCF (IPCC 2004) offer methodologies to account for changes in GHG-fluxes associated with the management of peatlands used for peat extraction, in order to close this reporting gap.

The purpose of this appendix is to discuss:
- How will GHG-fluxes be affected by different available options of peatland management, with or without including energy production?
- How can changes in GHG-fluxes due to management of peatlands be accounted for in relation to the Kyoto protocol?

There are a number of peatland management scenarios that would be relevant to investigate in this context:

Scenarios without energy production:
1. Drainage of natural mires
2. Afforestation of cut-offs
3. Re-wetting of cut-offs
4. Re-wetting of drained peatland for agriculture
5. Re-wetting of drained peatland for forestry
6. Conservation of natural mires

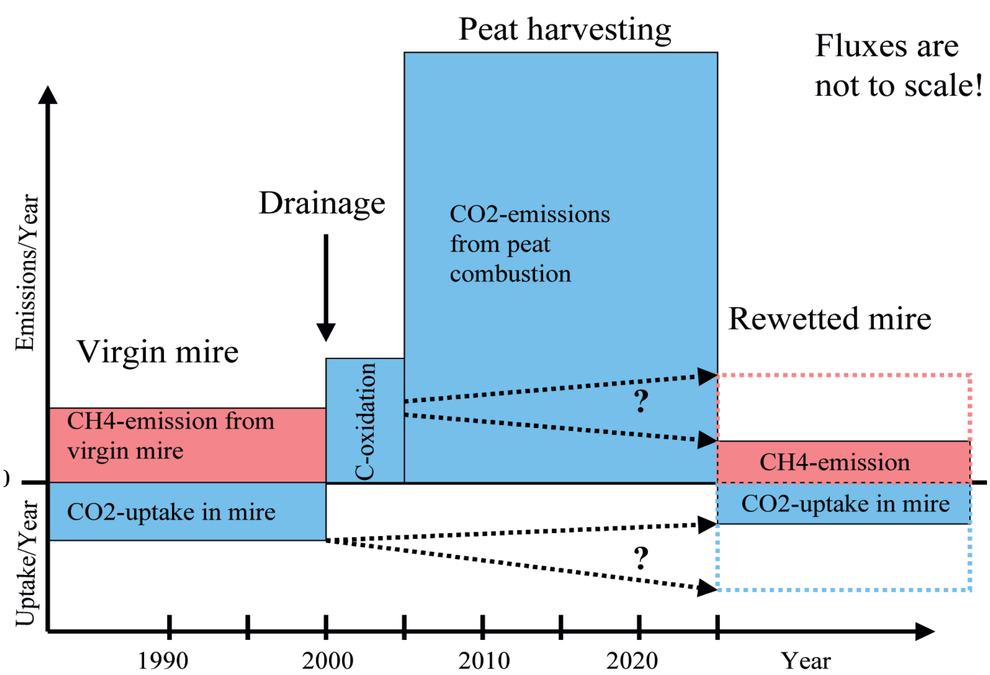
Scenarios including energy production:
7. Natural mire - > Drained mire -> Peat Energy production/Cut-off -> Re-wetting
8. Drained peatland that is forested -> Deforested area -> Peat Energy production/cut-off -> Afforestation

Due to the limited space available for this paper, we have chosen to focus on energy scenarios 7 and 8.

A.2. Scenario: Natural mire - > Drained mire -> Peat Energy production/Cut-off -> Re-wetting

A.2.1. *Schematic description of changes in GHG-fluxes and potential climate impact*

A.2.1.1. *Schematic description of how GHG-fluxes change over time due to this scenario*



a) Natural mire -> Drained mire: Methane emissions from virgin mire stop. Carbon uptake from virgin mire stops. Marginal effects on N_2O emissions in nutrient-poor bogs. Oxidation of ground carbon starts.
b) Drained mire -> Peat production: Combustion of peat leads to CO_2 emissions and marginal emissions of methane and N_2O. Marginal emissions connected to harvest machines and transport.
c) Harvested area (Cut-off) –> Re-wetting: Methane emissions and carbon uptake resume in re-wetted mire. Marginal effects on N_2O emissions in nutrient-poor bogs.

A.2.1.2. *Total climate impact*

If only combustion is considered the emissions from peat energy will be ca 90-110 g CO_2/MJ. If fluxes from land use are included in the analysis, both before and after harvesting the total climate impact can potentially be lower (or higher) than from combustion only. The result depends on if the total GHG fluxes from land use after harvesting have a lower climate impact than the total GHG fluxes before harvesting. The time scale will also be important for the result:

- Short time scale - year 2000-2025: The emissions from combustion will dominate.
- Long time scale – year 2025-2300: If the GHG fluxes after re-wetting have a lower climate effect than fluxes before harvesting then the net climate effect from harvesting peat will be lower than the climate effect from combustion of peat only. However, this scenario requires that low methane emissions are maintained for a long time in order to not return to previous conditions with high methane emissions. If we assume that the peatland returns to its original status, the net long-term difference is determined by the combustion emissions and the transient difference in GHG fluxes during site preparation, harvest, and transition to the original status and can be smaller or larger than combustion emissions alone.

A.2.2. *Accountability in relation to the Kyoto protocol*

a) **Natural mire -> Drained mire:** Methane emissions and carbon uptake from virgin mire decrease. Oxidation of carbon in peat leads to emissions of CO_2.
Accounting under the Kyoto protocol: Only anthropogenic emissions enter the GHG inventories, so only increased CO_2 and N_2O emissions would arise from this land use change. Therefore, this land use change is not eligible for acquiring credits (AAUs) for avoided methane emissions. According to the Marrakesh Accord, GHG mitigation measures should not compromise the goals of other conventions such as CBD and Ramsar. These conventions address the issues that drainage of peatlands has adverse effects on biodiversity, and regional water supply. Therefore, drainage of natural mires is neither accountable nor allowable under the Kyoto protocol.

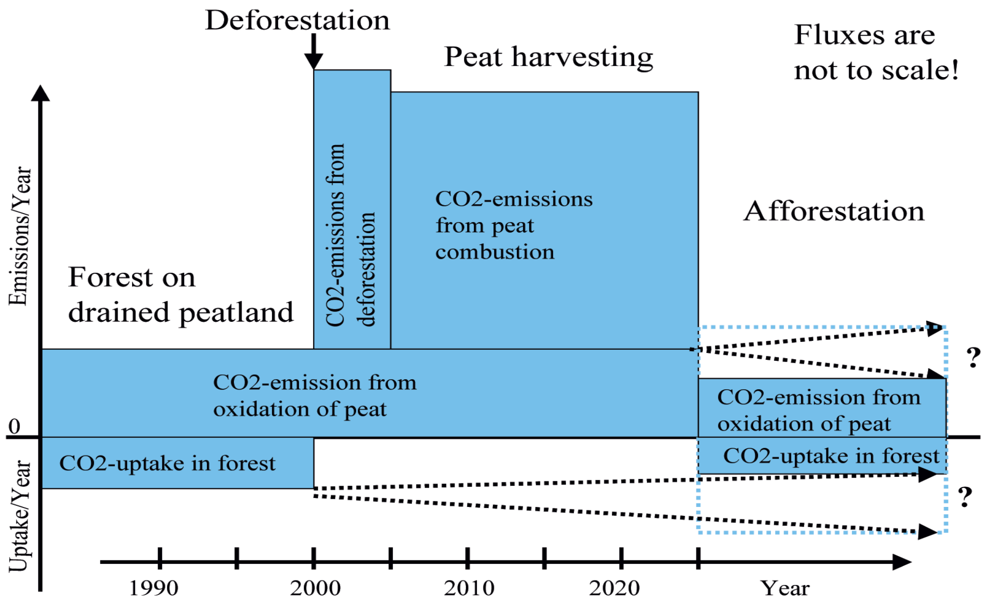
Drained mire -> Peat production/Cut-off: Combustion of peat leads to CO_2-emissions and marginal emissions of methane and N_2O. Marginal emissions connected to harvest machines and transports. Anthropogenic emissions of CO_2 and N_2O occur on the drained land.
Accounting under the Kyoto protocol: Combustion emissions are reported as emissions from fuels. Drainage emissions are reported in the category 5 LUCF in the inventory for UNFCCC. It is unclear whether or how they will enter in the inventory for the Kyoto Protocol because wetlands are not a defined Kyoto land category.

b) **Cut-off –> Re-wetting:** Methane emissions and carbon uptake are created in rewetted mire.
Accounting under the Kyoto Protocol: Reduced emissions and new sinks on rewetted land are reported in the category 5 LUCF in the inventory for UNFCCC. It is unclear whether or how they will enter in the inventory for the Kyoto Protocol because wetlands are not a defined Kyoto land category.

A.3.Drained peatland that is forested -> Deforested area -> Peat Energy production/cut-off -> Afforestation

A.3.1. ***Schematic description of changes in GHG-fluxes and potential climate impact***

A.3.1.1. ***Schematic description of how GHG-fluxes change over time due to this scenario***



For this scenario, the age dynamics of forest CO_2 uptake should be considered. The same is true for ground (previous peatland) emissions. In a young forest with low transpiration we will expect – CH_4 release or small CO_2 losses. In forest with high transpiration we will expect no CH_4, but increased CO_2 and N_2O emissions.

a) **Drained peatland that is forested -> Deforested area:** Deforestation leads to emissions of CO_2. CO2-uptake in forest ceases. Marginal effects from CH_4 and N_2O.

b) **Deforested area -> Peat Energy production/cut-off:** Combustion of peat leads to CO_2-emissions and marginal emissions of CH_4 and N_2O. Marginal emissions connected to harvest machines and transports. CO_2-emissions due to oxidation of ground carbon in drained peatland continue at the same or a lower rate.

c) **Peat Energy production -> Afforestation:** New biomass build-up is created. Oxidation of remaining ground carbon as peat continues at higher or lower level. These emissions cease when all peat carbon is oxidised, probably long time after harvesting. Marginal effects from CH_4 and N_2O.

A.3.1.2. ***Total climate impact***

If only combustion is considered the emissions from peat energy will be ca 90-110 g CO_2/MJ. If fluxes from land use are included in the analysis, both before and after harvesting the total climate impact can be lower or larger than from combustion only. The result depends on if the total GHG fluxes from land use during and after harvesting have a lower or higher climate impact than the total GHG fluxes before harvesting. The time scale will also be important for the result:

- Short time scale years 2000-2025: Emissions from deforestation and peat combustion will dominate and be higher than emissions from peat combustion only.

- Long time scale years 2025-2300: The most important effect is most likely the avoided ground carbon oxidation. Over several centuries the total accumulated emissions from the oxidation of ground carbon might be in the same order of magnitude as the emissions would be if the peat is combusted, since in both cases it's the same original amount of carbon that is lost to the atmosphere. Hence, the net GHG emissions from combustion of peat could be roughly the same as if the area continued to be used as forest on drained peatland with the difference that using the area for peat production will also render electricity and heat production. The forest biomass is lost during the 25-year period of peat harvesting, without altering too much the average biomass carbon stocks in the forest before and after this period. On a long time scale, the average forest biomass carbon stock is slightly reduced due to this interruption in the forest rotation cycle. In contrast, in a scenario in which forest is only moderately drained so that no peat carbon is lost or peat carbon even continues to accumulate (cf. emission factors in Section 8), the net effect is the CO2 from combustion and a small additional change due to the interruption of forestry use.

A.3.2 *Accountability in relation to the Kyoto protocol*

a) **Drained peatland that is forested -> Deforested area:** Biomass loss from deforestation leads to emissions of CO_2. CO_2-emissions due to oxidation of ground carbon as peat continue (in case of intensive drainage for previous forest) until peat is harvested. In the case of moderate drainage for previous forest the oxidation of peat is initiated.

   Accounting under the Kyoto protocol: Deforestation since 1990 is reported under Article 3.3 of the Kyoto Protocol. It should include the continuing CO_2 emissions from peat oxidation.

b) **Deforested area -> Peat Energy production/cut-off:** Combustion of peat leads to CO2-emissions and marginal emissions of CH4 and N2O. Marginal emissions connected to harvest machines and transports. Anthropogenic emissions of CO2 and N2O occur on the drained land.
   Accounting under the Kyoto protocol: Combustion emissions are reported as emissions from fuels. Continued drainage emissions are reported in the category 5 LUCF in the inventory for UNFCCC. It is unclear whether or how they will enter in the inventory for the Kyoto Protocol because wetlands are not a defined Kyoto land category.

c) **Peat Energy production -> Afforestation:** New biomass build-up is created.
   Anthropogenic emissions of CO2 and N2O continue on the drained land until the entire peat layer has been oxidised, or peat drainage ceases to moderate levels.

   Accounting under the Kyoto protocol: We assume that the peat cut period is shorter than 50 years. Deforestation (for peat cut) and reforestation since 1990 is reported under Article 3.3 of the Kyoto Protocol. It should include the continuing CO_2 emissions from peat oxidation. It is not expected that the peat layer will already be completely lost during the first commitment period. Reduced or abandoned drainage of forest peatlands will reduce peat carbon losses, but may have adverse effects on tree productivity. Given that the trees on the afforested cut-off lands are still younger than 22 years, the expected carbon credits are small.

The importance of time perspective

Assuming that the avoided ground carbon oxidation is accounted for it is clear that over a long time perspective (the time needed for the ground carbon to be fully oxidised) the emissions from ground oxidation are roughly equal to emissions from peat combustion. So peat combustion would be a favourable scenario in a very long-term perspective since the total GHG fluxes are the same as if the ground peat is left to oxidise, while peat combustion renders electricity and heat production, offsetting other fuel use. However, from a Kyoto point of view this effect will not be available in a short time perspective, for instance during the first commitment period 2008-2012. The net emissions in 2010 are actually higher than in 1990 if land use is included compared to only including emissions related to peat combustion, due to the deforestation. It will take many decades before the accumulated land use emissions are lower than the emissions would be if the forested peatland area was left untouched.

APPENDIX II: Tables

Table 11: *Peatland area (km²)*

Country	Table 3 (Joosten & Clarke, 2002) Peatland area 2002 (km^2)	Selin (1999) Peatland area 2002 (km^2)	Worldenergy[a] Peatland area 1999 (km^2)
Albania	179	NA	100
Andorra	5	NA	NA
Austria	200	200	220
Azores	1	NA	NA
Belarus	23500	23500	23970
Belgium	160	160	200
Bosnia and Herzegovina	150	NA	NA
Bulgaria	25	NA	30
Croatia	1	NA	NA
Czech Republic	200	200	270
Denmark	1400	1400	1420
Estonia	10000	10090	9020
Faroe Islands	30	NA	NA
Finland	85000	68740 [b]	89000
France	1500	1500	1000
FYRO Macedonia	30	NA	NA
Germany	16520 [c]	13000	14200
Greece	71	70	100
Hungary	330	330	1000
Iceland	8000	NA	10000
Ireland	11500	11500	11800
Italy	300	300	1200
Latvia	6600	6690	6400
Liechtenstein	1	NA	NA
Lithuania	3520	4820	4830
Luxembourg	3	10	NA
Moldova	10	NA	NA
Netherlands	2350	2350	2800
Norway	28000	28000	23700
Poland	12500	12500	12000
Portugal	20	20	200
Romania	1000	71	70
Russia European part	213000	213000	NA
Slovakia	26	NA	40
Slovenia	100	NA	1000
Spain	60	60	380
Sweden	66000	66000	64000
Switzerland	300	NA	220
Ukraine	8000	8000	10080
United Kingdom	17500	17500	19260
Yugoslavia	300	NA	NA

[a] (http://www.worldenergy.org/wec-geis/publications/reports/ser/peat/peat.asp), [b]Value not used for further analysis. Low area value may be based on different peatland definition assuming a greater depth of the peat layer. [c] New GIS-based estimate from German Federal Agricultural Research Centre (Freibauer et al., in prep.). High area value may be based on different peatland definition assuming a more shallow depth of the peat layer.

Table 12: *Peatland use (km²)*

Country	Joosten & Clarke (2002) Mire area (km2)	Joosten & Clarke (2002) Forestry area (km2)	Lappalainen (1996) Grass area (km2)	Lappalainen (1996) Crop area (km2)	Selin (1999) Peat cut area (km2)	Residual [a] (km²)
Albania	4	0	0	175	0	0
Andorra	2	0	0	3	0	0
Austria	100	0	0	0	0	100
Azores	1	0	0	0	0	0
Belarus	11412	2800	8667 [b, e]	963 [b, e]	120	-462
Belgium	3	0	144 [b]	16 [b]	0	-3
Bosnia and Herzegovina	10	0	0	140	0	0
Bulgaria	5	0	0	0	0	20
Croatia	1	0	0	0	0	0
Czech Republic	50	0	70	30	10	40
Denmark	50	0	1000	0	10	340
Estonia	3000	4600	1300	0	90	1010
Faroe Islands	25	0	0	0	0	5
Finland	32000	57200 [c]	2000	0	550	-6550
France	100	0	0	660	10	730
FYRO Macedonia	5	0	0	25	0	0
Germany	390 [d]	2189 [d]	8559 [d]	4724 [d]	300	358
Greece	13	0	0 [b]	71 [b]	0	0
Hungary	30	0	50	25	0	225
Iceland	3500	0	1300	0	0	3200
Ireland	2100	2100	1240 [b]	896 [b]	900	4264
Italy	30	0	0	270	0	0
Latvia	4663	600	0	1000	20	317
Liechtenstein	1	0	0	0	0	0
Lithuania	750	590	397 [b]	1000 [b]	20	763
Luxembourg	1	0	0	0	0	2
Moldova	1	0	0	0	0	9
Netherlands	150	0	2000 [b]	0 [b]	0	200
Norway	22000	0	1900	4200	20	-120
Poland	2000	1270	7565	55	10	1600
Portugal	2	0	0	18	0	0
Romania	500	0	0	0	0	500
Russia European part	150000	14250	23760 [b, e]	2640 [b, e]	160	22190
Slovakia	13	0	7	3	0	3
Slovenia	10	0	0	0	0	90
Spain	10	0	0	23	0	27
Sweden	55000	14100	0	3000	110	-1210
Switzerland	200	0	80	20	0	0
Ukraine	5800	0	0	5000	30	-30
United Kingdom	1000	6000	310 [b]	410 [b]	40	9740
Yugoslavia	50	0	0	250	0	0
Total	**305202**	**105699**	**60349**	**25617**	**2400**	**37358**

[a] Residual area to match total peatland area. Small negative areas result from the uncertainty in areas under other land uses. For the calculation, negative residual areas were set zero. [b] Literature value gives total agricultural area only. Distribution between grassland and cropland use based on expert judgement. [c] Minkkinen *et al.* 2002, GCB 8: 785-799. [d] New GIS-based estimate from German Federal Agricultural Research Centre (Freibauer et al., in prep.). High area value may be based on different peatland definition assuming a more shallow depth of the peat layer. [e] Distribution between grassland and cropland very uncertain. Significant contribution to uncertainty in European CO_2 and N_2O estimates.

Table 13: *Bog use (km²)*

Country	Lappalainen (1996) Total	Lappalainen (1996) Mire	Lappalainen (1996) Forest	Lappalainen (1996) Grass	Lappalainen (1996) Crop	Selin (1999) Peat cut	Residual
Albania [a, b]	0	0	0	0	0	0	0
Andorra [b]	0	0	0	0	0	0	0
Austria	100	50	0	0	0	0	50
Azores [b]	0	0	0	0	0	0	0
Belarus	4324	4324	0	0	0	0	0
Belgium	160	3	0	144	16	0	-3
Bosnia and Herzegovina [b]	0	0	0	0	0	0	0
Bulgaria	25	5	0	0	0	0	20
Croatia [b]	0	0	0	0	0	0	0
Czech Republic	158	39	0	70	0	10	38
Denmark	375	25	0	0	0	10	340
Estonia	4000	1110	1840	0	0	90	960
Faroe Islands	30	25	0	0	0	0	5
Finland	56950	32000	22800 [c]	0	0	550	1600
France	750	100	0	0	0	10	640
FYRO Macedonia [b]	0	0	0	0	0	0	0
Germany	4086 [d]	353 [d]	458 [d]	2163 [d]	787 [d]	300	25
Greece	0	0	0	0	0	0	0
Hungary	0	0	0	0	0	0	0
Iceland	0	0	0	0	0	0	0
Ireland	10575	2072	2100	1240	0	900	4263
Italy [b]	0	0	0	0	0	0	0
Latvia	3346	2454	600	0	0	20	272
Liechtenstein	1	1	0	0	0	0	0
Lithuania	735	157	123	397	0	20	38
Luxembourg	3	1	0	0	0	0	2
Moldova	0	0	0	0	0	0	0
Netherlands	0	0	0	0	0	0	0
Norway	28000	22000	0	1900	4200	20	-120
Poland	1000	160	353	129	0	10	348
Portugal [b]	0	0	0	0	0	0	0
Romania	14	7	0	0	0	0	7
Russia European part	110383	77734	14250	0	0	80	18318
Slovakia	6	3	0	1	0	0	2
Slovenia [b]	0	0	0	0	0	0	0
Spain [b]	0	0	0	0	0	0	0
Sweden [e]	23760	16600	7050	0	0	110	0
Switzerland	20	20	0	0	0	0	0
Ukraine	400	400	0	0	0	0	0
United Kingdom	16045	345	5501	310	205	40	9644
Yugoslavia [b]	0	0	0	0	0	0	0
Total	**265245**	**159988**	**55075**	**6354**	**5208**	**2170**	**36449**

Table 14: *Fen use (km²)*

Country	Lappalainen (1996) Total	Lappalainen (1996) Mire	Lappalainen (1996) Forest	Lappalainen (1996) Grass	Lappalainen (1996) Crop	Selin (1999) Peat cut	Residual
Albania [a, b]	179	4	0	0	175	0	0
Andorra	5	2	0	0	3	0	0
Austria	100	50	0	0	0	0	50
Azores [b]	1	1	0	0	0	0	0
Belarus	19176	7088	2800	8667	963	120	-462
Belgium	0	0	0	0	0	0	0
Bosnia and Herzegovina [b]	150	10	0	0	140	0	0
Bulgaria	0	0	0	0	0	0	0
Croatia [b]	1	1	0	0	0	0	0
Czech Republic	42	11	0	0	30	0	2
Denmark	1025	25	0	1000	0	0	0
Estonia	6000	1890	2760	1300	0	0	50
Faroe Islands	0	0	0	0	0	0	0
Finland	28050	0	34200 [c]	2000	0	0	-8150
France	750	0	0	0	660	0	90
FYRO Macedonia [b]	30	5	0	0	25	0	0
Germany	12434 [d]	370 [d]	1731 [d]	6396 [d]	3937 [d]	0	0
Greece	71	13	0	0	58	0	0
Hungary	330	30	0	50	25	0	225
Iceland	8000	3500	0	1300	0	0	3200
Ireland	925	28	0	0	896	0	1
Italy [b]	300	30	0	0	270	0	0
Latvia	3254	2254	0	0	1000	0	0
Liechtenstein	0	0	0	0	0	0	0
Lithuania	2785	593	467	0	1000	0	725
Luxembourg	0	0	0	0	0	0	0
Moldova	10	1	0	0	0	0	9
Netherlands	2350	150	0	2000	0	0	200
Norway	0	0	0	0	0	0	0
Poland	11500	1840	917	7436	55	0	1252
Portugal [b]	20	2	0	0	18	0	0
Romania	57	29	0	0	0	0	29
Russia European part	102617	72266	0	23760	2640	80	3872
Slovakia	20	10	0	6	3	0	1
Slovenia [b]	100	10	0	0	0	0	90
Spain [b]	60	10	0	0	23	0	27
Sweden [e]	42240	32190	7050	0	3000	0	0
Switzerland	280	180	0	80	20	0	0
Ukraine	7600	4570	0	0	3000	30	0
United Kingdom	1455	655	499	0	205	0	96
Yugoslavia [b]	300	50	0	0	250	0	0
Total	**252218**	**127867**	**50424**	**53995**	**18395**	**230**	**1306**

[a] According to Ramsar (http://www.ramsar.org/cop7_nr_albania.htm), most of wetlands are salt pans. [b] Distribution of land uses between bogs and fens assumed like in Greece. [c] Minkkinen et al. 2002, GCB 8: 785-799. [d] Freibauer et al., in prep.
[e] http://www.internat.naturvardsverket.se/, LUSTRA 2002 report

• Glossary

Acrotelm: The upper (usually <0.4 m in depth), aerated and more or less oxic layer in any undisturbed virgin peatland

Bog: Ombrotrophic (nutrient poor) mire

Catotelm: The underlying (usually >1 m) and anoxic layer in any undisturbed virgin peatland

Fen: Minerotrophic (nutrient rich) mire

GHG: Greenhouse Gas

GWP: Global Warming Potential

LORCA: The average long-term apparent rate of carbon accumulation

LULUCF: Land use, land use change and forestry

Mire: General term for non-tidal wetlands. In this report it is used for undisturbed currently peatforming mires

Peatland: An area with or without vegetation with a naturally accumulated peat layer at the surface. It includes disturbed or managed mires.

UNFCCC: The United Nations Framework Convention on Climate Change was agreed in 1992 and signatories are encouraged to conserve and enhance sinks of greenhouse gases. The Kyoto Protocol to the UNFCCC allows for carbon sequestration in vegetation and soils to be used to offset greenhouse gas emissions.

• Details on participants

Kenneth A. Byrne, Department of Civial and Environmental Engineering, University College Cork, Cork, Ireland

Bogdan Chojnicki, Department of agrometeorology, Poznan, Poland

Torben R. Christensen, GeoBiosphere Science Centre, Physical Geography and Ecosystems Analysis, Lund University, Sweden

Matthias Drösler, Department of Plant Ecology, Bayreuth, Germany

Annette Freibauer, Max-Planck-Institute for Biogeochemistry, Jena, Germany.

Thomas Friborg, Institute of Geography, Copenhagen, Denmark.

Steve Frolking, Institute for the Study of Earth, Oceans and Space, Durham NH, USA

Anders Lindroth, GeoBiosphere Science Centre, Physical Geography and Ecosystems Analysis, Lund University, Sweden

Julia Mailhammer, Technical University of Munich, Freising, Germany

Nils Malmer, Department of Ecology, Plant Ecology and Systematics, Lund University, Sweden

Pirkko Selin, Vapo Oy, Jyvaskyla, Finland

Jukka Turunen, Geological Survey of Finland (GTK), Kuopio Unit, Finland

Riccardo Valentini, Department of Forest Science and Environment, Viterbo, Italy

Lars Zetterberg, Swedish Environmental Research Institute, Stockholm, Sweden

• References

Alm, J., Saarnio, S., Nykanen, N., Silvola, J. & P. J. Martikainen, 1999. Winter CO2, CH4 and N2O fluxes on some natural and drained peatlands. *Biogeochemistry*, 44: 163–186.

Augustin, J., Merbach, W., Schmidt, W. & E. Reining, E., 1996. Effect of changing temperature and water table on trace gas emissions from minerotrophic mires. *Angew Bot,* 70: 45-51.

Augustin J., Merbach W., Käding H., Schalitz G. & W. Schmidt, 1998 a. Nordostdeutsche Niedermoore als Quelle und Senke klimarelevanter Spurengase. *Mitteilungen der Deutschen Bodenkundlichen Gesellschaft*, 88: 7-10.

Augustin, J., Merbach, W., Steffens, L. & B. Snelinski, 1998 b. Nitrous oxide fluxes of disturbed minerotrophic peatlands. *Agribiol. Res.*, 51: 47-57.

Augustin, J., 2001. Nordostdeutsche Niedermoore als Quelle klimarelevanter Spurengase. *In*: Succow, M. & H. Joosten (eds.). *Landschaftsökologische Moorkunde. 2. völlig neubearbeitete Auflage, Schweizerbart´sche Verlagsbuchhandlung*.

Aurela, M., Laurila, T., & J. P. Tuovinen, 2002. Annual CO2 balance of a subarctic fen in northern Europe: Importance of the wintertime efflux. *Journal of Geophysical Research-Atmospheres*, 107: art-4607.

Barber, K. E., 1981. *Peat Stratigraphy and Climate Change*. Rotterdam, Netherlands: Balkema.

Belyea, L. R. & N. Malmer, 2004. Carbon sequestration in peatland: patterns and mechanisms of response to climate change. *Global Change Biology,* 10: 1043-1052.

Belyea, L. R., 1996. Separating the effects of litter quality and microenvironment on decomposition rates in a patterned peatland. Oikos, 77: 529-539.

Botch, M. & V. Masing, 1979. Regionality of mire complex types in the USSR. Proceedings of the International Symposium on Classification of Peat and Peatlands Hyytiälä, Finland, September 17-21: 1-11.

Botch, M. S., Kobak, K. I., Vinson, T. S. & T. P. Kolchugina, 1995. Carbon pools and accumulation in peatlands of the Former Soviet Union. *Global Biogeochemical Cycles*, 9: 37-46.

Christensen, T.R., T. Friborg, M. Sommerkorn, J. Kaplan, L. Illeris, H. Søgaard, C. Nordstrøm, & S. Jonasson, 2000. Trace gas exchange in a high arctic valley 1: Variations in CO_2 and CH_4 flux between tundra vegetation types *Global Biogeochemical Cycles*, 14, 701-714.

Christensen ,T.R., Panikov, N., Mastepanov, M., Joabsson, A., Öquist, M., Sommerkorn, M., Reynaud, S. & Svensson, B. 2003. Biotic controls on CO_2 and CH_4 exchange in wetlands - a closed environment study *Biogeochemistry*, 64:337-354.

Christensen, T.R., Johansson, T., Åkerman, H.J., Mastepanov, M., Malmer, N., Friborg, T., Crill, P. & B.H. Svensson, 2004. Thawing sub-arctic permafrost: Effects on vegetation and methane emissions. *Geophysical Research Letters,* 31.

Clymo, R. S., 1984. The limits to peat bog growth. *Philosophical Transactions of the Royal Society* B, 303: 605-654.

Clymo, R. S., 1992. Models of peat growth. *Suo,* 43: 127-136.

Clymo, R. S. & D. M. E. Pearce, 1995. Methane and carbon dioxide production in, transport through, and efflux from a peatland. *Philosophical Transactions of the Royal Society A*, 350: 249-259.

Clymo R.S., Turunen J. & K. Tolonen, 1998. Carbon accumulation in peatland. *Oikos,* 81: 368–388.

Coulson, J. C. & J. Butterfield, 1978. An investigation of the biotic factors determining the rates of plant decomposition on blanket bogs. *Journal of Ecology,* 66: 631-650.

Drösler, M., Glatzel S. & J. Pfadenhauer, (in prep.). Net ecosystem CO_2 exchange in natural, degraded and restored bogs in Southern Germany. Submitted to *Global Change Biology*.

Drösler M., Munch C-J. & J. Pfadenhauer, (in prep.). Methane and nitrous oxide fluxes from natural, degraded and restored bog ecosystems in Southern Germany. In preparation for *Global Biogeochemical Cycles*.

Dörsch, P., Linde, J., Zaglauer, A. & H. Flessa, (in prep.). Nitrous oxide, methane, and carbon dioxide fluxes from a minerotrophic mire under long-term cultivation.

Finlayson, C. M., Davidson, N. C., Spiers, A. G. & N. J. Stevenson, 1999. Global wetland inventory: Current status and future priorities. *Marine and Freshwater Research*, 50, no. 8: 717-727.

Flessa, H., Wild, U., Klemisch, M. & J. Pfadenhauer, 1998. Nitrous oxide and methane fluxes from organic soils under agriculture. *European Journal of Soil Science*, 48: 327-335.

Freibauer, A., 2003. Regionalised inventory of biogenic greenhouse gas emissions from European agriculture. *European Journal of Agronomy,* 19: 135-160.

Freibauer A., Glatzel S., Borken W., Badeck, F., Baritz, R., Benndorf R., Butterbach-Bahl, K., Flessa H., Gensior A., Kuzyakov Y. & C. Wirth, (in prep.). Present Greenhouse Gas Budget and Mitigation in the German Biosphere.

Friborg, T., Soegaard, H., Christensen, T. R., Lloyd, C. R., & N. S. Panikov, 2003. Siberian wetlands: where a sink is a source. *Geophysical Research Letters*, 30.

Glatzel S. and Stahr, K, 2002: The greenhouse gas exchange of a pond margin in south Germany, pages 215-234. In: Broll, G., Merbach, W., Pfeiffer, E-M.,(eds.) Wetlands in central Europe. Springer, 2002, 244 pages.

Glatzel, S., Kalbitz, K., Dalva., M., & T. Moore, 2003. Dissolved organic carbon properties and their relationship to carbon dioxide efflux from restored peat bogs. *Geoderma,* 113: 397-411.

Gorham, E., 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. *Ecological Applications*, 1: 182-195.

Gunnarsson, U. & H. Rydin, 2000. Nitrogen fertilization reduces *Sphagnum* production in bog communities. *New Phytologist,* 147: 527-537.

Gunnarsson, U., Malmer, N. & H. Rydin, 2002. Dynamics or constancy in *Sphagnum* dominated mire ecosystems. A 40-year study. *Ecography,* 25: 685-704.

Harding, R. J., Gryning, S. E., Halldin, S., & C. R. Lloyd, 2001. Progress in understanding of land surface/atmosphere exchanges at high latitudes. *Theoretical and Applied Climatology*, 70: 5-18.

Heikkinen, J. E. P., Elsakov, V., & P. J. Martikainen, 2002. Carbon dioxide and methane dynamics and annual carbon balance in tundra wetland in NE Europe, Russia. *Global Biogeochemical Cycles*, 16: art-1115.

Hillebrand, K., 1993. The greenhouse effects of peat production and use compared with coal, oil, natural gas and wood. VTT Tiedotteita - Meddelanden – Research Notes 1494, Technical Research Centre of Finland, Espoo.

Höper H. & J. Blankenburg, 2000. Emissionen klimarelevanter Gase aus niedersächsischen Mooren und Möglichkeiten der Reduzierung. *NNA-Berichte*. 2: 100-117.

Houghton, J. T., Jenkins, G. J. & J. J. Ephraums, 1990. *Climate Change. The IPCC Scientific Assessment*. Cambridge University Press, Cambridge.

Ingram, H.A.P., 1982. Size and shape in raised mire ecosystems: a geophysical model. *Nature,* 297: 300-303.

IPCC, 2000. Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories Published for the IPCC by the Institute for Global Environmental Strategies, Japan.

IPCC, 2004. Good Practice Guidance for Land Use, Land Use Change and Forestry Published for the IPCC by the Institute for Global Environmental Strategies, Japan.

Ivanov, K.E., 1981. *Water Movement in Mirelands.* London: Academic Press.

Jaakkola, A. 1985. Denitrifikaatio viljapellosta ja nurmesta mitattuna asetyleeni-inhibitiomenetelmällä (Denitrification on cornfield and grass ley as determined by acetylene inhibition method), p. 76 Lannoite- ja kasviainestypen hyväksikäyttö ja typen häviö, Vol. 13. Biologisen typensidonnan ja ravinnetypen hyväksikäytön projekti, Helsinki.

Joabsson, A., Christensen, T. R. & B. Wallén, 1999. Vascular plant controls on methane emissions from northern peat forming wetlands. *Trends in Ecology and Evolution,* 14: 385-388.

Joabsson, A. & Christensen T. R., 2001. Methane emissions from wetlands and their relationship with vascular plants: an Arctic example. *Global Change Biology,* 7: 919-932.

Johnson, L. C. & A. W. H. Damman, 1993. Decay and its regulation in *Sphagnum* peatlands. - *Advances in Bryology,* 5: 249-296.

Jones, C. G., Lawton, J. H. & M. Shachak, 1994. Organisms as ecosystem engineers. *Oikos,* 69: 373-386.

Joosten, H. & D. Clarke (eds.), 2002. *Wise use of mires and peatlands, background and principles including a framework for decision-making*, pp. 1-304.

Kasimir Klemedtsson, Å., Klemedtsson, L., Berglund, K., Martikainen, P-J., Silvola, J. & O. Oenema, 1997. Greenhouse gas emissions from farmed organic soils: A review. *Soil Use and Management,* 13: 245-250.

Klarqvist, M., 2001. Peat growth and carbon accumulation rates during the Holocene in boreal mires. *Acta Universitatis Agriculturae Sueciae, Silvestria*, 203, 37 pp.

Klarqvist, M., Bohlin, E. & M. Nilsson, 2001. Long-term decline in apparent peat carbon accumulation in boreal mires in northern Sweden. *Acta Universitatis Agriculturae Sueciae, Silvestria* 203: III, 1-22.

Klemedtsson, L., Klemedtsson A.K., Moldan, F., & P. Weslien, 1997. Nitrous oxide emission from swedish forest soils in relation to liming and simulated increased N-deposition. *Biology & Fertility of Soils,* 25: 290-295.

Komulainen, V.-M., Nykänen, H., Martikainen, P-J. & J. Laine, 1998. Short-term effect of restoration on vegetation succession and methane emissions from peatlands drained for forestry in southern Finland. *Canadian Journal of Forest Research*, 28: 402-411
.

Kremenetski, K. V., Velichko, A. A., Borisova, O. K., MacDonald, G. M., Smith, L. C., Frey, K. E. & L. A. Orlova, 2003. Peatlands of the Western Siberian lowlands: current knowledge on zonation, carbon content and Late Quaternary history. *Quaternary Science Reviews,* 22: 703-723.

Laine J. & K. Minkkinen, 1996. Effect of forest drainage on the carbon balance of a mire - a case study. *Scandinavian Journal of Forest Research,* 11: 307-312.

Laine J., Silvola, J., Tolonen, K., Alm J., Nykanen, H., Vasander, H., Sallantaus, T., Savolainen, I., Sinisalo, J., & P.-J. Martikainen, 1996. Effect of water-level drawdown on global climatic warming – northern peatlands. *Ambio*, 25: 179-184.

Langeveld, C.A., R. Segers, B.O.M. Dirks, A. van den Pol-van Dasselaar, G.L. Velthof & A. Hensen, 1997. Emissions of CO_2, CH_4 and N_2O from pasture on drained peat soils in the Netherlands. *European Journal of Agronomy,* 7: 35-42.

Lappalainen, E. (ed.), 1996. Global Peat Resources, International Peat Society and Geological Survey of Finland: 57- 162.

Lappalainen, E. and P. Hänninen, 1993. Suomen turvevarat. Summary: The peat reserves of Finland. Report of Investigation 117, *Geological Survey of Finland*: 1-115.

LUSTRA, 2002. Land Use Strategies for Reducing Net Greenhouse Gas Emissions. Annual Report 2002, Uppsala, Sweden. *http://www.sml.slu.se/lustra/index.phtml*

Lütke Twenhöven, F., 1992. Untersuchungen zur Wirkung stickstoffhaltiger Niederschläge auf die Vegetationvon Hochmoooren. *Mitteilungen der Arbeitsgemeinschaft Geobotanik in Scleswig-Holstein und Hamburg*, 44.

Mäkilä, M., 1994. Suon energiasisällön laskeminen turpeen ominaisuuksien avulla. Summary: Calculation of the energy content of mires on the basis of peat properties. Report of Investigation 121, *Geological Survey of Finland*: 1-73.
Mäkilä, M., 1997. Holocene lateral expansion, peat growth and carbon accumulation on Haukkasuo, a raised bog in southeastern Finland. *Boreas*, 26: 1-14.

Mäkilä, M., Saarnisto, M. and T. Kankainen, 2001. Aapa mires as a carbon sink and source during the Holocene. *Journal of Ecology*, 89: 589-599.

Maljanen, M., Martikainen, P-J., Walden, J. & J. Silvola, 2001. CO2 exchange in an organic field growing barley or grass in eastern Finland. *Global Change Biology*, 7: 679-692.

Malmer, N. & E. Holm, 1984. Variation in the C/N-quotient of peat in relation to decomposition rate and age determination with ^{210}Pb. *Oikos*, 43: 171-182.

Malmer, N. & B. Wallén, 1993. Accumulation and release of organic matter in ombrotrophic bog hummocks - processes and regional variation. *Ecography,* 16: 193-211.

Malmer, N. & B. Wallén, 1996. Peat formation and mass balance in sub-arctic ombrotrophic peatlands around Abisko, northern Scandinavia. *Ecological Bulletins,* 45: 79-92.

Malmer, N., Svensson, G. & B. Wallén, 1997. Mass balance and nitrogen accumulation in hummocks on a South Swedish bog during Late Holocene. *Ecography,* 20: 535-549.

Malmer, N. & B. Wallén, 1999. The dynamics of peat accumulation on bogs: mass balance of hummocks and hollows and its variation throughout a millennium. *Ecography,* 22: 736-750.

Malmer, N., Albinsson,C., Svensson, B., & B. Wallén, 2003. Interferences between *Sphagnum* and vascular plants: effects on plant community structure and peat formation. *Oikos,* 100: 469-482.

Malmer, N. & B. Wallén, 2004. Input rates, decay losses and accumulation rates of carbon in bogs during the last millennium: internal processes and environmental changes. *The Holocene*, 14: 111-117.

Malmer, N., Moore, P. D. & M. C. F. Proctor, (in press). The Future of Cool Temperate Peatlands. In *The Future of Aquatic Ecosystems,* ed. Polunin, N. Cambridge University Press

Martikainen, P.J., Nykänen, H., Crill, P., & J. Silvola, 1993. Effect of water table on nitrous oxide fluxes from northern peatlands. *Nature*, 366: 51-53.

Martikainen, P.J., Nykänen, H., Alm, J., & J. Silvola, 1995. Change in fluxes of carbon dioxide, methane and nitrous oxide due to forest drainage of mire sites of different trophy. *Plant and Soil*, 168-169: 571-577.

Meyer, K., 1999. Die Flüsse der klimarelevanten Gase CO2, CH4 und N2O eines nordwestdeutschen Niedermoores unter dem Einfluß der Wiedervernässung. *Göttinger Bodenkundl. Ber.* 111: 1-134.

Milne, R. & T. A. Brown, 1997. Carbon in the Vegetation and Soils of Great Britain, *Journal of Environmental Management*, 49: 413-433.

Minkkinen, K. & J. Laine, 1998 a. Long-term effect of forest drainage on peat carbon stores of pine mires in Finland. *Canadian Journal of Forest Research*, 28: 1267-1275.

Minkkinen, K., Korhonen, R., Savolainen, I. & J. Laine, 2002. Carbon balance and radiative forcing of Finnish peatlands 1900-2100 – the impact of forestry drainage. *Global Change Biology*, 8: 785-799.

Moncrieff, J. B., Massheder, J. M., deBruin, H., Elbers, J., Friborg, T., Heusinkveld, B., Kabat, P., Scott, S., Soegaard, H., & A. Verhoef, 1997. A system to measure surface fluxes of momentum, sensible heat, water vapour and carbon dioxide. *Journal of Hydrology*, 189: 589-611.

Moore, P. D., 2002. The future of cool temperate bogs. *Environmental Conservation*, 29 (1): 3-20.

Mundel, G., 1976. Untersuchungen zur Torfmineralisation in Niedermooren. *Arch. Acker Pflanzenbau*, 20: 669-679.

Nilsson, M., Mikkela, C., Sundh, I., Granberg, G., Svensson, B. H., & B. Ranneby, 2001. Methane emission from Swedish mires: National and regional budgets and dependence on mire vegetation. *Journal of Geophysical Research-Atmospheres*, 106: 20847-20860.

Nykänen, H., Alm, J., Lång, K., Silvola, J. & P. J. Martikainen, 1995. Emissions of CH_4, N_2O and CO_2 from a virgin fen and a fen drained for grassland in Finland. *Journal of Biogeography*, 22: 351-357.

Nykänen, H., Alm, J., Silvola J., Tolonen, K. & P-J. Martikainen, 1998. Methane fluxes on boreal peatlands of different fertility and the effect of long-term experimental lowering of the water table on flux rates. *Global Biogeochemical Cycles*, 12 (1): 53-69.

Oechel, W. C., Hastings, S. J., Vourlitis, G., Jenkins, M., Riechers, G. & N. Grulke, 1993. Recent change of Arctic tundra ecosystems from a net carbon dioxide sink to a source. *Nature,* 361: 520-523.

Oechel, W. C., Cowles, S., Grulke, N., Hastings, S. J., Lawrence, B, Prudhomme, T., Riechers, G., Strain, B., Tiddue, D. & G. Vourlitis, 1994. Transient nature of CO2 fertilization in Arctic tundra. *Nature*, 371, 500-503.

Oechel, W. C., Vourlitis, G., Hastings, S. J. & S. A.Bocharev, 1995. Change in arctic CO_2 flux over 2 decades – effects of climate-change at Barrow, Alaska. *Ecological Applications, 5: 846-855.*

Oksanen, P. O., Kuhry, P. & R. N. Alekseeva, 2001. Holocene development of the Rogovaya River peat plateau, European Russian Arctic. *The Holocene*, 11: 25-40.

Olsrud, M. & Christensen, T.R. 2004. Carbon cycling in subarctic tundra; seasonal variation in ecosystem partitioning based on in situ ^{14}C pulse-labelling. *Soil Biology and Biochemistry*, 36: 245-253.

Pitkänen, A., Turunen, J. & K. Tolonen, 1999. The role of fire in the carbon dynamics of a mire, eastern Finland. *The Holocene*, 9: 453-462.

Post, W. M., Emanuel, W. R., Zinke, P. J. & A. G. Stangenberger, 1982. Soil carbon pools and world life zones. *Nature*, 298: 156-159.

Proctor, M. C. F., 1997. Aspects of water chemistry in relation to surface degradation on ombrotrophic mires. In *Blanket Mire Degradation: Causes, Consequences and Challenges,* eds. J.H. Tallis, R. Meade, & P. D. Hulme, pp. 140–152.
Aberdeen: Macaulay Land Use Research Institute.

Regina, K., Nykänen, H., Silvola, J. & P.-J. Martikainen, 1996. Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table level and nitrification capacity of the peat. *Biogeochemistry*, 35: 401-418.

Robinson, S. D. & T. R. Moore, 1999. Carbon and peat accumulation over the past 1200 years in a landscape with discontinuous permafrost, nortwestern Canada. *Global Biogeochemical Cycles* 13: 591-601.

Robinson S.D. & T. R. Moore, 2000. The influence of permafrost and fire upon carbon accumulation in high boreal peatlands, Northwest Territories, Canada. *Arct Antarct Alp Res,* 32: 155-166.

Selin, P., 1999. Industrial use of peatlands and the re-use of cut-away areas in Finland. University of Jyväskylä, 239 p.

Silvola, J., 1986. Carbon dioxide dynamics in mires reclaimed for forestry in eastern Finland. *Ann. Bot. Fennici*, 23: 59-67.

Ström, L., Ekberg, A. & Christensen, T.R., 2003. Species-specific effects of vascular plants on carbon turnover and methane emissions from a tundra wetland. *Global Change Biology*, 9:1185-1192.

Sundh, I., Nilsson, M., Mikkelä, C., Granberg, G. & B. H. Svensson, 2000. Fluxes of Methane and Carbon Dioxide on Peat-mining Areas in Sweden. *Ambio*, 29 (8): 499-503.

Tolonen, K, Davis, R. B. & L. Widoff, 1988. Peat accumulation rates in selected Maine peat deposits. Maine Geological Survey, Department of Conservation Bulletin 33: 1- 99.

Tolonen, K. & J. Turunen, 1996. Accumulation rates of carbon in mires in Finland and implications for climate change. *The Holocene*, 6: 171-178.

Tomlinson, R. & L. Davidson, 2000. Estimates of carbon stores in four Northern Irish lowland raised bogs. *Suo*, 51: 169-179.

Tuittila, E.-S., 2000. Restoring vegetation and carbon dynamics in a cut-away peatland. Publications in Botany from the University of Helsinki. No. 30.

Turunen, J., Tolonen, K., Tolvanen, S., Remes, M., Ronkainen, J. & H. Jungner, 1999. Carbon accumulation in the mineral subsoil of boreal mires. *Global Biogeochemical* Cycles, 13: 71-79.

Turunen, J., Pitkänen, A., Tahvanainen, T. & K. Tolonen, 2001. Carbon accumulation in West Siberian mires, Russia. *Global Biogeochemical Cycles*, 15: 285-296.

Turunen, J., Tomppo, E., Tolonen, K. & A. Reinikainen, 2002. Estimating carbon accumulation rates of undrained mires in Finland – application to boreal and subarctic regions. *The Holocene*, 12: 69-80.

Turunen, C. & J. Turunen, 2003. Development history and carbon accumulation of a slope bog in oceanic British Columbia, Canada. *The Holocene*, 13: 225-238.

Turunen, J. & T. R. Moore, 2003. Controls on carbon accumulation and storage in the mineral subsoil beneath peat in Lakkasuo mire, central Finland. *European Journal of Soil Science*, 52: 279-286.

Van den Pol-Van Dasselaar, A., van Beusichem, M.L. & O. Oenema, 1999. Effects of nitrogen input and grazing on methane fluxes of extensively and intensively managed grasslands in the Netherlands. *Biol. Fertil. Soils,* 29:24–30.

Velthof, G. L. & O. Oenema, 1993. Nitrous oxide emission from grasslands on sand, clay and peat soil in the Netherlands. In *Non-CO2 greenhouse gases*: Van Ham, J., et al. (eds.), Kluwer Academic Publishers, Doordrecht, 439-444.

Velthof, G. L. & O. Oenema, 1995. Nitrous oxide fluxes from grassland in the Netherlands: II. Effects of soil type, nitrogen fertilizer application and grazing. *European Journal of Soil Science,* 46: 541-549.

Velthof, G. L., Brader, A. B. & O. Oenema, 1996. Seasonal variations in nitrous oxide losses from managed grasslands in the Netherlands. *Plant and Soil,* 181: 263-274.

Virtanen, K., Hänninen, P., Kallinen, R.-L., Vartiainen, S., Herranen, T. & R. Jokisaari, 2003. Suomen turvevarat 2000. Summary: The peat reserves of Finland in 2000. Report of Investigation 156, *Geological Survey of Finland*: 1-101.

Vitt, D. A., Halsey, L. A., Bauer, I. E. & C. Campbell, 2000. Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. *Canadian Journal of Earth Science*, 37: 683-693.

Wallén, B., 1986. Above and below ground dry mass of the three main vascular plants on hummocks on a sub-arctic peat bog. *Oiko,s* 46: 51-56.

Weslien, P., Kasimir Klemedtsson, Å., Maljanen, M., Silvola, J., Martikainen, P-J., Priemé, A. & L. Klemedtsson, (in prep.). Greenhouse gas emissions from organic soils reforested with birch in Sweden and Finland.

Zoltai, S. C., Pollett, F. C., Jeglum, J. K. & G. D. Adams, 1975. Developing a wetland classification for Canada. *In*: Bernier B. & Winget, C.H. (eds.), *Proceedings, 4th North American Forest Soils Conference*, Laval University Press, Quebec, pp. 497-511.

WEBSITES:

http://www.internat.naturvardsverket.se
http://www.peatsociety.fi/
http://www.ramsar.org/cop7_nr_albania.htm
http://www.worldenergy.org/wec-geis/publications/reports/ser/peat/peat.asp
www.worldenergy.org